

**Florida State University Study on Nutrient Removal in Onsite Sewage Treatment  
and Disposal Systems in Karst Areas**

Prepared for:

Florida Department of Health  
Bureau of Onsite Sewage Programs  
4052 Bald Cypress Way BIN #A-08  
Tallahassee, Fl 32399-1713

Prepared by:

Department of Oceanography  
Florida State University  
Tallahassee, Florida 32306-4320  
Phone 850-644-6700

**Executive Summary**

Please note that this report is preliminary. The final nutrient data set was received from the lab on 6/22/2009 and this report was finalized on 6/30/2009 6 business days later. Preliminary conclusions are as follows:

1. The *Hydro-action* Nutrient Reducing Treatment System installed in 2004 at the Magnolia II campground exhibited total N-reduction ranging from 60 to 88%.
2. At Hickory Campground the *Micro-fast* Nutrient Reducing Treatment System exhibited Total N reduction ranging from 22 to 75%. This system was installed in March, 2009 and may have been still developing.
3. N-concentrations in campground wells were significantly lower than during phase I of the study, however the campgrounds were closed from April 13 to 29 due to flooding. Visitation was significantly reduced from May through early June

following the flood due to less than optimum conditions at the Park. Therefore nutrient loading was attenuated during this period by a lack of visitors.

## **1. Introduction**

The Suwannee River Water Management District (SRWMD) has determined that the level of total nitrogen exceeds the water quality standard for surface waters in many parts of the Suwannee River. Studies by the US Geological Survey identified that onsite sewage treatment and disposal systems (OSTDS) contribute to the nutrient load to the Suwannee River basin. The Lower Suwannee River is listed as impaired in regard to nutrients and dissolved oxygen.

The Lower Suwannee River Basin is underlain by Ocala limestone, which is karst in nature. Karst geology is typified by the presence of solution channels, sinkholes and springs that are formed when acidic rainfall dissolves the underlying calcium carbonate bedrock. These features have been shown to rapidly transport contaminants to the underlying groundwater. Phase I of the study demonstrated that rapid transport of nutrients from septic systems located at Manatee Springs State Park to the groundwater. The conventional systems studied in Phase I have been replaced with onsite nutrient reducing treatment systems (NRTSs) for Phase II.

The goal of this study is to assess the performance of two NRTSs. Since karst conditions occur throughout most of Florida, the results of the study will shed light on the effectiveness of NRTSs in much of the state. A secondary goal is to establish the relation of the observed nutrient concentrations in the groundwater to the septic effluent concentrations.

The conventional septic tanks at both campgrounds were both converted into the initial or trash tank, where the raw sewage flows into the system. At Hickory, a MicroFast system was installed in the treatment tank. The system was installed on 23 March 2009. In this system a blower aerates the sewage in the treatment tank and a fixed media provides surface area for nitrifying bacteria to grow. The sewage then flow into a pump tank. At Magnolia II, a Hydroaction system was installed. This system was installed in 2004. In this system, the sewage flows from the trash tank into the treatment tank where a blower aerates the sewage. The nitrifying bacteria grow in solution without

any fixed media. After the treatment tank, the sewage flows into a pump tank. At both sites, the original drain fields were replaced with mounded drain fields away from the bathhouses. A valve was installed into the plumbing to allow the effluent to be re-directed to the original drain fields for this study. The original drainfields were surrounded by a series of monitoring wells.

It should be noted that the sampling of the monitoring wells was affected by a flood at the Park during the sampling interval. This flooding had two effects. First, it likely flushed the groundwater in the wells, lowering nutrient values. This was especially true for the Magnolia campground bathhouse where the drainfield was standing in water. Second, the park campgrounds were closed from April 13 to April 29, 2009. Thus the septic systems were not in use during this period. Swimming was not opened at the Spring until May 7<sup>th</sup>. So even following reopening of the campground, visitation at the park was very light. During our June sampling event, the campgrounds were only about half full.

Current laws governing the installation of OSTDSs in the 10-year floodway require that alternative disposal techniques be used. A common method of complying with this provision is to install an aerobic treatment unit prior to the drain field of the OSTDS. While these systems enhance pathogen reduction, reduce total suspended solid and biological oxygen demand; they convert nitrogen to the nitrate form, which is highly mobile in the environment. This form of nitrogen, in combination with other nutrients increases the likelihood of eutrophication in waterways.

## **2. Methods**

### *2.1 Study Design*

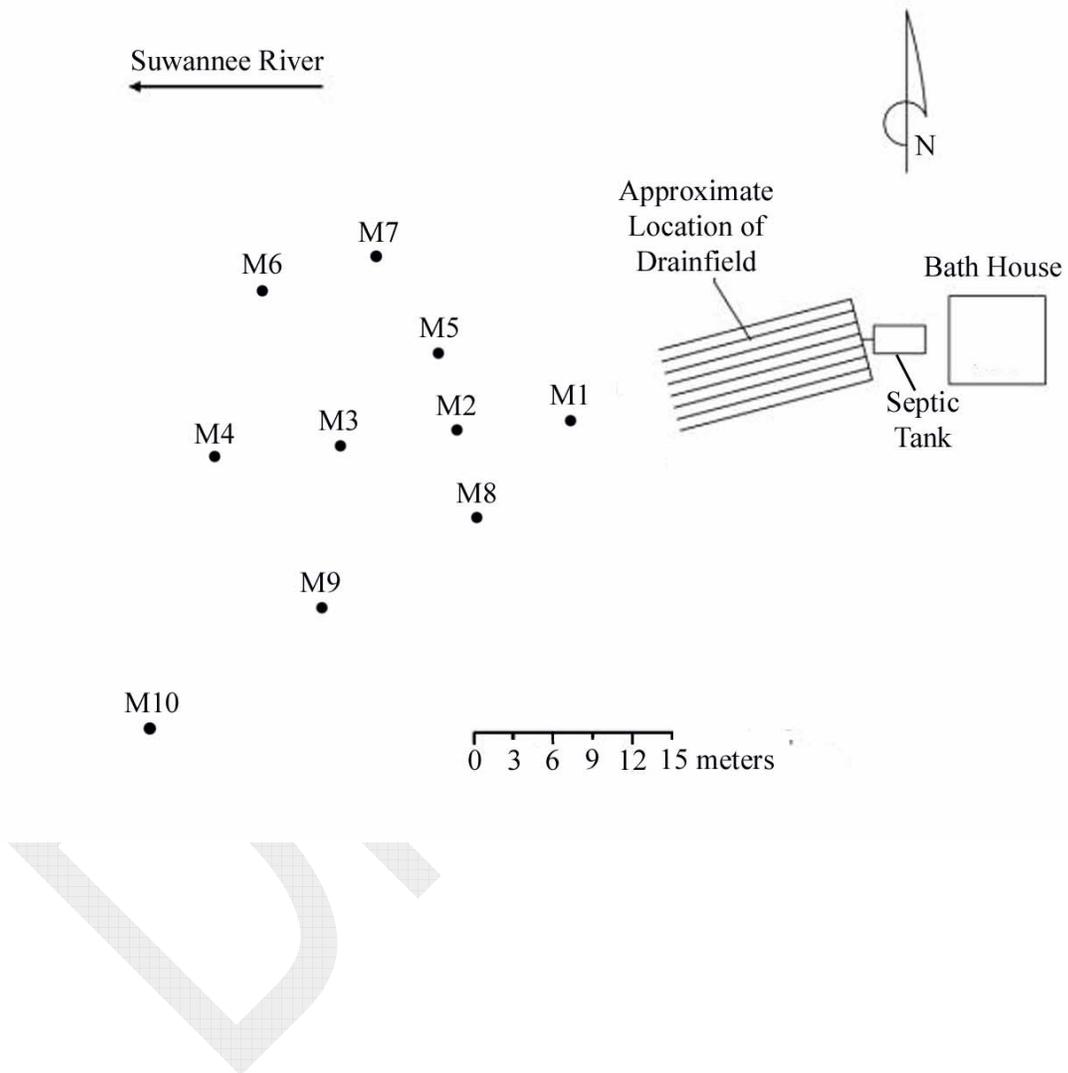
#### *Environmental sampling*

The conventional septic systems at Hickory and Magnolia were replaced by NRTS and the effluent pumped to new drain fields. A valve was installed so that the effluent could be re-directed to the original drain fields for this study. The first sampling event was timed to occur before the effluent was re-directed to the original drain fields in order to access the water quality in the wells surrounding the drain fields prior to the re-introduction of effluent. Three more sampling events occurred after effluent was diverted back to the original drain fields instrumented with the wells from Phase I. The same

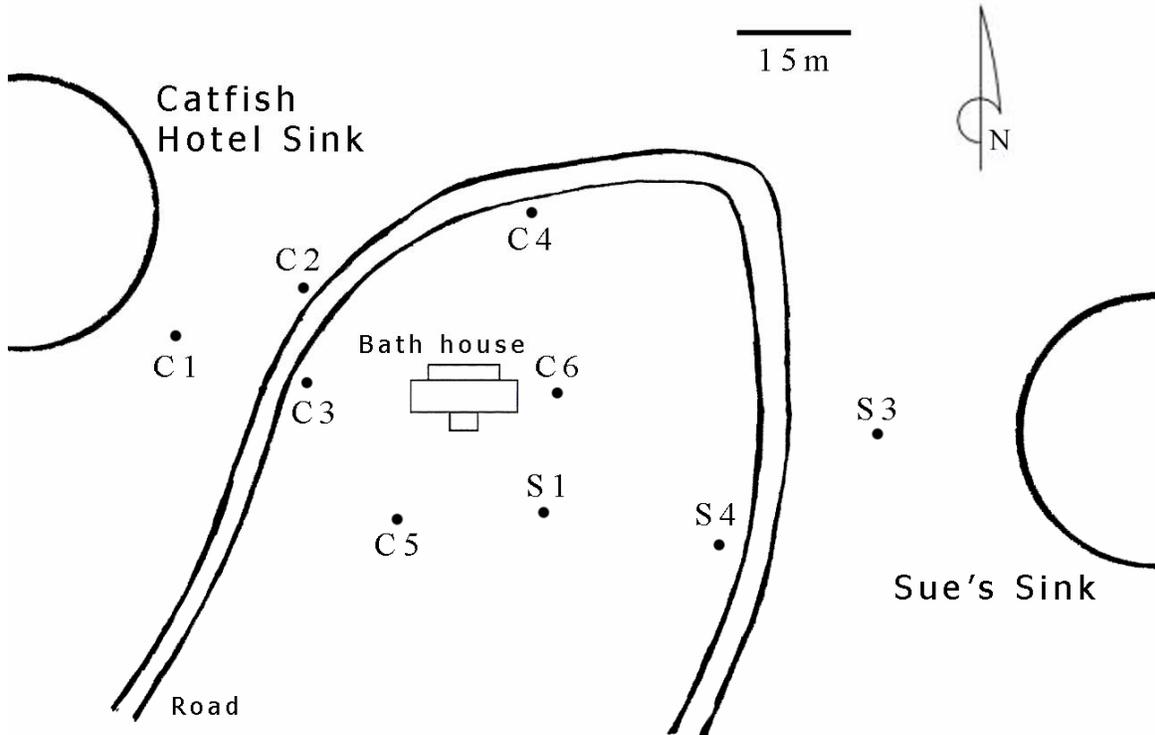
parameters measured in Phase I, nitrite, nitrate, ammonia, TKN, total phosphorus and fecal coliforms were measured in this study. Additionally, a YSI was used to collect field water quality data. The tracers released in Phase I were analyzed in samples from the first two sampling events occurring after the systems were finalized.

At Magnolia II, 8 wells were sampled, with two wells M4 and M10 being omitted (Figure 1). These wells were shown by the nutrient and tracer data to be outside the septic plume in Phase I. At Hickory 5 of the 10 monitoring wells were sampled. C6, one of the wells shown to elevated nitrate concentrations in Phase I, was damaged during construction of the new septic system. The 5 wells with the highest average nitrate concentrations in Phase I, S1, S2, C3, C4 and C5 were sampled in this study (Figure 2). The nitrate concentrations averaged less than 0.5 mg/L in the other 4 wells. One background well from Phase I was sampled in this study. The second background well from Phase I was substituted by SRWMD #4. The background well MB2 was shown to have elevated nitrate levels and therefore not an appropriate background sampling station. In addition to the groundwater samples, influent and effluent samples were taken from the two NRTS.

**Figure 1. Magnolia site behind the Magnolia II campground bathhouse. Well M1 is in the lower portion of the drain mound slope. Wells M10, M4, M6 are just in front of the cypress marsh adjacent to the Suwannee River.**



**Figure 2. Study site at Hickory campground. Well S1 is installed in a Paleo-sink hole as determined by the GPR study. S2 was installed at the end of a drainfield line. C1 was installed on the lip of the slope leading down to Catfish Hotel Sink. S3 was installed on the lip of the slope leading to Sue Sink.**



### *Performance assessment*

To assess diurnal and daily variability of performance, multiple samples of the influents and effluents were taken over a period of four consecutive days, consisting of both grab and 24 hour composite samples. The last environmental sampling event occurred on the last day of the performance assessment. YSI field measurements were also taken in addition to the same nutrients analyzed in the ground water samples.

### *2.2 Monitoring Well Sample Collection*

All samples were collected using a submersible purge pump. At least three well volumes were pumped prior to any sampling. Samples for nutrients and fecal coliforms were collected from the wells in containers provided by the analytical laboratory. Sulfur hexafluoride samples were collected in 30-mL serum vials. The vial was allowed to overflow for at least three bottle volumes, and was then sealed with a rubber septum and a crimp cap. Fluorescein samples were collected and stored in 100-mL amber polycarbonate containers.

### *2.3 Septic System Sample Collection*

Septic influent samples were taken from the first tank of the two systems; also know as a trash tank. At each campground, the original conventional septic tank served as the trash tank. A tube was placed in the filter chamber at the trash tank outflow pipe. Some treatment does occur in a conventional septic system, so these samples are a low estimate for the system inputs.

Septic tank effluent (STE) was sampled from a clean out between the pump tank and the drain field at the Hickory site. To take a sample the pump was manually turned on and a sample vessel was used to catch the flowing water. YSI measurements were taken by placing the probe into the collection vessel. At the Magnolia II site, the STE was sampled directly from the pump tank as no cleanout was available post pump tank. A peristaltic pump was used to take the sample through weighted tubing placed several inches below the effluent surface.

Only two automatic composite samplers were available at the time of the performance assessment. The other two composite samples consisted of 4 sub samples taken every 6 hours, held on ice and combined to make a composite sample.

#### *2.4 Nutrient and Fecal Coliform Analysis*

Samples were transported on ice to the laboratory and analyzed for fecal coliforms (SM9222D), total phosphorus (EPA 365.3), total ammonia (EPA 350.2), total Kjeldahl nitrogen (TKN) (EPA 351.3), nitrite-nitrogen (SM 4500NO2B), and nitrate-nitrogen (EPA 353.3).

#### *2.4 Sulfur Hexafluoride Sample Analysis*

Sulfur hexafluoride samples were extracted as described by Dillon et al. (1999) and Harden et al. (2003). A small headspace of 4 ml of ultra-high purity nitrogen was added to the samples using a syringe. Simultaneously, 3 mL of water from the sample was removed and discarded to allow room for the headspace. The serum vials were slightly over-pressurized with 1 cc of nitrogen to allow for several injection volumes (100 uL or less) for the gas chromatograph (GC). After shaking for at least two minutes, this method extracts 95+% of the SF<sub>6</sub> from a water sample. The lower limit of this technique is 0.1 pM (Dillon et al., 2000). Samples were analyzed with a Shimadzu model 8A gas chromatograph equipped with an electron capture detector as described in Harden et al. (2003). Head space concentrations in ppmv (parts per million by volume, = μL/L) of SF<sub>6</sub> were determined by reference to a 1.04 ppmv standard (Scott Specialty Gases). Headspace concentrations were converted to dissolved concentrations in pM.

#### *2.5 Fluorescein Dye Analysis*

The fluorescein samples were analyzed using a Turner Designs TD-700 Fluorometer, which provides exact concentrations after calibration. The fluorometer used a 10-089 blue mercury vapor lamp, 10-105 excitation filter (486 nm), and 10-109R-C emission filter (510-700 nm), as specified by the manufacturer. The fluorometer was initially calibrated using fluorescein standards made using DI water in the laboratory with

a lower detection limit of 0.0005 mg/L. Calibration was checked several times daily by use of solid state standards.

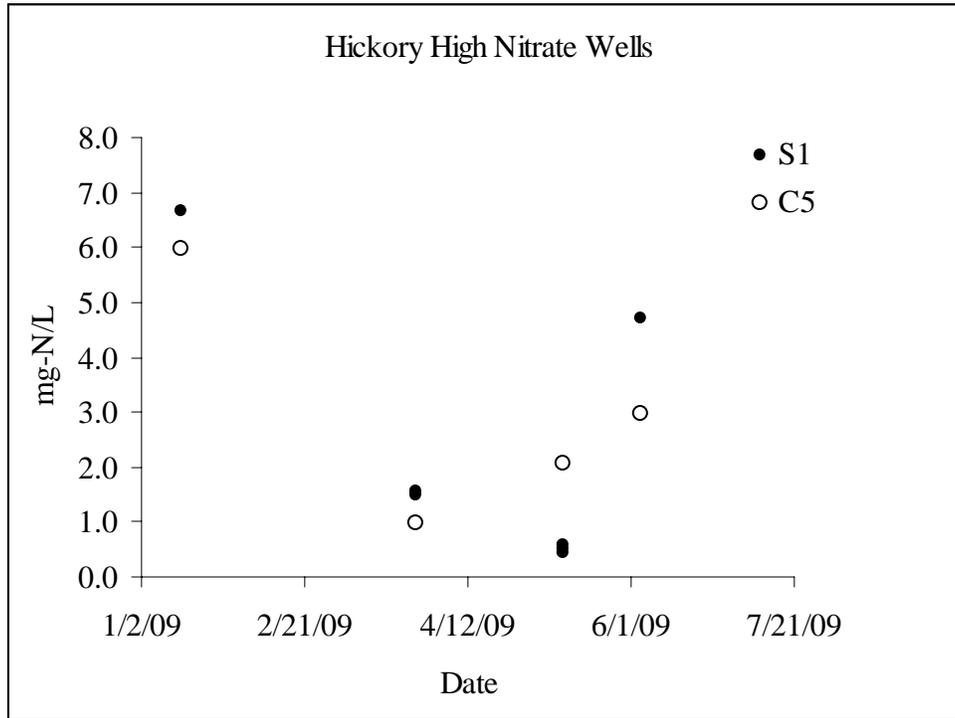
### **3. Results & Discussion**

#### ***3.1 Environmental Monitoring***

##### *Hickory Campground*

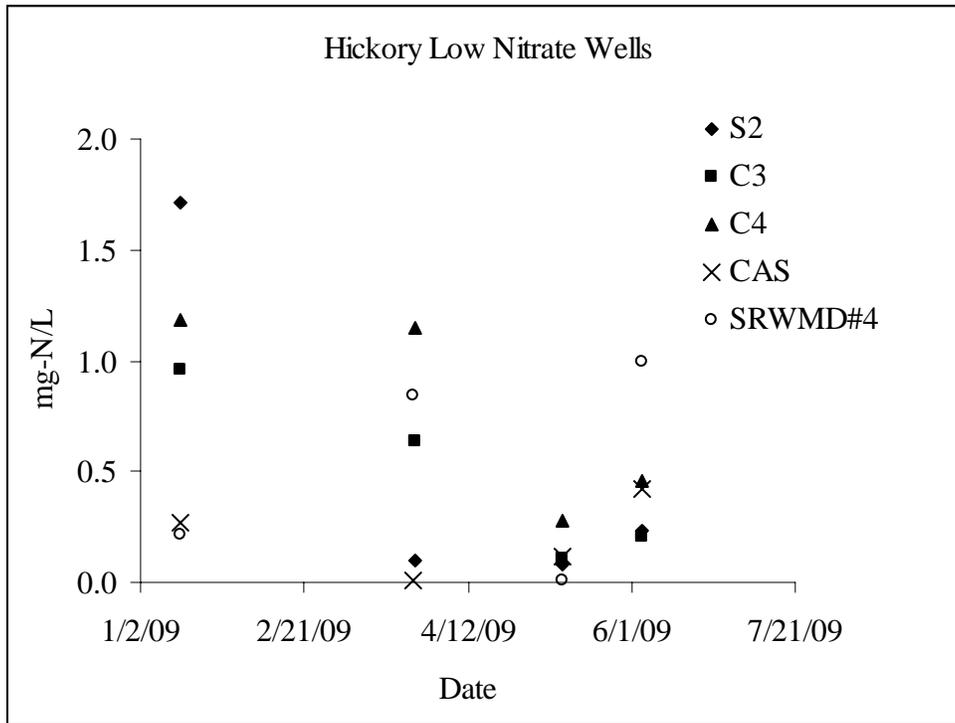
In all 5 wells at Hickory campground the highest nitrate levels were observed in samples from the first sampling event on 01/14/09 (Figure 3). This sampling event was intended to indicate the water quality surrounding the drain fields before the effluent was re-directed. Unfortunately, issues with the new systems required the old drain fields to be used during repairs prior to the start of the study. Wells S1 and C5 had the highest observed nitrate concentrations observed at the Hickory site. Nitrate concentrations were lower in the next two sampling events on 3/24/09 and 05/11/09. On the last sampling event on 06/04/09, the concentrations increased.

**Figure 3. Nitrate concentrations from wells S1 and C5 taken on 01/14/09, 3/27/09, 05/11/09 and 06/04/09. The Suwannee River flooded in early April closing the State Park campgrounds from April 13 to April 29.**



On 01/14/09, the nitrate concentrations in wells S2, C3, and C4 were above the levels found in the two background wells,  $0.20 \pm 0.18$  mg-N/L in CAS and  $0.52 \pm 0.48$  mg-N/L in SRWMD #4, yet much lower than S1 and C5 (Figure 4). Concentrations decreased on 3/27/09, but were still above background levels in wells C3 and C4. In the last two sampling events on 5/11/09 and 6/4/09 wells S2, C3, and C4 were near or below background well concentrations.

**Figure 4. Nitrate concentrations from wells S2, C3, C4 and background wells CAS and SRWMD #4 taken on 01/14/09, 3/27/09, 05/11/09 and 06/04/09. . The Suwannee River flooded in early April closing the State Park campgrounds from April 13 to April 29.**



Total phosphorus in wells S2, C3, C4, and C5 were less than half the concentration of  $0.13 \pm 0.14$  mg-N/L (n=8) observed in the two background wells. The total phosphorus observed in well S1 was  $0.76 \pm 0.40$  mg-N/L (n=4). The highest concentration of 1.32 mg-N/L was the only sample above background levels.

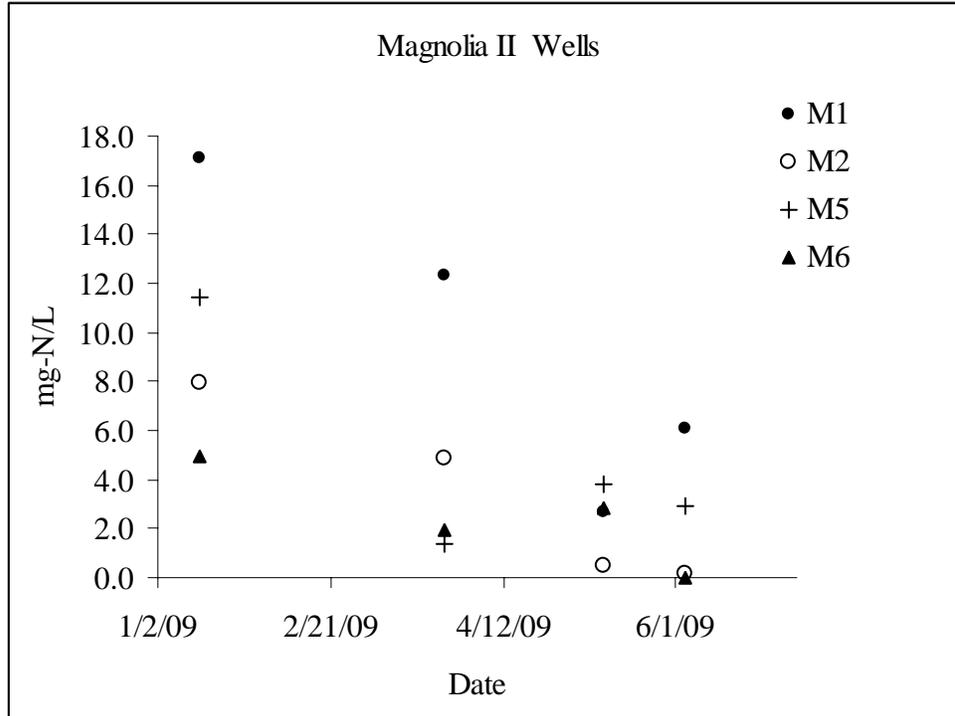
Fecal coliform was found sporadically throughout the well field. All wells had zero colonies in two of the four sampling events. The highest counts were 120 colonies/100 ml in C5, 76 colonies/100 ml in C4, and 18 and 25 colonies/100 ml in well S2. No fecal coliform colonies were found in the background wells

*Magnolia II Campground*

In wells M1, M2, M5 and M6 the highest concentrations were observed on the first sampling event, 01/14/09 and then concentrations decreased over the next three sampling events. In well M2, the nitrate concentrations observed on the last two

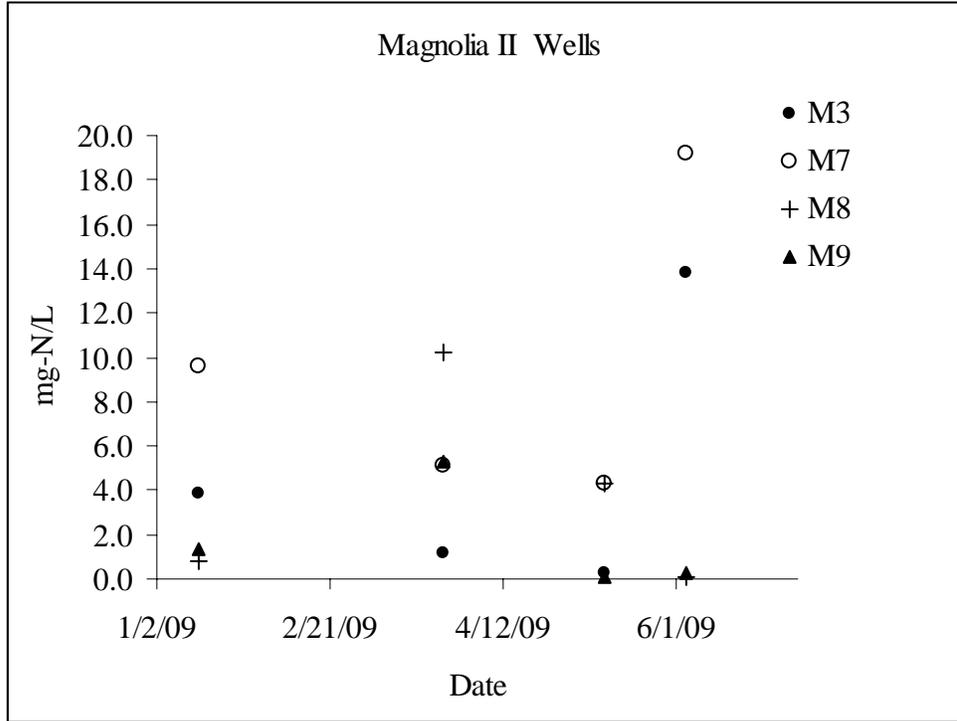
sampling events were below the levels in the background wells. Well M6, nitrate concentrations was also below background levels on the last sampling event on 6/47/09.

**Figure 5. Nitrate concentrations from wells M1, M2, M5 and M6 sampled on 01/14/09, 3/26/09, 05/11/09 and 06/04/09. The Suwannee River flooded in early April closing the park on April 13<sup>th</sup> and flooding the well field at Magnolia II. .**



On 01/14/09, nitrate levels observed in wells M3 and M7 were above background well concentrations, and wells M8 (0.77 mg-N/L) and M9 (1.36 mg-N/L) were near background levels. On 3/26/09, the nitrate concentrations in M8 and M9 increased and decrease in wells M3 and M7 to near background concentrations. On 05/11/09, concentrations of nitrate decreased to near background levels in wells M3 and M10, and also decreased in M7 and M8 but were well above background well concentrations. The highest concentrations of nitrate in wells M7 and M3 were observed on 06/04/09 and concentrations in M8 and M9 were below background well concentrations.

**Figure 6. Nitrate concentrations from wells M3, M7, M8 and M9 were taken on 01/14/09, 3/26/09, 05/11/09 and 06/04/09. The Suwannee River flooded in early April closing the State Park on April 13<sup>th</sup> and flooding the well field at Magnolia II. Campground usage was still below normal on the 05/11/09 and 06/04/09 sampling events.**



Total phosphorus in the wells at magnolia were between a low of  $0.09 \pm 0.03$  mg-P/L (n=4) in well M9 to a high of  $0.34 \pm 0.04$  mg-P/L (n=4) in well M2. All wells, except M9, were above the concentrations observed in the background wells,  $0.13 \pm 0.14$  mg-P/L (n=8).

Fecal coliform was found sporadically throughout the well field. All wells had zero colonies in at least one of the four sampling events. The highest counts were 93 colonies/100 ml in M9, 67 and 32 colonies/100 ml in M2, and 32 colonies/100 ml in well M6. No fecal coliform colonies were found in the background wells.

#### Comparison with Phase 1

At Hickory campground the wells most impacted by the septic system in Phase 1, S1 and C5 had the highest nitrate concentrations in Phase 2. The concentrations in these

wells were much lower Phase 2 than in Phase 1. The concentrations in wells S2, C3, and C4 had nitrate concentrations that were very similar to those in Phase 1 (Table 1).

At Magnolia II campground, the nitrate concentrations were much lower in Phase 2 than in Phase 1 (Table 1). In the weeks prior to the sampling event of 05/11/09, the well field was flooded from the Suwannee River. This flooding appears to have washed out much of the nitrate as indicated by the relatively low nitrate concentrations compared to the other sampling events. On 06/04/09, concentrations are again higher in many wells, especially M7, M3, M1 and M5 (Figures 5 and 6). Indicating these wells are most effected by the septic system at Magnolia II. The flow of septic effluent was altered during Phase 2. During construction of the NRTS at Magnolia the drain field mound was re-shaped. This is thought to have changed the flow away from well M8, which had low nitrate concentrations in the last two sampling events. Additionally, the septic effluent was pumped into the drain field, in the conventional system the effluent was gravity feed, likely changing the dispersal pattern in the drain filed mound.

**Table 1. The high and average nitrate concentrations from both Phase 1 and Phase 2 at Hickory campground. Nitrate concentrations are given in mg-N/L. The background well SRWMD #4 in Phase 2 was substituted for MB2 from Phase 1. Note that concentrations were generally greater in Phase I of the study. Interpretation of the data is not straight-forward however, due to flooding at the site during the middle of the study. We recommend re-sampling the groundwater wells.**

Well ID	High NO <sub>3</sub> Phase 1	Average NO <sub>3</sub> Phase 1		High NO <sub>3</sub> Phase 2	Average NO <sub>3</sub> Phase 2
	<b>Background</b>				
CA1	0.15	0.07 ± 0.06		0.42	0.20 ± 0.18
MB2	1.61	0.61 ± 0.74	SRWMD# 4	1.00	0.52 ± 0.48
	<b>Hickory</b>				
S1	56.1	21.9 ± 16.1		6.67	3.35 ± 2.84
S2	1.1	0.58 ± 0.38		1.71	0.53 ± 0.79
C3	1.1	0.59 ± 0.30		0.96	0.48 ± 0.40
C4	1.23	0.71 ± 0.33		1.18	0.77 ± 0.47
C5	41.3	11.7 ± 14.7		5.96	2.99 ± 2.14
	<b>Magnolia</b>				
M1	49.8 ± 18.8	29.4 ± 9.8		17.14	10.71 ± 6.44
M2	63.2	22.8 ± 17.5		8.15	3.36 ± 3.70
M3	33.9	14.9 ± 11.2		13.78	4.76 ± 6.21
M5	62.6	26.1 ± 15.2		11.47	4.89 ± 4.50
M6	51.4	17.5 ± 14.5		4.92	2.42 ± 2.04
M7	63.3	19.4 ± 17.7		19.22	9.54 ± 6.86
M8	54.6 ± 14.7	30.1 ± 17.1		10.26	3.87 ± 4.65
M9	35.9	6.9 ± 12.6		5.28	1.75 ± 2.42

### Tracers

Samples for tracers released in Phase I were sampled collected during the sampling events of 03/26-27/09 and 05/11/09. Neither tracer was detected in any of the samples, indicating that the tracers are no longer present in the groundwater.

### 3.2 Septic System Performance Assessment

The septic influent was sampled from the filter chamber in the outflow of the first tank in the system or trash tank. At Magnolia II the 4 trash tank composite samples were taken using an automatic sampler. The first composite trash tank sample at Hickory employed a composite sampler. The automatic composite sampler took a sample every three hours for 24 hours. Composite trash tank samples 2-4 at Hickory were manually combined and consisted of 4 100 ml samples, each taken 6 hours apart and stored on ice. Septic tank effluent (STE) was sampled directly form the pump tank at Magnolia II, with the first composite sample manually combined and composite samples 2-3 were sampled with an automatic sampler. At Hickory, the STE was sampled from a cleanout between the pump and drain field. All Hickory composite STE samples were manually combined. In addition to the composite samples, 2 grab samples were taken during each of 24 hour period over the 4 days. On the last day, the final environmental sampling event was performed which included an additional grab sample from each septic sampling station.

At Magnolia II, composite and grab sample results were very similar. The Magnolia II trash tank samples had total N values of  $143.33 \pm 25.27$  mg-N/L (n=4) for the composite samples and for the grab samples  $141.61 \pm 27.89$  mg-N/L (n=4). The STE values for total N at Hickory were also in close agreement between the 4 composite samples ( $44.52 \pm 12.53$  mg-N/L) and the 9 grab samples ( $42.49 \pm 8.24$  mg-N/L). The % reduction of total nitrogen calculated from the composite samples was 68.9% and from the grab samples 70.0%.

Higher TN concentrations were observed in the STE at Magnolia II on the environmental sampling events of 03/26/09 and 05/11/09, yet the influent TT samples were lower, yielding higher % reduction of TN (Table 2).

**Table 2. Nutrient results from the performance assessment composite and grab samples at Magnolia campground. TT indicates a trash tank sample, serving as influent concentrations. STE indicates a septic tank effluent sample. The composite samples are 24 hour samples taken over a 4 day period from 05/31/09 to 06/04/09. The grab samples include the septic samples from the final environmental sampling event. The septic grab samples from the other 3 environmental sampling events are also given. The % N reduction by the system is highlighted in yellow for the different sampling dates.**

			Nitrate+Nitrite	Ammonia		TKN	TN
			mg/L	mg/L	mg/L	mg/L	mg/L
<b>5/31 to 6/4</b>							
<b>Mag TT</b>	<b>Comp</b>	<b>Ave.</b>	<b>6.45</b>	<b>97.27</b>	<b>136.88</b>	<b>143.33</b>	<b>4.05</b>
		<i>Stdev</i>	12.60	20.91	27.38	25.27	0.30
<b>Mag STE</b>	<b>Comp</b>	<b>Ave.</b>	<b>18.39</b>	<b>21.38</b>	<b>26.13</b>	<b>44.52</b>	<b>3.47</b>
		<i>Stdev</i>	15.07	5.31	4.20	12.53	0.39
				<b>% TN Reduction</b>		<b>68.9</b>	
<b>5/31 to 6/4</b>							
<b>Mag TT</b>	<b>Grab</b>	<b>Ave.</b>	<b>3.59</b>	<b>103.31</b>	<b>138.02</b>	<b>141.61</b>	<b>3.85</b>
		<i>Stdev</i>	6.99	21.30	31.16	27.89	0.38
<b>Mag STE</b>	<b>Grab</b>	<b>Ave.</b>	<b>12.95</b>	<b>22.17</b>	<b>29.54</b>	<b>42.49</b>	<b>3.66</b>
		<i>Stdev</i>	10.18	6.56	14.74	8.24	0.30
				<b>% TN Reduction</b>		<b>70.0</b>	
<b>Mag STE</b>	<b>01/014/09</b>		<b>63.48</b>	<b>2.02</b>	<b>3.00</b>	<b>66.49</b>	<b>20.80</b>
<b>Mag TT</b>	<b>03/26/09</b>		<b>7.76</b>	<b>83.64</b>	<b>202.89</b>	<b>210.65</b>	<b>0.13</b>
<b>Mag STE</b>	<b>03/26/09</b>		<b>12.24</b>	<b>1.68</b>	<b>14.16</b>	<b>26.40</b>	<b>0.11</b>
				<b>% TN Reduction</b>		<b>87.5</b>	
<b>Mag TT</b>	<b>05/11/09</b>		<b>0.06</b>	<b>39.56</b>	<b>177.88</b>	<b>177.94</b>	<b>1.50</b>
<b>Mag STE</b>	<b>05/11/09</b>		<b>6.94</b>	<b>10.70</b>	<b>12.92</b>	<b>19.86</b>	<b>1.75</b>
				<b>% TN Reduction</b>		<b>88.8</b>	

At Hickory campground the difference between composite samples and grab samples was more pronounced. The trash tank composite samples had total nitrogen concentrations of  $79.83 \pm 17.64$  (n=4), while the total nitrogen in the grab samples,  $110.68 \pm 31.83$  mg-N/L (n=9), were more variable they were in the same range. The STE total nitrogen concentrations of the composite samples,  $61.34 \pm 9.97$  mg-N/L (n=4) and grab samples,  $50.09 \pm 20.79$  mg-N/L (n=9) were in good agreement. The % reduction of nitrogen by the system was less clear for the composite samples than the grab samples. The grab samples indicate a total nitrogen reduction of 54.7%. The % reduction of total nitrogen calculated from composite samples was less clear as the second set of composite samples yielded a higher effluent total nitrogen (61.0 mg-N/L) value than the effluent total nitrogen (52.5 mg-N/L), yielding a negative % reduction (Table 3). Note that this system was only installed on March 23, 2009.

**Table 3. Nutrient results from the performance assessment composite and grab samples at Hickory campground. TT indicates a trash tank sample, serving as influent concentrations. STE indicates a septic tank effluent sample. The composite samples are 24 hour samples. The grab samples include the septic samples from the final environmental sampling event. The septic grab samples from the environmental sampling events on 03/27/09 and 05/11/09 are also given. The % N reduction by the system is highlighted in yellow for the different sampling dates. The second set of composite samples, C2, had higher TN in the STE than the TT sample, yielding a negative % reduction.**

			Nitrate+Nitrite	Ammonia	TKN	TN	Total P
			mg/L	mg/L	mg/L	mg/L	mg/L
<b>5/31 to 6/4</b>							
<b>Hick TT</b>	<b>Comp</b>	<b>Ave.</b>	<b>0.52</b>	<b>65.85</b>	<b>78.41</b>	<b>78.93</b>	<b>3.50</b>
		<i>Stdev</i>	0.37	18.63	18.00	17.64	0.38
<b>Hick STE</b>	<b>Comp</b>	<b>Ave.</b>	<b>34.26</b>	<b>27.90</b>	<b>27.08</b>	<b>61.34</b>	<b>3.35</b>
		<i>Stdev</i>	18.79	17.04	15.44	9.91	0.42
				<b>% TN Reduction</b>		<b>22.3</b>	n=4
			<i>Without C2</i>	<b>% TN Reduction</b>		<b>30.7</b>	n=3
<b>5/31 to 6/4</b>							
<b>Hick TT</b>	<b>Grab</b>	<b>Ave.</b>	<b>3.08</b>	<b>80.54</b>	<b>107.60</b>	<b>110.68</b>	<b>4.05</b>
		<i>Stdev</i>	8.03	28.46	30.54	31.83	1.25
<b>Hick STE</b>	<b>Grab</b>	<b>Ave.</b>	<b>22.50</b>	<b>25.20</b>	<b>27.59</b>	<b>50.09</b>	<b>3.29</b>
		<i>Stdev</i>	21.41	16.92	19.40	20.79	0.32
				<b>% TN Reduction</b>		<b>54.7</b>	n=4
<b>Hick TT</b>	<b>03/27/09</b>		<b>0.12</b>	<b>61.70</b>	<b>173.31</b>	<b>173.43</b>	<b>13.50</b>
<b>Hick STE</b>	<b>03/27/09</b>		<b>2.42</b>	<b>42.34</b>	<b>106.70</b>	<b>109.13</b>	<b>12.99</b>
				<b>% TN Reduction</b>		<b>37.1</b>	n=4
<b>Hick TT</b>	<b>05/11/09</b>		<b>0.09</b>	<b>59.41</b>	<b>235.16</b>	<b>235.25</b>	<b>2.92</b>
<b>Hick STE</b>	<b>05/11/09</b>		<b>6.31</b>	<b>33.60</b>	<b>52.54</b>	<b>58.84</b>	<b>2.73</b>
				<b>% TN Reduction</b>		<b>75.0</b>	n=4

The field parameters measured with an YSI probe were very consistent over the course of the performance assessment. The influent trash tank samples were characterized by a high conductivity, low dissolved oxygen, and a highly negative oxidation/reduction potential compared to the STE samples with lower conductivity, higher dissolved oxygen, and a positive oxidation/reduction potential (Table 4).

**Table 4. Field parameters from the performance assessment of the septic systems conducted from 05/31/09 to 6/4/09.**

Site	Calc	Water T Celsius	COND μS/cm	COND mS/cm	Salinity ppt	%SAT %	DO mg/L	PH mg/L	ORP mg/L
<b>Hick TT</b>	<b>Ave.</b>	<b>25.62</b>	<b>1312.67</b>	<b>1.33</b>	<b>0.65</b>	<b>9.07</b>	<b>0.74</b>	<b>6.94</b>	<b>-141.55</b>
	<i>Stdev</i>	0.16	23.77	0.03	0.01	7.37	0.60	0.13	28.04
<b>Hick STE</b>	<b>Ave.</b>	<b>25.46</b>	<b>972.56</b>	<b>0.98</b>	<b>0.48</b>	<b>36.08</b>	<b>2.90</b>	<b>7.74</b>	<b>26.36</b>
	<i>Stdev</i>	0.49	82.65	0.09	0.04	10.73	0.85	0.21	17.69
<b>Mag TT</b>	<b>Ave.</b>	<b>27.15</b>	<b>1469.79</b>	<b>1.53</b>	<b>0.73</b>	<b>8.26</b>	<b>0.65</b>	<b>6.92</b>	<b>-142.48</b>
	<i>Stdev</i>	0.13	7.46	0.01	0.00	5.48	0.43	0.22	22.05
<b>Mag STE</b>	<b>Ave.</b>	<b>26.72</b>	<b>1123.13</b>	<b>1.16</b>	<b>0.55</b>	<b>41.64</b>	<b>3.32</b>	<b>6.82</b>	<b>19.53</b>
	<i>Stdev</i>	0.22	16.89	0.02	0.01	7.84	0.62	0.36	26.75

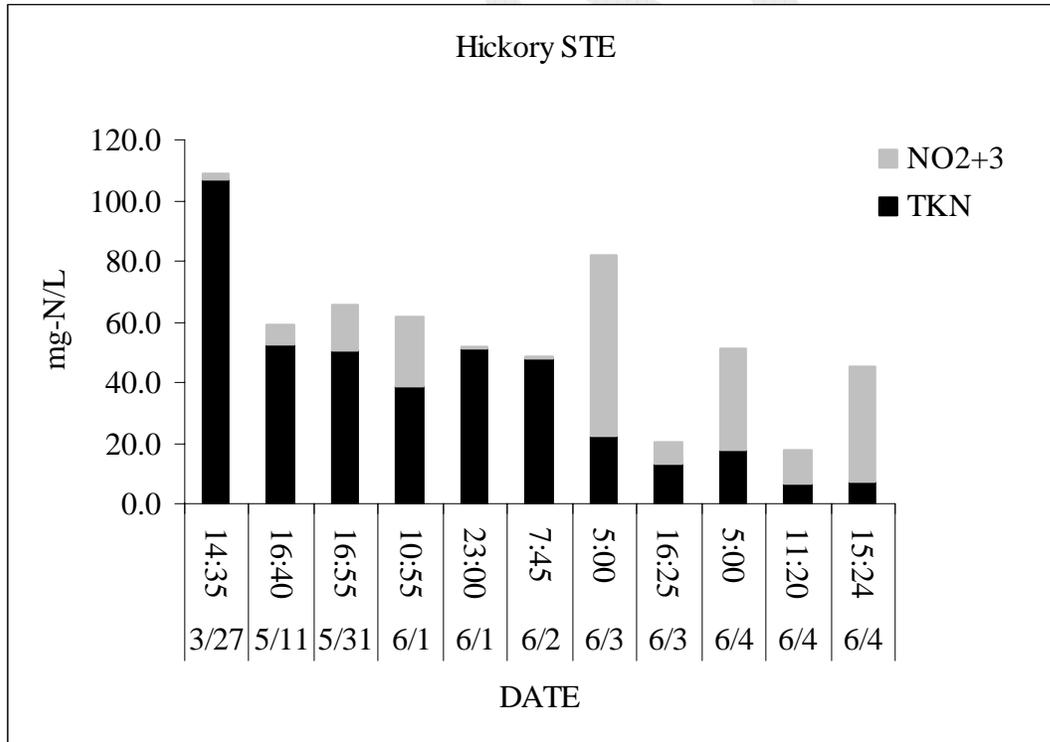
*Nitrification and De-nitrification in Septic Systems*

Nitrification occurs in the treatment tank of the NRTS by the addition of air into the septic effluent. In the two environmental STE grab samples on 03/27/09 and 05/11/09 and 5 grab samples of the performance assessment indicate that nitrification is limited, as the TKN values are greater than the nitrite + nitrate concentrations. The last 4 grab samples indicate that the majority of the TKN had been converted to nitrate. (Figure

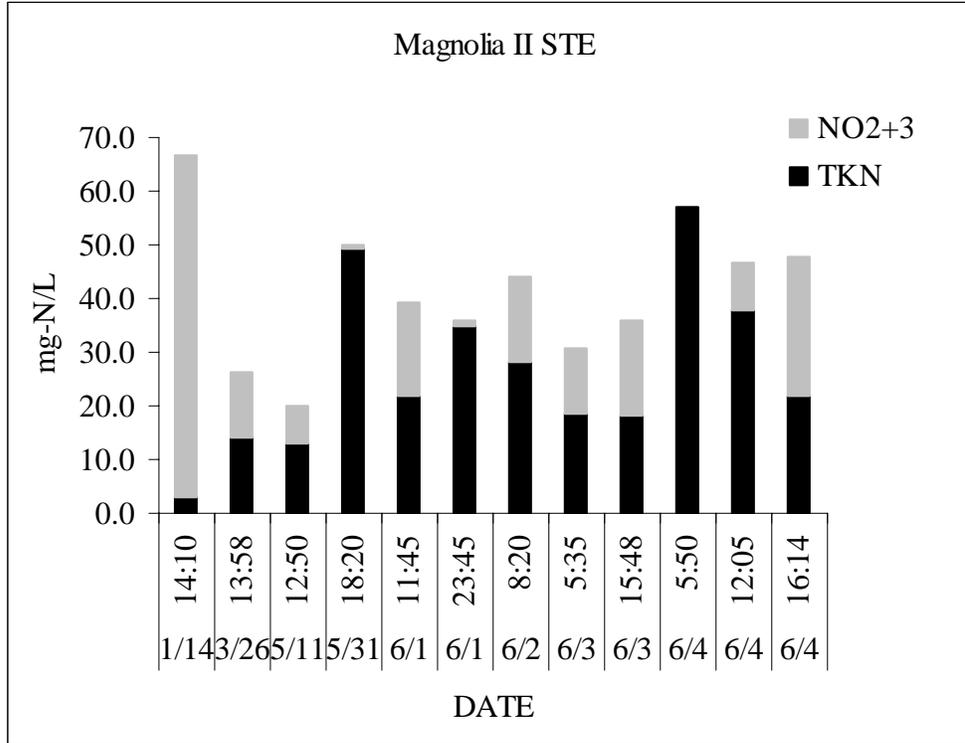
7). . Note the trend in the data towards lower Total N values and more oxidized forms of N in Figure 7. Nitrate concentration increase relative to TKN. Note that the system had only been running 3 days when the first sample was collected on 3/27.

At Magnolia II, nitrate seems to be limited in many of the samples, with TKN being the major N species in the effluent. A notable exception is the STE sample taken on 01/14/09, which is mostly nitrite + nitrate (Figure 8). This may be due to limitation of nitrification (nitrate formation) or it could have been due to consumption of nitrate (denitrification).

**Figure 7. TKN and nitrite + nitrate are given for the STE grab samples at Hickory campground. In samples with more TKN than nitrite + nitrate, nitrification is limited, or the else the nitrate was removed by denitrification. Note the trend in the data towards lower Total N values and more oxidized forms of N. Nitrate concentration increase relative to TKN. Note that the system had only been running 3 days when the first sample was collected on 3/27.**

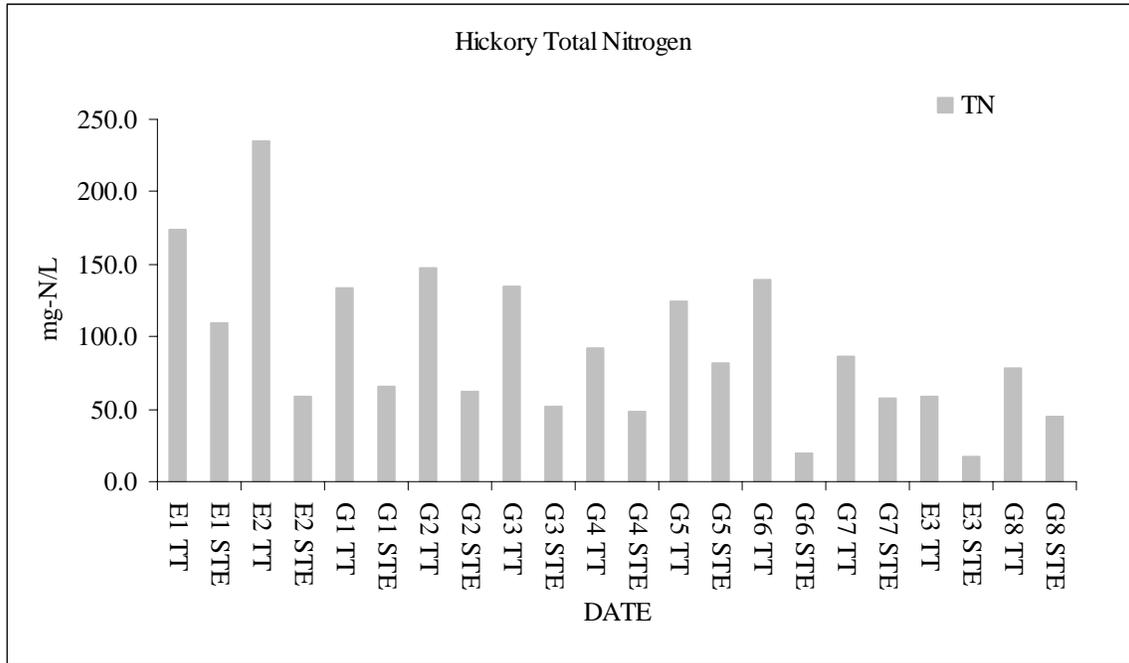


**Figure 8. TKN and nitrite + nitrate are given for the STE grab samples at Magnolia II campground. In samples with more TKN than nitrite + nitrate, nitrification is limited, or the else the nitrate was removed by denitrification.**

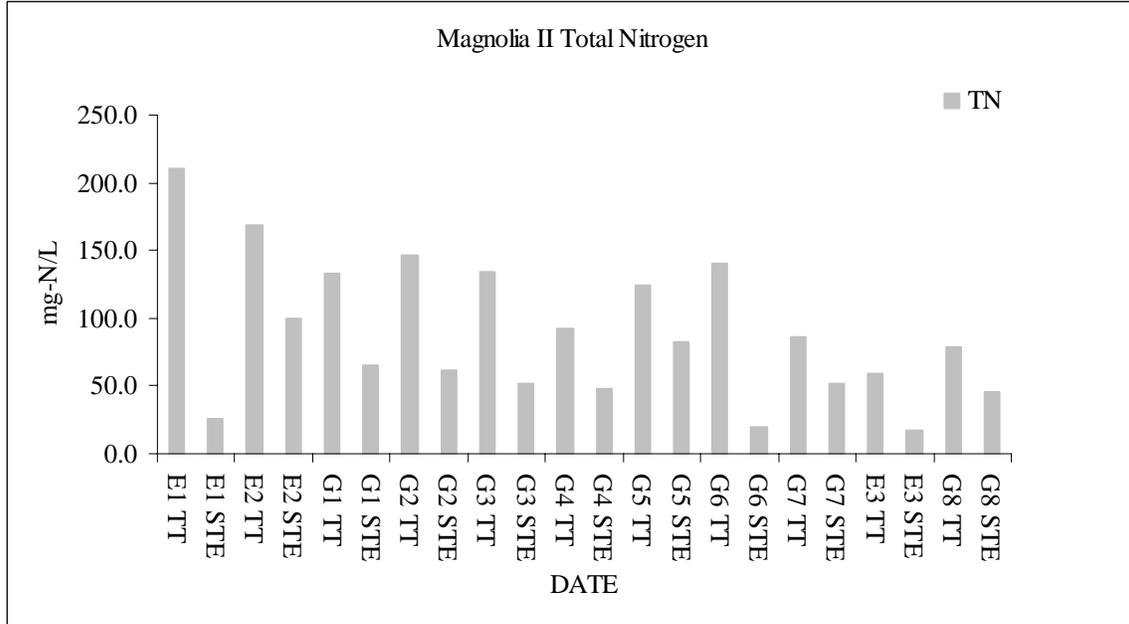


The significant reduction in total nitrogen during treatment, strongly suggests that de-nitrification is occurring (Figures 9 and 10). The dissolved oxygen numbers in the STE of approximately 40% saturation (Table 4) are not favorable for de-nitrification to occur. However micro-environments are likely to present which do allow for de-nitrification to occur.

**Figure 9. Total nitrogen concentrations of influent samples (TT) and effluent samples (STE) at Hickory campground. E are grab samples from the environmental monitoring and G are grab samples from the performance assessment. . Note that in every instance the TT bar is higher in concentration than the STE bar it is paired with. The amount of this decrease is the % N reduction in the system as reported in Table 3, % TN Reduction.**



**Figure 10. Total nitrogen concentrations of influent samples (TT) and effluent samples (STE) at Magnolia campground. E are grab samples from the environmental monitoring and G are grab samples from the performance assessment. Note that in every instance the TT bar is higher in concentration than the STE bar it is paired with. The amount of this decrease is the % N reduction in the system as reported in Table 2, % TN Reduction.**



Water meter data can be found in the appendix.

#### 4. Conclusions

Significant reduction of total nitrogen is occurring in both NRTS studied, greatly reducing the nutrient loading of the groundwater by the effluent from the two bath houses. At Magnolia, the % reduction in Total N ranged from 88 to 69%, while at Hickory it ranged from 22 to 75%. The Magnolia system, a hydroaction system, installed in 2004 seems to be operating quite well and reducing the N-loading at the site. It is unclear why Hickory has a lower level of performance. One possibility is that the Hickory site was problematic in its installation due to site limitations, and was finalized on 3/23, just prior to the 03/27/09 sampling event. However, as shown in Figure 7, the trend of the total N data was downward over the course of the study, and the nitrate content of the effluent increased relative to ammonia (TKN). This suggests that the system was coming on line. Issues with system have caused numerous delays in this

project and it may have been premature to evaluate the system at Hickory. Thus the current data set may not be the best possible for the evaluation of the performance of a MicroFast system.

Groundwater samples collected from wells were lower in the Phase II portion of this study, following the installation of the NRTS systems, relative to concentrations in Phase I. However interpretation of this result is somewhat tentative. The first sampling event was intended give nutrient concentrations in the groundwater prior the effluent being re-directed to the original drain fields, however effluent had been diverted to the drainfields in the vicinity of those wells prior to sampling. This was not supposed to have occurred. The flooding that occurred in the middle of the study also complicated the interpretation of the groundwater data as both campgrounds were closed and the Magnolia II well field submerged. The campgrounds were closed from April 13 to April 29, 2009. Thus the septic systems were not in use during this period. Swimming was not opened at the Spring until May 7<sup>th</sup>. So even following reopening of the campground, visitation at the park was very light. During our June sampling event, visitation was observed to be very light. We recommend re-sampling the groundwater wells during a period of active usage.

Appendix A Performance Assessment  
Field Data

Station	Date	Time <i>YSI</i>	Water T <i>YSI</i> Celsius	COND <i>YSI</i> µS/cm	COND <i>YSI</i> mS/cm	Salinity <i>YSI</i> ppt	DO %SAT <i>YSI</i> %	DO <i>YSI</i> mg/L	PH <i>YSI</i> mg/L	ORP <i>YSI</i> mg/L	PH <i>Probe</i> units
Hick STE	5/31/2009	17:06	27.13	1077	1.121	0.53	43.3	3.03	8.02	50.2	8.0
Hick STE	5/31/2009	22:56	25.48	1094	1.104	0.54	43.3	3.48	7.60	38.0	8.0
Hick STE	6/1/2009	4:55									7.7
Hick STE	6/1/2009	5:07	25.29	1117	1.124	0.55	44.2	3.36	7.15	13.0	7.7
Hick STE	6/1/2009	11:00	25.58	1076	1.088	0.53	49.3	3.98	7.07	63.1	8.1
Hick STE	6/1/2009	17:02	25.53	1053	1.064	0.52	53.7	4.39	7.99	59.1	8.0
Hick STE	6/1/2009	23:07	25.37	1015	1.022	0.50	39.1	3.19	8.09	29.9	7.9
Hick STE	6/2/2009	7:49	25.14	1003	1.006	0.49	32.1	2.64	8.29	14.9	7.3
Hick STE	6/2/2009	11:04	25.50	982	0.991	0.48	36.2	2.96	8.40	18.4	
Hick STE	6/2/2009	17:04	25.51	967	0.976	0.47	33.5	2.72	8.17	43.8	7.7
Hick STE	6/2/2009	23:03	25.43	944	0.951	0.46	38.2	3.12	8.20	10.3	7.7
Hick STE	6/3/2009	5:05	25.03	935	0.934	0.46	25.7	2.12	8.34	10.7	7.7
Hick STE	6/3/2009	11:03	25.73	922	0.935	0.45	21.6	1.75	8.57	8.5	7.6
Hick STE	6/3/2009	16:28	25.49	902	0.910	0.44	39.3	3.21	8.05	20.9	7.6
Hick STE	6/3/2009	17:03	25.44	900	0.908	0.44	29.9	2.44	8.14	6.4	7.6
Hick STE	6/3/2009	23:02	25.50	885	0.894	0.43	42.3	3.45	8.05	24.3	7.6
Hick STE	6/4/2009	5:07	24.62	885	0.879	0.43	15.5	1.27	7.93	26.0	
Hick STE	6/4/2009	11:24	25.41	874	0.881	0.43	45.0	3.68	7.77	26.3	
Hick STE	6/4/2009	15:30	25.15	875	0.878	0.43	17.2	1.42	7.70	10.6	7.6
	<b>average</b>		<b>25.46</b>	<b>972.56</b>	<b>0.98</b>	<b>0.48</b>	<b>36.08</b>	<b>2.90</b>	<b>7.97</b>	<b>26.36</b>	<b>7.74</b>
	<i>stdev</i>		<i>0.49</i>	<i>82.65</i>	<i>0.09</i>	<i>0.04</i>	<i>10.73</i>	<i>0.85</i>	<i>0.40</i>	<i>17.69</i>	<i>0.21</i>

Appendix A Performance Assessment  
Field Data (continued)

Station	Date	Time	Water T	COND	COND	Salinity	DO %SAT	DO	PH	ORP	PH
		<i>YSI</i>	<i>YSI</i> Celsius	<i>YSI</i> µS/cm	<i>YSI</i> mS/cm	<i>YSI</i> ppt	<i>YSI</i> %	<i>YSI</i> mg/L	<i>YSI</i> mg/L	<i>YSI</i> mg/L	<i>Probe</i> units
Hick TT	5/31/2009	17:23	25.62	1335	1.351	0.66	3.9	0.32	7.16	-203.0	7.2
Hick TT	5/31/2009	19:26	25.50	1330	1.342	0.66	4.0	0.33	6.98	-159.0	7.1
Hick TT	5/31/2009	22:37	25.51	1312	1.324	0.65	4.3	0.34	6.74	-177.8	7.1
Hick TT	6/1/2009	10:49	25.72	1271	1.289	0.63	2.9	0.24	7.00	-190.0	
Hick TT	6/1/2009	17:13	25.53	1261	1.275	0.63	2.9	0.24	7.03	-108.9	7.1
Hick TT	6/1/2009	23:18	25.39	1266	1.275	0.63	7.8	0.64	7.16	-139.0	6.9
Hick TT	6/2/2009	7:58	25.38	1295	1.304	0.64	4.1	0.33	7.51	-125.2	7.0
Hick TT	6/2/2009	11:15	25.58	1340	1.355	0.67	25.5	2.08	7.45	-130.6	
Hick TT	6/2/2009	17:13	25.54	1332	1.346	0.66	8.2	0.66	7.32	-136.2	6.9
Hick TT	6/2/2009	23:12	25.51	1324	1.337	0.66	6.8	0.55	7.28	-121.7	6.8
Hick TT	6/3/2009	5:13	25.40	1326	1.336	0.66	8.0	0.64	7.54	-119.0	6.9
Hick TT	6/3/2009	11:08	25.86	1323	1.345	0.66	4.3	0.35	7.55	-113.9	6.9
Hick TT	6/3/2009	16:42	25.73	1319	1.338	0.66	11.3	0.92	7.42	-116.0	6.9
Hick TT	6/3/2009	17:13	25.76	1323	1.342	0.66	9.2	0.75	7.44	-112.3	6.9
Hick TT	6/3/2009	23:15	25.86	1317	1.339	0.66	17.5	1.42	7.40	-164.0	6.8
Hick TT	6/4/2009	5:15	25.81	1323	1.343	0.66	11.4	0.93	7.23	-156.8	
Hick TT	6/4/2009	11:31	25.85	1309	1.330	0.65	27.4	2.22	6.98	-146.8	6.8
Hick TT	6/4/2009	15:36	25.63	1322	1.339	0.66	3.8	0.31	6.97	-127.7	6.8
Hick TT	<b>average</b>		<b>25.62</b>	<b>1312.67</b>	<b>1.33</b>	<b>0.65</b>	<b>9.07</b>	<b>0.74</b>	<b>7.23</b>	<b>-141.55</b>	<b>6.94</b>
	<i>stdev</i>		<i>0.16</i>	<i>23.77</i>	<i>0.03</i>	<i>0.01</i>	<i>7.37</i>	<i>0.60</i>	<i>0.24</i>	<i>28.04</i>	<i>0.13</i>

Appendix A Performance Assessment  
Field Data (continued)

Station	Date	Time	Water T	COND	COND	Salinity	DO %SAT	DO	PH	ORP	PH
		<i>YSI</i>	<i>YSI</i>	<i>YSI</i>	<i>YSI</i>	<i>YSI</i>	<i>YSI</i>	<i>YSI</i>	<i>YSI</i>	<i>YSI</i>	<i>Probe</i>
			Celsius	µS/cm	mS/cm	ppt	%	mg/L	mg/L	mg/L	units
Mag STE	5/31/2009	18:07	26.73	1168	1.207	0.58	30.8	2.45	6.72	79.6	
Mag STE	5/31/2009	23:48	26.90	1146	1.189	0.57	43.0	3.44	6.27	31.3	7.0
Mag STE	6/1/2009	5:48	26.65	1142	1.178	0.56	32.1	2.56	6.41	10.9	
Mag STE	6/1/2009	5:55									7.2
Mag STE	6/1/2009	11:56	26.61	1132	1.166	0.56	39.6	3.08	6.07	42.4	7.2
Mag STE	6/1/2009	18:02	26.85	1124	1.164	0.55	34.9	2.79	6.87	55.1	7.1
Mag STE	6/1/2009	23:46	26.76	1120	1.158	0.55	36.9	2.95	6.88	-2.9	7.1
Mag STE	6/2/2009	8:21	26.58	1117	1.151	0.55	40.0	3.21	7.26	3.6	7.0
Mag STE	6/2/2009	15:14	27.15	1112	1.158	0.55	57.7	4.56	7.00	24.4	
Mag STE	6/2/2009	16:10									7.2
Mag STE	6/2/2009	18:02	27.18	1111	1.157	0.55	48.9	3.86	6.74	48.9	6.6
Mag STE	6/3/2009	5:34	26.74	1111	1.147	0.55	46.0	3.68	6.94	-3.3	6.6
Mag STE	6/3/2009	15:51	26.56	1110	1.143	0.55	34.7	2.77	6.88	16.7	6.5
Mag STE	6/3/2009	16:10									7.0
Mag STE	6/3/2009	18:25	26.70	1111	1.148	0.55	41.2	3.29	6.87	5.0	6.5
Mag STE	6/4/2009	5:50	26.49	1114	1.146	0.55	41.9	3.36	6.41	-10.4	6.3
Mag STE	06/04/09	12:04	26.50	1113	1.145	0.55	55.7	4.47	6.21	-2.5	6.2
Mag STE	6/4/2009	16:13	26.41	1116	1.146	0.55	41.2	3.3	6.31	-5.9	
Mag STE	<b>average</b>		<b>26.72</b>	<b>1123.13</b>	<b>1.16</b>	<b>0.55</b>	<b>41.64</b>	<b>3.32</b>	<b>6.66</b>	<b>19.53</b>	<b>6.82</b>
	<i>stdev</i>		<b>0.22</b>	<b>16.89</b>	<b>0.02</b>	<b>0.01</b>	<b>7.84</b>	<b>0.62</b>	<b>0.35</b>	<b>26.75</b>	<b>0.36</b>

Appendix A Performance Assessment  
Field Data (continued)

Station	Date	Time	Water T	COND	COND	Salinity	DO %SAT	DO	PH	ORP	PH
		<i>YSI</i>	<i>YSI</i>	<i>YSI</i>	<i>YSI</i>	<i>YSI</i>	<i>YSI</i>	<i>YSI</i>	<i>YSI</i>	<i>YSI</i>	<i>Probe</i>
			Celsius	µS/cm	mS/cm	ppt	%	mg/L	mg/L	mg/L	units
Mag TT	5/31/2009	18:44	27.46	1458	1.527	0.73	1.8	0.14	6.93	-177.0	7.2
Mag TT	5/31/2009	23:23	27.39	1473	1.537	0.73	6.2	0.46	6.21	-176.9	7.1
Mag TT	6/1/2009	12:07	27.23	1472	1.535	0.73	3.2	0.25	5.78	-179.9	7.0
Mag TT	6/1/2009	18:12	27.13	1471	1.530	0.73	4.9	0.38	6.76	-137.6	6.9
Mag TT	6/1/2009	23:59	27.10	1465	1.524	0.73	3.5	0.28	6.77	-140.6	7.0
Mag TT	6/2/2009	8:33	27.04	1465	1.522	0.73	4.7	0.39	7.18	6.-126.9	6.9
Mag TT	6/2/2009	15:22	27.13	1473	1.533	0.74	15.6	1.23	7.13	-150.7	
Mag TT	6/2/2009	16:15									7.3
Mag TT	6/2/2009	18:15	27.09	1481	1.541	0.74	19.6	1.54	6.95	-134.6	6.7
Mag TT	6/3/2009	5:44	27.16	1477	1.538	0.74	6.8	0.54	7.09	-115.4	6.7
Mag TT	6/3/2009	15:40	27.11	1464	1.523	0.73	5.7	0.45	7.24	-128.8	6.7
Mag TT	6/3/2009	16:05									7.2
Mag TT	6/3/2009	18:13	27.05	1473	1.531	0.74	7.7	0.62	7.36	-130.6	6.7
Mag TT	06/04/09	11:54	27.07	1463	1.521	0.73	16.5	1.30	6.94	-125.9	6.7
Mag TT	6/4/2009	5:42	27.18	1460	1.521	0.73	11.8	0.93	6.97	-134.1	6.8
Mag TT	6/4/2009	15:58	26.96	1482	1.538	0.74	7.7	0.61	6.89	-120.2	
Mag TT	<b>average</b>		<b>27.15</b>	<b>1469.79</b>	<b>1.53</b>	<b>0.73</b>	<b>8.26</b>	<b>0.65</b>	<b>6.87</b>	<b>-142.48</b>	<b>6.92</b>
	<i>stdev</i>		<b>0.13</b>	<b>7.46</b>	<b>0.01</b>	<b>0.00</b>	<b>5.48</b>	<b>0.43</b>	<b>0.42</b>	<b>22.05</b>	<b>0.22</b>

Appendix A Performance Assessment  
Nutrient Data

	C	Composite sample			F Coli	Nitrite	Nitrate+Nitrite	Nitrate	Ammonia	TKN	TN	Total P
	G	Grab Sample			SM	EPA	EPA	calc	EPA	EPA		EPA
	E3	Event 3 sample			9222D	354.1	353.3		350.3	351.4	calc	365.2
				MDL	1	0.003	0.006	0.006	0.022	0.056	0.056	0.004
				PQL	4	0.012	0.024	0.024	0.088	0.224	0.224	0.016
Station	ID	Date	Time	Rep	Cts/100ml	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L
Hick STE	G1	05/31/09	16:55	1			14.917		40.114	50.612	65.529	3.535
Hick STE	C1	06/01/09	11:30	1			17.006		48.684	45.222	62.228	3.712
Hick STE	G2	06/01/09	10:55	1			23.271		41.359	38.587	61.857	3.364
Hick STE	G3	06/01/09	23:00	1			0.608		33.511	51.175	51.783	3.489
Hick STE	C2	06/02/09	12:30	1			26.526		34.401	34.423	60.949	3.659
Hick STE	G4	06/02/09	7:45	1			0.211		52.014	48.029	48.240	3.676
Hick STE	C3	06/03/09	12:00	1		2.695	32.794	30.099	17.775	16.175	48.969	2.836
Hick STE	G5	06/03/09	5:00	1		3.069	59.643	56.575	20.328	22.583	82.227	3.162
Hick STE	G6	06/03/09	16:25	1		3.848	7.140	3.292	14.535	12.977	20.116	3.071
Hick STE	G7	06/03/09	17:00	1		3.556	48.217	44.661	12.654	10.342	58.558	3.198
Hick STE	G7	06/03/09	17:00	F2		33.314	46.872	13.558	12.264	10.764	57.637	2.981
Hick STE	C4	06/04/09	13:30	1		0.583	62.333	61.749	10.685	11.341	73.674	2.958
Hick STE	C4	06/04/09	13:35	F2		0.398	59.103	58.704	10.790	13.646	72.749	3.396
Hick STE	E3	06/04/09	11:20	1	6200	4.607	10.785	6.179	5.759	6.783	17.569	2.668
Hick STE	G8	06/04/09	15:24	1		0.146	38.381	38.234	6.729	7.010	45.391	3.541

Appendix A Performance Assessment  
 Nutrient Data (continued)

	C	Composite sample			F Coli	Nitrite	Nitrate+Nitrite	Nitrate	Ammonia	TKN	TN	Total P
	G	Grab Sample			SM	EPA	EPA	calc	EPA	EPA		EPA
	E3	Event 3 sample			9222D	354.1	353.3		350.3	351.4	calc	365.2
				MDL	1	0.003	0.006	0.006	0.022	0.056	0.056	0.004
				PQL	4	0.012	0.024	0.024	0.088	0.224	0.224	0.016
Station	ID	Date	Time	Rep	Cts/100ml	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L
Hick TT	G1	05/31/09	19:15	1			0.226		27.374	133.164	133.390	4.311
Hick TT	C1	06/01/09	14:30	1			0.400		76.632	91.178	91.578	3.733
Hick TT	C1	06/01/09	14:30	F2			0.400		84.853	84.009	84.408	3.741
Hick TT	G2	06/01/09	10:50	1			0.878		129.518	146.016	146.895	3.959
Hick TT	G3	06/01/09	23:15	1				24.464	95.909	110.214	134.677	3.786
Hick TT	C2	06/02/09	12:45	1			1.052		50.668	51.474	52.526	3.896
Hick TT	G4	06/02/09	7:55	1			0.665		64.654	91.979	92.644	7.259
Hick TT	C3	06/03/09	11:50	1		0.135	0.211	0.076	83.149	88.735	88.946	3.270
Hick TT	G5	06/03/09	5:10	1		0.135	0.155	0.020	82.778	124.724	124.878	3.397
Hick TT	G6	06/03/09	16:34	1		0.151	0.161	0.010	96.803	139.712	139.873	3.397
Hick TT	G7	06/03/09	17:10	1		0.135	0.068	0.000	90.928	86.397	86.465	3.325
Hick TT	C4	06/04/09	13:14	1		0.117	0.401	0.283	56.022	93.552	93.953	2.759
Hick TT	C4	06/04/09	13:20	F2		0.112	0.401	0.289	41.630	78.160	78.561	3.409
Hick TT	E3	06/04/09	11:32	1	15400	0.115	0.099	0.000	61.506	58.459	58.558	3.321
Hick TT	G8	06/04/09	15:38	1		0.137	0.984	0.847	75.388	77.748	78.732	3.661

Appendix A Performance Assessment  
 Nutrient Data (continued)

	C	Composite sample			F Coli	Nitrite	Nitrate+Nitrite	Nitrate	Ammonia	TKN	TN	Total P
	G	Grab Sample			SM	EPA	EPA	calc	EPA	EPA		EPA
	E3	Event 3 sample			9222D	354.1	353.3		350.3	351.4	calc	365.2
				MDL	1	0.003	0.006	0.006	0.022	0.056	0.056	0.004
				PQL	4	0.012	0.024	0.024	0.088	0.224	0.224	0.016
Station	ID	Date	Time	Rep	Cts/100ml	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L
Mag STE	G1	05/31/09	18:20	1			0.806		24.848	49.333	50.139	3.864
Mag STE	C1	06/01/09	12:40	1			9.697		28.513	26.151	35.848	3.066
Mag STE	G2	06/01/09	11:45	1			17.528		28.368	21.862	39.390	4.002
Mag STE	G3	06/01/09	23:45	1			1.023		27.169	34.973	35.997	3.676
Mag STE	C2	06/02/09	16:10	1			12.090		19.490	33.525	45.615	4.037
Mag STE	C2	06/02/09	16:10	F2			10.028		27.528	26.149	36.176	3.888
Mag STE	G4	06/02/09	8:20	1			16.214		22.027	28.009	44.223	3.964
Mag STE	C3	06/03/09	16:10	1		3.069	6.356	3.287	18.177	27.590	33.946	3.470
Mag STE	G5	06/03/09	5:35	1		3.556	12.181	8.625	17.305	18.609	30.790	3.307
Mag STE	G6	06/03/09	15:48	1		4.156	17.559	13.402	14.931	18.240	35.798	3.252
Mag STE	C4	06/04/09	16:30	1		0.000	46.455	46.455	15.318	20.942	67.397	3.396
Mag STE	E3	06/04/09	12:05	1	6700	3.000	25.002	22.003	11.734	17.042	42.044	3.535
Mag STE	E3	06/04/09	12:05	F2	10700	0.022	25.330	25.308	11.848	14.407	39.737	3.314
Mag STE	G7	06/04/09	5:50	1		0.117	0.068	0.000	31.648	57.116	57.183	3.488
Mag STE	G8	06/04/09	16:14	1		0.002	26.000	25.998	21.432	21.962	47.962	3.966

Appendix A Performance Assessment  
Nutrient Data (continued)

	C	Composite sample			F Coli	Nitrite	Nitrate+Nitrite	Nitrate	Ammonia	TKN	TN	Total P
	G	Grab Sample			SM	EPA	EPA	calc	EPA	EPA		EPA
	E3	Event 3 sample			9222D	354.1	353.3		350.3	351.4	calc	365.2
				MDL	1	0.003	0.006	0.006	0.022	0.056	0.056	0.004
				PQL	4	0.012	0.024	0.024	0.088	0.224	0.224	0.016
Station	ID	Date	Time	Rep	Cts/100ml	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L
Mag TT	G1	05/31/09	18:35	1			0.168		97.367	156.863	157.031	4.215
Mag TT	C1	06/01/09	15:25	1			25.359		95.403	120.824	146.183	4.481
Mag TT	G2	06/01/09	12:05	1			20.138		76.243	89.330	109.468	4.567
Mag TT	G3	06/01/09	23:55	1			10.028		138.813	139.081	149.108	4.100
Mag TT	C2	06/02/09	16:15	1			0.039		77.420	121.218	121.257	3.820
Mag TT	G4	06/02/09	8:30	1			0.909		84.492	135.452	136.361	3.477
Mag TT	C3	06/03/09	16:05	1		0.087	0.130	0.043	89.717	177.664	177.794	3.886
Mag TT	G5	06/03/09	5:42	1		0.151	0.173	0.022	94.663	188.665	188.839	3.669
Mag TT	G6	06/03/09	15:35	1		0.175	0.214	0.038	86.564	130.690	130.904	3.524
Mag TT	G6	06/03/09	15:35	F2		0.161	0.155	0.000	89.316	143.493	143.647	3.796
Mag TT	C4	06/04/09	16:20	1		0.072	0.289	0.216	126.541	127.799	128.088	3.992
Mag TT	E3	06/04/09	11:55	1	102100	0.115	0.075	0.000	100.784	102.330	102.405	3.435
Mag TT	G7	06/04/09	5:40	1		0.118	0.068	0.000	127.163	169.553	169.621	3.597
Mag TT	G8	06/04/09	16:00	1		0.109	0.588	0.479	122.302	123.809	124.396	3.913

Appendix B Environmental Monitoring  
Field Data

Station	Date	Well <i>FSU</i> (m)	Time <i>YSI</i>	Water T <i>YSI</i> Celsius	COND <i>YSI</i> μS/cm	COND <i>YSI</i> mS/cm	Salinity <i>YSI</i> ppt	DO %SAT <i>YSI</i> %	DO <i>YSI</i> mg/L	PH <i>YSI</i> units	ORP <i>YSI</i> mg/L	PH <i>Probe</i> units
<b>Background</b>												
CAS	1/14/09	3.40	12:14	21.65	41.83			27.4	2.21	8.06		
CAS	03/26/09	6.40	13:31	21.00	41.83			27.4	2.2	8.06		
CAS	05/11/09	2.52	12:35	20.83	53				1.65	3.48		
CAS	06/04/09	2.79	11:04	21.33	277				2.19	4.48		
				<b>21.20</b>	<b>103.42</b>				<b>2.07</b>	<b>6.02</b>		
				<i>stdev</i>	<i>0.36</i>	<i>115.84</i>			<i>0.28</i>	<i>2.39</i>		
SRWMD #4	1/14/09	5.46	12:56	22.21	157.9			14.9	0.60	7.71		
SRWMD #4	03/26/09	6.77	13:12	22.21	157.9			14.9	0.6	7.71		
SRWMD #4	05/11/09	4.61	12:59	21.37	299				1.39	3.36		
SRWMD #4	06/04/09	4.90	11:43	21.48	206				1.45	8.46		
				<b>21.82</b>	<b>205.20</b>				<b>1.01</b>	<b>6.81</b>		
				<i>stdev</i>	<i>0.46</i>	<i>66.52</i>			<i>0.47</i>	<i>2.33</i>		

Appendix B Environmental Monitoring  
Field Data (continued)

Station	Date	Well <i>FSU</i> (m)	Time <i>YSI</i>	Water T <i>YSI</i> Celsius	COND <i>YSI</i> µS/cm	COND <i>YSI</i> mS/cm	Salinity <i>YSI</i> ppt	DO %SAT <i>YSI</i> %	DO <i>YSI</i> mg/L	PH <i>YSI</i> units	ORP <i>YSI</i> mg/L	PH <i>Probe</i> units
<b>Hickory</b>												
S1	1/14/09	4.36	9:44	21.86	634	0.596	0.31	21.7	1.89	7.57	219.1	
S1	03/27/09	4.20	14:20	21.96	589.00	0.555	0.29	28.2	2.46	6.75	101.9	
S1	05/11/09	3.75	16:19	21.57	645	0.602	0.31	5.6	0.49	4.00	36.3	
S1	06/04/09	3.73	11:09	21.62	660	0.618	0.32	3.2	0.29	7.37	29.7	7.2
	<b>average</b>			<b>21.75</b>	<b>632.00</b>	<b>0.59</b>	<b>0.31</b>	<b>14.68</b>	<b>1.28</b>	<b>6.42</b>	<b>96.75</b>	
	<i>stdev</i>			<i>0.19</i>	<i>30.58</i>	<i>0.03</i>	<i>0.01</i>	<i>12.20</i>	<i>1.06</i>	<i>1.65</i>	<i>87.84</i>	
S2	1/14/09	4.26	10:05	21.58	553	0.518	0.27	39.3	3.47	7.47	218.2	
S2	03/27/09	4.08	12:15	21.32	536	0.499	0.26	46.9	4.15	5.79	108.5	
S2	05/11/09	3.445	14:18	21.22	548	0.508	0.27	11.6	1.02	4.13	12.9	
S2	06/04/09	3.64	10:12	21.13	532	0.493	0.26	4.8	0.43	7.51	33.4	7.3
	<b>average</b>			<b>21.31</b>	<b>542.25</b>	<b>0.50</b>	<b>0.27</b>	<b>25.65</b>	<b>2.27</b>	<b>6.23</b>	<b>93.25</b>	
	<i>stdev</i>			<i>0.19</i>	<i>9.88</i>	<i>0.01</i>	<i>0.01</i>	<i>20.58</i>	<i>1.82</i>	<i>1.61</i>	<i>92.89</i>	
C3	1/14/09	4.30	10:55	21.46	588	0.548	0.29	57.0	5.03	7.34	243.7	
C3	03/27/09	4.12	13:30	21.07	566	0.523	0.27	62.1	5.51	6.60	105.0	
C3	05/11/09	3.475	15:24	20.85	555	0.511	0.27	31.3	2.79	4.03	35.6	
C3	06/04/09	3.66	10:44	20.83	550	0.506	0.27	27.0	2.50	7.41	32.6	7.2
	<b>average</b>			<b>21.05</b>	<b>564.75</b>	<b>0.52</b>	<b>0.28</b>	<b>44.35</b>	<b>3.96</b>	<b>6.35</b>	<b>104.23</b>	
	<i>stdev</i>			<i>0.29</i>	<i>16.88</i>	<i>0.02</i>	<i>0.01</i>	<i>17.76</i>	<i>1.53</i>	<i>1.59</i>	<i>98.82</i>	

Appendix B Environmental Monitoring  
Field Data (continued)

Station	Date	Well <i>FSU</i> (m)	Time <i>YSI</i>	Water T <i>YSI</i> Celsius	COND <i>YSI</i> µS/cm	COND <i>YSI</i> mS/cm	Salinity <i>YSI</i> ppt	DO %SAT <i>YSI</i> %	DO <i>YSI</i> mg/L	PH <i>YSI</i> units	ORP <i>YSI</i> mg/L	PH <i>Probe</i> units
C4	1/14/09	4.37	11:12	21.33	573	0.533	0.27	69.7	6.16	7.31	250.1	
C4	03/27/09	4.18	12:45	20.98	563	0.520	0.27	70.1	6.25	6.43	107.5	
C4	05/11/09	3.54	14:59	20:57	541	0.495	0.26	36.3	3.26	4.08	33.1	
C4	06/04/09	3.73	10:32	20.96	542	0.500	0.26	30.6	2.71	7.49	32.6	7.2
	<b>average</b>			<b>16.04</b>	<b>554.75</b>	<b>0.51</b>	<b>0.27</b>	<b>51.68</b>	<b>4.60</b>	<b>6.33</b>	<b>105.83</b>	
	<i>stdev</i>			<i>10.11</i>	<i>15.84</i>	<i>0.02</i>	<i>0.01</i>	<i>21.17</i>	<i>1.87</i>	<i>1.57</i>	<i>102.42</i>	
C5	1/14/09	4.30	10:35	21.91	696	0.655	0.34	40.6	3.55	7.39	240.8	
C5	03/27/09	4.13	13:50	21.86	639	0.600	0.31	28.1	2.46	6.67	96.6	
C5	05/11/09	3.475	15:50	21.33	641	0.596	0.31	10.9	0.96	4.00	34.7	
C5	06/04/09	3.66	10:59	21.41	658	0.613	0.32	32.4	2.87	7.38	31.7	7.2
	<b>average</b>			<b>21.63</b>	<b>658.50</b>	<b>0.62</b>	<b>0.32</b>	<b>28.00</b>	<b>2.46</b>	<b>6.36</b>	<b>100.95</b>	
	<i>stdev</i>			<i>0.30</i>	<i>26.41</i>	<i>0.03</i>	<i>0.01</i>	<i>12.52</i>	<i>1.10</i>	<i>1.61</i>	<i>97.91</i>	

Appendix B Environmental Monitoring  
Field Data (continued)

Station	Date	Well <i>FSU</i> (m)	Time <i>YSI</i>	Water T <i>YSI</i> Celsius	COND <i>YSI</i> µS/cm	COND <i>YSI</i> mS/cm	Salinity <i>YSI</i> ppt	DO %SAT <i>YSI</i> %	DO <i>YSI</i> mg/L	PH <i>YSI</i> units	ORP <i>YSI</i> mg/L	PH <i>Probe</i> units
M1	1/14/09	2.52	13:44	21.88	959	0.897	0.47	7.2	0.63	7.11	158.9	
M1	03/26/09	2.50										
M1	05/11/09	1.41	12:34	21.52	827	0.773	0.41	5.6	0.49	3.94	43.5	
M1	06/04/09	1.97	14:54	21.16	839	0.778	0.41	2.3	0.21	7.33	13.7	7.1
	<b>average</b>			<b>21.52</b>	<b>875.00</b>	<b>0.82</b>	<b>0.43</b>	<b>5.03</b>	<b>0.44</b>	<b>6.13</b>	<b>72.03</b>	
	<i>stdev</i>			<i>0.36</i>	<i>72.99</i>	<i>0.07</i>	<i>0.03</i>	<i>2.50</i>	<i>0.21</i>	<i>1.90</i>	<i>76.69</i>	
M2	1/14/09	2.04	13:12	19.64	719	0.645	0.35	18.0	1.64	7.09	165.7	
M2	03/26/09	2.04										
M2	05/11/09	0.93	12:04	19.95	492	0.445	0.24	5.4	0.49	3.98	23.1	
M2	06/04/09	1.45	14:31	21.04	616	0.569	0.3	3.3	0.29	7.48	10.3	7.1
	<b>average</b>			<b>20.21</b>	<b>609.00</b>	<b>0.55</b>	<b>0.30</b>	<b>8.90</b>	<b>0.81</b>	<b>6.18</b>	<b>66.37</b>	
	<i>stdev</i>			<i>0.74</i>	<i>113.66</i>	<i>0.10</i>	<i>0.06</i>	<i>7.95</i>	<i>0.73</i>	<i>1.92</i>	<i>86.26</i>	
M3	1/14/09	1.70	12:16	20.45	763	0.697	0.37	7.4	0.66	7.20	178.6	
M3	03/26/09	1.69										
M3	05/11/09	0.62	10:55	19:55	619	0.554	0.3	6.5	0.6	3.97	43.6	
M3	06/04/09	1.115	12:28	20.03	663	0.600	0.32	4.0	0.36	7.35	10.2	7.2
	<b>average</b>			<b>13.77</b>	<b>681.67</b>	<b>0.62</b>	<b>0.33</b>	<b>5.97</b>	<b>0.54</b>	<b>6.17</b>	<b>77.47</b>	
	<i>stdev</i>			<i>11.21</i>	<i>73.79</i>	<i>0.07</i>	<i>0.04</i>	<i>1.76</i>	<i>0.16</i>	<i>1.91</i>	<i>89.16</i>	
M5	1/14/09	2.00	13:00	21.36	886	0.824	0.44	8.0	0.71	7.10	163.8	
M5	03/26/09	1.98										
M5	05/11/09	0.91	11:54	20.64	748	0.686	0.37	4.2	0.38	3.93	43.0	
M5	06/04/09	1.39	14:21	20.59	745	0.682	0.36	4.9	0.44	7.76	14.0	7.2
	<b>average</b>			<b>20.86</b>	<b>793.00</b>	<b>0.73</b>	<b>0.39</b>	<b>5.70</b>	<b>0.51</b>	<b>6.26</b>	<b>73.60</b>	
	<i>stdev</i>			<i>0.43</i>	<i>80.55</i>	<i>0.08</i>	<i>0.04</i>	<i>2.02</i>	<i>0.18</i>	<i>2.05</i>	<i>79.45</i>	

Appendix B Environmental Monitoring  
Field Data (continued)

Station	Date	Well <i>FSU</i> (m)	Time <i>YSI</i>	Water T <i>YSI</i> Celsius	COND <i>YSI</i> µS/cm	COND <i>YSI</i> mS/cm	Salinity <i>YSI</i> ppt	DO %SAT <i>YSI</i> %	DO <i>YSI</i> mg/L	PH <i>YSI</i> units	ORP <i>YSI</i> mg/L	PH <i>Probe</i> units
M6	1/14/09	1.44	12:32	20.93	869	0.801	0.43	5.6	0.51	7.15	172.3	
M6	03/26/09	1.935										
M6	05/11/09	0.41	11:20	20.49	805	0.73	0.4	2	0.17	3.86	45.7	
M6	06/04/09	0.815	12:35	20.39	778	0.710	0.38	3.7	0.33	7.26	11.6	7.2
	<b>average</b>			<b>20.60</b>	<b>817.33</b>	<b>0.75</b>	<b>0.40</b>	<b>3.77</b>	<b>0.34</b>	<b>6.09</b>	<b>76.53</b>	
	<i>stdev</i>			<i>0.29</i>	<i>46.74</i>	<i>0.05</i>	<i>0.03</i>	<i>1.80</i>	<i>0.17</i>	<i>1.93</i>	<i>84.67</i>	
M7	1/14/09	1.63	12:46	21.36	894	0.832	0.44	7.1	0.63	7.12	165.8	
M7	03/26/09	1.605										
M7	05/11/09	0.565	11:37	20.32	810	0.738	0.4	2.7	0.24	3.80	43.4	
M7	06/04/09	1.03	12:43	20.57	744	0.681	0.36	4.3	0.37	7.33	11.5	7.2
	<b>average</b>			<b>20.75</b>	<b>816.00</b>	<b>0.75</b>	<b>0.40</b>	<b>4.70</b>	<b>0.41</b>	<b>6.08</b>	<b>73.57</b>	
	<i>stdev</i>			<i>0.54</i>	<i>75.18</i>	<i>0.08</i>	<i>0.04</i>	<i>2.23</i>	<i>0.20</i>	<i>1.98</i>	<i>81.45</i>	
M8	1/14/09	1.98	13:28	21.02	715	0.660	0.35	5.9	0.53	7.17	159.6	
M8	03/26/09	1.96										
M8	05/11/09	0.87	12:14	19.89	667	0.602	0.33	4.3	0.39	4.00	33.8	
M8	06/04/09	1.375	14:43	20.24	638	0.580	0.32	2.5	0.23	7.47	13.3	7.2
	<b>average</b>			<b>20.38</b>	<b>673.33</b>	<b>0.61</b>	<b>0.33</b>	<b>4.23</b>	<b>0.38</b>	<b>6.21</b>	<b>68.90</b>	
	<i>stdev</i>			<i>0.58</i>	<i>38.89</i>	<i>0.04</i>	<i>0.02</i>	<i>1.70</i>	<i>0.15</i>	<i>1.92</i>	<i>79.21</i>	
M9	1/14/09	1.84	11:50	20.67	689	0.633	0.34	11.3	1.02	7.39	180.7	
M9	03/26/09	1.82										
M9	05/11/09	0.735	10:44	19.65	562	0.505	0.27	6.9	0.63	4.12	42.6	
M9	06/04/09	1.24	12:11	20.22	566	0.515	0.27	4.0	0.36	7.46	9.9	7.1
	<b>average</b>			<b>20.18</b>	<b>605.67</b>	<b>0.55</b>	<b>0.29</b>	<b>7.40</b>	<b>0.67</b>	<b>6.32</b>	<b>77.72</b>	
	<i>stdev</i>			<i>0.51</i>	<i>72.20</i>	<i>0.07</i>	<i>0.04</i>	<i>3.68</i>	<i>0.33</i>	<i>1.91</i>	<i>90.67</i>	

Appendix B Environmental Monitoring  
Field Data (continued)

<b>Station</b>	<b>Date</b>	<b>Time</b>	<b>Water T</b>	<b>COND</b>	<b>COND</b>	<b>Salinity</b>	<b>DO %SAT</b>	<b>DO</b>	<b>PH</b>	<b>ORP</b>	<b>PH</b>
		<i>YSI</i>	<i>YSI</i>	<i>YSI</i>	<i>YSI</i>	<i>YSI</i>	<i>YSI</i>	<i>YSI</i>	<i>YSI</i>	<i>YSI</i>	<i>Probe</i>
			Celsius	µS/cm	mS/cm	ppt	%	mg/L	units	mg/L	units
Hick TT	03/27/09	14:45	23.29	1087	1.051	0.54	2.2	0.18	8.22	-79.6	
Hick TT	05/11/09	16:59	25.91	1611	1.639	0.81	8.9	0.72	4.40	-241.2	
Hick TT	06/04/09	11:31	25.85	1309	1.330	0.65	27.4	2.22	6.98	-146.8	6.8
Hick STE	05/11/09	16:44	25.96	1403	1.429	0.7	52.6	4.28	4.44	37.2	
Hick STE	06/04/09	11:24	25.41	874	0.881	0.43	45.0	3.68	7.77	26.3	
Mag TT	05/11/09	13:07	26.91	1430	1.482	0.71	1.6	0.11	3.78	-249.4	
Mag TT	06/04/09	11:54	27.07	1463	1.521	0.73	16.5	1.30	6.94	-125.9	6.7
Mag STE	1/14/09	14:10	18.50	1136	0.996	0.570	40.7	3.80	6.79	163.30	
Mag STE	05/11/09	12:48	26.15	1237	1.264	0.61	9.9	0.8	3.78	55.3	
Mag STE	06/04/09	12:04	26.50	1113	1.145	0.55	55.7	4.47	6.21	-2.5	6.2

Appendix B Environmental Monitoring  
Nutrient Data

				<b>F Coli</b>	<b>Nitrite</b>	<b>Nitrate+Nitrite</b>	<b>Nitrate</b>	<b>Ammonia</b>	<b>TKN</b>	<b>TN</b>	<b>Total P</b>
				SM	EPA	EPA	calc	EPA	EPA	calc	EPA
				9222D	354.1	353.3		350.3	351.4		365.20
			MDL	1	0.003	0.006	0.01	0.022	0.056	0.06	0.00
			PQL	4	0.012	0.024	0.02	0.088	0.224	0.22	0.02
<b>Station</b>	<b>Date</b>	<b>Time</b>	<b>Rep</b>	<b>Cts/100ml</b>	<b>mg/L</b>	<b>mg/L</b>	<b>mg/L</b>	<b>mg/L</b>	<b>mg/L</b>	<b>mg/L</b>	<b>mg/L</b>
CAS	01/14/09	12:14	1	0	0.041	0.310	0.27	0.120	0.448	0.76	0.01
CAS	03/26/09	13:31	1	0	0.131	0.139	0.01	0.012	0.172	0.31	0.01
CAS	05/11/09	12:35	1	0	0.042	0.155	0.11	0.053	0.073	0.23	0.32
CAS	06/04/09	11:00	1	0	0.052	0.476	0.42	0.407	0.832	1.31	0.10
	<b>average</b>				<b>0.067</b>	<b>0.270</b>	<b>0.20</b>	<b>0.148</b>	<b>0.381</b>	<b>0.65</b>	<b>0.11</b>
	<i>stdev</i>				<i>0.043</i>	<i>0.157</i>	<i>0.18</i>	<i>0.178</i>	<i>0.340</i>	<i>0.50</i>	<i>0.15</i>
SRWMD #4	01/14/09	12:56	1	0	0.010	0.226	0.22	0.123	0.088	0.31	0.01
SRWMD #4	03/26/09	13:12	1	4	0.019	0.863	0.84	0.014	0.130	0.99	0.01
SRWMD #4	05/11/09	12:59	1	0	0.009	0.020	0.01	0.045	0.095	0.11	0.27
SRWMD #4	06/04/09	11:30	1	0	0.033	1.032	1.00	0.466	0.655	1.69	0.29
	<b>average</b>				<b>0.018</b>	<b>0.535</b>	<b>0.52</b>	<b>0.162</b>	<b>0.242</b>	<b>0.78</b>	<b>0.14</b>
	<i>stdev</i>				<i>0.011</i>	<i>0.488</i>	<i>0.48</i>	<i>0.208</i>	<i>0.276</i>	<i>0.71</i>	<i>0.15</i>

Appendix B Environmental Monitoring  
 Nutrient Data (continued)

Station	Date	Time	Rep	F Coli	Nitrite	Nitrate+Nitrite	Nitrate	Ammonia	TKN	TN	Total P
				SM 9222D	EPA	EPA	calc	EPA	EPA	calc	EPA
				Cts/100ml	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L
			MDL	1	0.003	0.006	0.01	0.022	0.056	0.06	0.00
			PQL	4	0.012	0.024	0.02	0.088	0.224	0.22	0.02
S1	01/14/09	9:44	1	7	0.001	6.670	6.67	0.134	0.096	6.77	1.32
S1	03/27/09	14:20	1	1	0.006	1.504	1.50	0.006	0.048	1.55	0.39
S1	03/27/09	14:20	F2	1	0.006	1.562	1.56	0.006	0.053	1.61	0.39
S1	03/27/09	14:20	Ave	1	0.006	1.533	1.53	0.006	0.050	1.58	0.39
S1	05/11/09	16:10	1	0	0.005	0.575	0.57	0.041	0.072	0.65	0.57
S1	05/11/09	16:10	F2	0	0.006	0.453	0.45	0.040	0.069	0.52	0.55
S1	05/11/09	16:10	Ave	0	0.006	0.514	0.51	0.041	0.070	0.58	0.56
S1	06/04/09	11:10	1	0	0.133	4.841	4.71	0.264	0.565	5.41	0.78
			<b>average</b>		<b>0.036</b>	<b>3.390</b>	<b>3.35</b>	<b>0.111</b>	<b>0.195</b>	<b>3.59</b>	<b>0.76</b>
			<i>stdev</i>		<i>0.064</i>	<i>2.863</i>	<i>2.84</i>	<i>0.116</i>	<i>0.247</i>	<i>2.97</i>	<i>0.40</i>
S2	01/14/09	10:05	1	0	0.000	1.713	1.71	0.132	0.081	1.79	0.05
S2	03/27/09	12:15	1	18	0.009	0.104	0.10	0.008	0.169	0.27	0.07
S2	05/11/09	14:20	1	25	0.000	0.085	0.08	0.047	0.061	0.15	0.06
S2	06/04/09	10:14	1	0	0.010	0.245	0.23	0.335	0.445	0.69	0.02
			<b>average</b>		<b>0.005</b>	<b>0.537</b>	<b>0.53</b>	<b>0.131</b>	<b>0.189</b>	<b>0.73</b>	<b>0.05</b>
			<i>stdev</i>		<i>0.006</i>	<i>0.788</i>	<i>0.79</i>	<i>0.146</i>	<i>0.177</i>	<i>0.75</i>	<i>0.02</i>

Appendix B Environmental Monitoring  
Nutrient Data (continued)

Station	Date	Time	Rep	F Coli	Nitrite	Nitrate+Nitrite	Nitrate	Ammonia	TKN	TN	Total P
				SM 9222D	EPA	EPA	calc	EPA	EPA	calc	EPA
			MDL	Cts/100ml	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L
C3A	01/14/09	10:55	1	8	0.000	0.963	0.96	0.108	0.072	1.04	0.01
C3A	03/27/09	13:30	1	0	0.008	0.649	0.64	0.007	0.079	0.73	0.05
C3A	05/11/09	15:25	1	0	0.004	0.109	0.10	0.041	0.110	0.22	0.03
C3A	06/04/09	10:46	1	4	0.004	0.208	0.20	0.405	0.367	0.58	0.02
	<b>average</b>				<b>0.004</b>	<b>0.483</b>	<b>0.48</b>	<b>0.140</b>	<b>0.157</b>	<b>0.64</b>	<b>0.03</b>
	<i>stdev</i>				<i>0.003</i>	<i>0.397</i>	<i>0.40</i>	<i>0.181</i>	<i>0.141</i>	<i>0.34</i>	<i>0.02</i>
C4	01/14/09	11:12	1	0	0.000	1.180	1.18	0.109	0.084	1.26	0.02
C4	03/27/09	12:45	1	7	0.004	1.156	1.15	0.007	0.120	1.28	0.05
C4	05/11/09	15:05	1	0	0.001	0.281	0.28	0.044	0.113	0.39	0.05
C4	06/04/09	10:34	1	76	0.008	0.467	0.46	0.304	0.450	0.92	0.08
	<b>average</b>				<b>0.003</b>	<b>0.771</b>	<b>0.77</b>	<b>0.116</b>	<b>0.192</b>	<b>0.96</b>	<b>0.05</b>
	<i>stdev</i>				<i>0.004</i>	<i>0.465</i>	<i>0.47</i>	<i>0.132</i>	<i>0.173</i>	<i>0.41</i>	<i>0.03</i>
C5	01/14/09	10:35	1	120	0.026	5.984	5.96	0.118	0.220	6.20	0.03
C5	03/27/09	13:50	1	10	0.035	1.019	0.98	0.008	0.133	1.15	0.03
C5	05/11/09	15:50	1	0	0.003	2.064	2.06	0.040	0.081	2.15	0.05
C5	06/04/09	11:00	1	0	0.021	2.982	2.96	0.585	0.485	3.47	0.03
	<b>average</b>				<b>0.021</b>	<b>3.012</b>	<b>2.99</b>	<b>0.188</b>	<b>0.230</b>	<b>3.24</b>	<b>0.04</b>
	<i>stdev</i>				<i>0.014</i>	<i>2.137</i>	<i>2.14</i>	<i>0.269</i>	<i>0.180</i>	<i>2.19</i>	<i>0.01</i>

Appendix B Environmental Monitoring  
Nutrient Data (continued)

Station	Date	Time	Rep	F Coli	Nitrite	Nitrate+Nitrite	Nitrate	Ammonia	TKN	TN	Total P
				SM 9222D	EPA	EPA	calc	EPA	EPA	calc	EPA
			MDL	1	0.003	0.006	0.01	0.022	0.056	0.06	0.00
			PQL	4	0.012	0.024	0.02	0.088	0.224	0.22	0.02
M1	01/14/09	13:44	1	10	0.011	17.155	17.14	0.483	0.232	17.39	0.19
M1	03/26/09	13:40	1	7 *	0.006	13.243	13.24	0.116	0.361	13.60	0.13
M1	03/26/09	13:42	F2	8 *	0.008	11.372	11.36	0.117	0.348	11.72	0.14
M1	03/26/09	13:42	Ave	8	0.007	12.307	12.30	0.117	0.355	12.66	0.13
M1	05/11/09	12:38	1	0	0.008	2.698	2.69	0.050	0.120	2.82	0.39
M1	06/04/09	14:56	1	0	0.005	6.083	6.08	0.044	0.989	7.07	0.20
	<b>average</b>				<b>0.009</b>	<b>10.720</b>	<b>10.71</b>	<b>0.217</b>	<b>0.236</b>	<b>10.96</b>	<b>0.24</b>
	<i>stdev</i>				<i>0.002</i>	<i>6.440</i>	<i>6.44</i>	<i>0.209</i>	<i>0.389</i>	<i>6.37</i>	<i>0.11</i>
M2	01/14/09	13:12	1	67	0.005	8.157	8.15	0.127	0.136	8.29	0.37
M2	01/14/09	13:12	F2	67	0.005	7.674	7.67	0.128	0.155	7.83	0.33
M2	01/14/09	13:12	Ave	67	0.005	7.915	7.91	0.127	0.145	8.06	0.35
M2	03/26/09	12:45	1	32	0.013	4.846	4.83	0.016	0.161	5.01	0.34
M2	05/11/09	12:05	1	8	0.005	0.514	0.51	0.051	0.149	0.66	0.38
M2	06/04/09	14:32	1	0	0.004	0.184	0.18	0.042	0.665	0.85	0.29
	<b>average</b>				<b>0.006</b>	<b>3.365</b>	<b>3.36</b>	<b>0.059</b>	<b>0.280</b>	<b>3.65</b>	<b>0.34</b>
	<i>stdev</i>				<i>0.004</i>	<i>3.703</i>	<i>3.70</i>	<i>0.048</i>	<i>0.257</i>	<i>3.56</i>	<i>0.04</i>

Appendix B Environmental Monitoring  
Nutrient Data (continued)

Station	Date	Time	Rep	F Coli	Nitrite	Nitrate+Nitrite	Nitrate	Ammonia	TKN	TN	Total P
				SM 9222D	EPA	EPA	calc	EPA	EPA	calc	EPA
			MDL	Cts/100ml	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L
M3	01/14/09	12:16	1	0	0.023	3.899	3.88	0.125	0.237	4.14	0.38
M3	03/26/09	11:30	1	20	0.027	1.148	1.12	0.052	0.251	1.40	0.04
M3	05/11/09	11:00	1	5	0.002	0.266	0.26	0.042	0.109	0.38	0.25
M3	06/04/09	12:30	1	4	0.178	13.957	13.78	0.544	0.524	14.48	0.20
	<b>average</b>				<b>0.058</b>	<b>4.818</b>	<b>4.76</b>	<b>0.191</b>	<b>0.280</b>	<b>5.10</b>	<b>0.22</b>
	<i>stdev</i>				<i>0.081</i>	<i>6.286</i>	<i>6.21</i>	<i>0.238</i>	<i>0.175</i>	<i>6.45</i>	<i>0.14</i>
M5	01/14/09	13:00	1	0	0.006	11.474	11.47	0.124	0.127	11.60	0.42
M5	03/26/09	12:30	1	7	0.055	1.424	1.37	0.069	0.310	1.73	0.27
M5	05/11/09	11:55	1	0	0.003	3.787	3.78	0.051	0.138	3.92	0.18
M5	06/04/09	14:22	1	4	0.016	2.943	2.93	0.068	0.751	3.69	0.15
	<b>average</b>				<b>0.020</b>	<b>4.907</b>	<b>4.89</b>	<b>0.078</b>	<b>0.332</b>	<b>5.24</b>	<b>0.25</b>
	<i>stdev</i>				<i>0.024</i>	<i>4.486</i>	<i>4.50</i>	<i>0.032</i>	<i>0.292</i>	<i>4.35</i>	<i>0.12</i>
M6	01/14/09	12:32	1	0	0.081	5.005	4.92	0.125	0.333	5.34	0.09
M6	03/26/09	11:50	1	10	0.093	2.062	1.97	0.037	0.229	2.29	0.03
M6	05/11/09	12:50	1	5	0.064	2.864	2.80	0.051	0.112	2.98	0.58
M6	06/04/09	12:36	1	32	2.626	1.415	0.00	0.359	0.686	2.10	0.06
	<b>average</b>				<b>0.716</b>	<b>2.837</b>	<b>2.42</b>	<b>0.143</b>	<b>0.340</b>	<b>3.18</b>	<b>0.19</b>
	<i>stdev</i>				<i>1.274</i>	<i>1.562</i>	<i>2.04</i>	<i>0.149</i>	<i>0.248</i>	<i>1.49</i>	<i>0.26</i>

Appendix B Environmental Monitoring  
Nutrient Data (continued)

Station	Date	Time	Rep	F Coli	Nitrite	Nitrate+Nitrite	Nitrate	Ammonia	TKN	TN	Total P
				SM 9222D	EPA	EPA	calc	EPA	EPA	calc	EPA
			MDL	1	0.003	0.006	0.01	0.022	0.056	0.06	0.00
			PQL	4	0.012	0.024	0.02	0.088	0.224	0.22	0.02
M7	01/14/09	12:46	1	0	0.022	9.593	9.57	0.124	0.161	9.75	0.48
M7	03/26/09	12:15	1	29	0.063	5.151	5.09	0.037	0.229	5.38	0.09
M7	05/11/09	11:40	1	3	0.011	4.284	4.27	0.052	0.140	4.42	0.35
M7	06/04/09	12:45	1	14	0.202	19.425	19.22	0.373	0.562	19.99	0.22
	<b>average</b>				<b>0.075</b>	<b>9.613</b>	<b>9.54</b>	<b>0.146</b>	<b>0.273</b>	<b>9.89</b>	<b>0.29</b>
	<i>stdev</i>				<i>0.088</i>	<i>6.942</i>	<i>6.86</i>	<i>0.156</i>	<i>0.197</i>	<i>7.12</i>	<i>0.17</i>
M8	01/14/09	13:28	1	0	0.020	0.786	0.77	0.126	0.077	0.86	0.31
M8	03/26/09	13:20	1	2 *	0.026	10.284	10.26	0.015	0.157	10.44	0.09
M8	05/11/09	12:15	1	0	0.003	4.351	4.35	0.053	0.142	4.49	0.24
M8	06/04/09	14:44	1	0	0.002	0.102	0.10	0.036	0.705	0.81	0.17
	<b>average</b>				<b>0.013</b>	<b>3.881</b>	<b>3.87</b>	<b>0.058</b>	<b>0.270</b>	<b>4.15</b>	<b>0.20</b>
	<i>stdev</i>				<i>0.012</i>	<i>4.658</i>	<i>4.65</i>	<i>0.048</i>	<i>0.292</i>	<i>4.53</i>	<i>0.09</i>

Appendix B Environmental Monitoring  
 Nutrient Data (continued)

				<b>F Coli</b>	<b>Nitrite</b>	<b>Nitrate+Nitrite</b>	<b>Nitrate</b>	<b>Ammonia</b>	<b>TKN</b>	<b>TN</b>	<b>Total P</b>
				SM	EPA	EPA	calc	EPA	EPA	calc	EPA
				9222D	354.1	353.3		350.3	351.4		365.20
			MDL	<i>1</i>	<i>0.003</i>	<i>0.006</i>	<i>0.01</i>	<i>0.022</i>	<i>0.056</i>	<i>0.06</i>	<i>0.00</i>
			PQL	<i>4</i>	<i>0.012</i>	<i>0.024</i>	<i>0.02</i>	<i>0.088</i>	<i>0.224</i>	<i>0.22</i>	<i>0.02</i>
<b>Station</b>	<b>Date</b>	<b>Time</b>	<b>Rep</b>	<b>Cts/100ml</b>	<b>mg/L</b>	<b>mg/L</b>	<b>mg/L</b>	<b>mg/L</b>	<b>mg/L</b>	<b>mg/L</b>	<b>mg/L</b>
M9	01/14/09	11:50	1	13	0.009	1.370	1.36	0.110	0.127	1.50	0.08
M9	03/26/09	10:55	1	96	0.048	5.332	5.28	0.014	0.216	5.55	0.12
M9	03/26/09	10:55	L2	90	0.049	5.325	5.28	0.014	0.209	5.53	0.12
M9	03/26/09	10:55	Ave	93	0.049	5.329	5.28	0.014	0.213	5.54	0.12
M9	05/11/09	10:50	1	0	0.002	0.111	0.11	0.035	0.151	0.26	0.05
M9	06/04/09	12:15	1	2	0.017	0.239	0.22	0.419	0.562	0.80	0.16
M9	06/04/09	12:15	F2	2	0.019	0.297	0.28	0.401	0.471	0.77	0.06
M9	06/04/09	12:15	Ave	2	0.018	0.268	0.25	0.410	0.517	0.78	0.11
	<b>average</b>				<b>0.019</b>	<b>1.769</b>	<b>1.75</b>	<b>0.142</b>	<b>0.252</b>	<b>2.02</b>	<b>0.09</b>
	<i>stdev</i>				<i>0.021</i>	<i>2.438</i>	<i>2.42</i>	<i>0.183</i>	<i>0.180</i>	<i>2.40</i>	<i>0.03</i>

Appendix B Environmental Monitoring  
Nutrient Data (continued)

				<b>F Coli</b>	<b>Nitrite</b>	<b>Nitrate+Nitrite</b>	<b>Nitrate</b>	<b>Ammonia</b>	<b>TKN</b>	<b>TN</b>	<b>Total P</b>
				SM	EPA	EPA	calc	EPA	EPA	calc	EPA
				9222D	354.1	353.3		350.3	351.4		365.20
			MDL	<i>1</i>	<i>0.003</i>	<i>0.006</i>	<i>0.01</i>	<i>0.022</i>	<i>0.056</i>	<i>0.06</i>	<i>0.00</i>
			PQL	<i>4</i>	<i>0.012</i>	<i>0.024</i>	<i>0.02</i>	<i>0.088</i>	<i>0.224</i>	<i>0.22</i>	<i>0.02</i>
<b>Station</b>	<b>Date</b>	<b>Time</b>	<b>Rep</b>	<b>Cts/100ml</b>	<b>mg/L</b>	<b>mg/L</b>	<b>mg/L</b>	<b>mg/L</b>	<b>mg/L</b>	<b>mg/L</b>	<b>mg/L</b>
Hick TT	03/27/09	14:45	1	26000	0.054	0.116	0.063	61.696	173.313	173.429	13.496
Hick TT	05/11/09	17:00	1	60000	0.062	0.092	0.031	59.414	235.162	235.254	2.919
Hick TT	06/04/09	11:32	1	15400	0.115	0.099	0.000	61.506	58.459	58.558	3.321
Hick STE	03/27/09	14:35	1	8060	0.125	4.600	4.475	44.640	116.386	120.986	12.789
Hick STE	03/27/09	14:35	F2	6760	0.129	0.248	0.119	40.043	97.017	97.265	13.193
Hick STE	05/11/09	16:40	1	3350	1.327	6.307	4.979	33.595	52.537	58.844	2.731
Hick STE	06/04/09	11:20	1	6200	4.607	10.785	6.179	5.759	6.783	17.569	2.668

Appendix B Environmental Monitoring  
Nutrient Data (continued)

Station	Date	Time	Rep	F Coli	Nitrite	Nitrate+Nitrite	Nitrate	Ammonia	TKN	TN	Total P
				SM 9222D	EPA	EPA	calc	EPA	EPA	calc	EPA
			MDL	<i>1</i>	<i>0.003</i>	<i>0.006</i>	<i>0.01</i>	<i>0.022</i>	<i>0.056</i>	<i>0.06</i>	<i>0.00</i>
			PQL	<i>4</i>	<i>0.012</i>	<i>0.024</i>	<i>0.02</i>	<i>0.088</i>	<i>0.224</i>	<i>0.22</i>	<i>0.02</i>
				Cts/100ml	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L
Mag TT	03/26/09	14:20	1	10500	0.127	7.761	7.634	83.644	202.893	210.654	0.128
Mag TT	05/11/09	13:10	1	13520	0.071	0.055	0.000	38.718	168.989	169.045	1.842
Mag TT	05/11/09	13:10	F2	12410	0.070	0.070	0.000	40.402	186.769	186.839	1.150
Mag TT	06/04/09	11:55	1	102100	0.115	0.075	0.000	100.784	102.330	102.405	3.435
Mag STE	1/14/09	14:10	1	3780	0.371	63.484	63.113	2.025	3.003	66.487	20.802
Mag STE	03/26/09	13:58	1	2100	1.184	12.242	11.058	1.675	14.161	26.403	0.113
Mag STE	05/11/09	12:50	1	8000	1.512	6.943	5.431	10.199	13.160	20.102	1.743
Mag STE	05/11/09	12:50	F2	7610	1.819	6.943	5.124	11.211	12.675	19.618	1.763
Mag STE	06/04/09	12:05	1	6700	3.000	25.002	22.003	11.734	17.042	42.044	3.535
Mag STE	06/04/09	12:05	F2	10700	0.022	25.330	25.308	11.848	14.407	39.737	3.314

Appendix C Water meter data

<b>Date and time</b>	<b>Hickory meter</b>	<b>gal/day</b>	<b>Date/Time</b>	<b>Magnolia meter</b>	<b>gal/day</b>	
12/29/08 12:00	1013882		12/29/08 12:00	763991		Park staff, no time
1/5/09 12:00	1021590	1101	1/5/09 12:00	768874	698	Park staff, no time
1/12/09 12:00	1025574	569	1/12/09 12:00	771642	395	Park staff, no time
1/19/09 12:00	1028820	464	1/19/09 12:00	773974	333	Park staff, no time
1/26/09 12:00	1031736	417	1/26/09 12:00	777227	465	Park staff, no time
January 2009		638			473	
2/2/09 12:00	1036175		2/2/09 12:00	780145		Park staff, no time
2/9/09 12:00	1040051	554	2/9/09 12:00	783547	486	Park staff, no time
2/16/09 12:00	1044820	681	2/16/09 12:00	786390	406	Park staff, no time
2/23/09 12:00	1049580	680	2/23/09 12:00	789736	478	Park staff, no time
February 2009		638			457	
3/2/09 12:00	1053595		3/2/09 12:00	793866		Park staff, no time
3/9/09 12:00	1059670	868	3/9/09 12:00	799027	737	Park staff, no time
3/16/09 12:00	1066030	909	3/16/09 12:00	803084	580	Park staff, no time
3/23/09 12:00	1072963	990	3/23/09 12:00	807278	599	Park staff, no time
3/27/09 14:55	1077512	1104	3/26/09 10:00	808471	409	FSU
3/30/09 12:00	1079730	771	3/30/09 12:00	810920	600	Park staff, no time
March 2009		933			609	

Appendix C Water meter data  
(continued)

Hickory			Magnolia			
Date and time	meter	gal/day	Date/Time	meter	gal/day	
4/6/09 12:00	1088049		4/6/09 12:00	817026		Park staff, no time
4/14/09 12:00	1095200	894	4/14/09 12:00	823783	845	Park staff, no time
4/22/09 12:00	1095200	0	4/22/09 12:00	823900	15	Park staff, no time
April 2009		447			430	
5/4/09 12:00	1097500		5/4/09 12:00	826000		Park staff, no time
5/11/09 12:00	1102320	689	5/11/09 13:20	834365	1186	FSU
5/11/09 17:10	1102463	664	5/11/09 14:00	834440	2700	Park staff, no time
5/25/09 12:00	1111500	656	5/25/09 12:00	839262	346	Park staff, no time
5/31/09 16:30	1114825	537	5/31/09 18:30	841188	307	FDOH
5/31/09 19:30	1114900	600	5/31/09 19:06	841213	1000	FDOH
5/31/09 22:30	1115085	1480	5/31/09 23:30	842078	4718	FDOH
May 2009		641			585	
6/1/09 12:00	1115200		6/1/09 12:00	842200		Park staff, no time
6/4/09 15:28	1116695	475	6/4/09 16:11	842781	183	FSU
6/8/09 12:00	1119500	728	6/8/09 12:00	844820	533	Park staff, no time
6/15/09 12:00	1125600	871	6/15/09 12:00	849130	616	Park staff, no time
6/22/09 12:00	1133100	1071	6/22/09 12:00	854000	696	Park staff, no time
June 2009		852			562	
<b>Total cumulative</b>		<b>654</b>			<b>501</b>	

Appendix C Water meter data during performance assessment.

**Performance Assessment**

Date and time	Hickory		Date / time	Magnolia		
	meter	gal/hr		meter	gal/hr	
5/31/09 16:30	1114825		5/31/09 18:30	841188		FDOH
5/31/09 19:30	1114900	25	5/31/09 19:06	841213	42	FDOH
5/31/09 22:30	1115085	62	5/31/09 23:30	842078	197	FDOH
6/1/09 5:10	1115188	15	6/1/09 5:43	842085	1	FSU
6/1/09 10:46	1115195	1	6/1/09 11:47	842160	12	FSU
6/1/09 11:00	1115202	30	6/1/09 12:07	842160	0	FSU
6/1/09 12:30	1115242	27	6/1/09 15:02	842176	5	FSU
6/1/09 17:01	1115293	11	6/1/09 15:18	842199	86	FSU
6/1/09 17:11	1115296	18	6/1/09 18:00	842206	3	FSU
6/1/09 23:04	1115517	38	6/1/09 18:10	842206	0	FSU
6/1/09 23:16	1115530	65	6/1/09 23:46	842288	15	FSU
6/2/09 5:05	1115616	15	6/1/09 23:59	842288	0	FSU
6/2/09 7:45	1115690	28	6/2/09 8:18	842305	2	FSU
6/2/09 7:55	1115692	12	6/2/09 8:21	842305	0	FSU
6/2/09 11:02	1115747	18	6/2/09 8:33	842320	75	FSU
6/2/09 11:15	1115754	32	6/2/09 15:11	842372	8	FSU
6/2/09 17:02	1115828	13	6/2/09 15:20	842379	47	FSU
6/2/09 17:11	1115830	13	6/2/09 16:10	842382	4	FSU
6/2/09 23:01	1116012	31	6/2/09 17:58	842398	9	FSU
6/2/09 23:12	1116020	44	6/2/09 18:13	842398	0	FSU
6/3/09 5:03	1116053	6	6/3/09 5:32	842521	11	FSU
6/3/09 5:13	1116053	0	6/3/09 15:38	842566	4	FSU
6/3/09 11:03	1116257	35	6/3/09 15:49	842566	0	FSU
6/3/09 11:16	1116258	5	6/3/08 18:10	842608	0	FSU
6/3/09 16:26	1116328	14	6/3/09 18:20	842624	0	FSU
6/3/09 16:42	1116329	4	6/3/09 18:29	842632	53	FSU
6/3/09 17:01	1116335	19	6/4/09 5:38	842711	7	FSU
6/3/09 17:09	1116335	0	6/4/09 5:50	842711	0	FSU
6/3/09 23:02	1116453	20	6/4/09 11:52	842766	9	FSU
6/3/09 23:10	1116453	0	6/4/09 12:02	842766	0	FSU
6/4/09 5:05	1116459	1	6/4/09 15:56	842781	4	FSU
6/4/09 5:13	1116459	0	6/4/09 16:11	842781	0	FSU
6/4/09 11:22	1116649	31				FSU
6/4/09 11:29	1116658	77				FSU
6/4/09 15:28	1116695	9				FSU
<b>Performance</b>	<b>gal/hr</b>	<b>20</b>			<b>17</b>	
<b>Performance</b>	<b>gal/day</b>	<b>473</b>			<b>408</b>	

Appendix D Sampling Event on 01/14/09

Nutrient Data

				<b>F Coli</b>	<b>Nitrite</b>	<b>Nitrate+Nitrite</b>	<b>Nitrate</b>	<b>Ammonia</b>	<b>TKN</b>	<b>TN</b>	<b>Total P</b>
				SM	EPA	EPA	calc	EPA	EPA	calc	EPA
				9222D	354.1	353.3		350.3	351.4		365.2
			MDL	1	0.003	0.006	0.006	0.022	0.056	0.056	0.004
			PQL	4	0.012	0.024	0.024	0.088	0.224	0.224	0.016
<b>Station</b>	<b>Date</b>	<b>Time</b>	<b>Rep</b>	<b>Cts/100ml</b>	<b>mg/L</b>	<b>mg/L</b>	<b>mg/L</b>	<b>mg/L</b>	<b>mg/L</b>	<b>mg/L</b>	<b>mg/L</b>
<b>Background Wells</b>											
CAS	1/14/09	12:14	1	0	0.041	0.310	0.269	0.120	0.448	0.758	0.005
SRWMD#4	1/14/09	12:56	1	0	0.010	0.226	0.216	0.123	0.088	0.314	0.013
<b>Hickory</b>											
S1	1/14/09	9:44	1	7	0.001	6.670	6.669	0.134	0.096	6.766	1.316
S2	1/14/09	10:05	1	0	0.000	1.713	1.713	0.132	0.081	1.795	0.050
C3A	1/14/09	10:55	1	8	0.000	0.963	0.963	0.108	0.072	1.036	0.011
C4	1/14/09	11:12	1	0	0.000	1.180	1.180	0.109	0.084	1.264	0.016
C5	1/14/09	10:35	1	120*	0.026	5.984	5.958	0.118	0.220	6.203	0.032
<b>Magnolia</b>											
M1	1/14/09	13:44	1	10	0.011	17.155	17.144	0.483	0.232	17.386	0.186
M2	1/14/09	13:12	1	67*	0.005	8.157	8.152	0.127	0.136	8.293	0.374
M2	1/14/09	13:12	F2	67*	0.005	7.674	7.669	0.128	0.155	7.829	0.329
M3	1/14/09	12:16	1	0	0.023	3.899	3.876	0.125	0.237	4.136	0.377
M5	1/14/09	13:00	1	0	0.006	11.474	11.468	0.124	0.127	11.601	0.417
M6	1/14/09	12:32	1	0	0.081	5.005	4.924	0.125	0.333	5.338	0.090
M7	1/14/09	12:46	1	0	0.022	9.593	9.571	0.124	0.161	9.754	0.478
M8	1/14/09	13:28	1	0	0.020	0.786	0.766	0.126	0.077	0.862	0.311
M9	1/14/09	11:50	1	13*	0.009	1.370	1.361	0.110	0.127	1.497	0.085
Mag STE	1/14/09	14:10	1	3780	0.371	63.484	63.113	2.025	3.003	66.487	20.802

\*colony development was late

## Appendix D Sampling Event on 01/14/09

## Field Data

Station	Date	Time	Depth	Water T	COND	COND	Salinity	%SAT	DO	PH	ORP
			Meters	Celsius	YSI µS/cm	YSI mS/cm	YSI ppt	YSI %	YSI mg/L	YSI mg/L	YSI mg/L
<b>Background Wells</b>											
CAS	1/14/09	3.40	12:14	21.65	41.83			27.4	2.21	8.06	
SRWMD #4	1/14/09	5.46	12:56	22.21	157.9			14.9	0.60	7.71	
<b>Hickory</b>											
S1	1/14/09	4.36	9:44	21.86	634	0.596	0.31	21.7	1.89	7.57	219.1
S2	1/14/09	4.26	10:05	21.58	553	0.518	0.27	39.3	3.47	7.47	218.2
C3A	1/14/09	4.30	10:55	21.46	588	0.548	0.29	57.0	5.03	7.34	243.7
C4	1/14/09	4.37	11:12	21.33	573	0.533	0.27	69.7	6.16	7.31	250.1
C5	1/14/09	4.30	10:35	21.91	696	0.655	0.34	40.6	3.55	7.39	240.8
<b>Magnolia</b>											
M1	1/14/09	2.52	13:44	21.88	959	0.897	0.47	7.2	0.63	7.11	158.9
M2	1/14/09	2.04	13:12	19.64	719	0.645	0.35	18.0	1.64	7.09	165.7
M3	1/14/09	1.70	12:16	20.45	763	0.697	0.37	7.4	0.66	7.20	178.6
M5	1/14/09	2.00	13:00	21.36	886	0.824	0.44	8.0	0.71	7.10	163.8
M6	1/14/09	1.44	12:32	20.93	869	0.801	0.43	5.6	0.51	7.15	172.3
M7	1/14/09	1.63	12:46	21.36	894	0.832	0.44	7.1	0.63	7.12	165.8
M8	1/14/09	1.98	13:28	21.02	715	0.660	0.35	5.9	0.53	7.17	159.6
M9	1/14/09	1.84	11:50	20.67	689	0.633	0.34	11.3	1.02	7.39	180.7
Mag STE	1/14/09		14:10	18.50	1136	0.996	0.570	40.7	3.80	6.79	163.30

Appendix E Sampling Event on 03/26/09 and 03/27/09  
 Nutrient Data

				<b>F Coli</b>	<b>Nitrite</b>	<b>Nitrate+Nitrite</b>	<b>Nitrate</b>	<b>Ammonia</b>	<b>TKN</b>	<b>TN</b>	<b>Total P</b>
				EPA	EPA	EPA	calc	EPA	EPA	calc	EPA
				FC	354.1	353.3		350.3	351.4		365.2
			MDL	1	0.003	0.006	0.006	0.022	0.056	0.056	0.004
			PQL	4	0.012	0.024	0.024	0.088	0.224	0.224	0.016
<b>Station</b>	<b>Date</b>	<b>Time</b>	<b>Rep</b>	<b>Cts/100ml</b>	<b>mg/L</b>	<b>mg/L</b>	<b>mg/L</b>	<b>mg/L</b>	<b>mg/L</b>	<b>mg/L</b>	<b>mg/L</b>
<b>Background</b>											
CAS	03/26/09	13:31	1	0	0.131	0.139	0.008	0.012	0.172	0.312	0.007
SRWMD #4	03/26/09	13:12	1	4	0.019	0.863	0.845	0.014	0.130	0.993	0.008
<b>Hickory</b>											
S1	03/27/09	14:20	1	1	0.006	1.504	1.498	0.006	0.048	1.551	0.391
S1	03/27/09	14:20	L2	1	0.006	1.562	1.556	0.006	0.053	1.615	0.391
S2	03/27/09	12:15	1	18	0.009	0.104	0.095	0.008	0.169	0.273	0.067
C3A	03/27/09	13:30	1	0	0.008	0.649	0.641	0.007	0.079	0.728	0.052
C4	03/27/09	12:45	1	7	0.004	1.156	1.152	0.007	0.120	1.275	0.052
C5	03/27/09	13:50	1	10	0.035	1.019	0.984	0.008	0.133	1.152	0.032
Hick TT	03/27/09	14:45	1	26000	0.054	0.116	0.063	61.696	173.313	173.429	13.496
Hick STE	03/27/09	14:35	1	8060	0.125	4.600	4.475	44.640	116.386	120.986	12.789
Hick STE	03/27/09	14:35	F2	6760	0.129	0.248	0.119	40.043	97.017	97.265	13.193

Appendix E Sampling Event on 03/26/09 and 03/27/09  
 Nutrient Data (continued)

Station	Date	Time	Rep	F Coli	Nitrite	Nitrate+Nitrite	Nitrate	Ammonia	TKN	TN	Total P
				EPA FC	EPA 354.1	EPA 353.3	calc	EPA 350.3	EPA 351.4	calc	EPA 365.2
			MDL	1	0.003	0.006	0.006	0.022	0.056	0.056	0.004
			PQL	4	0.012	0.024	0.024	0.088	0.224	0.224	0.016
Station	Date	Time	Rep	Cts/100ml	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L
<b>Magnolia II</b>											
M1	03/26/09	13:40	1	7 *	0.006	13.243	13.236	0.116	0.361	13.604	0.130
M1	03/26/09	13:42	F2	8 *	0.008	11.372	11.364	0.117	0.348	11.720	0.138
M2	03/26/09	12:45	1	32	0.013	4.846	4.834	0.016	0.161	5.008	0.339
M3	03/26/09	11:30	1	20	0.027	1.148	1.122	0.052	0.251	1.399	0.039
M5	03/26/09	12:30	1	7	0.055	1.424	1.369	0.069	0.310	1.734	0.266
M6	03/26/09	11:50	1	10	0.093	2.062	1.969	0.037	0.229	2.292	0.027
M7	03/26/09	12:15	1	29	0.063	5.151	5.088	0.037	0.229	5.380	0.091
M8	03/26/09	13:20	1	2 *	0.026	10.284	10.259	0.015	0.157	10.441	0.091
M9	03/26/09	10:55	1	96	0.048	5.332	5.284	0.014	0.216	5.549	0.123
M9	03/26/09	10:55	L2	90	0.049	5.325	5.276	0.014	0.209	5.534	0.120
Mag TT	03/26/09	14:20	1	10500	0.127	7.761	7.634	83.644	202.893	210.654	0.128
Mag STE	03/26/09	13:58	1	2100	1.184	12.242	11.058	1.675	14.161	26.403	0.113

\*colony development was late



Appendix E Sampling Event on 03/26/09 and 03/27/09  
 Field Data (continued)

<b>Station</b>	<b>Date</b>	<b>Time</b>	<b>Rep</b>	<b>Depth</b> Meters	<b>Water T</b> <i>YSI</i> Celsius	<b>COND</b> <i>YSI</i> µS/cm	<b>COND</b> <i>YSI</i> mS/cm	<b>Salinity</b> <i>YSI</i> ppt	<b>%SAT</b> <i>YSI</i> %	<b>DO</b> <i>YSI</i> mg/L	<b>PH</b> <i>YSI</i> mg/L	<b>ORP</b> <i>YSI</i> mg/L
<b>Magnolia II</b>												
M1	03/26/09	13:40	1	2.50								
M2	03/26/09	12:45	1	2.04	<b>No YSI Data</b>							
M3	03/26/09	11:30	1	1.69								
M5	03/26/09	12:30	1	1.98								
M6	03/26/09	11:50	1	1.935								
M7	03/26/09	12:15	1	1.605								
M8	03/26/09	13:20	1	1.96								
M9	03/26/09	10:55	1	1.82								
Mag TT												
Mag STE												

Appendix F Sampling Event on 05/11/09  
Nutrient Data

				<b>F Coli</b>	<b>Nitrite</b>	<b>Nitrate+Nitrite</b>	<b>Nitrate</b>	<b>Ammonia</b>	<b>TKN</b>	<b>TN</b>	<b>Total P</b>	<b>pH</b>
				SM	EPA	EPA	calc	EPA	EPA		EPA	EPA
				9222D	354.1	353.3		350.3	351.4	calc	365.2	150.1
				1	0.003	0.006	0.006	0.022	0.056	0.056	0.004	0.011
				4	0.012	0.024	0.024	0.088	0.224	0.224	0.016	0.04
<b>Station</b>	<b>Date</b>	<b>Time</b>	<b>Rep</b>	<b>Cts/100ml</b>	<b>mg/L</b>	<b>mg/L</b>	<b>mg/L</b>	<b>mg/L</b>	<b>mg/L</b>	<b>mg/L</b>	<b>mg/L</b>	<b>units</b>
<b>Background</b>												
CAS	05/11/09	2:52	1	0	0.042	0.155	0.113	0.053	0.073	0.228	0.322	7.15
SRWMD #4	05/11/09	4:61	1	0	0.009	0.020	0.011	0.045	0.095	0.115	0.269	7.40
<b>Hickory</b>												
S1	05/11/09	16:10	1	0	0.005	0.575	0.570	0.041	0.072	0.648	0.567	7.66
S1	05/11/09	16:10	F2	0	0.006	0.453	0.447	0.040	0.069	0.522	0.547	7.79
S2	05/11/09	14:20	1	25	0.000	0.085	0.085	0.047	0.061	0.145	0.060	7.80
C3A	05/11/09	15:25	1	0	0.004	0.109	0.105	0.041	0.110	0.219	0.034	7.68
C4	05/11/09	15:05	1	0	0.001	0.281	0.281	0.044	0.113	0.394	0.047	7.72
C5	05/11/09	15:50	1	0	0.003	2.064	2.062	0.040	0.081	2.145	0.052	7.94
Hick TT	05/11/09	17:00	1	60000	0.062	0.092	0.031	59.414	235.162	235.254	2.919	7.67
Hick STE	05/11/09	16:40	1	3350	1.327	6.307	4.979	33.595	52.537	58.844	2.731	7.80

Appendix F Sampling Event on 05/11/09  
 Nutrient Data (continued)

				F Coli	Nitrite	Nitrate+Nitrite	Nitrate	Ammonia	TKN	TN	Total P	pH
				SM	EPA	EPA	calc	EPA	EPA		EPA	EPA
				9222D	354.1	353.3		350.3	351.4	calc	365.2	150.1
				1	0.003	0.006	0.006	0.022	0.056	0.056	0.004	0.011
				4	0.012	0.024	0.024	0.088	0.224	0.224	0.016	0.04
Station	Date	Time	Rep	Cts/100ml	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	units
<b>Magnolia</b>												
M1	05/11/09	12:38	1	0	0.008	2.698	2.690	0.050	0.120	2.818	0.386	7.69
M2	05/11/09	12:05	1	8	0.005	0.514	0.509	0.051	0.149	0.663	0.384	7.57
M3	05/11/09	11:00	1	5	0.002	0.266	0.264	0.042	0.109	0.375	0.251	7.62
M5	05/11/09	11:55	1	0	0.003	3.787	3.784	0.051	0.138	3.925	0.175	7.73
M6	05/11/09	12:50	1	5	0.064	2.864	2.800	0.051	0.112	2.976	0.577	7.69
M7	05/11/09	11:40	1	3	0.011	4.284	4.273	0.052	0.140	4.424	0.353	7.74
M8	05/11/09	12:15	1	0	0.003	4.351	4.348	0.053	0.142	4.493	0.244	7.63
M9	05/11/09	10:50	1	0	0.002	0.111	0.109	0.035	0.151	0.261	0.047	7.62
MAG TT	05/11/09	13:10	1	13520	0.071	0.055	0.000	38.718	168.989	169.045	1.842	7.29
MAG TT	05/11/09	13:10	F2	12410	0.070	0.070	0.000	40.402	186.769	186.839	1.150	7.32
Mag STE	05/11/09	12:50	1	8000	1.512	6.943	5.431	10.199	13.160	20.102	1.743	7.52
Mag STE	05/11/09	12:50	F2	7610	1.819	6.943	5.124	11.211	12.675	19.618	1.763	7.52

Appendix F Sampling Event on 05/11/09  
Field Data

	<b>Date</b>	<b>Depth</b>	<b>Time</b>	<b>Water T</b>	<b>COND</b>	<b>COND</b>	<b>Salinity</b>	<b>DO %SAT</b>	<b>DO</b>	<b>PH</b>	<b>ORP</b>
			<i>YSI</i>	<i>YSI</i>	<i>YSI</i>	<i>YSI</i>	<i>YSI</i>	<i>YSI</i>	<i>YSI</i>	<i>YSI</i>	<i>YSI</i>
		<b>Meters</b>		<b>Celsius</b>	<b>µS/cm</b>	<b>mS/cm</b>	<b>ppt</b>	<b>%</b>	<b>mg/L</b>	<b>mg/L</b>	<b>mg/L</b>
<b>Background</b>											
CAS	05/11/09	2.52	12:35	20.83	53				1.65	3.48	
SRWMD #4	05/11/09	4.61	12:59	21.37	299				1.39	3.36	
<b>Hickory</b>											
S1	05/11/09	3.75	16:19	21.57	645	0.602	0.31	5.6	0.49	4.00	36.3
S1	05/11/09										
S2	05/11/09	3.445	14:18	21.22	548	0.508	0.27	11.6	1.02	4.13	12.9
C3A	05/11/09	3.475	15:24	20.85	555	0.511	0.27	31.3	2.79	4.03	35.6
C4	05/11/09	3.54	14:59	20:57	541	0.495	0.26	36.3	3.26	4.08	33.1
C5	05/11/09	3.475	15:50	21.33	641	0.596	0.31	10.9	0.96	4.00	34.7
Hick TT	05/11/09		16:59	25.91	1611	1.639	0.81	8.9	0.72	4.40	-241.2
Hick STE	05/11/09		16:44	25.96	1403	1.429	0.7	52.6	4.28	4.44	37.2

Appendix F Sampling Event on 05/11/09  
 Field Data (continued)

	<b>Date</b>	<b>Depth</b>	<b>Time</b>	<b>Water T</b>	<b>COND</b>	<b>COND</b>	<b>Salinity</b>	<b>DO %SAT</b>	<b>DO</b>	<b>PH</b>	<b>ORP</b>
			<i>YSI</i>	<i>YSI</i>	<i>YSI</i>	<i>YSI</i>	<i>YSI</i>	<i>YSI</i>	<i>YSI</i>	<i>YSI</i>	<i>YSI</i>
Station	Date	Meters		Celsius	µS/cm	mS/cm	ppt	%	mg/L	mg/L	mg/L
<b>Magnolia</b>											
M1	05/11/09	1.41	12:34	21.52	827	0.773	0.41	5.6	0.49	3.94	43.5
M2	05/11/09	0.93	12:04	19.95	492	0.445	0.24	5.4	0.49	3.98	23.1
M3	05/11/09	0.62	10:55	19:55	619	0.554	0.3	6.5	0.6	3.97	43.6
M5	05/11/09	0.91	11:54	20.64	748	0.686	0.37	4.2	0.38	3.93	43.0
M6	05/11/09	0.41	11:20	20.49	805	0.73	0.4	2	0.17	3.86	45.7
M7	05/11/09	0.565	11:37	20.32	810	0.738	0.4	2.7	0.24	3.80	43.4
M8	05/11/09	0.87	12:14	19.89	667	0.602	0.33	4.3	0.39	4.00	33.8
M9	05/11/09	0.735	10:44	19.65	562	0.505	0.27	6.9	0.63	4.12	42.6
MAG TT	05/11/09		13:07	26.91	1430	1.482	0.71	1.6	0.11	3.78	-249.4
Mag STE	05/11/09		12:48	26.15	1237	1.264	0.61	9.9	0.8	3.78	55.3

Appendix G Sampling Event on 06/04/09  
Nutrient Data

				<b>F Coli</b>	<b>Nitrite</b>	<b>Nitrate+Nitrite</b>	<b>Nitrate</b>	<b>Ammonia</b>	<b>TKN</b>	<b>TN</b>	<b>Total P</b>
				SM	EPA	EPA	calc	EPA	EPA		EPA
				9222D	354.1	353.3		350.3	351.4	calc	365.2
			MDL	1	0.003	0.006	0.006	0.022	0.056	0.056	0.004
			PQL	4	0.012	0.024	0.024	0.088	0.224	0.224	0.016
				<b>Cts/100m</b>							
<b>Station</b>	<b>Date</b>	<b>Time</b>	<b>Rep</b>	<b>1</b>	<b>mg/L</b>	<b>mg/L</b>	<b>mg/L</b>	<b>mg/L</b>	<b>mg/L</b>	<b>mg/L</b>	<b>mg/L</b>
<b>Background</b>											
CAS	06/04/09	11:00	1	0	0.052	0.476	0.424	0.407	0.832	1.308	0.097
SRWMD #4	06/04/09	11:30	1	0	0.033	1.032	0.999	0.466	0.655	1.687	0.285
<b>Hickory</b>											
S1	06/04/09	11:10	1	0	0.133	4.841	4.709	0.264	0.565	5.407	0.782
S2	06/04/09	10:14	1	0	0.010	0.245	0.235	0.335	0.445	0.690	0.024
C3A	06/04/09	10:46	1	4	0.004	0.208	0.205	0.405	0.367	0.575	0.022
C4	06/04/09	10:34	1	76	0.008	0.467	0.459	0.304	0.450	0.916	0.079
C5	06/04/09	11:00	1	0	0.021	2.982	2.961	0.585	0.485	3.468	0.028
Hick STE	06/04/09	11:20	1	6200	4.607	10.785	6.179	5.759	6.783	17.569	2.668
Hick TT	06/04/09	11:32	1	15400	0.115	0.099	0.000	61.506	58.459	58.558	3.321

Appendix F Sampling Event on 06/04/09  
 Nutrient Data (continued)

				<b>F Coli</b>	<b>Nitrite</b>	<b>Nitrate+Nitrite</b>	<b>Nitrate</b>	<b>Ammonia</b>	<b>TKN</b>	<b>TN</b>	<b>Total P</b>
				EPA	EPA	calc	EPA	EPA		EPA	
				9222D	354.1	353.3		350.3	351.4	calc	365.2
			MDL	1	0.003	0.006	0.006	0.022	0.056	0.056	0.004
			PQL	4	0.012	0.024	0.024	0.088	0.224	0.224	0.016
<b>Station</b>	<b>Date</b>	<b>Time</b>	<b>Rep</b>	<b>Cts/100ml</b>	<b>mg/L</b>	<b>mg/L</b>	<b>mg/L</b>	<b>mg/L</b>	<b>mg/L</b>	<b>mg/L</b>	<b>mg/L</b>
M1	06/04/09	14:56	1	1	0	0.005	6.083	6.078	6.127	0.945	7.072
M2	06/04/09	14:32	1	1	0	0.004	0.184	0.180	0.226	0.623	0.849
M3	06/04/09	12:30	1	4	0.178	13.957	13.779	0.544	0.524	14.481	0.196
M5	06/04/09	14:22	1	1	4	0.016	2.943	2.927	3.011	0.683	3.694
M6	06/04/09	12:36	1	32	2.626	1.415	0.000	0.359	0.686	2.101	0.056
M7	06/04/09	12:45	1	14	0.202	19.425	19.223	0.373	0.562	19.987	0.220
M8	06/04/09	14:44	1	1	0	0.002	0.102	0.099	0.137	0.669	0.807
M9	06/04/09	12:15	1	2	0.017	0.239	0.221	0.419	0.562	0.801	0.159
M9	06/04/09	12:15	F2	2	0.019	0.297	0.277	0.401	0.471	0.767	0.056
Mag STE	06/04/09	12:05	1	6700	3.000	25.002	22.003	11.734	17.042	42.044	3.535
Mag STE	06/04/09	12:05	F2	10700	0.022	25.330	25.308	11.848	14.407	39.737	3.314
Mag TT	06/04/09	11:55	1	102100	0.115	0.075	0.000	100.784	102.330	102.405	3.435

Appendix F Sampling Event on 06/04/09  
Field Data

Station	Date	Depth Meters	Time YSI	Water T YSI Celsius	COND YSI µS/cm	COND YSI mS/cm	Salinity YSI ppt	DO %SAT YSI %	DO YSI mg/L	PH YSI mg/L	ORP YSI mg/L	PH Probe units
<b>Background</b>												
CAS	06/04/09	2.79	11:04	21.33	277				2.19	4.48		
SRWMD #4	06/04/09	4.9	11:43	21.48	206				1.45	8.46		
<b>Hickory</b>												
S1	06/04/09	3.73	11:09	21.62	660	0.618	0.32	3.2	0.29	7.37	29.7	7.2
S2	06/04/09	3.64	10:12	21.13	532	0.493	0.26	4.8	0.43	7.51	33.4	7.3
C3A	06/04/09	3.66	10:44	20.83	550	0.506	0.27	27.0	2.50	7.41	32.6	7.2
C4	06/04/09	3.73	10:32	20.96	542	0.500	0.26	30.6	2.71	7.49	32.6	7.2
C5	06/04/09	3.66	10:59	21.41	658	0.613	0.32	32.4	2.87	7.38	31.7	7.2
Hick STE	06/04/09		11:24	25.41	874	0.881	0.43	45.0	3.68	7.77	26.3	
Hick TT	06/04/09		11:31	25.85	1309	1.330	0.65	27.4	2.22	6.98	-146.8	6.8
<b>Magnolia</b>												
M1	06/04/09	1.97	14:54	21.16	839	0.778	0.41	2.3	0.21	7.33	13.7	7.1
M2	06/04/09	1.45	14:31	21.04	616	0.569	0.3	3.3	0.29	7.48	10.3	7.1
M3	06/04/09	1.115	12:28	20.03	663	0.600	0.32	4.0	0.36	7.35	10.2	7.2
M5	06/04/09	1.39	14:21	20.59	745	0.682	0.36	4.9	0.44	7.76	14.0	7.2
M6	06/04/09	0.815	12:35	20.39	778	0.710	0.38	3.7	0.33	7.26	11.6	7.2
M7	06/04/09	1.03	12:43	20.57	744	0.681	0.36	4.3	0.37	7.33	11.5	7.2
M8	06/04/09	1.375	14:43	20.24	638	0.580	0.32	2.5	0.23	7.47	13.3	7.2
M9	06/04/09	1.24	12:11	20.22	566	0.515	0.27	4.0	0.36	7.46	9.9	7.1
Mag STE	06/04/09		12:04	26.50	1113	1.145	0.55	55.7	4.47	6.21	-2.5	6.2
Mag TT	06/04/09		11:54	27.07	1463	1.521	0.73	16.5	1.30	6.94	-125.9	6.7



# Florida Onsite Sewage Nitrogen Reduction Strategies Study

Task A.2

## Literature Review of Nitrogen Reduction Technologies for Onsite Sewage Treatment Systems

### Final Report

June 2009

44237-001

**HAZEN AND SAWYER**  
Environmental Engineers & Scientists

In association with



**AET**  
Applied Environmental Technology

**OTIS  
ENVIRONMENTAL  
CONSULTANTS, LLC**

# **Florida Onsite Sewage Nitrogen Reduction Strategies Study**

## **TASK A.2 FINAL REPORT**

### **Literature Review of Nitrogen Reduction Technologies for Onsite Sewage Treatment Systems**

#### **Prepared for:**

Florida Department of Health  
Division of Environmental Health  
Bureau of Onsite Sewage Programs  
4042 Bald Cypress Way Bin #A-08  
Tallahassee, FL 32399-1713

FDOH Contract CORCL

**June 2009**

#### **Prepared by:**

**HAZEN AND SAWYER**  
Environmental Engineers & Scientists

In Association With:

**AET**  
Applied Environmental Technology

**Otis Environmental  
Consultants, LLC**



## Table of Contents

---

Section 1.0	Study Background .....	1-1
Section 2.0	Nitrogen in the Environment .....	2-1
	2.1 Nitrogen Fixation .....	2-2
	2.2 Nitrogen Uptake .....	2-3
	2.3 Nitrogen Mineralization (Ammonification) .....	2-3
	2.4 Nitrification .....	2-3
	2.5 Denitrification .....	2-4
Section 3.0	Nitrogen in Wastewater .....	3-1
Section 4.0	Wastewater Nitrogen Reduction Technologies.....	4-1
	4.1 Biological Nitrification / Denitrification Processes .....	4-3
	4.1.1 Mixed Biomass with Alternating Aerobic / .....	4-5
	Anoxic Environments (simultaneous)	
	4.1.2 Mixed Biomass Recycling Systems .....	4-5
	4.1.3 Two-stage External Electron.....	4-6
	Donor Denitrification	
	4.1.4 Anaerobic Ammonium Oxidation .....	4-6
	4.2 Physical / Chemical Nitrogen Removal Processes .....	4-6
	4.3 Source Separation .....	4-8
	4.4 Natural Systems.....	4-8
	4.5 Passive Nitrogen Removal.....	4-9
Section 5.0	Review of Onsite Nitrogen Reducing Technologies and Practices.....	5-1
	5.1 Source Separation .....	5-1
	5.1.1 Urine Separation and Recovery.....	5-5
	5.1.1.1 Urine Separation .....	5-5
	5.1.1.2 Urine Treatment .....	5-7

o:\44237-001\R0071\Wpdocs\Report\Final

- 5.1.1.3 Direct Nitrification ..... 5-7
- 5.1.1.4 Precipitation ..... 5-8
- 5.1.2 Greywater Collection and Reuse ..... 5-9
  - 5.1.2.1 Greywater Treatment ..... 5-10
- 5.1.3 Black Water Separation and Treatment..... 5-11
- 5.2 Primary Treatment (Septic Tank)..... 5-12
- 5.3 Biological Nitrification / Denitrification Processes ..... 5-12
  - 5.3.1 Mixed Biomass Nitrification / Denitrification .... 5-15
    - 5.3.1.1 Suspended Growth ..... 5-15  
(Activated Sludge) Reactors
    - 5.3.1.2 Recirculating Media Filters ..... 5-16
    - 5.3.1.3 Integrated Fixed-Film Activated .... 5-23  
Sludge (IFAS)
  - 5.3.2 Segregated Biomass (Two Stage)..... 5-23  
Denitrification
    - 5.3.2.1 Anoxic Packed Bed Reactors..... 5-24
    - 5.3.2.2 Heterotrophic Denitrification ..... 5-26
    - 5.3.2.3 Autotrophic Denitrification ..... 5-27
- 5.4 Physical / Chemical Nitrogen Reduction Processes ..... 5-29
  - 5.4.1 Membrane Processes ..... 5-29
- 5.5 Natural Systems..... 5-30
  - 5.5.1 Soil Infiltration ..... 5-30
  - 5.5.2 Constructed Wetlands ..... 5-38
  - 5.5.3 Evapotranspiration and Vegetative Uptake .... 5-38
- 5.6 Modifications to Conventional Onsite..... 5-39  
Treatment Systems

o:\44237-001\R0071\Wpdocs\Report\Final

Section 6.0 OSTDS Nitrogen Reduction Strategies in Florida.....6-1

- 6.1 Nitrogen Reducing Technologies..... 6-1
- 6.2 Categories of Technologies ..... 6-1
- 6.3 Process Applications for OSTDS ..... 6-2
- 6.4 Process Performance Limitations ..... 6-3
- 6.5 Emerging Technologies ..... 6-4
- 6.6 Nitrogen Reduction Implementation Strategies ..... 6-5
  - 6.6.1 Establishing Nitrogen Reduction Standards ..... 6-5
  - 6.6.2 Technology Selection ..... 6-6
  - 6.6.3 Management and Enforcement ..... 6-6

Section 7.0 References ..... 7-1

Appendix A Glossary..... A-1

o:\44237-001R007\Wpdocs\Report\Final

**List of Tables**

Table 3.1 Nitrogen Species Concentrations in Raw Wastewater and Septic Tank Effluent by Source (Lowe, Rothe et al., 2006) ..... 3-2

Table 3.2 Daily Septic Tank Effluent Flows by Source in Gallons/Day (Lowe, Rothe et al., 2006) ..... 3-3

Table 5.1 Per Capita Volume and Constituent Loading in U.S. Domestic Sewage ..... 5-3

Table 5.2 Volume and Constituent Concentrations of Domestic Sewage Wastestreams for a Four Person Household in the U.S. .... 5-4

Table 5.3 Biological Denitrification Processes and Typical Nitrogen Reduction Limits of OSTDS ..... 5-14

Table 5.4 Summary of Media Filter Performance ..... 5-19

Table 5.5 Summary of Saturated Anoxic Media Reactors ..... 5-25

Table 5.6 Total Nitrogen Removals below Soil Infiltration Zones (Siegrist and Jenssen 1989) ..... 5-32

Table 5.7 Estimates of TN Removal Based on Soil Texture Below a Traditional Household Wastewater Infiltration System (Long 1995) ..... 5-32

Table 5.8 Total Nitrogen Removals Found in Various Studies of OWTS ..... 5-33

Table 5.9 NRCS Drainage Classes, Descriptions and Expected Impacts on Denitrification ..... 5-34

o:\44237-001R007\Wpdocs\Report\Final

**List of Figures**

Figure 2-1 The Nitrogen Cycle (Harrison, 2003).....2-1

Figure 2-2 Nitrogen Transformation in Biological Processes .....2-2  
(Eckenfelder and Argaman, 1991)

Figure 2-3 Recent Increases in Anthropogenic N Fixation.....2-3  
in Relation to “Natural” N Fixation (Harrison, 2003)

Figure 2-4 Illustration of the Classic Heterotrophic Denitrification .....2-5  
and Anammox Denitrification (Butler and Richardson, 2005)

Figure 3-1 Cumulative Frequency of Total Nitrogen Concentrations in .....3-2  
Septic Tank Effluent (Lowe, Rothe et al., 2006)

Figure 4-1 Treatment Options for Reducing Nitrogen in Household Sewage .....4-2

Figure 4-2 Onsite Treatment Technology Categories for Biological .....4-3  
Nitrification/Denitrification Processes

Figure 4-3 Alternating Oxidic / Anoxic Reactor Denitrification .....4-5

Figure 4-4 Mixed Biomass Recycling Denitrification Process .....4-5

Figure 4-5 External Electron Donor Denitrification Process.....4-6

Figure 4-6 Onsite Treatment Technology Categories for .....4-7  
Physical/Chemical Processes

Figure 4-7 Nitrogen Source Separation Categories .....4-8

Figure 4-8 Categories of Natural Systems for Nitrogen Reduction .....4-9

Figure 4-9 Passive Two-Stage Denitrification System .....4-11

Figure 5-1 Domestic Wastestream Components .....5-2

Figure 5-2 Two Swedish Urine Separating Toilets.....5-6  
(EcoSan and Novaquatis)

Figure 5-3 Relative Complexities of Nitrification / Denitrification.....5-15  
Unit Operations

Figure 5-4: Common Integrated Fixed Film Activated Sludge .....5-23  
(IFAS) Processes)

o:\44237-001\R0071\Wpdocs\Report\Final

## Section 1.0

### Study Background

---

The quality of Florida's surface and groundwater resources is increasingly being threatened by anthropogenic sources of pollutants. Nitrogen is one of these pollutants, which is both an environmental and drinking water concern. As little as one milligram per liter of nitrogen has been shown to lead to algae growth in Florida's springs. In concentrations greater than 10 mg/L, it also is a drinking water concern.

Onsite sewage treatment and disposal systems (OSTDS) are one of the sources of nitrogen. These systems are used for household wastewater treatment where sewers are unavailable. The systems discharge partially treated wastewater into the soil where further treatment is achieved as the water percolates to groundwater. Approximately one-third of Florida's population is served by OSTDS representing approximately 2.5 million systems (Briggs, Roeder et al. 2007). This number is expected to increase with rising population in the state. Consequently, OSTDS are one of the largest artificial groundwater recharge sources in Florida. However, few OSTDS are designed to remove nitrogen. Consequently, nitrogen can reach drinking water wells or surface water raising concerns over risks to human health and the environment.

In 2008, the Florida Department of Health was directed by the State Legislature to develop a comprehensive program to examine nitrogen reduction strategies for OSTDS in Florida. To comply with this directive, the Department initiated the *Florida Onsite Sewage Nitrogen Reduction Strategies (FOSNRS) Study*, to develop strategies for nitrogen reduction that complement the use of conventional OSTDS. The study includes four primary tasks:

Task A: Identification of available and emerging nitrogen reduction technologies suitable for use in OSTDS and to rank the systems for field testing priority;

Task B: Evaluation of performance of the selected systems under actual field conditions and associated costs of such OSTDS nitrogen reduction strategies in comparison to conventional and existing technologies;

Task C: Evaluation of naturally occurring nitrogen reduction in soil and groundwater below OSTDS; and

Task D: Development of a simple predictive model of nitrogen reduction in unsaturated soil and shallow water table under and downgradient of OSTDS.

This report presents the results from the first task of this study. It incorporates, updates and expands the scope of the literature review that was prepared as part of the "*Florida Passive Nitrogen Removal Study (PNRS) Final Report*" (Smith, Otis et al. 2008). This current update also reviews the broader range of nitrogen reduction technologies to include both passive and active systems.

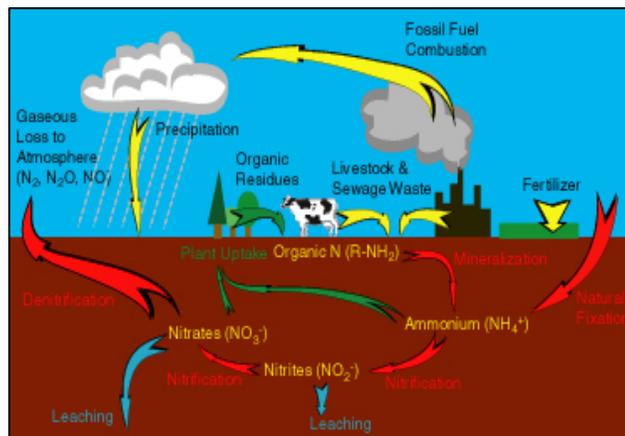
o:\44237-00\FR007\Wpdocs\Report\Final

## Section 2.0

### Nitrogen in the Environment

Nitrogen is ubiquitous in the environment. It is an essential component of DNA, RNA, and proteins, which are the building blocks of life that all organisms require to live and grow. Approximately, 78 percent of the earth's atmosphere is  $N_2$ , but this is unavailable for use by organisms because of the strong triple bond between the two N atoms of the molecule, which makes it relatively inert. In order for plants and animals to be able to use nitrogen,  $N_2$  gas must first be converted to a more chemically available form such as ammonium ( $NH_4^+$ ), nitrate ( $NO_3^-$ ), or organic nitrogen (e.g. urea -  $(NH_3)_2CO$ ). Because of the inert nature of  $N_2$  biologically available nitrogen is often in short supply in natural ecosystems, limiting plant growth and biomass accumulation.

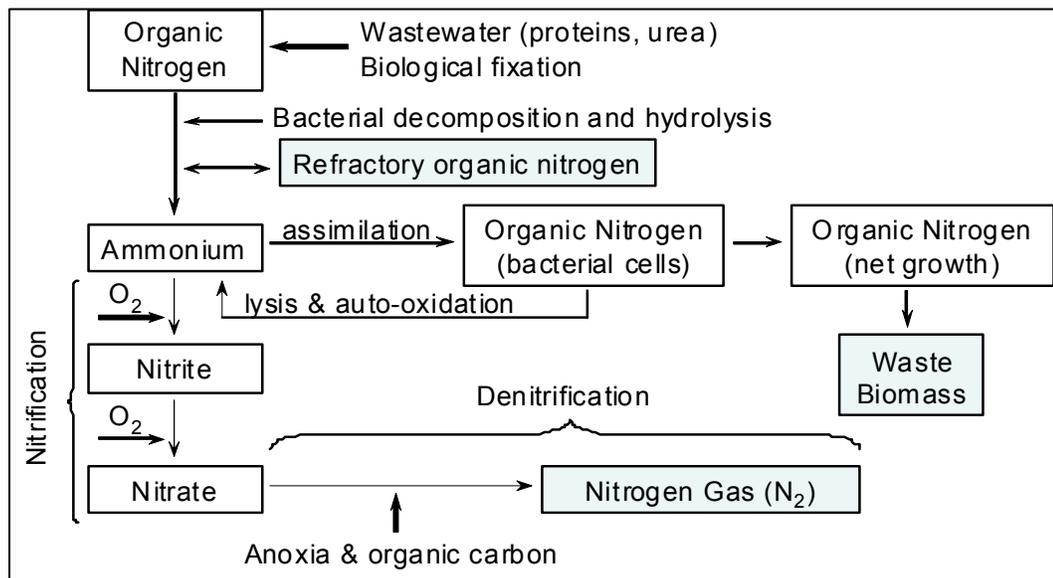
Nitrogen takes many forms, both inorganic and organic. It also exists in many different oxidation states as well. It cycles between the atmosphere, biosphere and geosphere in different forms or species (Figure 2-1). Like other biogeochemical cycles such as carbon, the nitrogen cycle consists of various "storage pools" and processes by which the "pools" exchange nitrogen (arrows in Figure 2-1).



**Figure 2-1: The Nitrogen Cycle (Harrison, 2003)**

(Yellow arrows indicate human sources; red arrows indicate microbial transformations; blue arrows indicate physical forces acting on nitrogen; green arrows indicate natural, non-microbial processes affecting the form and fate of nitrogen.)

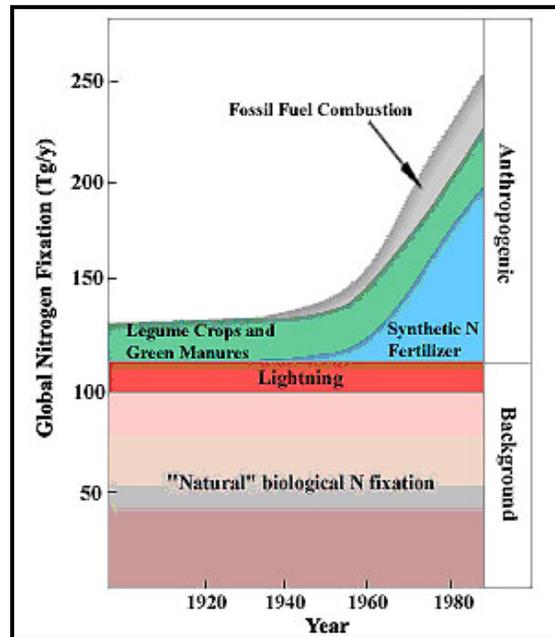
Five principal processes cycle nitrogen through the environment: nitrogen fixation, nitrogen uptake (incorporation by organisms), nitrogen mineralization (decay), nitrification, and denitrification (Figure 2-2). Microorganisms, particularly bacteria, play major roles in all of the principal nitrogen transformations. As microbially mediated processes, the rates of these nitrogen transformations are affected by environmental factors that influence microbial activity, such as temperature, moisture, and resource availability.



**Figure 2-2: Nitrogen Transformation in Biological Processes**  
(Eckenfelder and Argaman, 1991)

## 2.1 Nitrogen Fixation

Nitrogen fixation is the only way organisms can obtain nitrogen directly from the atmosphere. This process converts nitrogen gas,  $N_2$ , to ammonium,  $NH_4^+$ . Bacteria from the genus *Rhizobium* are the only organisms that can fix nitrogen directly from the atmosphere through metabolic processes. Other natural processes that can fix nitrogen are high-energy events such as lightning and forest fires. While significant, the amounts are much smaller than biological fixation. The annual natural fixation of gaseous nitrogen is only a small amount relative to the local stores of previously fixed nitrogen, which cycles within ecosystems. However in the last century, anthropogenic activities such as the burning of fossil fuels and the use of synthetic fertilizers have doubled the amount of fixed nitrogen to where today it exceeds the combined total of all natural sources (Figure 2-3).



**Figure 2-3: Recent Increases in Anthropogenic N Fixation in Relation to “Natural” N Fixation**  
(Harrison, 2003)

## 2.2 Nitrogen Uptake

The ammonia produced by nitrogen fixing bacteria is in the form of ammonium ions, which are positively charged and consequently adsorbed to negatively charged clay particles and soil organic matter. The adsorbed ammonium is thereby held in the soil until it is taken up by plants and organisms for incorporation into organic biomass or conversion to nitrate.

## 2.3 Nitrogen Mineralization (Ammonification)

After nitrogen is incorporated into organic matter, it can be converted back into inorganic nitrogen by a process called nitrogen mineralization or by decomposition of dead organisms. Mineralization converts the organic nitrogen back into ammonium, which makes the nitrogen available for use by plants or for further transformation into nitrate ( $\text{NO}_3^-$ ) through nitrification.

## 2.4 Nitrification

Nitrification is a biological process that converts ammonium into nitrate. This process is used by chemoautotrophic bacteria to acquire the energy released by the conversion of ammonium to produce their own food from other inorganic compounds. This can only be

done in the presence of oxygen. Since the conversion produces hydrogen ions, the pH can be lowered to a point where the nitrifying bacteria can no longer thrive. Therefore, sufficient alkalinity is needed to buffer the pH so that acidic conditions do not occur to inactivate the nitrifiers and prevent complete nitrification. Also, the nitrifying bacteria are very sensitive to cold temperatures, which can slow the reactions. Though nitrate can be utilized by organisms for growth, the nitrate produced is negatively charged, which in soils is not adsorbed but travels with the soil water until captured, taken up by plant roots or denitrified as described in the next section.

## 2.5 Denitrification

Denitrification also is a biological process that converts nitrate to reduced forms of nitrogen. Biological denitrification is the only nitrogen transformation that removes nitrogen from ecosystems. Once converted to  $N_2$ , the nitrogen is not likely to be reconverted to a biologically available form except through nitrogen fixation.

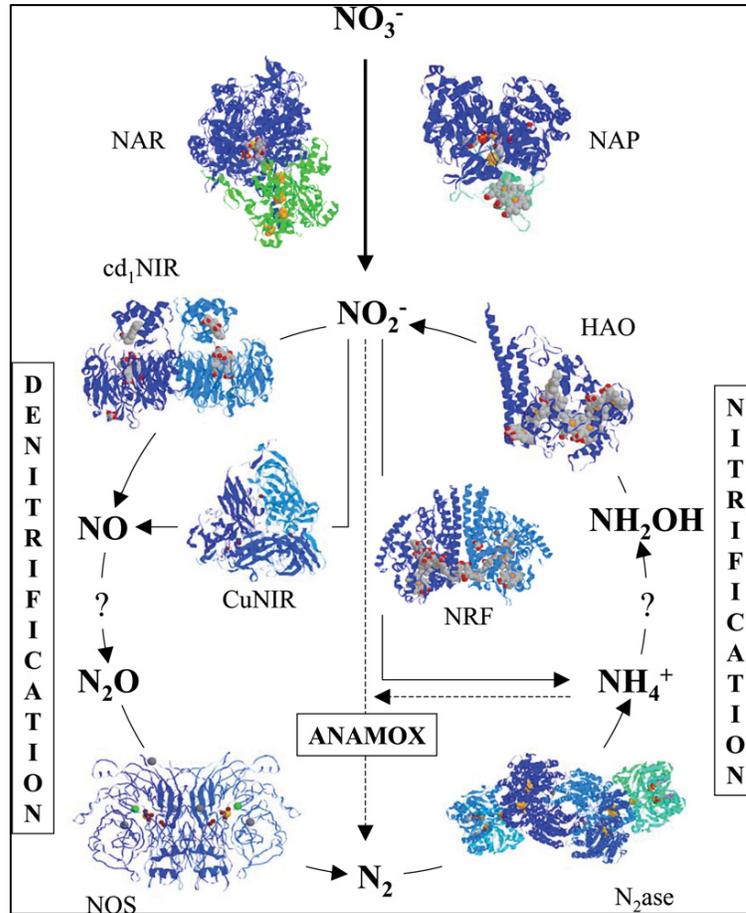
At least two biologically mediated denitrification processes are known to occur. The one considered dominant and well understood is performed by facultative heterotrophic or autotrophic bacteria under anoxic conditions (no free oxygen). The heterotrophs use organic carbon as an electron donor and the oxygen from the nitrate molecule and its resulting breakdown compounds as the electron acceptors to obtain energy necessary for their growth. This process reduces the nitrate to nitrogen gas following the sequence of  $NO_3^- \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2$ . If the process is interrupted before the sequence is complete, nitric oxide (NO) and nitrous oxide ( $N_2O$ ) can be released, which contribute to smog and greenhouse gases respectively.

Autotrophs use inorganic compounds such as sulfur, iron and hydrogen as electron donors in place of organic carbon to obtain their energy for growth. The combined oxygen on the nitrate molecule and its breakdown compounds are still used as the electron acceptors. The advantage of using autotrophs over heterotrophs is primarily in the management of the electron donors. Inorganic compounds are easier to manage and maintain than organic carbon.

The other biologically mediated denitrification process has been recognized only recently and is still poorly understood. However, it appears to be a significant factor in the conversion of nitrogen compounds to nitrogen gas in soils, wetlands, and marine, freshwater, and estuarine sediments. It is a two step biochemical process in which ammonia-oxidizing bacteria (*Nitrosomonas sp.*) partially oxidize ammonium to nitrite ( $NO_2^-$ ) followed by the conversion of the remaining ammonium directly to  $N_2$  by Anammox bacteria, which use the ammonium as an electron donor and nitrite as the electron acceptor. Organic carbon is not necessary as an electron donor in this pathway as it is in hetero-

trophic denitrification. It is quite likely that the two denitrification processes occur together in environments where aerobic and anoxic conditions fluctuate.

These two denitrification processes are illustrated in Figure 2-4 below.



**Figure 2-4: Illustration of the Classic Heterotrophic Denitrification and Anammox Denitrification (Structures of the enzymes are shown in each step; question marks represent unsolved structures) (Butler and Richardson, 2005)**

o:\44237-001\R007\Wpdocs\Report\Final

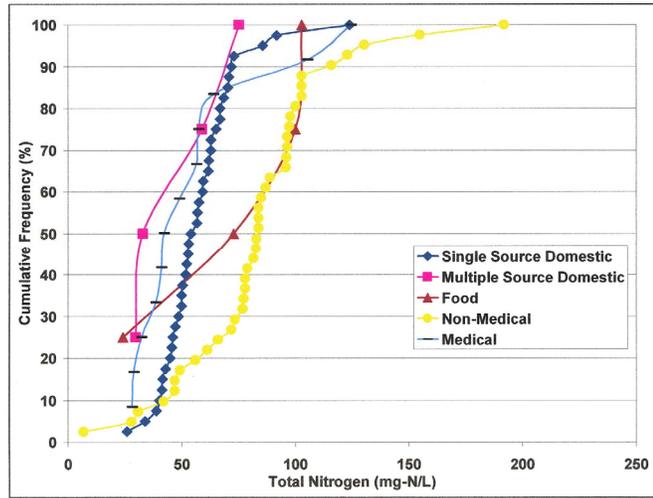
## Section 3.0

### Nitrogen in Wastewater

---

Sizing and design of a nitrification/denitrification treatment system depends in part, on the mass of nitrogen in the wastewater to be removed. Our diets largely determine the amount of nitrogen discharged daily into an OSTDS. On average each person in the U.S. discharges approximately 11.2 grams of nitrogen into wastewater each day (EPA, 2002). 70 to 80 percent of this is discharged as toilet wastes (Lowe, Rothe et al., 2006; U.S. EPA, 2002). Another 15 percent is primarily from food preparation, which enters the waste stream via kitchen sinks and dishwashers. Various household products contain nitrogen compounds but these contribute only minor amounts of nitrogen. Commercial establishments will have different wastewater nitrogen loadings based on their use (Figure 3-1 and Table 3.1).

The concentration of TN in household wastewater will depend on the number of residents in the home. As the number increases, water use per capita typically decreases but the nitrogen loading does not. Consequently, homes with more residents often have higher total nitrogen concentrations in their wastewater. Therefore, using TN concentration without good flow estimates based on expected occupancy of the home can result in under or over sizing of the OSTDS. Measured average per capita daily wastewater flows show that they typically range from 50 to 70 gpd per person (Brown&Caldwell, 1984; Anderson and Siegrist 1989; Anderson, Mulville-Friel et al. 1993; Mayer, DeOreo et al. 1999), which result in a raw wastewater nitrogen concentration of 59 to 42 mg-N/L respectively. In commercial establishments, the daily wastewater flow will vary by use (Table 3.2).



**Figure 3-1: Cumulative Frequency of Total Nitrogen Concentrations in Septic Tank Effluent**  
(Lowe, Rothe et al., 2006)

**Table 3.1**  
**Nitrogen Species Concentrations in Raw Wastewater and Septic Tank Effluent by Source (Lowe, Rothe et al., 2006)**

		Median		Average		Standard Deviation		Range		Number of Reported Values	
		Raw	STE	Raw	STE	Raw	STE	Raw	STE	Raw	STE
Total nitrogen	Single-Source Domestic	63	55.4	87.0	57.7	45.2	17.1	44.1-189	26-124	11	43
	Multiple-Source Domestic	-	46	-	49.3	-	21.7	-	29.8-75.3	2	4
	Food	-	86.5	-	75.0	-	36.5	-	24.2-103	-	4
	Non-Medical	-	84.0	-	83.8	-	33.0	-	7-192	1	41
	Medical	-	45.6	-	55.8	-	30.2	-	28.3-125	-	12
Kjeldahl nitrogen	Single-Source Domestic	62	52	78.0	54.2	40.1	14.8	43-123.9	27-94.4	5	25
	Multiple-Source Domestic	-	-	-	-	-	-	-	-	2	2
	Food	-	71	-	65.6	-	17.3	-	30-82	-	7
	Non-Medical	-	100	-	233	-	257	-	30-830	3	26
	Medical	-	-	-	-	-	-	-	-	-	-
Ammonia nitrogen	Single-Source Domestic	47.5	36.1	53.4	37.2	37.7	14.8	8.8-154	0-96.2	12	80
	Multiple-Source Domestic	-	30	-	34.2	-	13.68	-	20.1-55	-	7
	Food	-	-	-	-	-	-	-	-	-	-
	Non-Medical	178	83	289	186	345	229	32.2-767	19.8-890	4	37
	Medical	-	-	-	-	-	-	-	-	-	-
Nitrate nitrogen	Single-Source Domestic	0.16	0.20	0.49	0.82	0.56	1.9	0.05-1.1	0-10.3	5	45
	Multiple-Source Domestic	-	-	-	-	-	-	-	-	-	3
	Food	-	-	-	-	-	-	-	-	-	-
	Non-Medical	-	0.23	-	0.45	-	0.53	-	0-1.4	1	7
	Medical	-	-	-	-	-	-	-	-	-	-

- value not reported or calculated for 3 or less reported data values.

o:\44237-001R007\Wpdocs\Report\Final

**Table 3.2**  
**Daily Septic Tank Effluent Flows by Source in Gallons/Day**  
**(Lowe, Rothe et al. 2006)**

	<b>Median</b>	<b>Average</b>	<b>Standard Deviation</b>	<b>Range</b>	<b>Number of Reported Values</b>
Single-Source Domestic	161	184	84.8	62.9-388	30
Multiple-Source Domestic	-	-	-	-	3
Food	353	814	1,079	73.2-3,791	12
Non-Medical	234	1,554	3,056	30-14,100	26
Medical	-	-	-	-	-

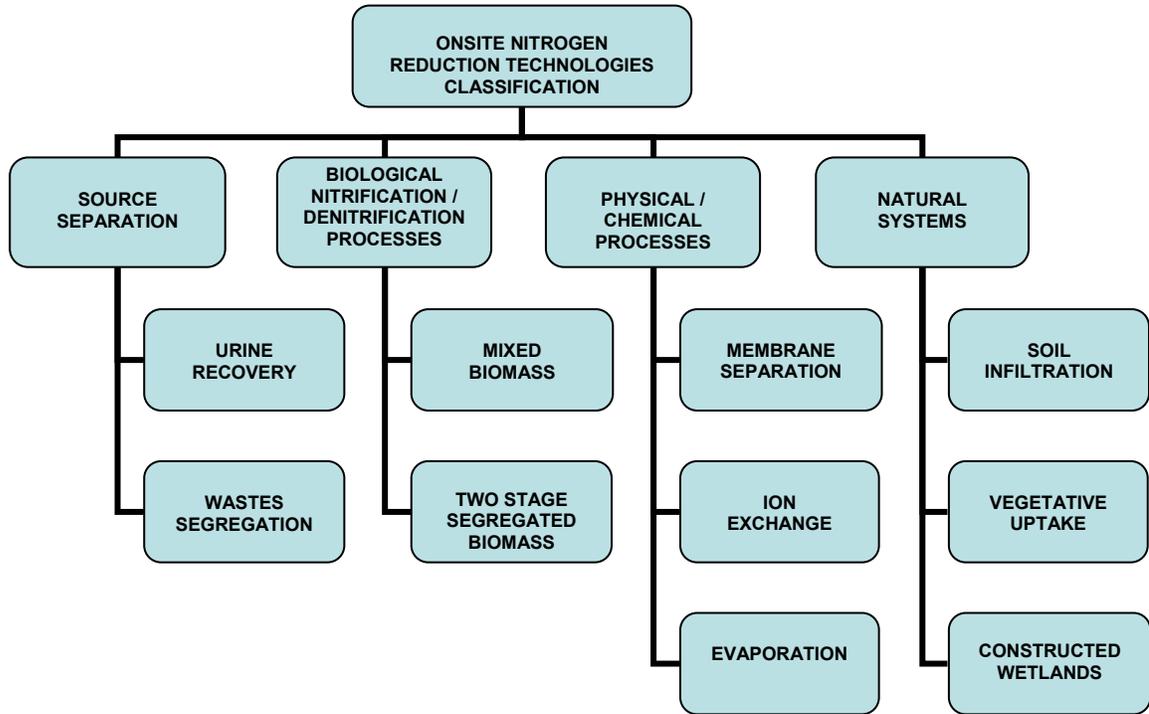
- value not reported or calculated for 3 or less reported data values.

## **Section 4.0**

### **Wastewater Nitrogen Reduction Technologies**

---

A variety of nitrogen reduction technologies exist and are available for use with onsite treatment systems. The technologies can be grouped into four general process categories; source separation, physical/chemical processes, biological nitrification/denitrification, and natural systems (Figure 4-1). Natural systems, which primarily rely on the assimilative capacity of the receiving environment, have been the most prevalent of the systems used to protect public health and our water resources. They are passive systems that are simple in design, easy to use, and require little attention by the owner. However, their treatment performance is difficult to monitor which raises concerns in nitrogen sensitive environments. In these environments, biological nitrification/denitrification has been the preferred method for most applications. Physical/chemical reduction methods have been generally less favored because of the greater need for operator attention, greater chemical and energy costs and larger volumes of residuals that may be generated. Source separation is an emerging option as the technologies improve and the nutrients recovered are increasingly valued. Each of these categories is briefly described here.

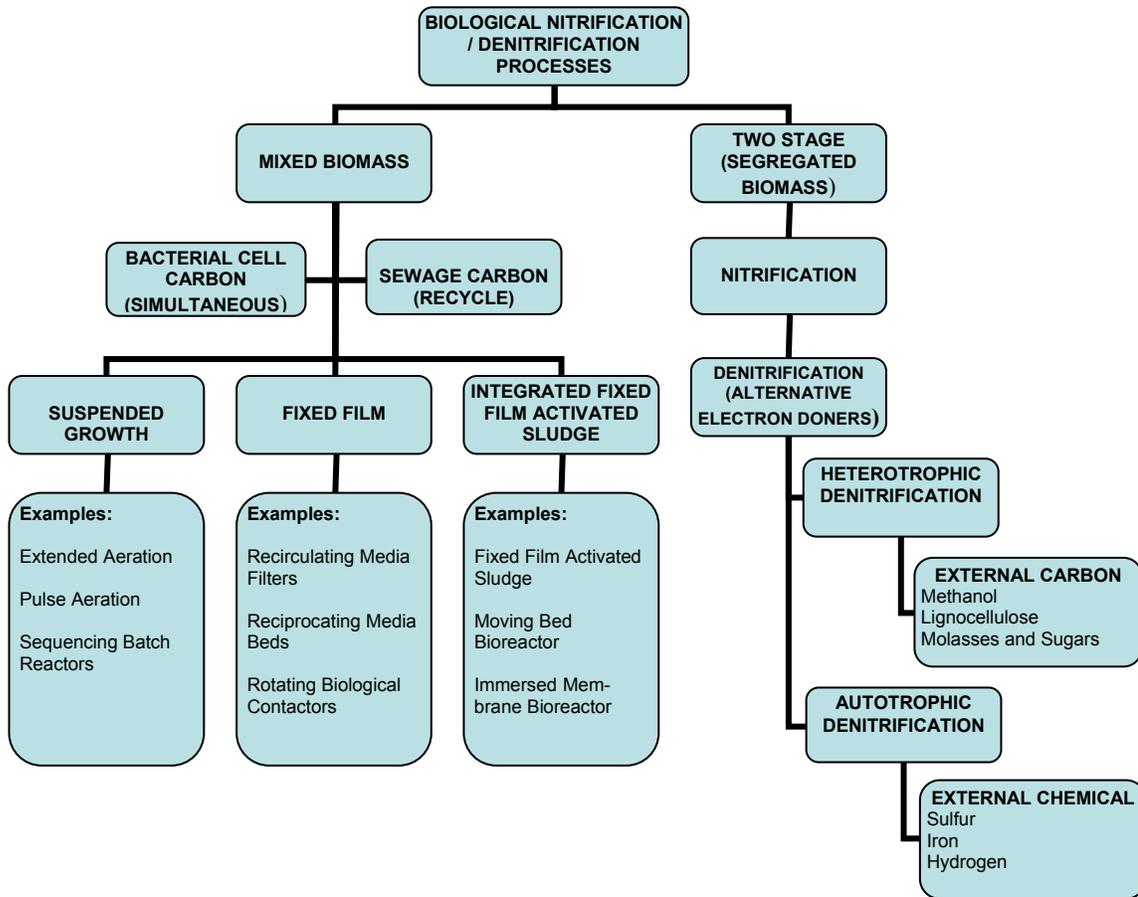


**Figure 4-1: Treatment Options for Reducing Nitrogen in Household Sewage**

o:\44237-00\FR007\Wpdocs\Report\Final

### 4.1 Biological Nitrification / Denitrification Processes

There are many different nitrification/denitrification technologies available. Figure 4-2 lists commonly used groups of systems for each of the biological nitrification/denitrification processes described here.



**Figure 4-2: Onsite Treatment Technology Categories for Biological Nitrification/Denitrification Processes**

To effect biological denitrification in wastewater, treatment works must provide the requisite environmental conditions to sustain the biological mediated processes from organic nitrogen mineralization through nitrification and denitrification. Each of these steps is mediated by different groups of bacteria that require different environments. Many different wastewater treatment trains have been developed to provide the necessary conditions in the necessary sequence to achieve biological nitrification and denitrification, but they all generally fit into three process types: 1) mixed biomass with alternating ox-ic/anoxic environments (simultaneous denitrification), 2) mixed biomass with recycle

o:\44237-001R007\Wpdocs\Report\Final

back to the treatment headworks, and 3) two-stage (separated biomass) using external electron donors (Figures 4-3 through 4-5).

“Biomass” in the context of this review refers to the active microorganisms that provide treatment in the process. In the mixed biomass processes, the active microorganisms are a mixture of autotrophs (nitrifiers) and facultative heterotrophs (organic degraders & denitrifiers) while in the two-stage system, the two groups of microorganisms are segregated in separate reactors.

In each of these processes, treatment is achieved as result of bacteria respiration, which transfers electrons from an electron donor to an electron acceptor that releases energy needed for their growth. The donor compound is oxidized while the acceptor compound is reduced during this transfer. In nitrification and denitrification, electron donors are typically carbonaceous organics, though other donors can be used. The differences between the three process types are the source of the electron donors. In a single stage process using alternating aerobic and anoxic environments, the process is heavily dependent on microbial cell carbon for the electron donor during denitrification. A single stage process with recycle relies heavily on the organic carbon from the fresh incoming wastewater as the electron donor for denitrification. As a result of the recycle loop to acquire organic carbon as an electron donor complete nitrification is not possible in mixed biomass processes. In a two stage process, external electron donors are necessary in the second stage (denitrification) because the organic carbon is removed during the first stage (nitrification) however, nitrification is more complete, which results in more complete denitrification than is possible in mixed biomass systems.

Reactor pH has a significant affect on nitrification. If the reactor is too acidic, nitrification may cease. Therefore, it is important that the pH be controlled during treatment. The optimum pH range is 6.5 to 8.0 (USEPA 1993). The pH is often controlled naturally by alkalinity in the wastewater itself. However, the nitrification reactions consume approximately 7 mg of alkalinity (as  $\text{CaCO}_3$ ) for every mg of ammonium oxidized because of the hydrogen ions released by the oxidation reaction. Thus, there is a risk in low alkalinity waters that the pH could become too acidic and inhibit biochemical nitrification. Typical household wastewater nitrogen (organic and ammonium as N) concentrations range from 40 to as much as 70 mg/L, which would require 300 to up to 500 mg/L of alkalinity respectively for complete nitrification (Oakley 2005). Where alkalinity is too low, it would be necessary to add alkalinity to control the pH if low total nitrogen concentrations in the treated water are required.

#### 4.1.1 Mixed Biomass with Alternating Aerobic/Anoxic Environments (simultaneous)

This nitrification/denitrification process combines the aerobic and anoxic reactors of the mixed biomass recycling system into one reactor (Figure 4-3). Periods of aeration when cBOD oxidation and nitrification occur alternate with periods of no aeration during which the active biomass is allowed to deplete the oxygen to create anoxic conditions for denitrification. The treatment performance is similar to the mixed biomass recycling process.

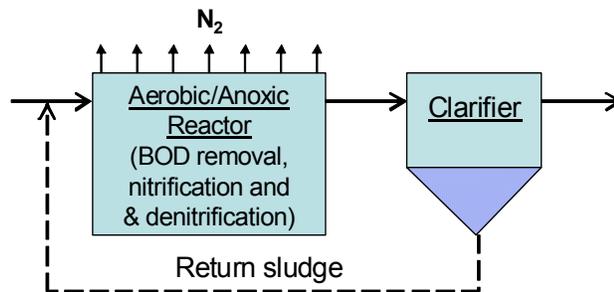


Figure 4-3: Alternating Oxidic / Anoxic Reactor Denitrification

#### 4.1.2 Mixed Biomass Recycling Systems

Mixed biomass systems combine nitrification and denitrification using a mixed active biomass with alternating aerobic and anoxic environments. Typically raw wastewater enters through an anoxic reactor, a septic tank in onsite systems, where the carbonaceous organics (cBOD) are reduced, which releases ammonium and organic nitrogen (Figure 4-4). From this reactor, the wastewater flows to the aerobic reactor where the ammonium and organic nitrogen are nitrified. As the nitrified effluent exits the aerobic reactor, it is split with a small fraction directed to the final discharge while the majority is directed back to the anoxic tank where the nitrate can be reduced to nitrogen gas using the incoming wastewater cBOD as the electron donor. Also, the alkalinity consumed by nitrification is recovered during denitrification thereby reducing the alkalinity requirements. However, total nitrogen removal cannot be achieved with this process because “new” nitrogen is continuously introduced into the flow from fresh raw influent of which a portion is not recycled but discharged from the system. The amount of nitrate that can be removed by onsite systems utilizing this process ranges from approximately 40 to 75 percent.

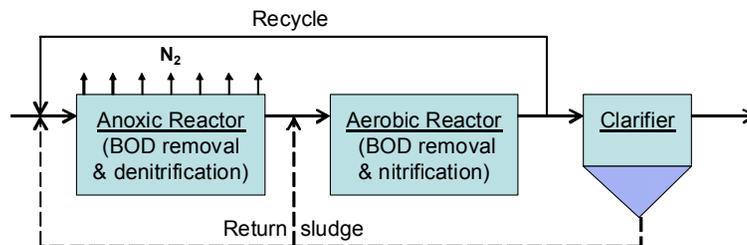
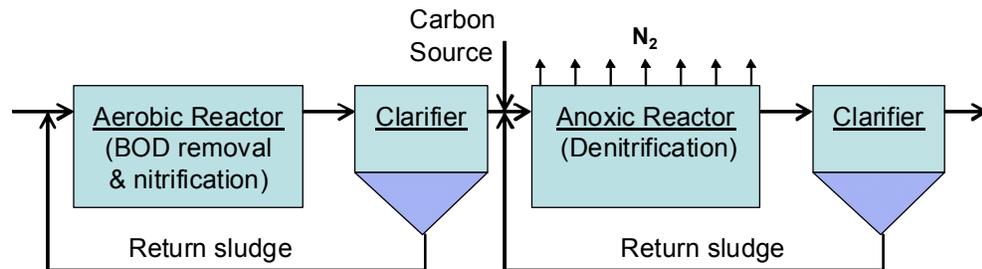


Figure 4-4: Mixed Biomass Recycling Denitrification Process

### 4.1.3 Two-Stage External Electron Donor Denitrification

The two-stage process cultivates two separate bacteria populations; one for nitrification and the other for denitrification (Figure 4-5). This configuration allows nearly complete nitrogen removal because nitrate cannot by-pass denitrification as it can in the mixed biomass options. However, during the nitrification step nearly all the organic carbon in the raw wastewater can be oxidized. As a result, carbon is not available as an electron donor in denitrification thus requiring a donor from an external source to be added directly into the denitrification reactor. A number of organic carbon sources have been used successfully. For larger treatment systems, liquid sources are typically used. The more popular are methanol, ethanol, and acetate. For smaller systems where less operation attention is possible or desired, solid reactive media have been used such as lignocellulose and elemental sulfur.



**Figure 4-5: External Electron Donor Denitrification Process**

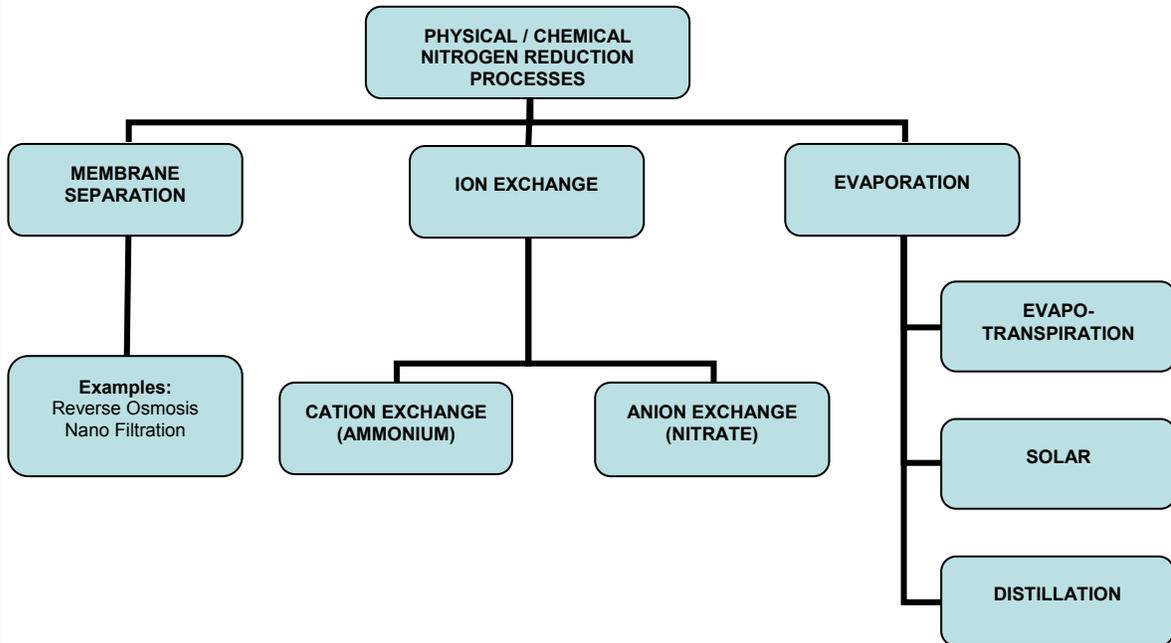
### 4.1.4 Anaerobic Ammonium Oxidation

A fourth biological process called “anammox” has recently been recognized. It is a naturally occurring anaerobic ammonium oxidation pathway in which nitrite and ammonium are converted directly into  $N_2$  gas. It was first recognized in marine environments. The bacteria that are able to use this pathway belong to the bacterial phylum planctomycetes. This is a group of autotrophs, which need no organic carbon. This process only requires partial oxidation of the ammonium to nitrite, which the planctomycetes can then use to reduce the ammonium under anoxic or anaerobic conditions (Gable and Fox 2000; Ahn 2006; Kalyuzhnyi, Gladchenko et al. 2006; Chamchoi, Nitisravut et al. 2008; Wallace and Austin 2008). Because this process has yet to be considered for development of a treatment unit for onsite use, it is not included in this technology review.

## 4.2 Physical / Chemical Nitrogen Removal Processes

Physical/chemical (P/C) processes use non-biochemical approaches to wastewater nitrogen reduction. A fundamental difference from biological processes is that biological nitrification/denitrification converts the biodegradable organic nitrogen to ammonium prior to nitrification; P/C processes typically do not make this conversion, which can

make reduction of total nitrogen to very low concentrations more difficult. Though P/C processes were equally acceptable initially, they have been essentially abandoned in municipal wastewater treatment because they were found to be more problematic (USEPA, 1993). P/C process options that might be appropriate for onsite sewage treatment are shown in Figure 4-6.



**Figure 4-6: Onsite Treatment Technology Categories for Physical/Chemical Processes**

There are several P/C options that are capable of reducing total nitrogen in wastewater. However, many are not practical for household applications including ammonia stripping and breakpoint chlorination. The more suitable P/C options for household use are 1) membrane separation, 2) ion exchange, and 3) evaporation. Membrane separation requires substantial and costly pretreatment, and therefore is most commonly used for drinking water treatment at the household level. Ion exchange also requires pretreatment and commercial regeneration of the exchange resins. Evapotranspiration can be effective in warm climates with year round growing seasons, but require periodic removal and appropriate disposal of the evaporates. Distillation is an emerging option for households, but it is early in its development.

### 4.3 Source Separation

The source of the majority of nitrogen in household wastewater is the toilet, which accounts for 70 to 80 percent of the total daily discharge of nitrogen (Univ. of Wisconsin, 1978; U.S. EPA, 2002; Lowe, Rothe et al., 2006). Nitrogen from food wastes that are discharged through the kitchen sink or dishwasher account for an additional 15 percent. These sources can be segregated from the total household waste flows for separate treatment and handling. For common separation options, see Figure 4-7.

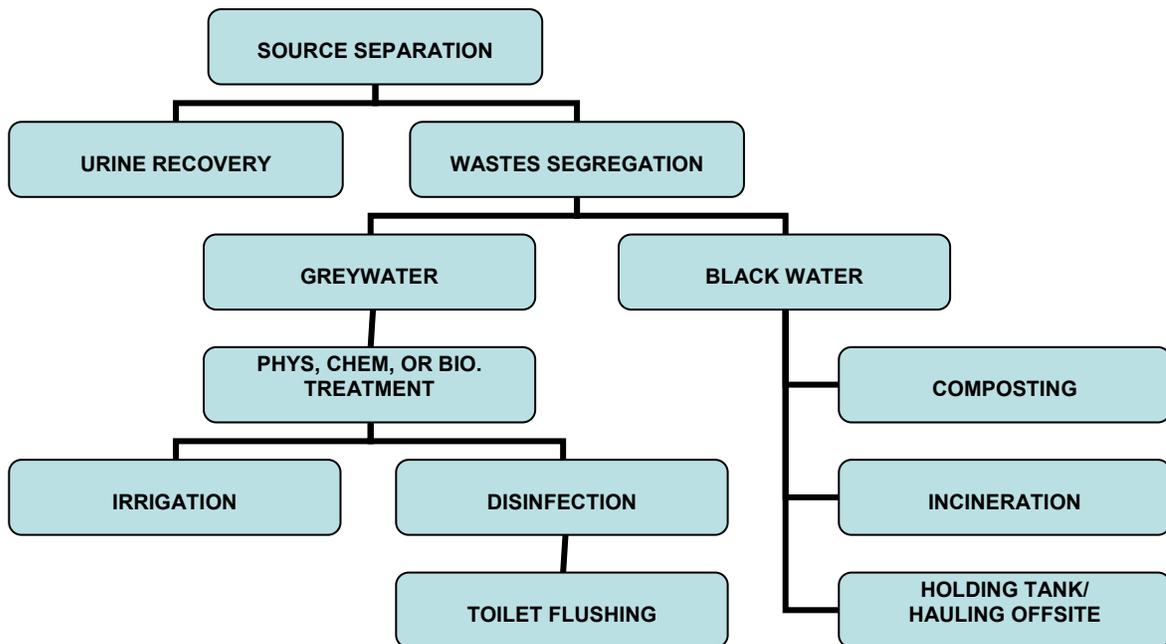
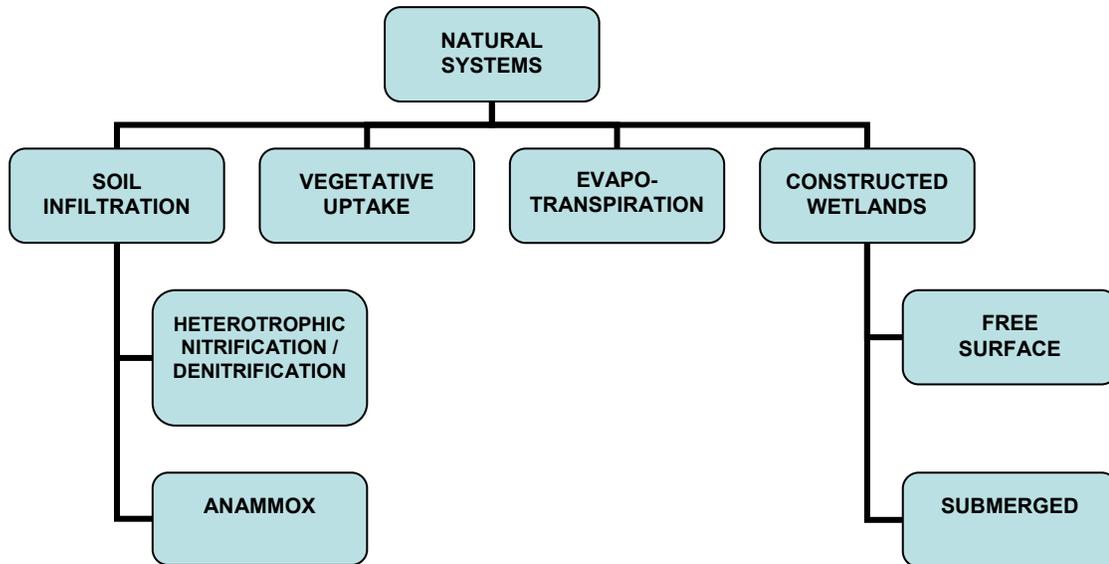


Figure 4-7: Nitrogen Source Separation Categories

### 4.4 Natural Systems

Natural systems are included as a separate classification because they are capable of significant nitrogen reduction. They utilize a combination of physical, chemical and biological processes that occur naturally in the environment. Natural biological processes can mimic both single and two-stage processes depending on the soil conditions (Briggs, Roeder et al., 2007; Otis, 2007). Categories of technologies that are practical for onsite sewage treatment are presented in Figure 4-8.



**Figure 4-8: Categories of Natural Systems for Nitrogen Reduction**

#### 4.5 Passive Nitrogen Removal

Treatment systems can be either “passive” or “active”. Passive systems are generally preferred for onsite wastewater treatment because if well designed, they run largely on their own without the need for frequent inspection or servicing. By design, they have a minimum of moving parts to avoid breakdowns typically using hydraulics of the influent water as the driving force through the system. With limited inputs of external energy, passive systems tend to be designed conservatively large because there are few operational remedial measures that can be taken if undersized. Consequently, capital costs can be more expensive and/or systems require more land area than “active” systems that rely more on external energy inputs. If the treatment process is upset however, passive systems may take longer to recover and are also generally more difficult to upgrade to improve performance. Active systems are easier to upset but also easier to reestablish treatment performance. However, to be effective, regular operation and maintenance of active systems is necessary.

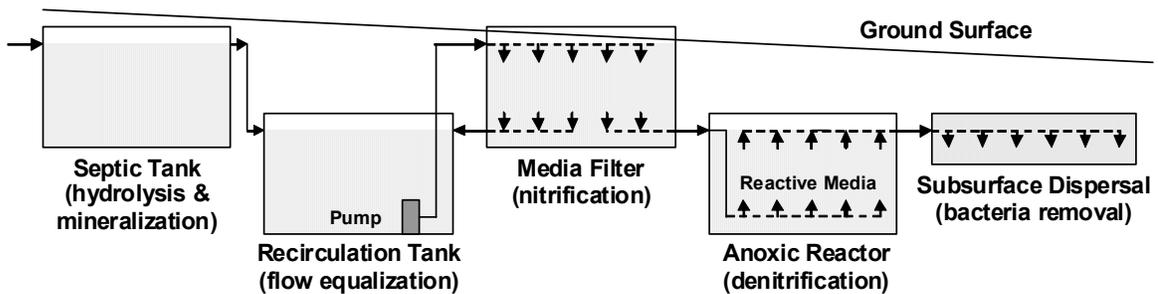
The Florida Department of Health (FDOH) favors passive systems for household and small commercial and cluster systems. To make a clear distinction between passive and active systems, FDOH has defined “passive” strictly as, *“a type of onsite sewage treatment and disposal system that excludes the use of aerator pumps and includes no more than one effluent dosing pump with mechanical and moving parts and uses a reactive*

*media to assist in nitrogen removal.*” Reactive media is defined as media that reacts with wastewater to reduce nitrogen concentrations.

This definition precludes most nitrogen reduction options primarily because of the requirement for reactive media. Only biological two-stage systems would qualify as passive under this definition (Figure 4-4). Cation exchange ( $\text{NH}_4^+$ ), a physical/chemical process is another reactive medium process but to be effective, pre-filtration and treatment is necessary to prevent resin fouling, which may require additional mechanical components beyond one pump and would eliminate it as a passive system. In any event, the added cost of the pretreatment would likely make ion exchange impractical for household applications. Most mixed biomass systems would be “passive” except for the requirement for reactive media, but these systems have less ability to meet very low total nitrogen concentrations. Where the total nitrogen requirements are above 10 mg N/L, these systems could be acceptable options. Mixed biomass systems also have the advantage that they recycle the alkalinity, which may be important in areas with low alkalinity in drinking water. The FDOH definition of “passive” is followed in describing and comparing the different nitrogen reduction processes and technologies in this review.

A two-stage denitrification system for household use that meets the FDOH “passive” definition probably would consist of a septic tank, recirculating media filter, anoxic denitrification reactor followed by soil infiltration. An example of such a system is shown in Figure 4-9. In the septic tank, proteins are hydrolyzed releasing the organic nitrogen, which is oxidized to ammonium. Any nitrate or nitrite present in the influent is denitrified because of the anoxic environment and the availability of ample organic carbon. The media filter is an unsaturated aerobic media, which removes most of the BOD, nitrifies the ammonium and removes up to 50 percent of the total nitrogen. Where low total nitrogen concentrations are necessary the filtrate must be returned to the recirculation tank to be recycled onto the media filter since nitrification may not be complete after a single pass through the filter. This requires a pump and a passive filtrate flow splitter that can divert the flow for recycling or discharge to the next treatment stage. The advantage of using the pump here is three fold. First, it can dose the media filter based on time (rather than demand) and under pressure, which achieves uniform distribution over the filter surface both spatially and temporally significantly enhancing treatment performance. Second, it provides flow control (equalization) through the remainder of the system, which also enhances system performance. Third, it can be used to raise the hydraulic grade line through the remainder of the system so that flow through the system occurs by gravity, which eliminates the need for additional pumps. The nitrified filtrate flows to the anoxic reactor, which is filled with saturated reactive media that provides the electron donors for denitrification to occur. After this reactor, the treated wastewater is discharged for subsurface dispersal where bacteria in the water are removed by processes in the soil as the water percolates to the groundwater.

Availability of alkalinity is an important consideration in any nitrification/denitrification treatment process. It is an important buffering agent that is necessary to maintain pH concentrations in an acceptable range for nitrifying organisms to thrive. During nitrification, hydrogen ions are created and if not controlled by a buffering agent, will increase the acidity of the water to the point that nitrification ceases. Nitrification consumes approximately 7.14 grams of alkalinity as  $\text{CaCO}_3$  per gram ammonia N nitrified. Typical individual home domestic wastewater averages approximately 60 mg-N/L of total nitrogen, most of which is organic and ammonium (Lowe, Rothe et al. 2006). Alkalinity over 400 mg/L, as  $\text{CaCO}_3$ , would be necessary to nitrify all of the TN. The wastewater itself can add 60-120 mg/L alkalinity as  $\text{CaCO}_3$  (Crites and Tchobanoglous, 1998) but there may be many areas where sufficient alkalinity is unavailable for nitrification.



**Figure 4-9: Passive Two-Stage Denitrification System**

Water conservation trends will limit alkalinity availability further. Since the alkalinity is not recovered in two-stage systems as it is in mixed biomass systems, augmentation of alkalinity to the media filter using crushed limestone or oyster shells may be necessary and must be addressed during design. A benefit of using a recirculating media filter for nitrification is that the recycled filtrate will undergo as much as 50 percent denitrification in the recirculation tank using the influent organic carbon as an electron donor, which will restore some of the alkalinity consumed during nitrification.

Denitrification using reactive media under saturated conditions has not been studied extensively particularly in passive applications. The reactive media is added to the anoxic reactor as a solid. Dissolution of the reactive material is necessary to release the electron donors needed in denitrification. Ideally, the rate of media dissolution should equal the rate of denitrification. If the dissolution is too rapid, media longevity and the effluent quality will be reduced by excess dissolution product which would require more frequent media replacement. If the rate of dissolution is too slow, denitrification would be incomplete. Balancing these rates between dissolution and consumption is problematic under

passive conditions and with intermittent flows typical of household OSTDS. Over time with continuous operation, flow channeling in the media can occur allowing short circuiting through the media, which decreases retention time in the reactor, allows less contact of the wastewater with the media resulting in decline of performance. Careful selection of the media and attention to design of the reactor and selection of media are critical to success.

One cautionary note concerning any denitrification system when TN effluent concentrations below 5 mg-N/L are required is how to deal with refractory organic nitrogen in the effluent. Refractory organic nitrogen is dissolved organic nitrogen (DON) that is resistant to decay. As much as 2-3 mg-N/L can be found in denitrified effluent, which can result in exceedences of effluent limits (Mulholland, Love et al. 2007). Since it is not readily bioavailable and easily adsorbed by the soil, there is good cause not to include DON in the TN limit. Currently, the Water Environment Research Foundation is studying this issue because of challenges to its inclusion by municipal treatment plants (WERF 2008).

## **Section 5.0**

### **Review of Onsite Nitrogen Reducing Technologies and Practices**

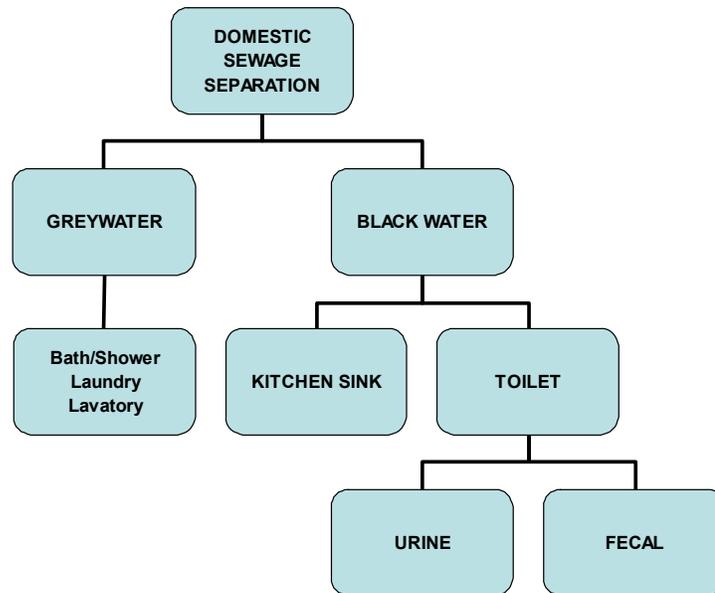
---

The following is a review of what are considered technically and economically feasible nitrogen reduction technologies and practices suitable for single households and small commercial establishments. In this review, the technologies and practices are presented in an order that they would appear in an onsite wastewater nitrogen reduction system.

#### **5.1 Source Separation**

Traditionally onsite domestic sewage treatment has focused on systems that receive the entire combined stream of household waste discharges. Future trends are likely to place increasing emphasis on concepts of water sustainability and resource recovery, entailing water infrastructure that maintains segregation of individual wastestreams for treatment, recovery and reuse. Wastewater segregation of greywater for reuse has been practiced predominately in water short areas for some time. More recently, recovery of urine for its nutrient content through the use of urine separating toilets is gaining attention as a sustainable solution to reported worldwide shortages of nutrients, particularly phosphorus. Since the source of 70 to 80 percent of all the nitrogen discharged from households are from toilets, the recovery of urine could reduce total nitrogen discharges from domestic wastewater by 50 to 75 percent.

Domestic sewage can be subdivided into two to four separate wastestreams based on options for segregation that are likely to provide most appropriate treatment and reuse combinations. The domestic wastestreams typically considered for separation are illustrated in Figure 5-1.



**Figure 5-1: Domestic Wastestream Components**

The quantity and constituent mass of these wastestreams are summarized from published data for typical U.S. households in Table 5.1 (Mayer, DeOreo et al. 1999; USEPA 2002; Tchobanoglous, Burton et al. 2003). Four waste source groupings are shown based on quality characteristics of the wastestreams representing typical U.S. conditions (Crites and Tchobanoglous 1998; Lens and Lettinga 2001; Davison, Pont et al. 2006; Makropoulos, Natsis et al. 2008; Benetto, Nguyen et al. 2009; Mah, Bong et al. 2009).

**Table 5.1**  
**Per Capita Volume and Constituent Loading in U.S. Domestic Sewage**

Source Designation	Water Source	Daily Volume (gpcd)	Gram / person-day			
			C-BOD <sub>5</sub>	TSS	Total N (as N)	Total P (as P)
<b>A</b>	Non-kitchen sinks, clothes washer, shower, bathtubs	32	11.4	5.2	0.8	0.2
<b>B</b>	Kitchen sinks, dishwasher, garbage grinder	10.3	35.1	38.5	1.7	0.3
<b>C</b>	Toilet: non-urine	17.5	12.5	80	1.1	0.4
<b>D</b>	Toilet: urine	0.6	4.2	0.1	10.9	1.2
<b>Sum</b>		60.4	63.2	124	14.5	2.0

(Crites and Tchobanoglous, 1998; Lens and Lettinga, 2001; Davison, Pont et al., 2006; Lowe, Rothe et al., 2006; Makropoulos, Natsis et al., 2008; Benetto, Nguyen et al., 2009; Mah, Bong et al., 2009)

Wastestream segregation increases the options available for nutrient reduction by separating wastestreams with differing constituents and characteristics to facilitate separate storage, treatment and reuse of each segregated stream. Storage and onsite or offsite recovery and reuse of nitrogen is possible for wastestreams with small volumes and high nitrogen concentrations. Separation of wastestream components with relatively low pollutant concentrations enables onsite reuse with limited treatment, which reduces the mass and volume of the remaining, more concentrated wastestreams that require smaller sized treatment units. Thus, waste segregation can reduce nitrogen loading to the environment through recovery and beneficial use of nutrients in the wastestreams and by decreased nitrogen loadings to onsite soil treatment and dispersal units.

Components of domestic wastestreams are shown in Table 5.2 for a four person household in the U.S. based on the Table 5.1 data. The daily volume and constituent concentrations for the entire wastestream (A+B+C+D) are subdivided according to degree of source separation, resulting in functional wastestream component designations that vary significantly in daily volume and constituent concentration. The Table 5.2 designations can be applied to analysis and selection of nitrogen reduction technologies that are advantageous for different source separation options.

**Table 5.2**  
**Volume and Constituent Concentrations of Domestic**  
**Sewage Wastestreams for a Four Person Household in the U.S.**

Description	Components	Daily Volume (gallons)	Constituent concentration (mg/L)				% of Total Constituent Mass			
			C-BOD <sub>5</sub>	TSS	Total N (as N)	Total P (as P)	C-BOD <sub>5</sub>	TSS	Total N (as N)	Total P (as P)
Domestic Sewage	A+B+C+D	241	277	542	63	8.8	100	100	100	100
Greywater	A	128	94	43	6	1.2	18	4	5	8
Black Water	B+C+D	113	483	1,105	128	17	82	96	95	93
Domestic Sewage w/o Urine	A+B+C	239	261	547	16	3.5	93	100	25	40
Black Water w/o Urine	B+C	111	453	1,128	27	6.2	75	96	19	33
Urine	D	2.4	1,838	35	4,808	528	7	0.065	75	60

(Mayer, DeOreo et al. 1999; Günther 2000; Lens and Lettinga 2001; Lens, Zeeman et al. 2001; USEPA 2002; Tchobanoglous, Burton et al. 2003; Memon 2005; Lowe, Rothe et al. 2006; Magid, Eilersen et al. 2006; Makropoulos, Natsis et al. 2008; Benetto, Nguyen et al. 2009)

Typically, domestic sewage is separated into greywater (A) and black water (B+C+D) (Table 5.2). Here, the kitchen wastestream should not be included in the greywater designation because of its association with production and consumption of food and the BOD, TSS and pathogens that may be found in kitchen waste. Greywater comprises over half of the water volume while contributing relatively small fractions of total pollutant mass. With lower constituent concentrations, greywater requires less intensive treatment than black water to meet a given level of water quality. Greywater may be rendered suitable for onsite reuse (irrigation or indoor toilet flushing) with relatively simple aerobic biological treatment.

Urine (D) accounts for very small volumes but high fractions of nitrogen and phosphorus. Separation and recovery of urine as a concentrated nutrient source provides benefits for both onsite nitrogen reduction and beneficial nutrient recovery. Urine separation can be accomplished with or without the separation of greywater and black water, resulting in typical domestic wastestreams minus urine (A+B+C) or a black water wastestream minus urine (B+C).

Black water (B+C+D) contains a majority of the constituent mass but less than half of the volume of the whole domestic wastestream (A+B+C+D), resulting in higher constituent

concentrations (Table 5.2). Treatment of black water would require generally similar treatment as combined domestic wastestreams, although the necessary treatment system capacity required to achieve a similar level of effluent quality could be smaller. Removal of urine from domestic wastestreams (A+B+C) or from black water (B+C) has relatively minor effect on total daily volume and BOD and TSS concentrations (Table 5.2). The treatment plant required for removal of BOD and TSS would not be greatly affected, but the required nitrogen reduction treatment capacity would be reduced.

The primary options for household source separation are recovery of urine and segregation of greywater for reuse. Urine separation removes a majority of the nitrogen and a small fraction of the volume of total household wastestream (Larsen, Peters et al. 2001). The remaining household wastestream has a similar daily volume but only 20 to 30 percent of the total nitrogen. Recovery of the nitrogen and phosphorus content of urine can provide beneficial reuse of these macronutrients. In many cases the life cycle energy expenditure of converting urine nutrients into solids for application as agricultural fertilizer may be lower than the cost of industrial nutrient production and biological nutrient reduction of wastewater (Maurer et al., 2003). Where located in a centralized service area, the costs of centralized wastewater treatment plants can be reduced (Wilsenach and Loosdrecht 2006). For distributed infrastructure (i.e. individual residences and cluster systems), urine separation results in a much reduced nitrogen concentration in the effluent stream (Table 5.2). Beneficial use of urine could also provide a future funding mechanism for onsite treatment infrastructure.

## 5.1.1 Urine Separation and Recovery

### 5.1.1.1 Urine Separation

Urine separation systems include urine separating toilets and waterless urinals. Urine separation technologies include toilets with separate collection bowls (Figure 5-2) and effluent lines for urine and feces, and waterfree urinals with a single effluent line. The urine from the toilets and urinals is conveyed through a small pipe to a storage tank, which is periodically emptied. The feces are either directed into the building sewer or into a composting bin.



**Figure 5-2: Two Swedish Urine Separating Toilets (EcoSan and Novaquatis)**

Several studies have described monitoring urine collection systems under actual usage. Vinneras and Jonsson (Vinnerås and Jönsson 2002a) describe the performance of a urine collection system for a urine separating toilet. Annually, 125 gallons of urine were collected per person with a coefficient of variation of 11 percent. When combined with feces collection, 60 percent of the nitrogen was recovered from the wastewater. In Switzerland, urine separating toilets and waterless urinals were tested in four households (Rossi, Lienert et al. 2009). Water recovery was 0.036 gal/flush in households and 0.059 gal/use with waterfree urinals. Mean urine collection rates in households were 1.68 gpd on weekdays and 2.44 gpd on weekends. Urine recovery in households was maximally 70 to 75 percent of the physiologically expected quantity.

A modeling framework was developed to predict pharmaceutical concentrations in human urine and to support risk assessments of urine recovery and beneficial use (Winker, Tettenborn et al. 2008b). The model showed that model predictions are adequate when the collection system is used by a sufficiently large number of people. The concentrations of 28 pharmaceuticals in the urine were compared to the same pharmaceuticals in municipal wastewater. This comparison showed that the majority of pharmaceuticals are excreted in urine.

The overall urine separation system must include provision for management of material removed from the storage tank. The collected urine may be transported offsite as a liquid by truck or pipeline (Justyna Czemieli Berndtsson 2006). The collected urine can be used as a liquid fertilizer or treated in a centralized facility (Borsuk, Maurer et al. 2008). The urine can be used on the owner's own property if there is sufficient nutrient demand. If used onsite, the benefits of separating the urine from other household sewage may be limited. The proximity of agricultural nutrient demand to urine generation would influence the most advantageous approach.

Adoption of urine separating toilets requires broad public acceptance if it is to have significant impact (Lienert and Larsen 2006). Further development of urine separating toilet technology may be required to increase public acceptance and adoption (Borsuk, Maurer et al. 2008; Rossi, Lienert et al. 2009).

For a single family residence, urine separation installation would require purchase of system components including an urine separating toilet, water-free urinal or both, a storage tank, plumbing and appurtenances. The components are commercially available but currently urine separating systems are not in widespread use in the U.S. Providing for removal of material from the storage tank and its management must also be considered. Field evaluations have concluded that current urine separation technology is in need of improvement. Realizing the nutrient recovery benefits of urine separation would require treatment onsite or offsite treatment with technologies that are generally still under development. Centralized offsite treatment and recovery would require a system infrastructure and management entity for collection and treatment.

#### **5.1.1.2 Urine Treatment**

A number of urine treatment processes could be used for removal and recovery of nitrogen and other constituents, including evaporation, freeze-thaw, nanofiltration, reverse osmosis, precipitation, ion exchange, ammonia stripping, and electro dialysis/ozonation, and electrochemical treatment (Lind, Ban et al. 2001; Maurer, Pronk et al. 2006; Pronk, Palmquist et al. 2006; Ikematsu, Kaneda et al. 2007; Pronk, Zuleeg et al. 2007). Research presently being conducted suggests that practical applications of these processes are limited.

Nitrogen in human urine is predominantly urea. Urine storage leads to hydrolysis of urea, which leads to the release of ammonia, increase in pH, and the onset of precipitation (Udert, Larsen et al. 2003a; Liu, Zhao et al. 2008c). Complete urea hydrolysis may require two days or longer in undiluted urine (Wilsenach and Loosdrecht 2006), while some studies indicate longer times (Hotta and Funamizu 2008). Time to achieve complete hydrolysis is decreased at higher temperature and by mixing fresh urine with previously hydrolyzed urine (Liu, Zhao et al. 2008b).

#### **5.1.1.3 Direct Nitrification**

A packed column treating urine achieved 95 percent nitrification when pH was artificially maintained at 8, whereas only 50 percent of ammonia was nitrified without pH adjustment (Feng, Wu et al. 2008).

#### 5.1.1.4 Precipitation

In undiluted urine, nitrogen precipitates as magnesium ammonium phosphate  $[(\text{NH}_4)\text{MgPO}_4 \cdot 6\text{H}_2\text{O}]$ , a mineral called struvite, which has direct use as plant fertilizer (Ronteltap, Maurer et al. 2007a; Yetilmezsoy and Sapci-Zengin 2009). Hydroxyapatite  $[\text{Ca}_{10}(\text{PO}_4)_6(\text{OH})_2]$  and other non-nitrogen containing precipitates are also formed (Udert, Larsen et al. 2003b). The maximum precipitation potential of undiluted urine may be reached in 4 hours or less (Udert, Larsen et al. 2003a).

Factors that affect the struvite precipitation process are reactor pH, hydraulic retention time, mixing, the degree of supersaturation, and molar ratios of magnesium to phosphorus, nitrogen to phosphorus, and calcium to magnesium (Stratful, Scrimshaw et al. 2001; Pastor, Mangin et al. 2008; Saidou, Korchef et al. 2009). In addition, the surface roughness of materials in contact with the liquid may influence struvite precipitation (Doyle and Parsons 2002a). A high fractional removal of phosphorus can be achieved, which is accompanied by nitrogen removal; magnesium supplementation may increase removal efficiencies in some cases (Jaffer, Clark et al. 2002). Batch struvite crystallization experiments were conducted on human urine and analog human urine, and crystallization occurred within 30 to 50 minutes (Lind, Ban et al. 2000). Liu et al. (Liu, Zhao et al. 2008c) reported 5 to 96 percent recovery efficiency for ammonia nitrogen and 85 to 98 percent recovery efficiency for phosphate in batch precipitation experiments with human urine. The higher ammonia removal efficiencies occurred when the urine was supplemented with magnesium and phosphate salts, and a maximum ammonia reduction from 6,266 mg/L to 269 mg/L was achieved (Liu, Zhao et al. 2008c).

Various reactor configurations have been proposed with the goal of optimizing efficiency of nutrient capture, minimizing contact time, and minimizing energy input. Design features that affect the precipitation process include pH, temperature, molar ratios of Mg/N/P/Mg, and mixing energy (Liu, Zhao et al. 2008a). Struvite precipitation can be conducted in fluidized bed reactors, pellet reactors, and complete mix reactors (Doyle and Parsons 2002; Wilsenach, Schuurbiens et al. 2007; Pastor, Mangin et al. 2008). Liu et al. (Liu, Zhao et al. 2008a) reported on an internal recycle seeding reactor (IRSR) to enhance performance at low nutrient concentrations. The process employs recirculation of struvite crystals from a sedimentation zone to a separate crystallization zone.

The levels of urine microconstituents that precipitate in struvite are an important consideration for fertilizer use. A recent study reported that hormones and non-ionic, acidic and basic pharmaceuticals generally remain in solution with struvite precipitation from urine and that heavy metals levels in struvite were several orders of magnitude less than commercial fertilizers (Ronteltap, Maurer et al. 2007b). Pathogen levels in source separated urine are of concern for public health. Transmissible pathogens originate mainly

from cross-contamination by feces. Twenty two to 37 percent of urine storage tank samples were found to be contaminated using fecal sterols in lieu of indicator bacteria (Schönning, Leeming et al. 2002). Urine and urea can reduce survival of indicators organisms (Schönning, Leeming et al. 2002; Vinnerås and Jönsson 2002a).

The mass ratio of nitrogen to phosphorus in domestic sewage and urine ranges from 4 to 11 (Maurer, Pronk et al. 2006). However struvite has a 1:1 molar ratio of nitrogen to phosphorus and as a result only partial nitrogen removal is achieved by precipitation of struvite from unamended urine. Additional treatment options to increase nitrogen reduction include stoichiometric addition of phosphate to the influent of the struvite precipitation reactor, ion exchange, ammonia stripping, and reverse osmosis. Removal of ammonium ion with zeolites can be integrated with struvite precipitation in the same reactor or alternatively, ion exchange can be applied as a post treatment process following the precipitation reactor.

The efficiency of nitrogen removal from human urine by struvite precipitation was increased from 5 to 95 percent by addition of magnesium and phosphate salts (Liu, Zhao et al. 2008c). This approach has the disadvantage of requiring additional phosphate and magnesium. Ammonium ion removal can be accomplished with ion adsorptive materials with high ammonium affinity including clinoptilolite, a naturally occurring zeolite (Lind, Ban et al. 2000; Lind, Ban et al. 2001; Jorgensen and Weatherley 2003; Smith 2008; Smith, Otis et al. 2008); the mineral wollastonite (Lind, Ban et al. 2001), and polymeric ion exchange resins (Jorgensen and Weatherley 2003). Ion exchange can be applied as post treatment following struvite precipitation or as an integrated precipitation/ion exchange process. A combined process consisting of magnesium enhanced struvite crystallization and ion exchange adsorption was evaluated in laboratory experiments. Up to 80 percent of the nitrogen content of a synthetic human urine was removed (Lind et al., 2001). In theory, post treatment ion exchange could achieve very high nitrogen reduction efficiencies and the ion exchange material regenerated by a biological process.

### **5.1.2 Greywater Collection and Reuse**

Since greywater contains only a small portion of the nitrogen in household sewage, the total impact of greywater separation on nitrogen reduction is limited. However, it does reduce the amount of organic carbon available to potential electron donors during denitrification of the black water.

A universally accepted definition of greywater does not exist. Excluding kitchen waste from greywater is consistent with Florida requirements. Separate collection of effluent from all kitchen and toilet sources is typical. Some greywater definitions include kitchen waste, which would increase pollutant concentrations and lead to greater nuisance po-

tential and greater requirement for treatment. Kitchen wastes have been further subdivided, where all wastes except garbage grinder wastes are included in greywater. Including kitchen wastes in greywater would necessitate more intensive treatment processes which would duplicate black water treatment processes and reduce the advantage of separating greywater. In reviewing any reports on system performance and feasibility, the composition of the greywater stream should be determined.

Rational for separate greywater collection is to reuse or dispose of the less polluted greywater onsite, through irrigation, application on land or indoor non-potable reuse. Modeling predicted that a 40 percent savings in potable water demand could result with greywater recycling in an urbanized area, although no attention was given to nitrogen reduction (Mah, Bong et al. 2009). Greywater recycling in a multi-story residential building for toilet flushing reduced potable water use by 29 to 35 percent and had a payback period of less than 8 years. Nitrogen reduction was not reported (Ghisi and Ferreira 2007). A stochastic model of urine generation over multiple contributing individuals was used to predict strategies for reducing ammonia loadings at centralized treatment plants (Rauch, Brockmann et al. 2003).

Guidelines for the safe use of greywater were presented by the World Health Organization (WHO 2006). The composition of greywater was found to depend on the source. Household and personal care product usage was reviewed as it pertained to the composition of greywater. Over 900 different synthetic organic compounds were identified as possible greywater constituents (Eriksson, Auffarth et al. 2002). Prevalence of pathogens in the population and fecal load in greywater formed the basis of a screening level quantitative microbial risk assessment (QMRA), which was applied to simulated greywater exposure scenarios for direct contact, irrigation of sport fields and groundwater recharge (Ottoson and Stenström 2003). Rotavirus risks were unacceptably high in all exposure scenarios, which provided an argument for additional greywater treatment. The mass flows of selected hazardous substances in greywater and black water were monitored from ordinary Swedish households (Palmquist and Hanæus 2005). Over 90 percent of the measured inorganic elements were found in both greywater and black water while 46 out of 81 organic substances were detected in greywater. Generally, the specific sources of household wastes that contributed the individual chemicals could not be distinguished.

### 5.1.2.1 Greywater Treatment

Greywater treatment has been examined by several investigators with a variety of treatment technologies applied in many different schemes for overall water recycling (Eriksson, Andersen et al. 2008; Ramona, Green et al. 2004; Benetto, Nguyen et al. 2009; Gual, Moià et al. 2008; Günther 2000; Kim, Song et al. 2009; Misra and Sivongxay 2009;

Nolde 1999; Pidou, Avery et al. 2008; Jefferson, Burgess et al. 2001; Widiastuti, Wu et al. 2008; Winward, Avery et al. 2008a; Elmitwalli and Otterpohl 2007; Friedler, Kovalio et al. 2005; Winward, Avery et al. 2008b; Schäfer, Nghiem et al. 2006).

Varying local and state regulatory codes may discourage adoption of greywater systems in the U.S. According to one website, packaged greywater storage and recycling systems are difficult to find in the U.S. ([www.greywater-systems.com](http://www.greywater-systems.com)). Some systems include simple outdoor holding tanks, under sink systems, and systems with filtration and disinfection. California guidance on a standard greywater irrigation system design includes a surge tank, filter, pump, and irrigation system (CSWRCB 1995). Guidance can be found on installing these systems ([www.greywater.net](http://www.greywater.net)) but there appears to be limited documentation on measured system performance. To be effective for outdoor irrigation reuse over many years of operation, application of greywater would likely require very simple systems with low operation and maintenance needs. One source recommends mulch type planting beds (<http://oasisdesign.net/greywater>).

Storage of greywater is an important element of all greywater recycling systems. Greywater quality has been found to be affected by storage; sedimentation, aerobic microbial oxidation, anaerobic microbial processes in settled solids, and reaeration (Dixon, Butler et al. 2000). Storing greywater for a 24 hr period led to improved quality due to the reduction of suspended solids, but dissolved oxygen is depleted after 48 hrs which can result in odor problems. These results suggest that practical greywater systems could benefit from low intensity aerobic treatment, such as mild or intermittent aeration. In Australia, greywater collection systems are required to use disinfection (UV or chlorine) if greywater is held for longer than 24 hrs. This would serve to oxidize BOD in the influent greywater, and oxidize organics and odors that are released from underlying settled solids.

The preferred practice for separate disposal of residential greywater are mulch filled basins supplied by drain or a branched drain network, with pipes a few inches above the mulch or in appropriately sized underground chambers if subsurface discharge is required (*Builder's Grey Water Guide*). The preferred practice for reuse is to plumb the system in such a way that there is some certainty where the water is being applied so that adjustments can be made as necessary. Simple designs would likely be needed and be most effective.

### 5.1.3 Black Water Separation and Treatment

Different techniques were examined for separation of fecal material from flush water. The Aquatron system uses surface tension, gravitation and a whirlpool effect to produce a solids stream that contains 70 to 80 percent of the incoming dry matter thereby reco-

vering the majority of nitrogen (Vinnerås and Jönsson 2002a). Black water treatment was investigated using anaerobic biotreatment followed by filtration using commercial nano-filtration and reverse osmosis membranes (van Voorthuizen, Zwijnenburg et al. 2005). Ortho P removals were 74 to 99 percent while ammonia removals were 21 to 94 percent. Onsite anaerobic treatment of black water (Luostarinen and Rintala 2005) is similar to treatment of whole domestic sewage, albeit with higher constituent concentrations. Three combinations of biological treatment and membrane filtration were compared for separate black water treatment: a UASB followed by membrane filtration, anaerobic MBR, and aerobic MBR (van Voorthuizen, Zwijnenburg et al. 2008). All three systems exhibited high nutrient conservation and effluent with low TSS and high soluble COD in the effluent.

## 5.2 Primary Treatment (Septic Tank)

A septic tank is commonly used as the first treatment step in an OSTDS. Its principal function is to remove, store, and digest settleable and floatable suspended solids in the raw wastewater. These solids collect as sludge and scum within the tank where the organic carbon is degraded via hydrolysis, acidogenesis, acetogenesis and methanogenesis. During hydrolysis, the protein molecules are broken apart to release the organic nitrogen, much of which is converted to ammonium. Nitrate in the influent is quickly denitrified by the heterotrophic denitrifiers. Consequently, the form of nitrogen in domestic septic tank effluent is approximately 70 percent ammonium and 30 percent organic nitrogen (Wisconsin 1978; Lowe, Rothe et al. 2006). Nitrate is typically negligible. About 15 percent of the influent nitrogen is retained in the tank within the sludge and scum (Otis 2007).

In denitrification systems, the septic tank is often used as a carbon source for heterotrophic denitrification of nitrified wastewater returned from downstream nitrification processes. The nitrified wastewater is returned to the septic tank inlet to mix with the influent and septage in the tank. Up to 70 percent reduction of the total nitrogen in the wastewater can be achieved with recycle (USEPA 2002). The increased throughput of the septic tank due to recycling will increase the rate of flow through the septic tank and reduce the residence time in the tank. This must be taken into account in sizing the tank during design.

## 5.3 Biological Nitrification / Denitrification Processes

Two classes of biological nitrification/denitrification processes that are most practical and commonly used for onsite sewage treatment are mixed biomass (single stage) and segregated biomass (two stage). The principal difference between the two is the source of the electron donor used by the denitrifying microorganisms. The mixed biomass systems

use organic carbon that is available in the wastewater being treated; either microbial cell carbon and/or wastewater carbon. Segregated biomass systems require external sources of organic carbon or chemical donors.

Management of wastewater carbon is critical to successful denitrification. This is difficult in mixed biomass systems because nitrification must be achieved first. Since nitrification is an aerobic process, much of the organic carbon is oxidized during nitrification, which can leave an insufficient amount for subsequent denitrification under anoxic conditions. This is particularly true in OSTDS where small and intermittent sewage discharges into the treatment system can easily result in extended periods of aeration during low or no flow periods with the result that the organic carbon is oxidized before the denitrification step. Consequently, without careful carbon management, OSTDS that use mixed biomass processes are less likely to achieve low total nitrogen effluent concentrations, particularly those using processes that rely on microbial cell carbon as the electron donor in denitrification. Table 5.3 summarizes total nitrogen removal results from OSTDS using mixed biomass and segregated biomass, which shows the differences in treatment capability due to the source of the electron donor. System complexity is also impacted by the unit operation chosen for nitrification/denitrification (Figure 5-3).

**Table 5.3**  
**Biological Denitrification Processes and**  
**Typical Nitrogen Reduction Limits of OSTDS**

Process	Mixed Biomass (Simultaneous)	Mixed Biomass (with Recycle)	Segregated Biomass (Two Stage)
<b>Electron Donor</b>	Organic carbon from bacterial cells	Organic carbon from influent wastewater	External electron donor (Organic carbon; Ligno-cellulose; Sulfur; Iron, Other)
<b>Typical N Reductions</b>	40 to 65%	45 to 75%	70 – 96%
<b>Typical Technologies</b>	<ul style="list-style-type: none"> <li>• Extended aeration<sup>1</sup></li> <li>• Pulse aeration<sup>2</sup></li> <li>• Recirculating media filters<sup>3</sup></li> <li>• Sequencing batch reactors<sup>4</sup></li> <li>• Reciprocating media beds<sup>5</sup></li> <li>• Membrane bioreactor<sup>6</sup></li> </ul>	<ul style="list-style-type: none"> <li>• Extended aeration with recycle back to septic tank</li> <li>• Recirculating media beds with recycle back to septic tank<sup>7</sup></li> <li>• Moving bed bioreactor</li> </ul>	<ul style="list-style-type: none"> <li>• Heterotrophic suspended growth<sup>8</sup></li> <li>• Heterotrophic packed bed fixed film</li> <li>• Autotrophic packed bed fixed film<sup>9</sup></li> </ul>

<sup>1</sup> Leverenz, et al., (2002); USEPA (2002)

<sup>2</sup> California State Water Resources Control Board (2002)

<sup>3</sup> USEPA (2002)

<sup>4</sup> Ayres Associates (1998)

<sup>5</sup> Behrends, et al. (2007)

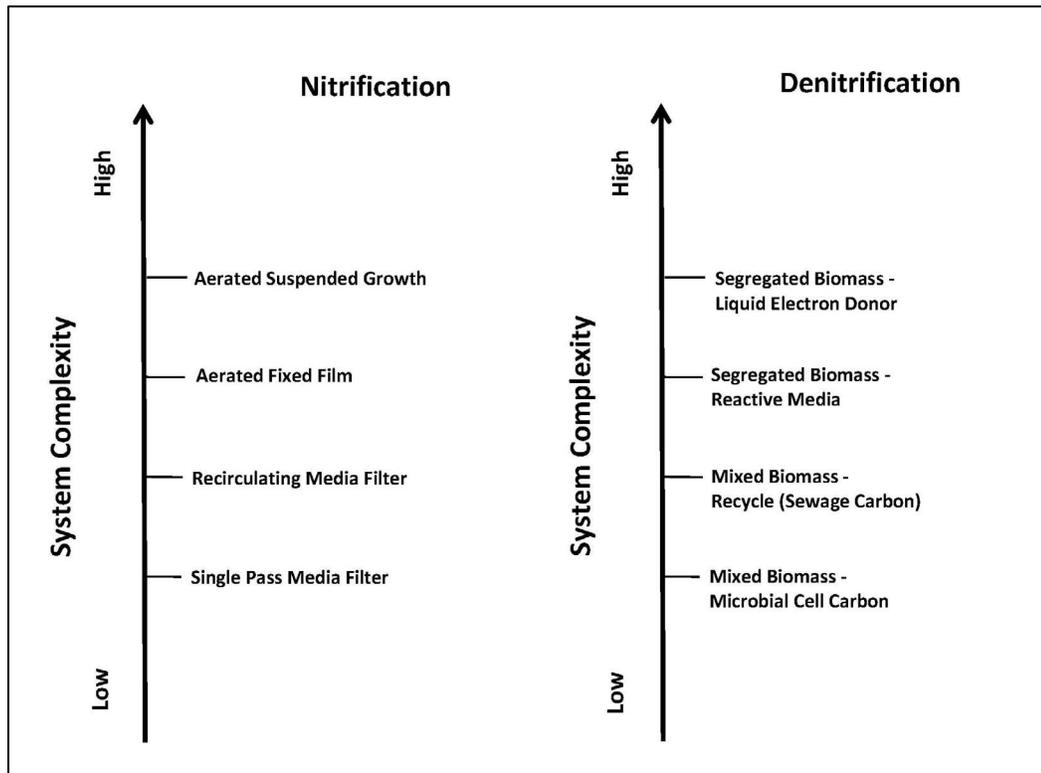
<sup>6</sup> Abbeggen, et al., (2008); Sarioglu, et al. (2009)

<sup>7</sup> Ronayne, et al. (1982); Gold, et al. (1992); Piluk and Peters (1994); Roy and Dube (1994)

California Regional Water Quality Control Board (1997); Ayres Associates (1998); Loudon et al. (2005)

<sup>8</sup> USEPA, (1993)

<sup>9</sup> Rich (2007); Heufelder et al. (2008)



**Figure 5-3: Relative Complexities of Nitrification / Denitrification Unit Operations**

### 5.3.1 Mixed Biomass Nitrification / Denitrification

#### 5.3.1.1 Suspended Growth (Activated Sludge) Reactors

Activated sludge processes are well developed and have proven capabilities to remove total nitrogen from sewage to very low concentrations via biological nitrification/denitrification (USEPA 1993). Many manufacturers offer suspended growth treatment units for onsite use. Most were developed to provide additional treatment after septic tanks to remove BOD<sub>5</sub> to reduce clogging of the infiltrative surface in the drainfield. Most of the manufactured units use the extended aeration process because of its simplicity and lower sludge production. Extended aeration is similar to conventional activated sludge and complete mix processes except the hydraulic residence times are one to more than two days as compared to less than 10 hours for the conventional and complete mix systems. The extended reaction times are used to maximize endogenous respiration, which reduces the amount of sludge accumulation.

More recently sequencing batch reactors (SBR) have been manufactured for onsite use, which are more complex in operation but can be easily automated. This process uses two or more reactor tanks in which aeration, sedimentation and decanting occur in each reactor. This allows the treatment to occur in batches. A decanted reactor (active biomass is retained in the reactor after decanting) is filled. Once filled, it receives no more influent and is allowed to aerate and settle on a timed cycle. In the meantime, another reactor is filled. When the treatment period is complete, the supernatant is discharged.

Both of these processes can achieve complete nitrification because of the extended aeration times. Also they are used to denitrify but denitrification by these processes requires careful management of the organic carbon during treatment. Both extended aeration and SBR processes can incorporate recycling back to the septic tank to reduce TN but during recycling TKN is added, which will not be completely denitrified and will enter the discharge stream. If only microbial cell carbon is relied upon, addition of TKN is avoided but without attention to carbon oxidation, sufficient carbon may not be available to support denitrification. Pulse or intermittent aeration can be an effective way to reduce the loss of organic carbon during nitrification (AyresAssociates 1998; Habermeyer and Sánchez 2005).

### 5.3.1.2 Recirculating Media Filters

Media filters are unsaturated, aerobic fixed film bioreactors, which accept settled raw wastewater or septic tank effluent for treatment. They consist of a lined excavation or container filled with a bed of porous media that is placed over an underdrain system. The wastewater is dosed onto the surface of the bed through a distribution network where it is allowed to percolate through the porous media to the underdrain system. The underdrain system discharges the filter percolate for further processing or discharge. The filter surface may be left open or covered.

The porous media is typically inert with sand and fine gravel being the most common materials, but peat, textile and open cell foam are also prevalent. Other media materials that are used are crushed glass, slag, tire chips, polystyrene, expanded shale, natural zeolites (hydrous aluminum silicates) and coir (fibrous material from coconut husks) see Table 5.4. Most filters using media other than sand or gravel are proprietary systems.

Aerobic biochemical transformations and physical filtration are the dominant treatment mechanisms within media filters, but chemical sorption also can be significant depending on the media selected. Oxygen is supplied by diffusion and mass flow of air behind wetting fronts through pore spaces in the media. Bio-slimes from the growth of microorganisms develop as films on the porous media. The microorganisms in the slimes absorb soluble and colloidal waste materials in the wastewater as it percolates over the surfaces

of the media. The absorbed materials are incorporated into new cell mass or degraded under aerobic conditions to carbon dioxide and water. The BOD is nearly completely removed if the wastewater retention times in the media are sufficiently long for the microorganisms to absorb the waste constituents. With depleting carbonaceous BOD in the percolating wastewater, nitrifying microorganisms thrive deeper in the surface layer where nitrification readily occurs.

“Single pass” and “recirculating” filters are used. With single pass or “intermittent” filters, the wastewater passes through the filter media only once before being discharged for further treatment or dispersal. Because only one pass is made, intermittent filters are not used where nitrogen reduction is needed. Recirculating filters recycle the filtrate through the filter several times. The recirculation provides the needed wastewater residence times in the media to achieve nitrification. Recycling provides more control of treatment process by adjustments that can be made to recycle ratios and dosing frequencies. BOD and TSS removals are somewhat greater than those achieved by single pass filters and nitrification is nearly complete. The mixing of the return filtrate with fresh influent in the recirculation tank results in significant nitrogen removal. Also, the filtrate can be recycled back to the treatment head works to mix with undiluted raw wastewater or to an anoxic reactor between the septic tank and recirculation tank to increase nitrogen removal significantly. Summaries of media filter applications, design, operation and performance can be found elsewhere (Crites and Tchobanoglous 1998; Leverenz, Tchobanoglous et al. 2002; USEPA 2002; Jantrania and Gross 2006).

Treatment performance of media filters using various media types is presented in Table 5.4. Typical filter effluent concentrations treating domestic wastewater treatment are <10/10 mg/L for BOD and TSS respectively and approximately 50 percent total nitrogen removal. With recycle back to the septic tank, total nitrogen removal can increase up to 75 percent (USEPA 2002).

Recirculating sand filters (RSF) are capable of achieving ammonia removals of 98 and Total N removals of 40 to over 70 percent (Piluk and Peters 1994; Kaintz and Snyder 2004; Loudon, Bounds et al. 2004; Richardson, Hanson et al. 2004). Effluent ammonia levels of 3 mg/L are typical (USEPA 2002; Urynowicz, Boyle et al. 2007). Low temperatures typically inhibit nitrification but recirculating media filters appear to overcome the effects of low temperatures by increasing residence time in the filters through recirculation. Regardless, adverse temperature effects should be of limited significance in the Florida climate.

Peat filters can achieve ammonia nitrogen removal efficiencies of 96 percent or greater from septic tank effluent, with effluent NH<sub>3</sub>-N in some cases reduced to 1 mg/L or less (Lacasse, Bélanger et al. 2001; Lindbo and MacConnel 2001; Loomis, Dow et al. 2004;

Patterson 2004; Rich 2007). Peat filters can also bind phosphorus (Kõiv, Vohla et al. 2009). TN reductions of 29 to 41 percent have been reported in modular recirculating peat filters (Monson Geerts, McCarthy et al. 2001a); 54 percent in peat filters using pressurized dosing (Patterson 2004).

Recirculating textile filters were shown to achieve 44 to 47 percent TN reduction (Loomis, Dow et al. 2004) from septic tank effluent. In some cases, textile filters treating septic tank effluent have produced effluents with  $\text{NH}_3\text{-N}$  levels of less than 1 mg/L (Rich 2007). Textile filters also produce nitrified effluents (McCarthy, Monson Geerts et al. 2001; Wren, Siegrist et al. 2004; Rich 2007) and are often operated at higher hydraulic loading rates (Table 5.4).

A variety of different media were tested in laboratory columns including slag, polonite (a calcium silicate based mineral material), limestone, opoka, and sand. Greater than 98 percent ammonia transformation to nitrate was achieved in all columns (Renman, Hylander et al. 2008). Stratified sand biofilters were used to treat synthetic dairy wastewater for > 300 days at loading rates of 0.16 to 1.46 gal/ft<sup>2</sup>-day and 0.0045 to 0.0119 lb BOD<sub>5</sub>/ft<sup>2</sup>-day; over 90 percent removal of reduced nitrogen was achieved (Rodgers, Healy et al. 2005). A horizontal flow bioreactor system using parallel plastic sheets as support media for microbial growth removed reduced nitrogen species by over 90 percent when operated at 3.8 gal/ft<sup>2</sup>-day (Rodgers, Lambe et al. 2006).

**Table 5.4**  
**Summary of Media Filter Performance**

Media Type	Features	Typical Performance Range
Sand (recirculating) <sup>1</sup>	1.5 - 3 mm media 18 - 36 in. depth 3 - 5 gal/ft <sup>2</sup> -day 40 - 120 dose/day	TN: Removal: 40 to 75% Effluent: 15 to 30 mg/L NH3-N: Effluent: 1 to 5 mg/L
Textile <sup>2</sup>	2 - 3 in. cubes 36 - 72 in. depth 8 - 17 gal/ft <sup>2</sup> -day 80 - 140 dose/day	TN Removal: 20 to 60% Effluent: 10 to 60 mg/L NH3-N: Effluent: 1.7 to 5.9 NO3-N: Effluent: 11 mg/L
Peat (single pass or recirculation) <sup>3</sup>	24 - 36 in. depth 3 to 6 gal/ft <sup>2</sup> -day 12 to 120 dose/day	TN: Removal: 10 to 75% Effluent: 10 to 60 mg/L TKN: Removal: 90 to 95% NH3-N: Effluent: 1 mg/L NO3-N: Effluent: 20 to 50
Open Cell Foam (single pass or recirculation) <sup>4</sup>	3 - 4 in. cube media 48 in. depth 11 gal/ft <sup>2</sup> -day	TN: Removal: 62% Effluent: 14 mg/L NH3-N: Effluent: 2.4 mg/L NO3-N: Effluent: 10 mg/L
Zeolite <sup>5</sup>	20 - 30 in. depth 6.1 gal/ft <sup>2</sup> -day	NH3-N: Removal: 98.6% Influent: 70 mg/L Effluent: 1 mg/L NO3-N: Effluent: 57 mg/L
Zeolite <sup>6</sup>	24 in. media depth Stratified media size 8 in. 2.3-4.8 mm 8 in. 1.2-2.4 mm 6 in. 0.5-1.2 mm 2.9 gal/ft <sup>2</sup> -day	TN: Removal: 36.1% Influent: 72.2 mg/L Effluent: 43.6 mg/L NH3-N: Removal: 99.9% Influent: 63.4 mg/L Effluent: 0.036 mg/L NO3-N: Effluent: 38.8 mg/L
Expanded Clay <sup>7</sup>	24 in. media depth Stratified media size 8 in. 3-5 mm 8 in. 1.0 - 2.0 mm 6 in. 0.5 -1.0 mm 2.9 gal/ft <sup>2</sup> -day	TN: Removal: 16.4% Influent: 72.2 mg/L Effluent: 59.7 mg/L NH3-N: Removal: 99.8% Influent: 63.4 mg/L Effluent: 0.13 mg/L NO3-N: Effluent: 58.9 mg/L

o:\44237-001R007\Wpdocs\Report\Final

**Table 5.4**  
**Summary of Media Filter Performance**

Media Type	Features	Typical Performance Range
Coir <sup>8</sup>	Coconut coir media 18 gal/ft <sup>2</sup> -day 5.88 gal/ft <sup>3</sup> -day	TN: Removal: 55% Influent: 38 mg/L Effluent: 17 mg/L TKN: Removal: 83% Influent: 38 mg/L Effluent: 6.5 mg/L
Aerocell <sup>9</sup>	2 in. cube media 18 gal/ft <sup>2</sup> -day 5.88 gal/ft <sup>3</sup> -day	TN: Removal: 77 % Influent: 40 mg/L Effluent: 9.3 mg/L TKN: Removal: 87% Influent: 40 mg/L Effluent: 5.4 mg/L
Polystyrene <sup>10</sup>	24 in media depth Polystyrene sphere media 2.5 - 4.5 mm 6.6 gal/ft <sup>2</sup> -day	NH3-N: Removal: 97.7% Influent: 92.5 mg/L Effluent: 2.1 mg/L

<sup>1</sup> Mueller, Sperandio et al. 1985; Sandy, Sack et al. 1987; Wakatsuki, Esumi et al. 1993; Boyle, Otis et al. 1994; Bruen and Piluk 1994; Duncan, Reneau et al. 1994; Mote and Ruiz 1994; Oseseck, Shaw et al. 1994; Piluk and Peters 1994; Crites and Tchobanoglous 1998; Jantrania, Sheu et al. 1998; Kanter, Tyler et al. 1998; Venhuizen, Wiersma et al. 1998; Christopherson, Anderson et al. 2001; Ebeling, Tsukuda et al. 2001; Lindbo and MacConnel 2001; MacQuarrie, Sudicky et al. 2001; Costa, Heufelder et al. 2002; Jaynes, Kaspar et al. 2002; Richardson, Hanson et al. 2004; Tsukuda, Ebeling et al. 2004; Horiba, Khan et al. 2005

<sup>2</sup> (McKee and Brooks 1994; Jantrania, Sheu et al. 1998; Lindbo and MacConnel 2001; Darby and Leverenz 2004; Loudon, Bounds et al. 2004; Wren, Siegrist et al. 2004; Horiba, Khan et al. 2005; Rich 2007

<sup>3</sup> Rock, Brooks et al. 1984; Lamb, Gold et al. 1987; Winkler and Veneman 1991; Boyle, Otis et al. 1994; McKee and Brooks 1994; Jantrania, Sheu et al. 1998; Ebeling, Tsukuda et al. 2001; Mergaert, Boley et al. 2001; Patterson, Davey et al. 2001; Monson Geerts, McCarthy et al. 2001b; Darby and Leverenz 2004; Loudon, Bounds et al. 2004; Patterson 2004; Tsukuda, Ebeling et al. 2004; Horiba, Khan et al. 2005; Patterson and Brennan 2006; Rich 2007

<sup>4</sup> NSF-International 2003e

<sup>5</sup> Philip and Vassel 2006

<sup>6</sup> Smith et al. 2008 Smith, 2008

<sup>7</sup> Smith et al. 2008 Smith, 2008

<sup>8</sup> (NSF-International 2006; Sherman 2006; Talbot, Pettigrew et al. 2006; Sherman 2007)137,180,181,196

<sup>9</sup> (NSF-International 2005)136

<sup>10</sup> E-Z Treat Company, 2009

The hydraulic, organic and nitrogen loading rates are critical operating parameters for recirculating media filters, particularly as they relate to the functioning of the physical and biological processes within the media. Key elements for successful treatment in a media filter are surface area for attachment of microorganisms and for sorption of dissolved and colloidal constituents in the wastewater, the need for sufficient pore space for assimilation of solid materials and their biodegradation between doses, the water retention capacity of the media, and the pore space that is available for aeration. The performance of any unsaturated media filter is determined by the interactions of media characteristics (Table 5.3) with system parameters (Table 5.4). A significant interaction that occurs is between the water retention capacity of the media and the hydraulic application rate. The water retention capacity is important for prolonging the wastewater retention time in the media to achieve adequate treatment. The water retention capacity of the media must exceed the hydraulic application rate per dose to prevent saturated flow to prevent rapid movement of the applied wastewater through the filter. However, if the water content in the soil exceeds 50 – 60 percent of the porosity, anoxic conditions will result (Bremner and Shaw, 1956; Christensen, et al., 1990; Cogger, et al., 1998; Donahue, et al., 1983; Pilot and Patrick, 1972; Reneau, 1979; Singer and Munns, 1991; Tucholke, et al., 2007).

Organic overloading to porous media biofilters leads to development of excessive biomass near the application surface, reduction in reaeration rates and media clogging that reduces treatment capacity (USEPA 2002; Kang, Mancl et al. 2007). A highly critical factor to optimum functioning of unsaturated media filters is the reaeration capacity of the filter media. Unsaturated media filters are four phase systems: solid media, attached microbial film, percolating wastewater, and gas phase. The total porosity (excluding internal pore spaces within the media) must be shared between attached biofilm, percolating water, and gas phase. A media with a high total porosity will more likely allow sufficient oxygen transfer throughout the filter bed, providing more effective utilization of the total media surface area for aerobic treatment. If media size becomes too small, a larger fraction of the pores may remain saturated and become inaccessible to oxygen transfer. For example, sand with a total porosity of 38 percent could have an aeration porosity of only 2.5 percent of the total media volume, depending on sand size, uniformity and the hydraulic application rate. Such conditions could decrease nitrification effectiveness but increase denitrification within microzones. Denitrification within an unsaturated filter would improve total nitrogen removal but could result in less efficient nitrification and higher effluent ammonia concentrations.

Media with significant ion exchange capacity may offer a method of superior removal of ammonia nitrogen in flowing systems. Natural zeolites provide excellent surfaces for biofilm attachments, and have relatively high porosities (Philip and Vassel 2006; Smith 2006; Zhang, Wu et al. 2007; Smith 2008; Smith, Otis et al. 2008). Sorption of ammonium ions

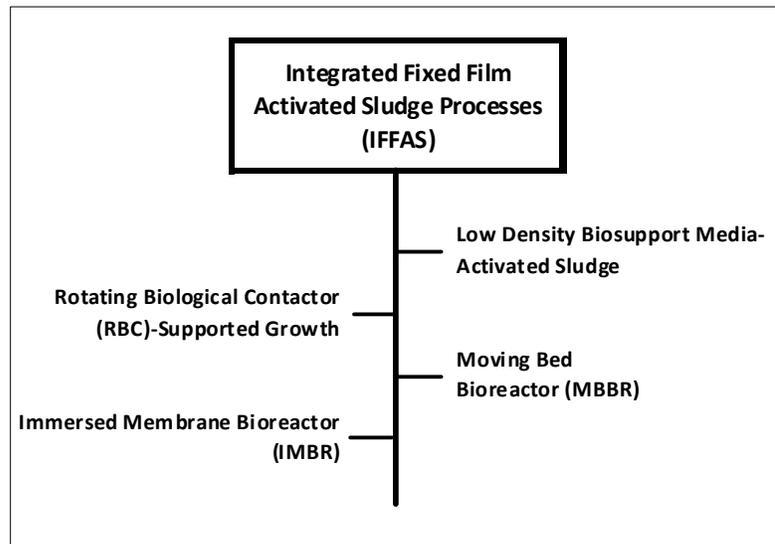
onto zeolite media can sequester ammonium ions from the water and provide enhanced contact with attached nitrifying organisms under steady flow conditions. Sorption also provides a buffer when loading rates are high or other factors inhibit nitrification, resulting in increased resiliency of the treatment process. Ammonia ion exchange adsorption onto zeolites is reversible, and microorganisms can biologically regenerate the zeolite media in periods of lower loading. A zeolite filter for onsite wastewater treatment removed 98.6 percent of ammonia and produced an effluent ammonia nitrogen concentration of 1 mg/L when operated at 6.1 gal/ft<sup>2</sup>-day (Philip and Vassel 2006). In an eight month bench scale study, a clinoptilolite media biofilter treating septic tank effluent and operated at 2.8 gal/ft<sup>2</sup>-day and 48 dose per day reduced ammonia by an average of 99.9 percent (Smith 2008; Smith, Otis et al. 2008). In these studies, the filters were able to sustain a BOD<sub>5</sub> surface loading rate of 0.0037 to 0.0041 lb/ft<sup>2</sup>-day without surface ponding or observable material accumulations of the media surface, which contrasts to reported COD loadings of 0.0039 lb/ft<sup>2</sup>-day which caused media clogging in sand filters (Healy, Rodgers et al. 2007). Other bench scale and pilot studies have demonstrated the ability of zeolite filters to maintain high ammonia removal under high non-steady loadings of ammonia nitrogen (Smith 2006). Expanded mineral media may also have significant sorption potential for ammonium ions (Kietlinska and Renman 2005; Hinkle, Böhlke et al. 2008). An expanded clay biofilter reduced ammonia by 99.9 percent when operated on septic tank effluent at 2.9 gal/ft<sup>2</sup>-day with dosing every 30 min.

Coconut coir is a natural, renewable material that is a waste product from coconut production. Coir has many of the same properties of peat that make it a desirable treatment media, including high surface area, high water retention, and high porosity (Talbot, Pettigrew et al. 2006), and has been successfully used as a planting media in greenhouses. While most coir is produced in Asia, Florida contains abundant coconut palm trees that could potentially provide a sustainable material source. An onsite wastewater treatment system using coconut coir has been reported (Sherman 2006; Sherman 2007).

Candidate media for the unsaturated media filter should possess many of the desirable characteristics that have been discussed above. Zeolite filters also have promise for unsaturated flow filters for passive systems. The interaction of cation exchange media with microbial reactions appears to offer potential for passive treatment with enhanced performance. Other candidate media include expanded clays, expanded shales, and tire crumb.

### 5.3.1.3 Integrated Fixed-Film Activated Sludge (IFAS)

IFAS is a group of technologies that combine both fixed film and suspended growth microbial communities. The combination of these communities results in very stable treatment processes that achieve more reliable and consistent performance than other mixed biomass processes. The more commonly used processes in this group are listed in Figure 5-4. All have been adapted for use in onsite treatment.



**Figure 5-4: Common Integrated Fixed Film Activated Sludge (IFAS) Processes**

The most common process design immerses low density biosupport media in a portion of the reactor tank through which the reactor contents are recirculated vertically down through the media. The recycle operation also mixes the entire reactor to keep the unattached biomass in suspension.

Moving bed bioreactors (MBBR) and immersed membrane bioreactors (IMBR) are two IFAS technologies that recently have been introduced to the onsite market and show promising performance.

### 5.3.2 Segregated Biomass (Two Stage) Denitrification

Segregated biomass processes consist of two separate stages of treatment that segregate the nitrification from denitrification. This type of process eliminates the problem of nitrate “leakage” in the discharge, which can occur in mixed biomass systems due to recycling. Consequently, a high degree of treatment is achieved more effectively. However, organic carbon that is used in single stage (mixed biomass) processes does not

reach the second anoxic stage requiring that an external donor be supplied to the second stage. Also alkalinity, which is recovered during denitrification, cannot be recycled to buffer the nitrification stage in a two stage system. If it is necessary to buffer the nitrification stage, an external source of alkalinity would be needed.

Two groups of processes are used for denitrification. Heterotrophic denitrification uses organic carbon as the electron donor, which may be added as a liquid or as a solid reactive medium. Autotrophic denitrification uses chemical compounds for electron donors, which are added as solid reactive media.

### 5.3.2.1 Anoxic Packed Bed Reactors

Anoxic packed bed reactors are filled with various kinds of “reactive” media, which is submerged and saturated. The “reactive” media provide a slowly dissolving source of electron donor for reduction of nitrate and nitrite by microbial denitrification. Denitrifying microorganisms grow predominantly attached to the media surfaces. Water flows by advection through the media pores, where the oxidized nitrogen species is consumed by attached microorganisms. Water saturation of the pores prevents ingress of oxygen, which could interfere with nitrate reduction.

Hydraulic and nitrogen loading rates, surface area of media, pore size, and flow characteristics within the reactor are important considerations. The media is consumed by dissolution, and this process must be sufficiently rapid to supply electron equivalents for nitrate reduction and other possible reactions. On the other hand, rapid dissolution would reduce the longevity of the media. Too rapid a dissolution rate could also lead to the presence of excess dissolution products in the effluent (e.g. BOD for wood-based filters; sulfate for sulfur-based filters). Geometry of the column could affect flow patterns and potential channeling; the later effects could be overcome by use of larger systems. The effects of flow channeling on performance deterioration could require maintenance or media replacement at time scales appreciably shorter than longevities based on theoretical stoichiometric requirements of electron donor for denitrification. A summary of performance of passive anoxic denitrification filters is shown in Table 5.5.

**Table 5.5**  
**Summary of Saturated Anoxic Media Reactors**

System Type	Description	Features	Treatment Performance
Sulfur/oyster shell filter (bench scale) <sup>1</sup>	1 liter bench column synthetic wastewater upflow single pass	Sulphur/oyster shell media (75/25% by volume) Sulphur: 4.7 mm	anoxic only NO <sub>3</sub> -N Removal: 80% Influent: 50 mg/L Effluent: 10 mg/L
Sulfur/oyster shell filter (bench scale) <sup>2</sup>	0.70 liter bench column septic tank effluent pre-treated in aerobic biofilter horizontal flow single pass	Sulphur/oyster shell media (75/25% by volume) Sulphur: 2 to 5 mm mm 11.1 gal/ft <sup>2</sup> -day	anoxic only NO <sub>3</sub> -N Removal: 99.9% Influent: 38.8 mg/L Effluent: 0.030 mg/L
Sulfur/oyster shell filter (bench scale) <sup>3</sup>	0.70 liter bench column septic tank effluent pre-treated in aerobic biofilter horizontal flow single pass 18 hr. HRT	Sulfur/oyster shell/expanded shale media (60/20/20% by volume) Sulphur: 2 to 5 mm mm 11.8 gal/ft <sup>2</sup> -day	anoxic only NO <sub>3</sub> -N Removal: 99.9% Influent: 58.8 mg/L Effluent: 0.031 mg/L
Sulfur/oyster shell filter (bench scale) <sup>4</sup>	0.70 liter bench column septic tank effluent pre-treated in aerobic biofilter horizontal flow single pass 18 hr. HRT	Sulfur/oyster shell/expanded shale media (45/15/40% by volume) Sulphur: 2 to 5 mm mm 10.8 gal/ft <sup>2</sup> -day	anoxic only NO <sub>3</sub> -N Removal: 89.9% Influent: 47.7 mg/L Effluent: 4.3 mg/L
Sulfur/limestone column <sup>5</sup>	22.4 gal. column Simulated groundwater upflow single pass Residence time: 24 to 48 hr.	Sulfur/limestone media (75/25% by volume) Sulfur: 5 to 10 mm 5 to 10 gal/ft <sup>2</sup> -day	anoxic only NO <sub>3</sub> -N Removal: >95% Influent: 60 mg/L Effluent: < 1 mg/L NO <sub>2</sub> -N Effluent: < 1 mg/L
Sulfur/oyster shell filter <sup>6</sup>	185 gal. column aerobic effluent upflow single pass 18 hr. HRT	Sulfur/oyster shell media (75/25% by volume) 47 gal/ft <sup>2</sup> -day	anoxic only NO <sub>3</sub> -N Removal: 88% Influent: 20 mg/L Effluent: 2.4 mg/L
Sulfur/limestone column <sup>7</sup>	237 gal. column groundwater upflow single pass Residence time: 13 hr.	Sulfur/limestone media (67/33% by volume) 63 gal/ft <sup>2</sup> -day Sulfur: 2.5 to 3.0 mm Limestone: 2.38 to 4.76 mm	anoxic only NO <sub>3</sub> -N Removal: 96% Influent: 64 mg/L Effluent: 2.4 mg/L NO <sub>2</sub> -N Effluent: 0.2 mg/L

**Table 5.5  
Summary of Saturated Anoxic Media Reactors**

System Type	Description	Features	Treatment Performance
Nitrex™ <sup>8</sup>	aerobic effluent gravity flow upflow single pass	Nitrex wood-based media 24 to 30 inch media depth (est.) 4.6 gal/ft <sup>2</sup> -day (est.)	aerobic+anoxic  TN Removal: 79 to 96% Effluent: 3 to 18 mg/L NO <sub>3</sub> -N Effluent: 0.3 to 8 mg/L
Black& Gold™ <sup>9</sup>	wood-based media single pass downflow gravity	Influent: STE 280 gal. column Sand/tire crumb/woodchip (85/11/5% by volume) 8.3 gal/ft <sup>2</sup> -day	aerobic+anoxic  TN Removal: 98% Influent: 414 mg/L Effluent: 7.1 mg/L NH <sub>3</sub> -N Effluent: 4.4 mg/L NO <sub>3</sub> -N Effluent: 0.05 mg/L

<sup>1</sup> Sengupta and Ergas 2006

<sup>2</sup> Smith et al. 2008, Smith, 2008

<sup>3</sup> Smith et al. 2008, Smith, 2008

<sup>4</sup> Smith et al. 2008 Smith, 2008

<sup>5</sup> (Moon, Shin et al. 2008)

<sup>6</sup> Brighton 2007

<sup>7</sup> Darbi, Viraraghavan et al. 2003a

<sup>8</sup> Long 1995; Robertson, Blowes et al. 2000; Dupuis, Rowland et al. 2002; Loomis, Dow et al. 2004; Robertson, Ford et al. 2005; EPA 2007; Rich 2007; Vallino and Foreman 2007

<sup>9</sup> Shah 2007

### 5.3.2.2 Heterotrophic Denitrification

Passive heterotrophic denitrification systems use solid phase carbon sources including woodchips (Robertson and J. A. Cherry 1995; Robertson, Blowes et al. 2000; Cooke, Doheny et al. 2001; Jaynes, Kaspar et al. 2002; Kim, Seagren et al. 2003; Robertson, Ford et al. 2005; Greenan, Moorman et al. 2006; van Driel, Robertson et al. 2006), sawdust (Kim, Seagren et al. 2003; Eljamal, Jinno et al. 2006; Greenan, Moorman et al. 2006; Jin, Li et al. 2006; van Driel, Robertson et al. 2006; Eljamal, Jinno et al. 2008), cardboard (Greenan, Moorman et al. 2006), paper (Kim, Hwang et al. 2003; Jin, Li et al. 2006), and agricultural residues (Cooke, Doheny et al. 2001; Kim, Seagren et al. 2003; Greenan, Moorman et al. 2006; Jin, Li et al. 2006; Ovez 2006a; Ovez, Ozgen et al. 2006b; Xu, Shao et al. 2009). Limited studies have also been conducted using other carbon sources such as cotton (Della Rocca, Belgiorna et al. 2005), poly( $\epsilon$ -caprolactone) (Horiba, Khan et al. 2005), and bacterial polyesters (Mergaert, Boley et al.

o:\44237-001R007\Wpdocs\Report\Final

2001). Cellulosic-based systems using wood agricultural residues, particularly corn cobs, are the most common. Such systems have produced average TN removals of 88 to 96 percent from septic tank effluent, with average effluent NO<sub>3</sub>-N concentrations of 2 to 5.4 mg/L (WDOH 2005; Rich 2007). In another study, a subsurface leaching chamber was installed beneath an active parking lot for on-site sewage treatment, using sawdust as carbon source (St. Marseille and Anderson 2002). At a loading of 1.22 gal/ft<sup>2</sup>-day; the effluent NO<sub>3</sub>-N averaged 0.6 mg/L. Chang et al. (2009a) reported initial results for septic tank effluent treatment using a lined drainfield that contained a layer of lignocellulosic-based electron donor media underneath a layer of sand. The systems were operated at a surface loading rate of ca 0.5 gal/ft<sup>2</sup>-day, with an influent total nitrogen of 46.3 mg/L. Ammonia removals were 85 to 90 percent in the two monitoring samples, while the corresponding total nitrogen removals were 60 and 85 percent. Other heterotrophic denitrification systems have been successfully tested at laboratory scale.

### 5.3.2.3 Autotrophic Denitrification

The autotrophic denitrification systems that have received the most attention are elemental sulfur-based media filters, which are under development. Sulfur-based denitrification filters have employed limestone or oyster shell as a solid phase alkalinity source to buffer the alkalinity consumption of the sulfur-based biochemical denitrification (Flere and Zhang 1998; Shan and Zhang 1998; Koenig and Liu 2002; Nugroho, Takanashi et al. 2002; Zhang 2002; Kim, Hwang et al. 2003; Darbi, Viraraghavan et al. 2003a; Darbi and Viraraghavan 2003b; Zhang 2004; Zeng and Zhang 2005; Sengupta and Ergas 2006; Zhang and Zeng 2006; Brighton 2007; Sengupta, Ergas et al. 2007; Sierra-Alvarez, Beristain-Cardoso et al. 2007; Smith 2008; Smith, Otis et al. 2008). The use of solid phase sulfur obviates the need for careful dosing control of sulfur donor that would pertain for liquid sulfur sources (Campos, Carvalho et al. 2008). Furthermore, dissolution of solid phase alkalinity sources will add bicarbonate and buffer the pH, ostensibly leading to more stable operation for autotrophic denitrifiers (Ghafari, Hasan et al. 2009). Nitrate can also act as electron acceptor for sulfide species as well as elemental sulfur (Mahmood, Zheng et al. 2007; Li, Zhao et al. 2009).

A pilot scale filter containing elemental sulfur and oyster shell at a 3:1 ratio was operated for 11 months at the Massachusetts Alternative Septic System Test Center (Brighton 2007). The filter received the effluent from an aerobic fixed film treatment system that received septic tank effluent. The sulfur/oyster shell filter removed 82 percent of influent TN, while the aerobic/sulfur treatment train removed 89.5 percent TN from the septic tank effluent. A pilot scale elemental sulfur/limestone column was operated for 6 months on a well water containing 65 mg/L NO<sub>3</sub>-N; nitrate removal averaged 96 percent and average effluent NO<sub>3</sub>-N was 2.4 mg/L (Darbi, Viraraghavan et al. 2003a). A 22.5 gallon upflow column packed with sulfur/limestone at a 3:1 volume ratio treated a simulated

groundwater at 0.9 to 1.8 gal/ft<sup>2</sup>-day surface loading rate and removed greater than 95 percent of nitrate that was at 60 mg/L NO<sub>x</sub>-N in the influent (Moon, Shin et al. 2008). A laboratory sulfur/oyster shell column was operated at an empty bed contact time of 0.33 to 0.67 days and removed 80 percent of influent nitrate (Sengupta and Ergas 2006). Three saturated denitrification biofilters containing sulfur and oyster shell media were operated for eight months on septic tank effluent that was pretreated with unsaturated media filters that provided ammonification, nitrification, and carbonaceous biochemical oxygen demand reduction (Smith, 2008; Smith, 2008). Average NO<sub>x</sub> reductions were 99.9, 99.9 and 88.9 percent respectively for treatment of effluent from unsaturated biofilters containing clinoptilolite, expanded clay, and granular rubber media, respectively. Corresponding average effluent NO<sub>x</sub>-N were 0.03, 0.031 and 4.3 mg/L. These denitrification filters operated at hydraulic loading rates of 4.9 gal/ft<sup>2</sup>-day and at average NO<sub>x</sub>-N loadings of 0.003 to 0.005 lb/ft<sup>2</sup>-day, which are similar to loading rates applied to acetic acid amended sand denitrification filters that achieved 94 to 99 percent NO<sub>x</sub> reduction (Aslan and Cakici 2007).

Design factors for sulfur-based denitrification filters include filter size and aspect ratio, water residence time, media size and shape, and the fraction of media for alkalinity supply. Smaller media particle size has been shown to result in higher volumetric denitrification rate constants, ostensibly due to higher surface area for sulfur dissolution and biochemical reaction (Moon, Chang et al. 2006). Factors that affect the long term performance of sulfur-based autotrophic denitrification filters include the long term availability of electron donor supply for the wastestream being treated, the physical structure of the biodegradable components of the media, reduction in external porosity due to solids accumulation, and continued availability of phosphorus as a nutrient for autotrophic microorganisms (Moon, Shin et al. 2008). As for any packed bed, biologically active media filter deployed over extended periods of time, the long term hydraulics of the unit are a concern. Accumulation of biological and inorganic solids could lead over time to the development of preferential flow paths within the filter, reducing average residence time and wastewater contact with the media. To the extent that these processes occur, deterioration of performance could result. The timescales of media replacement, maintenance and supplementation and the practical aspects of these activities must be considered. Another factor is the release of sulfate as water passes through the filter, and possible odors through hydrogen sulfide generation.

Several candidate media can be suggested for the saturated media filter which forms the second stage of a passive onsite nitrogen removal system for Florida. Media should possess many of the desirable characteristics that have been previously discussed. Both elemental sulfur and lignocellulosic based treatment systems are readily available and economical candidates. Crushed oyster shell is readily available. These alkalinity sources could also be used in a single pass, unsaturated first stage filter if nitrification

would otherwise be inhibited. Anion exchange media, and its interaction with microbial mediated denitrification reactions, offers the potential to increase denitrification performance in passive filtration systems (Samatya, Kabay et al. 2006; Matos, Sequeira et al. 2009). Expanded shales with anion exchange capacity are commercially available and could be used in mixed media to increase the resiliency and performance of second stage anoxic denitrification filters.

## 5.4 Physical / Chemical Nitrogen Reduction Processes

Physical/chemical processes have not been widely used for onsite treatment systems in the U.S. primarily because of their complexity and associated costs. Preliminary research has been done on various processes that could have application, but full development for onsite treatment systems has not been achieved. The primary processes of interest have been membranes, ion exchange and evaporation.

### 5.4.1 Membrane Processes

While membranes are used for water and wastewater treatment, they have not been applied effectively for nitrogen removal in onsite wastewater. Membranes are a separation technology based on filtration through synthetic membranes. However, most are not capable of removing nitrogen molecules from water. Reverse osmosis is one membrane process that is capable of nitrogen removal and is used in wastewater treatment, but has not been applied to onsite treatment. It is used for treatment of household drinking water however.

Membrane bioreactors (MBR), also called immersed membrane bioreactors (IMBR), have gained widespread application in municipal treatment facilities and recently have been introduced to the onsite treatment market. These membranes are used in activated sludge processes as a replacement for the final clarifier. The membranes retain the volatile suspended solids in the treatment vessel through filtration rather than sedimentation, which allows significantly higher mixed liquor concentrations that facilitate simultaneous denitrification. Because the membranes themselves do not remove the nitrogen but rather support more effective biological denitrification, this type of process is reviewed under "Biological Nitrification / Denitrification Processes".

Ion exchange for removal of either  $\text{NH}_4^+$  or  $\text{NO}_3^-$  nitrogen from wastewater has been studied by several investigators. The natural zeolite clinoptilolite has been shown to have a high selectivity for ammonium with a total exchange capacity of approximately 2 meq/g. It can be regenerated with sodium chloride or an alkaline reagent such as sodium or calcium hydroxide. However, without prior treatment, the zeolite is easily fouled (University of Wisconsin, 1978; Eckenfelder, 1991). Wu, et al. (2008) found that the addition of powdered zeolite added to a contact stabilization activated sludge plant was ef-

fective in removing ammonium and during the anoxic stage was biologically regenerated. However, the powdered zeolite was continuously lost from the system. Removal of low concentrations typical found in municipal wastewater were not effective (Zhang, 2007).

Distillation is another process that has been considered for onsite sewage treatment. Efforts to develop an effective proprietary mechanical distillation unit have been attempted but have not been marketed. Disposal of the residuals containing nitrogen have not been addressed.

## 5.5 Natural Systems

Natural treatment systems represent a group of technologies and practices that rely heavily on the assimilative capacity of the receiving environment to effect the required treatment. These systems tend to be passive and typically have larger land area requirements. With OSTDS, the soil matrix with its myriad of physical, chemical, and biological processes that it supports is how most treatment is achieved, and this can vary with soil characteristics, climate, and method of wastewater application. The intrinsic values of natural systems are their operational and mechanical simplicity. They tend to absorb perturbations in influent flows with little operator attention or loss of performance. However, their potential liability is the unpredictability of the many natural processes that effect the needed treatment due to fluctuating environmental conditions. Therefore, design of natural systems needs to be more forgiving of changes by including recycle loops, load-splitting, and operation flexibility.

Natural systems are the traditional methods of onsite wastewater treatment. Historically however, the basis of their design was the hydraulic loading to the soil with the objective of avoiding wastewater surfacing and exposure to the public. Today, groundwater and surface water contamination is equally a concern. Designed properly, there are several natural systems that have application for onsite sewage treatment and are able to meet the more stringent water quality requirements except in the most sensitive of environments. These include soil infiltration, vegetative uptake, evapotranspiration and constructed wetlands.

### 5.5.1 Soil Infiltration

Biological denitrification in soils below wastewater infiltration systems readily occurs where the requisite conditions exist. To define these requirements, the heterotrophic denitrification process model was used. Using these, it is clear that the most critical conditions include the soils natural drainage, depth to saturated conditions, and the availability of organic carbon. Internal drainage provides a measure of the soil's permeability and the extent of time that it may be unsaturated. Unsaturated conditions are necessary to

aerate the soils to allow the autotrophs to nitrify the ammonium nitrogen to nitrate. The shallower the depth to the water table, the more likelihood organic matter will be leached to where the soil moisture is sufficiently high to restrict soil reaeration to the point that aerobic organic matter decomposition is inhibited, which preserves the organic carbon for heterotrophic denitrification. Insufficient organic carbon will limit the extent of denitrification that can occur.

The capacity of the soil to denitrify varies depending on the specific environmental conditions at the particular site and the design and operation of the OSTDS. Numerous investigations into the fate of nitrogen below OSTDS have been undertaken. However, the results are quite variable even for sites that appear similar. Gold and Sims (2000) point out the dynamic and open nature of OSTDS designs that create uncertainties with in-situ studies of the fate of nitrogen in soil. The affects of dispersion, dilution, special variability in soil properties, wastewater infiltration rates, inability to identify a plume, uncertainty of whether the upstream and downstream monitoring locations are in the same flow path, and temperature impacts are a few of the problems that challenge the in-situ studies. As a result, even when small differences in concentrations are observed, the spatial and temporal variability can result in large changes in estimates of the mass loss of nitrogen.

Several investigators have performed rather thorough reviews of the fate of nitrogen below soil water infiltration systems. Siegrist and Jennsen (Siegrist and Jennsen 1989) reviewed national and international literature for both laboratory and field studies of nitrogen removal for soil infiltration. Laboratory studies using soil columns showed removals of TN from less than 1 to 84 percent. Hydraulic loadings varied from 1.23 to 8.66 gal/ft<sup>2</sup>-day and influent TN concentrations from 16 to 74 mg/L. The field studies were performed on systems installed in sands. As in the case of most field studies, influent flows and TN concentrations were not always accurately known. Estimates of TN removal in these studies ranged from 0 to 94 percent. The investigators noted that high TN removals have been observed but that reasonably comparable studies showed limited removals. Based on their review, they provided a table of what they thought were “achievable nitrogen removal efficiencies” below soil water infiltration zones (Table 5.6).

**Table 5.6**  
**Total Nitrogen Removals below Soil Infiltration Zones**  
**(Siegrist and Jenssen 1989)**

Soil Water Infiltration Type	Achievable N Removals	
	Typical	Range
Traditional In-Ground	20%	10 – 40%
Mound/Fill	25%	15 – 60%
Systems with Cyclic Loading	50%	30 – 80%

Long (Long 1995) reviewed studies of nitrogen transformations in OSTDS to develop a methodology for predicting OSTDS nitrogen loadings to the environment. Long also found that in-situ studies were confounded with many known and unknown variables that made data interpretation complicated. His review of the data indicated that soil treatment removes between 23 to 100 percent of the nitrogen. He correlated greater removals with finer grained soils because anoxic conditions would be achieved more frequently, which also would help to preserve available organic carbon for denitrification. Using this correlation, he estimated TN removals as shown in Table 5.7.

**Table 5.7**  
**Estimates of TN Removal Based on Soil Texture Below**  
**a Traditional Household Wastewater Infiltration System (Long 1995)**

Soil Texture	Estimated TN Removal	Comments
Coarse grained sands	23%	Soils promote rapid carbon and nitrogen oxidation leaving insufficient carbon for denitrification. If anoxic conditions and a source of carbon are available, such as a high or fluctuating water table, TN removal would increase.
Medium grained sands	40%	Soils restrict gas transfer during bulk liquid flow periods to create anoxic conditions.
Fine grained sands	60%	Soils restrict gas transfer for longer periods after bulk flow periods.
Silt or clay	70%	Soils further restrict gas transfer and retain nutrients higher in the soil profile.

Gable and Fox (Gable and Fox 2000) and Woods et al. (Woods, Bouwer et al. 1999) suspect that the Anammox process could explain why nitrogen removal below large soil aquifer treatment systems (SAT) exceeds what can be attributed to heterotrophic nitro-

gen removal alone because the organic carbon to nitrogen ratio is typically too low to sustain heterotrophic denitrification. Crites (Crites 1985) reports that denitrification below seven large scale SAT systems in the US were observed to achieve total nitrogen removals of 38 to 93 percent. While Anammox quite likely could contribute substantially to the reduction of nitrogen below OSTDS, little is known about the conditions under which it is likely to occur. Until the Anammox process is better understood, estimating the extent of denitrification via the Anammox process is difficult. Such data were not available so the estimates of nitrogen removal below OSTDS reported in this study may underestimate the actual removals.

In a study investigating the effects of effluent type, effluent loading rate, dosing interval, and temperature on denitrification under soil water infiltration zones, Degen, et al. (Degen, Reneau et al. 1991) and (Stolt and R. B. Reneau 1991) reviewed published results of other studies that measured denitrification in OSTDS. They found denitrification removals varied substantially depending on the type of pretreatment and the design of the soil water infiltration system (Table 5.8).

**Table 5.8**  
**Total Nitrogen Removals Found in Various Studies of OWTS**

<b>System Type</b>	<b>TN Removal</b>
Traditional	0-35% <sup>1</sup>
Sand Filter	71-97% <sup>2</sup>
Low Pressure Dosing Shallow	46% <sup>3</sup>
Low Pressure Dosing At-Grade	98% <sup>4</sup>
Mound	44-86% <sup>5</sup>

<sup>1</sup> Ritter and Eastburn, 1988

<sup>2</sup> Wert and Path, 1985

<sup>3</sup> Brown and Thomas, 1978

<sup>4</sup> Stewart and Reneau, 1988

<sup>5</sup> Harkin, Duffy et al., 1979

The more significant environmental factors that determine whether nitrogen removal occurs and to what extent include the soil's texture, structure, and mineralogy, soil drainage and wetness, depth to a saturated zone and the degree to which it fluctuates, and amount of available organic carbon present. OSTDS design and operation factors include the species of nitrogen discharged to the soil infiltration zone, the depth and geometry of the infiltrative surface, the daily hydraulic loading and its method of application, whether it is dosed and, if so, its frequency.

Soil drainage class has been found to be a good indicator of a soil's capacity to remove nitrogen (Gold, Addy et al. 1999). The Natural Resources Conservation Service (NRCS) uses seven drainage classes to describe the "quality" of the soil that allows the downward flow of excess water through it (USDA 1962). The classes reflect the frequency and duration of periods of soil saturation with water, which are determined in part, by the texture, structure, underlying layers, and elevation of the water table in relation to the addition of water to the soil. Table 5.9 provides a brief description of each of the classes and their expected impacts on denitrification.

**Table 5.9**  
**NRCS Drainage Classes, Descriptions and Expected Impacts on Denitrification**

<b>Drainage Class</b>	<b>Description</b>	<b>Expected Impact on Heterotrophic Denitrification</b>
Excessively drained	Water is removed from the soil very rapidly. The soils are very porous. These soils tend to be droughty.	<ul style="list-style-type: none"> <li>● Well aerated soil capable of achieving complete nitrification of applied TKN</li> <li>● Provides little organic carbon and will likely degrade any added organic matter within the aerobic zone</li> <li>● Short retention time</li> </ul>
Somewhat excessively drained	Water is removed from the soils rapidly. The soils are sandy and very porous. These soils tend to be droughty but can support some agricultural crops without irrigation.	<ul style="list-style-type: none"> <li>● Well aerated soil capable of achieving complete nitrification of applied TKN</li> <li>● Provides little organic carbon and will likely degrade any added organic matter within the aerobic zone</li> <li>● Short retention time</li> </ul>
Well drained	Water is removed from the soil readily but not rapidly. The soils are commonly intermediate in texture and retain optimum amounts of moisture for plant growth after rains.	<ul style="list-style-type: none"> <li>● Sufficiently aerated soil capable of achieving complete nitrification</li> <li>● May allow some organic matter to reach a saturated zone where it would be available for denitrification if a shallow water table is present</li> </ul>
Moderately well drained	Water is removed from the soil somewhat poorly so that the profile is wet for a small but significant period of time. The soils commonly have a slowly permeable layer within or immediately beneath the solum and/or a shallow water table.	<ul style="list-style-type: none"> <li>● Sufficiently aerated soil capable of achieving complete nitrification</li> <li>● Denitrification would be enhanced with a fluctuating water table for a "two sludge" process or with slow drainage for a "single sludge" process</li> </ul>

o:\44237-001R007\Wpdocs\Report\Final

**Table 5.9  
NRCS Drainage Classes, Descriptions and Expected Impacts on Denitrification**

<b>Drainage Class</b>	<b>Description</b>	<b>Expected Impact on Heterotrophic Denitrification</b>
Somewhat poorly drained	Water is removed from the soil slowly enough to keep it wet for significant periods of time. These soils commonly have a slowly permeable layer within the profile and/or a shallow water table. The growth of crops is restricted to a marked degree unless artificial drainage is provided.	<ul style="list-style-type: none"> <li>● Ample organic matter for a carbon source and to create anoxic conditions in saturated zones for significant nitrogen reduction</li> <li>● Insufficiently aerated soil to nitrify TKN requiring nitrification of the wastewater prior to application to the soil</li> </ul>
Poorly drained	Water is removed so slowly that the soil remains wet for a large part of the time. The water table is commonly at or near the soil surface for a considerable part of the year. They tend to be mucky.	<ul style="list-style-type: none"> <li>● Ample organic matter for a carbon source and to create anoxic conditions in saturated zones for significant nitrogen reduction</li> <li>● Insufficiently aerated soil to nitrify TKN requiring nitrification of the wastewater prior to application to the soil</li> </ul>
Very poorly drained	Water is removed from the soil so slowly that the water table remains at or on the surface the greater part of the year. They commonly have mucky surfaces.	<ul style="list-style-type: none"> <li>● Ample organic matter for a carbon source and to create anoxic conditions in saturated zones for significant nitrogen reduction</li> <li>● Insufficiently aerated soil to nitrify TKN requiring nitrification of the wastewater prior to application to the soil</li> </ul>

Poorly drained and very poorly drained soils can have a high capacity for nitrogen removal because the saturated zone is shallow, carbon enriched and anoxic while moderately well and well drained soils have a very limited denitrification capacity (Parkin and Meisinger 1989; Groffman, Gold et al. 1992; Simmons, Gold et al. 1992; Hanson, Groffman et al. 1994; Nelson, Gold et al. 1995). Groundwater in moderately well drained or well drained soils typically flows deeper within the subsoil and does not intersect the reduced and organic enriched surface horizons.

Heterotrophic bacterial denitrification is often limited by the availability of sufficient quantities of organic matter (Burford and Bremner 1975; Gambrell, Gilliam et al. 1975; Christensen, Simkins et al. 1990; Bradley, Fernandez et al. 1992). Sources of organic matter

o:\44237-001R007\Wpdocs\Report\Final

in soil are either natural, which is continuously replenished in the soil from the decay of vegetative materials or supplied by the wastewater itself.

The amount of organic matter in the soil is greatest in the root zone (Starr and Gillham 1993; Paul and Zebarth 1997). Roots regularly exude carbonaceous materials and die and decay. Much of the organic carbon is degraded in the vadose zone through natural degradation within 2-3 ft of the ground surface. Organic matter is typically very low (<1%) below about 3 ft in most soils with a deep vadose zone. There are some cases of soil horizons that are lower in the soil profile and that contain organic matter, iron and aluminum. An example is spodic soils which are common in some locations, which contain organic matter that would be available for heterotrophic denitrifiers.

Water tables or perched saturated zones restrict reaeration of the soil. With organic matter present, the saturated zone will become anoxic or anaerobic. This will inhibit nitrification and if nitrate and organic matter are present, will support denitrification. When the air-filled porosity drops below 11 to 14 percent or the moisture content is greater than 60 to 75 percent of the soil's water holding capacity, reaeration is sufficiently restricted to allow anoxic conditions to develop (Bremner and Shaw 1956; Pilot and Patrick 1972; Reneau 1977; Donahue, Miller et al. 1983; Christensen, Simkins et al. 1990; Singer and Munns 1991; Cogger, Hajjar et al. 1998; Tucholke, McCray et al. 2007).

If the water table is deep, little denitrification seems to occur. In soils with thick unsaturated zones, organic matter may not reach the saturated zone because it is oxidized before it can leach to the water table. Where the ground water depths exceed about three feet, denitrification is greatly reduced (Starr and Gillham 1993; Barton, McLay et al. 1999). However, a shallow, fluctuating water table can create the conditions for simultaneous denitrification. This occurs when a seasonally high water table prevents nitrification of the ammonium, which will adsorb to negatively charged clay particles in the soil. The ammonium is held by the soil and after draining and reaerating, the ammonium is nitrified. If organic matter is present and the soil nears saturation again, the nitrate can be denitrified and the newly applied ammonium is adsorbed as before, repeating the process. (Walker, Bouma et al. 1973a; Reneau 1977; Cogger 1988).

The type of infiltration system used can affect the soil's potential for nitrogen removal. Traditional in-ground trench systems are installed with their infiltrative surfaces typically below the A horizon and thus below where organic matter can be expected to be the highest. At-grade and mound systems are typically installed above the O and A horizon thereby gaining the advantage of having a high organic layer available to create anoxic conditions with organic carbon available (Harkin, Duffy et al. 1979; Converse 1999). However, in Florida, the OSTDS rules for mound construction require the removal of the O and A horizons, which removes most of the available organic carbon. Also, "digouts",

which are systems on sites where a restrictive horizon in the soil profile is removed, can result in reducing a particular soil's nitrogen removal potential because quite often the restrictive horizon removed is a spodic layer, which can have a sufficiently high organic content and be restrictive enough to create a saturated zone where anoxic conditions may be created for denitrification.

Modifying the method by which sewage is applied to the soil has been shown to enhance nitrogen removal in soil infiltration systems. By dosing septic tank effluent on timed cycles into the drainfield, alternating aerobic and anoxic conditions are created in the biomat and upper layer of the drainfield's soil infiltrative surface. With each dose the infiltrative surface becomes saturated during which time the soil can become anoxic due to the depletion of oxygen created by facultative heterotrophic bacteria degrading the organic matter. With the creation of anoxic conditions, nitrification of the ammonium ceases and the ammonium ion, which is positively charged, is adsorbed onto the negatively charged soil particles. As the soil drains and reaerates, the ammonium is nitrified but is not able to percolate downward because the soil has drained and is no longer saturated. However, the next dose adds fresh organic matter, which causes anoxic conditions to return creating the necessary conditions to enable the heterotrophic bacteria to denitrify the nitrate using the fresh septic tank effluent carbon as an electron donor. This intermittent dosing of septic tank effluent has been shown by several studies to reduce the total nitrogen applied.

A controlled field study was conducted at the Colorado School of Mines to investigate the fate of nitrogen in septic tank effluent that is applied to soil using drip dispersal (Parzen, et al., 2007). The study showed that ammonium decreased with depth but it did not disappear completely. The ammonium apparently was nitrified when aerobic conditions were present between doses. Nitrate however decreased with depth that could not be explained by dilution based on bromide tracer tests performed at the test site. Denitrification appeared to be responsible for the reduction. The total reduction of nitrogen was not quantified in this study however.

Similar studies were performed at Delaware Valley College. One was a study conducted in two phases over eight years in sandy loam to loam soil (Hayes & Moore, 2007). Four treatment sites were monitored. Groundwater samples were taken up and down gradient of each drainfield. Results showed that the median concentration values of total nitrogen in the groundwater below the test site were consistently less than the water quality standards. These results suggested that systems can perform well in areas where the seasonally high water tables are less than 50 cm (20 in) below ground surface provided that a 30 cm (12 in) separation between the drip tubing and the seasonally high water table is maintained.

A second study was designed to measure the reductions of bacteriologic and chemical constituents in septic tank effluent with soil depth using drip dispersal for the effluent application (Hepner, et al., 2007). Three drip dispersal systems of 1,200 lineal feet of drip tubing each were dosed with 400 gpd septic tank treated wastewater at a hydraulic loading rate of 0.17 gal/ft<sup>2</sup>-day. Zero tension lysimeters were installed at 1, 2, 3, and 4 feet beneath the surface to capture gravity water moving through the soil. Samples were analyzed for fecal coliform, fecal strep, BOD<sub>5</sub>, NH<sup>3</sup>-N, and NO<sup>3</sup>-N. Median value reductions of 99 percent for fecal coliform, 99 percent for fecal strep, 86 percent for BOD<sub>5</sub>, and 85 percent for NH<sup>3</sup>-N + NO<sup>3</sup>-N. Based on these trials 1 foot of aerobic soil appeared to provide significant treatment of septic tank wastewater when loaded at 0.17 gal/ft<sup>2</sup>-day with a landscape linear load of approximately 6 gal/ft-day.

### 5.5.2 Constructed Wetlands

Subsurface flow constructed wetlands are another natural system that has been used for single family and commercial applications. This system consists of a submerged rock bed that may be planted with wetland vegetation. Initially claimed to remove nitrogen from septic tank effluent, studies have shown that wetland plant roots do not supply excess oxygen to nitrify ammonium in septic tank effluent (Austin & Nivala, 2009; Behrands & Bailey, 2007; Burgan & Sievers, 1994; Huyang and Reneau, 1994; Johns, et al., Kavanagh and Keller, 2007; McIntyre and Riha, 1991; USEPA, 2000). Nitrification seldom exceeds 50 percent, which limits denitrification. However, denitrification does reduce nearly all the nitrate that is available. Providing recirculating gravel filters or vertical wetlands to pre-nitrify the effluent has been successful in increasing total nitrogen reductions in subsurface vegetated beds up to nearly 90 percent (Askew & Hines, 1994; Kantawanichkul, et al., 2001; White, 1995). Anammox may be an alternative pathway for removing nitrogen in wetlands without the need for denitrification. Several alternative biochemical pathways may be involved, but development work is needed to optimize wetland design to successfully apply this process (Wallace & Austin, 2008). Design guidelines may be found in USEPA's manual, *Constructed Wetlands Treatment of Municipal Wastewaters* (2000).

### 5.5.3 Evapotranspiration and Vegetative Uptake

Lined evapotranspiration beds and vegetative uptake are two other methods that have been promoted for nitrogen removal. Both rely on plants to either transpire the water and uptake nitrogen for incorporation into the plants. However, the loss of water through evapotranspiration leaves a nutrient and salts rich liquid that must be removed periodically to prevent toxic conditions for the plants. Also the plants must be continually harvested to remove the nutrients taken up from the system. Studies have found that nitrogen removal is achieved by these systems but that other systems perform as well or better in removing nitrogen from the wastewater (Atkins & Christensen, 2001; Barton, et al.,

2005; Taylor et al., 2006). While promoted heavily in the 1970's and early 1980's as an option for areas with slowly permeable soils or shallow water tables, evapotranspiration beds are infrequently used and seem to have been replaced by constructed wetlands. However, in southwestern states of the US they are primarily employed to reduce the hydraulic load on the drainfield (Rainwater, et al., 2005).

## 5.6 Modifications to Conventional Onsite Treatment Systems

Modifications to conventional OSTDS entail the in-situ addition of a permeable reactive media that supports denitrification through the release of carbon or electron donor. Wastewater (septic tank effluent) would initially pass through an unsaturated layer or zone (of sand for example), where nitrification occurs. Following passage through the unsaturated zone, the wastewater would pass through a permeable denitrification layer or zone. Denitrification media could be placed as an underlayment beneath the unsaturated soil, or as a subdivided treatment zone within a drainfield through which effluent from the aerobic zone must pass.

A patented method of rejuvenating ponded conventional septic tank drainfields using forced air also was found to enhance total nitrogen removal (Amador, et al., 2005; Amador, et al. 2007; Amador, et al., 2008 Potts, et al., 2004). In this method air is blown into the drainfield every 2 hours for 30 minutes. At traditional hydraulic loadings of septic tank effluent, 10 to 50 percent of the total nitrogen was found to be lost in the soil below the drainfield. When the hydraulic loading was increased, the total nitrogen reduction was increased up to 70 percent. The reason postulated for the increase was the increased organic carbon loading that prolonged the anoxic conditions favorable to biological denitrification. This method of operation was suggested to be similar to a sequencing batch reactor, which according to the investigators, would need regular attention if it were to be optimized for nitrogen removal.

Another approach to increasing the nitrogen reduction capacity of soil infiltration systems is to install permeable horizontal "barriers" consisting of cellulosic materials such as sawdust or woodchips below the systems, which provide reactive media for electron donors for denitrification (Robertson, et al., 2000; Robertson & Cherry, 1995). These barriers have a high water retention capacity to keep the media near saturation so that anoxic conditions are created as the septic tank effluent percolates through. An unsaturated layer of sand or other porous media is placed above the reactive barrier where the septic tank effluent is nitrified. Nitrogen reductions of 60 to 100 percent were achieved in four field trials.

Chang et al. (2009b) performed a comparative evaluation of two Florida drainfield sands (astatula sand and washed building sand) which received a common septic tank effluent

that had been pretreated in a recirculating sand filter. The total nitrogen in the influent to the septic tank was 46.1 mg/L. Suction lysimeters were employed to sample nitrogen levels at several depths in the drainfields. At the lowest sample depth of 24 in., total nitrogen concentrations were 9.6 and 5.7 mg/L respectively in astatula sand and washed building sand.

A modified drainfield design using a sulfur/limestone layer beneath a sand layer provided greater than 95 percent TN removal in laboratory scale columns receiving primary effluent from a municipal wastewater treatment plant (Shan and Zhang 1998). Nitrification occurred in the upper sand layer, and the lower denitrification layer was not maintained in a satu- rated condition.

A wood based system using a mixture of sand, wood chips, and tire crumb (85/11/4 percent by mass), was examined in bench scale columns to simulate treatment that would occur in a separate reactive media treatment zone established within a drainfield (Shah 2007). In this system, septic tank effluent would first pass through an unsaturated sand layer, and then through the treatment zone containing the reactive media. Laboratory column experiments with septic tank effluent supplied at a hydraulic residence time of 24 hours resulted in 98 percent TN removal. Average effluent ammonia and nitrate nitrogen concentrations were 4.4 and 0.05 mg/L, respectively.

Other studies, conducted in the laboratory for the most part, have demonstrated an increase in total nitrogen removal using modified drainfield designs with carbon substrates (usually wood chips or sawdust) or inorganic electron donors (elemental sulfur). The general concepts are similar to the drainfield modifications presented above. Issues of concern for modified drainfields include media longevity, replacement intervals, and hydraulic issues related to preferential flow paths. Replacement of in-situ denitrification media could require disturbing or removing the entire drainfield, so the life of the reactive media in the denitrification zone would need to be at least as long as the other drainfield components. However, Robertson and Vogan (2008) report that after 15 years of use, a barrier consisting of a mixture of sawdust and sand was still achieving denitrification of septic tank effluent.

## Section 6.0

### OSTDS Nitrogen

### Reduction Strategies in Florida

---

The goal of the *Florida Onsite Sewage Nitrogen Reduction Strategies Study* is to develop cost-effective strategies for nitrogen reduction by OSTDS. This first phase of the study provides a review and critical assessment of available literature on nitrogen reduction practices, treatment processes and existing technologies that appear suitable for use in individual home and small commercial onsite sewage treatment and disposal systems (OSTDS). The review catalogued well over 600 papers, proceedings, reports, and manufacturers' technical materials regarding existing and emerging technologies, which can be accessed on the database CD accompanying this report. A summary of the findings and recommendations for application of nitrogen reduction strategies in Florida are provided in this section. Supplements to this report include a technology classification scheme to allow comparisons of an array of technologies, a ranking scheme to allow relative rankings of technologies based on nitrogen reduction and treatment performance, system reliability and consistency, complexity of operation and maintenance, costs, aesthetics, and stage of development criteria, and a priority listing of the technologies for further testing and evaluation.

#### 6.1 Nitrogen Reducing Technologies

Many nitrogen reducing technologies are available for OSTDS applications. Most are based on well established treatment processes that have proven effective in municipal treatment applications. However, requirements for nitrogen reduction in sewage from individual homes and small commercial facilities are relatively new. Consequently, the capabilities of these systems to reduce nitrogen are not fully known. Available test results indicate that substantial variations exist between technologies.

#### 6.2 Categories of Technologies

To simplify evaluation, the available technologies were grouped by the treatment processes used to achieve nitrogen reduction. Four major categories were identified; source separation, biological nitrification/denitrification, physical/chemical, and natural systems. Each of these categories was broken down further based on distinct process variations within a group (see Figure 4-1).

The most prevalent nitrogen reduction processes used for onsite sewage treatment were found to be biological nitrification/denitrification and natural systems. Significant overlap exists between these two process types but because natural systems are not typically confined to treatment vessels but instead rely primarily on the natural assimilative capacity of the receiving environment over which control is limited, natural systems were given their own category. Biological nitrification/denitrification treatment processes are typically contained in treatment vessels, which allow access to observe and modify operation. Natural systems effect treatment from combinations of biochemical processes that occur within the soil matrix, evapotranspiration, and vegetative uptake. Constructed wetlands, which are designed based on mimicking ecological communities, are also included within this group.

Physical/chemical and source separation are the other two primary process groups, but these are used infrequently, if at all, for onsite sewage treatment. Physical/chemical processes, which do not rely on biological processes, are easier to control and are more consistent in treatment achieved but they require more operator attention and are more costly. Originally thought to be more effective for municipal treatment, they were mostly abandoned as biological processes became better understood and controlled.

Source separation on the other hand, is an emerging option for nitrogen removal. A promising practice is urine separation and recovery. Urine recovery can remove 70 to 80 percent of household generated nitrogen by installing urine separating toilets, which if the infrastructure for urine collection and use as fertilizer is developed offers an effective, reliable and easy to implement option that is low in cost compared to the other identified nitrogen reduction technologies. It also provides a readily available source of fertilizer rich in nitrogen and phosphorus.

### **6.3 Process Applications for OSTDS**

OSTDS technologies are available for most biological nitrification/denitrification and natural systems processes. The majority is proprietary units, but some public domain designs exist. Nearly all of the treatment technologies designed for nitrogen removal can achieve close to 50 percent total nitrogen reduction but as removal requirements increase, fewer technologies are available. For example, most mixed biomass (single stage) technologies are unable to consistently achieve stringent total nitrogen effluent concentration limits that are set to meet the drinking water standard of 10 mg/L or below before discharging to the soil. Below this limit, only segregated biomass (two stage) processes appear to be able to meet this requirement reliably.

o:\44237-001R007\Wpdocs\Report\Final

## 6.4 Process Performance Limitations

Two biological nitrification/denitrification processes are commonly used; mixed biomass (single stage) and segregated biomass (two stage). The single stage process is the most frequently used process because it relies on organic carbon in the wastewater to be the electron donor during denitrification as opposed to the two stage process, which requires an external source of organic carbon. The single stage process has been shown to achieve high removals of nitrogen in municipal wastewater treatment but this process does not perform as well in OSTDS. The reason for this seems to be that sufficient organic carbon is not reaching the denitrification stage in OSTDS thus limiting the amount of nitrogen reduction that can be achieved. This may be an inherent problem with most OSTDS that use the single stage process.

Carbon management is critical to mixed biomass nitrification/denitrification processes. Intermittent influent sewage flows with variable nitrogen content are common in OSTDS. This coupled with the conservative design flows prescribed by state rules leads to extended hydraulic residence times in which the wastewater is over aerated resulting in the excessive loss of organic carbon. This phenomenon can be seen in the performance of OSTDS that use different methods of carbon management in the system. Those OSTDS that rely on organic carbon released by dying microorganisms in the active biomass of the system typically achieve 40-60 percent total nitrogen removal while OSTDS that regularly recycle nitrified wastewater back to the anoxic septic tank to mix with organic carbon present in the raw wastewater typically achieve 60-80 percent total nitrogen reduction. Segregated biomass or two stage processes, which do not rely on organic carbon in the system but rather adds carbon to the denitrification stage from an external source, can achieve nearly complete removal of nitrate by metering the carbon into the denitrification reactor based on the nitrate concentration it receives. Sequencing batch reactors (SBR), which follow a mixed biomass process, should be able to manage the organic carbon better than most of the mixed biomass technologies because of the ability of SBR's to control the aeration and wastewater residence times in the treatment reactors. Limited data suggest that this does occur but to be able to perform to strict limits, operation requirements would increase dramatically. Because of the intermittent flows and need for increased surveillance, a segregated biomass (two stage) biological nitrification/denitrification process would be necessary where strict total nitrogen limits that require more than 70 percent removal prior to discharge to the soil.

Natural systems, which include the traditional OSTDS, also have inherent performance limitations. Application of septic tank effluent to unsaturated soil results in excellent cBOD and fecal coliform removals. However, nitrogen removals in traditional OSTDS are typically less than 40 percent. Siting requirements and design flows that are prescribed by the OSTDS rules are significant causes of the low removals. Soils with moderate to

high hydraulic permeability with unsaturated (vadose) zones several feet deep below the system infiltrative surface are favored by the rules. Such soils are well aerated, which provide efficient and nearly complete nitrification of the influent nitrogen, but as the result of the aerobic soil atmosphere, the vadose zone is unable to retain organic carbon. Slowly permeable soils, shallow organic soils, and soils with shallow perched saturated zones, which typically are not permitted for OSTDS would favor greater denitrification if nitrification were to be provided upstream of the infiltration system. Infiltration systems such as mound and at-grade systems, which are constructed above the ground surface with the soil's O and A horizons left intact, can provide both nitrification through the sand fill so that the organic layers below, if anoxic, can be used to supply electron donors for denitrification.

System design flows that are prescribed to be based on the size of the house to be served also create conditions that prevent mixed biomass technologies from achieving nitrogen removals greater than 50-70 percent. For the average home, the average daily flow is typically less than half the prescribed design flow. Studies have shown daily household flow to range between 150 and 230 gal/day with little weekly or seasonal variation (Thrasher, 1988; WEF, 2008; WERF, 2006). Using the inflated prescribed design flows in sizing system components results in excessive residence times in the treatment reactors, which causes over aeration and loss of carbon for denitrification. Two stage denitrification avoids this problem.

Timed dosing of septic tank effluent with drip dispersal is a method that can enhance nitrogen reduction because of the wetting and drying cycles that occur below the drip emitters as a result of the intermittent dosing. The alternating aerobic and anoxic soil conditions in the presence of the carbon rich septic tank effluent results in nitrification and denitrification. However, if the timed dosing is set for the daily flow prescribed by rule rather than the actual daily flow, nitrogen reduction will be less. Soil infiltration systems, particularly those that use drip dispersal, can also be constructed to create large "footprints" parallel to the lot's contours, which reduce the mass of nitrogen loading per square foot of area to avoid unacceptable concentrations in the underlying groundwater. Like any of the natural systems though, carbon management is problematic and because the discharges are below the ground surface, compliance monitoring is difficult and costly. Therefore OSTDS are usually only favored where strict nitrogen limits are not required.

## 6.5 Emerging Technologies

Few emerging technologies were identified in the literature. Most of those that were found have been variants to well-established processes such as various media for use in media filters or different component designs or applications. Others could be considered

new technologies for onsite treatment such as distillation or ion exchange but these technologies are early in their development stages and are not yet proven effective.

The most promising new technology is urine recovery. This method of nitrogen reduction is already practiced in Scandinavia where urine separating toilets are commercially available. Implementation of this method of nitrogen reduction would be highly effective and far less costly if the necessary servicing and urine reuse infrastructure could be built and public objections to the idea of urine recovery could be overcome or avoided. In addition to ease of use and lower costs, urine recovery also has the added benefit of reducing phosphorus discharges.

## **6.6 Nitrogen Reduction Implementation Strategies**

### **6.6.1 Establishing Nitrogen Reduction Standards**

The need for nitrogen reduction is not likely to be the same for all receiving environments. Therefore, because most nitrogen reduction options are more costly than traditional OSTDS, more complex, and require more attention to operate, the requirements for nitrogen reduction should be carefully considered. Attainment of end-of-pipe concentrations less than 10 mg-N/L are more costly and operation intensive than the traditional OSTDS.

An appropriate analysis procedure to evaluate risks to receiving environments should be developed to assign the appropriate treatment requirement and the variations around that standard that will be allowed. The recent report of a fresh water lake study indicated that limiting nitrogen additions to the lake where phosphorus was present did not result in a decrease in the rate of eutrophication because nitrogen-fixing cyanobacteria produced sufficient available nitrogen to allow biomass to be produced in proportion to the phosphorus in the lake (Shindler, et al., 2008). This study suggests that much is still to be learned about nitrogen impacts on water quality; therefore setting conservatively high nitrogen reduction standards might have less impact on water quality than anticipated.

In addition to establishing risk-based nitrogen reduction standards, the point of the standard's application can impact the choice of a nitrogen reducing technology. Several options exist. They can include the end-of-pipe prior to discharge to the soil, the point below the system that the percolate enters the groundwater, at a property boundary, and/or at a point of use, e.g. a well, or a surface water. End-of-pipe points of application deny further treatment that might be attained in the soil, which can add considerable construction and operating costs.

### 6.6.2 Technology Selection

The variety of available nitrogen reduction technologies and performance capabilities allows selection of a system design that will best meet the particular site conditions and nitrogen reduction requirements established for the area. For example, where the density of housing is low and far from high value surface waters, natural systems might be appropriate. If the soil underlying the system contains organic matter, nitrogen reduction achieved could be more than 75 percent (Briggs et al., 2007). If poorly drained, a component designed to nitrify the wastewater before discharging to the soil could be added. In areas where surface waters are not considered threatened but preventive measures are considered prudent, a technology using a mixed biomass nitrification/denitrification process that is capable of removing at least 50 percent might be most practical. In sensitive areas where protection of ground and surface waters is a high priority a two stage nitrification/denitrification process could be the only acceptable alternative.

### 6.6.3 Management and Enforcement

Implementation of nitrogen reduction technologies will expand the Department of Health's monitoring and enforcement operations and the owners' responsibilities toward their systems. Thought must be given to how nitrogen reduction standards are to be stated and how compliance monitoring is to be performed. Nitrogen reduction standards may be stated as concentration limits or as percent removals. Concentration standards will require water quality sampling to confirm compliance. Alternatively, standards stated as percent removal while less accurate are more flexible. Rather than water quality sampling, compliance could be based on proper technology selection (technologies with processes that are known to meet the desired removal) and routine maintenance and/or inspections to ensure the technology is functioning as intended. This latter approach to stating standards would likely be much less costly to monitor. If concentration standards are used, watershed monitoring rather than individual system monitoring to observe the aggregate impact of OSTDS water resources could be an effective alternative and a more accurate approach for compliance monitoring. Since impacts to watersheds have many sources and are tracked by multiple agencies, costs of monitoring could be shared between state and local water quality agencies. Regardless of the choices made, system performance and maintenance tracking, inspections, monitoring and enforcement procedures should be developed and available for deployment prior to permitting nitrogen reduction systems.

Needed service provider qualifications and certification programs and sufficient service provider capacity also should be developed before system implementation. A public awareness program is also needed. Without these programs, requirements for nitrogen reduction systems are not likely to achieve the intended goals.

## Section 7.0

### References

---

Abegglen, C., M. Ospelt, et al. (2008). "Biological nutrient removal in a small-scale MBR treating household wastewater." Water Research **42**(1-2): 338-346.

Ahn, Y.-H. (2006). "Sustainable nitrogen elimination biotechnologies: A review." Process Biochemistry **41**(8): 1709-1721.

Anderson, D. L., D. M. Mulville-Friel, et al. (1993). The Impact of Water Conserving Plumbing Fixtures on Residential Water Use Characteristics in Tampa, Florida. Conserv93 Conference, American Water Works Association.

Anderson, D. L. and R. L. Siegrist (1989). "The Performance of Ultra Low-Volume Flush Toilets in Phoenix." Journal of American Water Works **81**(3).

Aslan, S. and H. Cakici (2007). "Biological denitrification of drinking water in a slow sand filter." Journal of Hazardous Materials **148**(1-2): 253-258.

AyresAssociates (1998). Florida Keys Onsite Wastewater Nutrient Reduction Systems Demonstration Project - Phase II Addendum: Report to the Florida Department of Health Onsite Sewage Program: 28.

Barton, L., C. D. A. McLay, et al. (1999). "Denitrification rates in a wastewater irrigated forest in New Zealand." Journal of Environmental Quality **28**: 2008-2014.

Behrends, L. L., L. Houke, et al. (2007). ReCip® Water Treatment System with U.V. Disinfection for Decentralized Wastewater Treatment: Part II: Water Quality Dynamics. NOWRA 16th Annual Technical Education Conference & Exposition. Baltimore, Maryland, National Onsite Wastewater Recycling Association.

Benetto, E., D. Nguyen, et al. (2009). "Life cycle assessment of ecological sanitation system for small-scale wastewater treatment." Science of The Total Environment **407**(5): 1506-1516.

Borsuk, M., M. Maurer, et al. (2008). "Charting a Path for Innovative Toilet Technology Using Multicriteria Decision Analysis." Environ. Sci. Technol. **42**(6): 1855-1862.

Boyle, W. C., R. J. Otis, et al. (1994). Nitrogen removal from domestic wastewater in unsewered areas. On-Site Wastewater Treatment - Seventh International Symposium on Individual and Small Community Sewage Systems. E. Collins. Atlanta, Georgia, American Society of Agricultural Engineers. **7**: 485-498.

Bradley, P. M., J. M. Fernandez, et al. (1992). "Carbon limitation of denitrification rates in an anaerobic groundwater system." Environmental Science and Technology **26**(12): 2377-2381.

Bremner, J. M. and K. Shaw (1956). "Denitrification in soil: II. Factors affecting denitrification." Journal of Agricultural Science **51**: 40-52.

Briggs, G. R., E. Roeder, et al. (2007). Nitrogen Impact of Onsite Sewage Treatment and Disposal Systems in the Wekiva Study Area, Florida Department of Health, Bureau of Onsite Sewage Programs, Division of Environmental Health.

Brighton, W. (2007). Wastewater Alternatives Performance Summary.

Brown, K. W. and J. C. Thomas (1978). "Uptake of Nitrogen by Grass from Septic Fields in Three Soils." Agronomy Journal **70**.

Brown&Caldwell (1984). Residential Water Conservation Projects. Washington, D.C., US Department of Housing and Urban Development, Office of Policy Development.

Bruen, M. G. and R. J. Piluk (1994). Performance and costs of on-site recirculating sand filters. On-Site Wastewater Treatment - 7th International Symposium on Individual and Small Community Sewage Systems, Atlanta, Georgia, American Society of Agricultural Engineers.

Burford, J. R. and J. M. Bremner (1975). "Relationships between the denitrification capacities of soils and total, water-soluble, and readily decomposable soil organic matter." Soil Biology and Biochemistry **7**: 389-394.

Campos, J. L., S. Carvalho, et al. (2008). "Kinetics of denitrification using sulphur compounds: Effects of S/N ratio, endogenous and exogenous compounds." Bioresource Technology **99**(5): 1293-1299.

Chamchoi, N., S. Nitorisavut, et al. (2008). "Inactivation of ANAMMOX communities under concurrent operation of anaerobic ammonium oxidation (ANAMMOX) and denitrification." Bioresource Technology **99**(9): 3331-3336.

Chang, N., M. Wanielista, et al. (2009a). "Nutrient and Pathogen Removal with an Innovative Passive Underground Drainfield for On-site Wastewater Treatment." World Environmental and Water Resource Congress.

Chang, N., M. Wanielista, et al. (2009b). "Comparative Study of Two Standard Septic Tank Drainfields Using Different Sand with Recirculation for Nutrient Removal." World Environmental and Water Resource Congress.

Christensen, S., S. Simkins, et al. (1990). "Spatial variation in denitrification: Dependency of activity centers on the soil environment." Soil Science Society of America Journal **54**: 1608-1613.

Christopherson, S. H., J. L. Anderson, et al. (2001). Evaluation of recirculating sand filters in Minnesota. On-Site Wastewater Treatment - Ninth National Symposium on Individual and Small Community Sewage Systems. K. Mancl. Fort Worth, Texas, American Society of Agricultural Engineers. **9**: 207-214.

Cogger, C. G. (1988). "Onsite septic systems: The risk of groundwater contamination." Journal of Environmental Health **51**.

Cogger, C. G., L. M. Hajjar, et al. (1998). "Septic system performance on a coastal barrier island." Journal of Environmental Quality **17**: 401-408.

Converse, J. C. (1999). Nitrogen as it relates to onsite wastewater treatment with emphasis on pretreatment removal and profiles beneath dispersal units. 10th Northwest On-Site Wastewater Treatment Short Course, University of Washington, Seattle, WA, College of Engineering, University of Washington.

Cooke, R., A. Doheny, et al. (2001). Bio-reactors for edge-of-field treatment of tile outflow. 2001 ASAE Annual Meeting. Sacramento, CA American Society of Agricultural and Biological Engineers, St. Joseph, Michigan

Costa, J., G. Heufelder, et al. (2002). "Nitrogen removal efficiencies of three alternative septic technologies and a conventional septic system." Environment Cape Cod **5**(1): 15-24.

Crites, R. and G. Tchobanoglous (1998). Small Scale and Decentralized Wastewater Management Systems. Boston, MA, WCB/McGraw Hill.

Crites, R. W. (1985). "Nitrogen removal in rapid infiltration systems." Journal of Environmental Engineering Division, American Society of Civil Engineers **111**(6): 865-873.

CRWQCB (1997). Evaluation of Alternative Onsite Treatment Systems for the Removal of Nitrogen from Wastewater. Scaramento, CA, California Regional Water Quality Control Board, Central Coast Region.

CSWRCB (1995). California Greywater Guide. Sacramento, CA, California State Water Resources Control Board.

CSWRCB (2002). Evaluation of Alternative Onsite Treatment Systems for the Removal of Nitrogen from Wastewater. Scaramento, CA, California State Water Resources Control Board.

Darbi, A. and T. Viraraghavan (2003b). "A Kinetic Model for Autotrophic Denitrification using Sulphur: Limestone Reactors." Water Qual. Res. J. Canada **38**(1): 183-193.

Darbi, A., T. Viraraghavan, et al. (2003a). "Pilot-Scale Evaluation of Select Nitrate Removal Technologies." Journal of Environmental Science and Health Part A—Toxic/Hazardous Substances & Environmental Engineering **A38**(9): 1703-1715.

Darby, J. and H. Leverenz (2004). Virus, phosphorus, and nitrogen removal in onsite wastewater treatment processes, University of California Water Resources Center Technical Completion Reports (University of California, Multi-Campus Research Unit).

Davison, L., D. Pont, et al. (2006). "Dealing with nitrogen in subtropical Australia: Seven case studies in the diffusion of ecotechnological innovation." Ecological Engineering **28**(3): 213-223.

Degen, M. B., J. R. B. Reneau, et al. (1991). Denitrification in Onsite Wastewater Treatment and Disposal Systems. Blacksburg, Virginia, Virginia Polytechnic Institute and State University.

Della Rocca , C., V. Belgiorna, et al. (2005). "Cotton-supported heterotrophic denitrification of nitrate-rich drinking water with a sand filtration post-treatment." Water SA **31**(2): 229-236.

Dixon, A., D. Butler, et al. (2000). "Measurement and modelling of quality changes in stored untreated grey water." Urban Water **1**(4): 293-306.

Donahue, R. L., R. W. Miller, et al. (1983). Soils: An Introduction fo Soils and Plant Growth. Englewood Cliffs, NJ, Prentice Hall, Inc.

Doyle, J. D. and S. A. Parsons (2002). "Struvite formation, control and recovery." Water Research **36**(16): 3925-3940.

Doyle, J. D. and S. A. Parsons (2002a). "Struvite formation, control and recovery." Water Research **36**(16): 3925-3940.

Duncan, C. S., J. R. B. Reneau, et al. (1994). Impact of effluent quality and soil depth on renovation of domestic wastewater. On-Site Wastewater Treatment: Seventh International Symposium on Individual and Small Community Sewage Systems. E. Collins. Atanta, Georgia, American Society of Agricultural Engineers. **7**: 219228.

Dupuis, R., S. Rowland, et al. (2002). Nitrogen Removal Performance of Three Alternative On-Site Wastewater Treatment Systems in Montana. Helena, Montana, Department of Natural Resources and Conservation.

Ebeling, J., S. Tsukuda, et al. (2001). Evaluation and Real-Time Monitoring of a Recirculating Sand and Peat Filter. Third NSF International Symposium on Small Drinking Water and Wastewater Systems. Washington DC.

Eckenfelder, W. W. and Y. Argaman (1991). Principles of Biological and Physical/Chemical Nitrogen Removal. Phosphorus and Nitrogen Removal from Municipal Wastewater - Principles and Practice. R. Sedlak, Lewis Publishers: 3-42.

Eljamal, O., K. Jinno, et al. (2006). "Denitrification of Secondary Wastewater Using Sawdust."

Eljamal, O., K. Jinno, et al. (2008). "Modeling of Solute Transport with Bioremediation Processes using Sawdust as a Matrix." Water Air Soil Pollution **195**: 115-127.

Elmitwalli, T. A. and R. Otterpohl (2007). "Anaerobic biodegradability and treatment of grey water in upflow anaerobic sludge blanket (UASB) reactor." Water Research **41**(6): 1379-1387.

EPA (2007). "Innovative Technology Inventory (ITI) University of Waterloo NITREX TM." from [http://epa.gov/region1/assistance/ceit\\_iti/tech\\_cos/waterloo.html](http://epa.gov/region1/assistance/ceit_iti/tech_cos/waterloo.html).

Eriksson, E., H. R. Andersen, et al. (2008). "Greywater pollution variability and loadings." Ecological Engineering In Press, Corrected Proof.

Eriksson, E., K. Auffarth, et al. (2002). "Characteristics of grey wastewater." Urban Water **4**(1): 85-104.

Feng, D., Z. Wu, et al. (2008). "Nitrification of human urine for its stabilization and nutrient recycling." Bioresource Technology **99**(14): 6299-6304. Entration, mainly short nitrification (from NH<sub>4</sub>-N to NO<sub>2</sub>-N) occurred.

Flere, J. and T. Zhang (1998). "Sulfur-Based Autotrophic Denitrification Pond Systems for In-Situ Remediation of Nitrate-Contaminated Surface Water." Water Science and Technology **38**(1): 15-22.

Friedler, E., R. Kovalio, et al. (2005). "On-site greywater treatment and reuse in multi-storey buildings." Water Science and Technology **51**(10): 187-194.

Gable, J. E. and P. Fox (2000). Nitrogen removal during soil aquifer treatment by anaerobic ammonium oxidation (ANAMMOX), San Antonio, TX, Water Environment Federation and American Water Works Association.

Gambrell, R. P., J. W. Gilliam, et al. (1975). "Denitrification in Subsoils of the North Carolina Coastal Plain as Affected by Soil Drainage." J Environ Qual **4**(3): 311-316.

Ghafari, S., M. Hasan, et al. (2009). "Effect of carbon dioxide and bicarbonate as inorganic carbon sources on growth and adaptation of autohydrogenotrophic denitrifying bacteria." Journal of Hazardous Materials **162**(2-3): 1507-1513.

Ghisi, E. and D. F. Ferreira (2007). "Potential for potable water savings by using rainwater and greywater in a multi-storey residential building in southern Brazil." Building and Environment **42**(7): 2512-2522.

Gold, A. J., K. Addy, et al. (1999). Nitrate removal in shallow groundwater. 10th Northwest On-Site Wastewater Treatment Short Course and Equipment Exhibition.

R. W. Seabloom. University of Washington, Seattle, WA, College of Engineering, University of Washington.

Gold, A. J., B. E. Lamb, et al. (1992). "Wastewater Renovation in Buried and Recirculating Sand Filters." J Environ Qual **21**(4): 720-725.

Gold, A. J. and J. T. Sims (2000). A risk-based approach to nutrient contamination. National Research Needs Conference: Risk-Based Decision Making for Onsite Wastewater Treatment, Washington University, St. Louis, MO, Electric Power Research Institute, Palo Alto, CA.

Greenan, C., T. Moorman, et al. (2006). "Comparing Carbon Substrates for Denitrification of Subsurface Drainage Water " Journal of Environmental Quality **35**: 824-829.

Groffman, P. M., A. J. Gold, et al. (1992). "Nitrate Dynamics in Riparian Forests: Microbial Studies." J Environ Qual **21**(4): 666-671.

Gual, M., A. Moià, et al. (2008). "Monitoring of an indoor pilot plant for osmosis rejection and greywater reuse to flush toilets in a hotel." Desalination **219**(1-3): 81-88.

Günther, F. (2000). "Wastewater treatment by greywater separation: Outline for a biologically based greywater purification plant in Sweden." Ecological Engineering **15**(1-2): 139-146.

Habermeyer, P. and A. Sánchez (2005). "Optimization of the Intermittent Aeration in a Full-Scale Wastewater Treatment Plant Biological Reactor for Nitrogen Removal." Water Environment Research **77**(May/June): 229-233.

Hanson, G. C., P. M. Groffman, et al. (1994). "Denitrification in riparian wetlands receiving high and low groundwater nitrate inputs." Journal of Environmental Quality **23**.

Harkin, J. M., C. P. Duffy, et al. (1979). Evaluation of mound systems for purification of septic tank effluent. Madison, WI, University of Wisconsin Water Resources Center.

Harrison, J. A. (2003). "The Nitrogen Cycle: Of Microbes and Men " Visionlearning Vol. EAS-2 (4), 2003. from [http://www.visionlearning.com/library/module\\_viewer.php?mid=98](http://www.visionlearning.com/library/module_viewer.php?mid=98).

Healy, M. G., M. Rodgers, et al. (2007). "Performance of a stratified sand filter in removal of chemical oxygen demand, total suspended solids and ammonia nitrogen from high-strength wastewaters." Journal of Environmental Management **83**(4): 409-415.

Heufelder, G., S. Rask, et al. (2008). Performance of Innovative Alternative Onsite Septic Systems for the Removal of Nitrogen in Barnstable County, Massachusetts 1999-2007. Onsite Wastewater Management: Planning for the Future - 3rd Northeast Onsite Wastewater Treatment Short Course and Equipment Exhibition. Groton, Connecticut, New England Interstate Water Pollution Control Commission.

Hinkle, S. R., J. K. Böhlke, et al. (2008). "Mass balance and isotope effects during nitrogen transport through septic tank systems with packed-bed (sand) filters." Science of The Total Environment **407**(1): 324-332.

Horiba, Y., S. Khan, et al. (2005). "Characterization of the Microbial Community and Culturable Denitrifying Bacteria in a Solid-phase Denitrification Process Using Poly( $\epsilon$ -caprolactone) as the Carbon and Energy Source." Microbes Environ. **20**(1): 25-33.

Hotta, S. and N. Funamizu (2008). "Evolution of ammonification potential in storage process of urine with fecal contamination." Bioresource Technology **99**(1): 13-17.

Ikematsu, M., K. Kaneda, et al. (2007). "Electrochemical treatment of human urine for its storage and reuse as flush water." Science of The Total Environment **382**(1): 159-164.

Jaffer, Y., T. A. Clark, et al. (2002). "Potential phosphorus recovery by struvite formation." Water Research **36**(7): 1834-1842.

Jantrania, A. and M. Gross (2006). Advanced Onsite Wastewater Systems Technologies. Boca Raton, Florida, CRC Press/Taylor and Francis.

Jantrania, A. R., K. C. Sheu, et al. (1998). Performance evaluation of alternative systems - Gloucester, MA, demonstration project. On-site Wastewater Treatment - Eighth National Symposium on Individual and Small Community Sewage Systems. D. M. Sievers. Orlando, Florida, American Society of Agricultural Engineers. **8**: 480-489.

Jaynes, D., T. Kaspar, et al. (2002). Subsurface Drain Modifications to Reduce Nitrate Losses in Drainage. Annual Meeting, American Society of Agricultural Engineers.

Jefferson, B., J. E. Burgess, et al. (2001). "Nutrient addition to enhance biological treatment of greywater." Water Research **35**(11): 2702-2710.

Jin, Z., W. Li, et al. (2006). "Methods for nitrate removal from underground water." Technology of Water Treatment **32**(8): 34-37.

Jorgensen, T. C. and L. R. Weatherley (2003). "Ammonia removal from wastewater by ion exchange in the presence of organic contaminants." Water Research **37**(8): 1723-1728.

Justyna Czemieli Berndtsson (2006). "Experiences from the implementation of a urine separation system: Goals, planning, reality." Building and Environment **41**(4): 427-437.

Kaintz, R. F. and W. A. Snyder (2004). Performance evaluation of alternative on-site PA small flow treatment facilities in two state parks. On-Site Wastewater Treatment - Tenth National Symposium on Individual and Small Community Sewage Systems. K. R. Mankin. Sacramento, California, American Society of Agricultural Engineers. **10**: 318-324.

Kalyuzhnyi, S., M. Gladchenko, et al. (2006). "New anaerobic process of nitrogen removal." Water Science & Technology **54**(8): 163-170.

Kang, Y. W., K. M. Mancl, et al. (2007). "Treatment of turkey processing wastewater with sand filtration." Bioresource Technology **98**(7): 1460-1466.

Kanter, R. D., E. J. Tyler, et al. (1998). A denitrification system for domestic wastewater using sulfur oxidizing bacteria. On-Site Wastewater Treatment - Eighth National Symposium on Individual and Small Community Sewage Systems, Orlando, Florida, American Society of Agricultural Engineers.

Kietlinska, A. and G. Renman (2005). "An evaluation of reactive filter media for treating landfill leachate." Chemosphere **61**(7): 933-940.

Kim, H., E. Seagren, et al. (2003). "Engineered Bioretention for Removal of Nitrate from Stormwater Runoff." Water Environment Research **75**(4): 355-367.

Kim, J., Y. Hwang, et al. (2003). "Nitrification and denitrification using a single biofilter packed with granular sulfur." Water Science and Technology **47**(11): 153-156.

Kim, J., I. Song, et al. (2009). "A laboratory-scale graywater treatment system based on a membrane filtration and oxidation process -- characteristics of graywater from a residential complex." Desalination **238**(1-3): 347-357.

Koenig, A. and L. Liu (2002). "Use of limestone for pH control in autotrophic denitrification: continuous flow experiments in pilot-scale packed bed reactors." Journal of Biotechnology **99**(10/267630): 161-171.

Köiv, M., C. Vohla, et al. (2009). "The performance of peat-filled subsurface flow filters treating landfill leachate and municipal wastewater." Ecological Engineering **35**(2): 204-212.

Lacasse, R., G. Bélanger, et al. (2001). A Denitrification Process Based on a New Filtering Media for Onsite Wastewater Treatment. On-Site Wastewater Treatment, Proc. Ninth Natl. Symp. on Individual and Small Community Sewage Systems. K. Mancl. Fort Worth, Texas, USA, ASAE. **IX**: 235-244

Lamb, B., A. J. Gold, et al. (1987). Evaluation of nitrogen removal systems for on-site sewage disposal. On-Site Wastewater Treatment - Fifth National Symposium on Individual and Small Community Sewage Systems, Chicago, Illinois, American Society of Agricultural Engineers.

Larsen, T., I. Peters, et al. (2001). "Re-engineering the toilet for sustainable wastewater management." Environ. Sci. Technol. **35**(9): 192A–197A.

Lens, P. and G. Lettinga, Eds. (2001). Decentralized Sanitation and Reuse Concepts, Systems and Implementation. Integrated Environmental Technology Series. London, IWA Publishing.

Lens, P., G. Zeeman, et al., Eds. (2001). Decentralized Sanitation and Reuse Concepts, systems and implementation. Integrated Environmental Technology Series. London, IWA Publishing.

Leverenz, H., G. Tchobanoglous, et al. (2002). Review of Technologies for the Onsite Treatment of Wastewater in California, Center for Environmental and Water Resources Engineering, Department of Civil and Environmental Engineering, University of California, Davis, California.

Leverenz, H. L., G. Tchobanoglous, et al. (2009). "Clogging in intermittently dosed sand filters used for wastewater treatment." Water Research **43**(3): 695-705.

Li, W., Q.-I. Zhao, et al. (2009). "Sulfide removal by simultaneous autotrophic and heterotrophic desulfurization-denitrification process." Journal of Hazardous Materials **162**(2-3): 848-853.

Lienert, J. and T. Larsen (2006). "Considering User Attitude in Early Development of Environmentally Friendly Technology: A Case Study of NoMix Toilets." Environmental Science and Technology **40**(16): 4838-4844.

Lind, B.-B., Z. Ban, et al. (2000). "Nutrient recovery from human urine by struvite crystallization with ammonia adsorption on zeolite and wollastonite." Bioresource Technology **73**(2): 169-174.

Lind, B.-B., Z. Ban, et al. (2001). "Volume reduction and concentration of nutrients in human urine." Ecological Engineering **16**(4): 561-566.

Lindbo, D. and V. MacConnel (2001). Evaluation of a Peat Biofilter Treatment System. Ninth National Symposium on Individual and Small Community Sewage Systems, American Society of Agricultural and Biological Engineers.

Liu, Z., Q. Zhao, et al. (2008a). "Enhancing phosphorus recovery by a new internal recycle seeding MAP reactor." Bioresource Technology **99**(14): 6488-6493.

Liu, Z., Q. Zhao, et al. (2008b). "Urea hydrolysis and recovery of nitrogen and phosphorous as MAP from stale human urine." Journal of Environmental Sciences **20**(8): 1018-1024.

Liu, Z., Q. L. Zhao, et al. (2008c). "Comparison between complete and partial recovery of N and P from stale human urine with MAP crystallization." Journal of Environmental Engineering & Science **7**(3): 223-228.

Long, T. (1995). Methodology to predict nitrogen loading from on-site sewage treatment systems, University of Washington, Seattle, WA, College of Engineering, University of Washington, Seattle, WA.

Loomis, G., D. Dow, et al. (2004). Long-term Treatment Performance of Innovative Systems. On-Site Wastewater Treatment X, American Society of Agricultural and Biological Engineers.

Loudon, T. L., T. R. Bounds, et al. (2004). Nitrogen Removal And Other Performance Factors In Recirculating Sand Filters On-Site Wastewater Treatment X. e. Richard

Cooke. Sacramento, California, American Society of Agricultural and Biological Engineers. **X**: 451-459.

Loudon, T. L., T. R. Bounds, et al. (2005). Nitrogen Removal and Other Performance Factors in Recirculating Sand Filters. 13th NW Onsite Wastewater Treatment Short Course. R. W. Seabloom. University of Washington, Seattle, WA, College of Engineering, University of Washington.

Lowe, K. S., N. k. Rothe, et al. (2006). Influent Constituent Characteristics of the Modern Waste Stream from Single Sources: Literature Review, Water Environment Research Foundation.

Luostarinen, S. A. and J. A. Rintala (2005). "Anaerobic on-site treatment of black water and dairy parlour wastewater in UASB-septic tanks at low temperatures." Water Research **39**(2-3): 436-448.

MacQuarrie, K., E. Sudicky, et al. (2001). "Numerical simulation of a fine-grained denitrification layer for removing septic system nitrate from shallow groundwater." Journal of Contaminant Hydrology **52**: 29-55.

Magid, J., A. M. Eilersen, et al. (2006). "Possibilities and barriers for recirculation of nutrients and organic matter from urban to rural areas: A technical theoretical framework applied to the medium-sized town Hillerød, Denmark." Ecological Engineering **28**(1): 44-54.

Mah, D. Y. S., C. H. J. Bong, et al. (2009). "A conceptual modeling of ecological greywater recycling system in Kuching City, Sarawak, Malaysia." Resources, Conservation and Recycling **53**(3): 113-121.

Mahmood, Q., P. Zheng, et al. (2007). "Anoxic sulfide biooxidation using nitrite as electron acceptor." Journal of Hazardous Materials **147**(1-2): 249-256.

Makropoulos, C. K., K. Natsis, et al. (2008). "Decision support for sustainable option selection in integrated urban water management." Environmental Modelling & Software **23**(12): 1448-1460.

Matos, C. T., A. M. Sequeira, et al. (2009). "Nitrate removal in a closed marine system through the ion exchange membrane bioreactor." Journal of Hazardous Materials **In Press, Corrected Proof**.

Maurer, M., W. Pronk, et al. (2006). "Treatment processes for source-separated urine." Water Research **40**(17): 3151-3166.

Mayer, P. W., W. B. DeOreo, et al. (1999). Residential End Uses of Water. A. W. W. R. Foundation. Denver, CO.

McCarthy, B., S. Monson Geerts, et al. (2001). Performance of a Textile Filter, Polishing Sand Filter and Shallow Trench System for the Treatment of Domestic Wastewater at the Northeast Regional Correction Center. Duluth, MN, Natural Resources Research Institute University of Minnesota – Duluth, 5013 Miller Trunk Highway, Duluth, MN 55811: 28.

McKee, J. A. and J. L. Brooks (1994). Peat filters for on-site wastewater treatment, Atlanta, Georgia, American Society of Agricultural Engineers.

Memon, F. A., Butler, D., Ed. (2005). Water Demand Management. IWA Publishing. London, IWA Publishing.

Mergaert, J., A. Boley, et al. (2001). "Identity and Potential Functions of Heterotrophic Bacterial Isolates from a Continuous-Upflow Fixed-Bed Reactor for Denitrification of Drinking Water with Bacterial Polyester as Source of Carbon and Electron Donor." Systematic and Applied Microbiology **24**: 303-310.

Misra, R. K. and A. Sivongxay (2009). "Reuse of laundry greywater as affected by its interaction with saturated soil." Journal of Hydrology **366**(1-4): 55-61.

Monson Geerts, S., B. McCarthy, et al. (2001a). Performance of Pre-engineered Modular Peat Filters for the Treatment of Domestic Wastewater at the Northeast Regional Correction Center. Duluth, MN, Natural Resources Research Institute

Monson Geerts, S., B. McCarthy, et al. (2001b). Performance of Peat Filters in the Treatment of Domestic Wastewater in Minnesota. On-Site Wastewater Treatment - Ninth National Symposium on Individual and Small Community Sewage Systems K. Mancl. Fort Worth, Texas, ASABE. **9**: 295-304

Moon, H. S., S. W. Chang, et al. (2006). "Effect of reactive media composition and co-contaminants on sulfur-based autotrophic denitrification." Environmental Pollution **144**(3): 802-807.

Moon, H. S., D. Y. Shin, et al. (2008). "A long-term performance test on an autotrophic denitrification column for application as a permeable reactive barrier." Chemosphere **73**(5): 723-728.

Mote, C. R. and E. E. Ruiz (1994). Design and operating criteria for nitrogen removal in a recirculating sand filter, Atlanta, Georgia, American Society of Agricultural Engineers.

Mueller, W., A. Sperandio, et al. (1985). "Denitrification with mineralizable substrates as carriers in advanced waste-water purification." Landwirtsch Forsch **38**(1-2): 132-138.

Mulholland, M. R., N. G. Love, et al. (2007). Bioavailability of Organic Nitrogen from Treated Wastewater, Chesapeake Bay Program.

Nelson, W. M., A. J. Gold, et al. (1995). "Spatial and temporal variation in groundwater nitrate removal in a riparian forest." Journal of Environmental Quality **24**.

Nolde, E. (1999). "Greywater reuse systems for toilet flushing in multi-storey buildings - over ten years experience in Berlin." Urban Water **1**(4): 275-284.

NSF-International (2003e). Waterloo Biofilter Environmental Technology Verification Statement, NSF International, Ann Arbor, Michigan.

NSF-International (2005). Aerocell Model ATS-SCAT-8-AC-C500 Standard 40-Residential Wastewater Treatment Systems, NSF International, Ann Arbor, Michigan.

NSF-International (2006). Bio-Coir Model ATS-SCAT-8-BC-C500 NSF-ANSI Standard 40-Residential Wastewater Treatment Systems.

Nugroho, R., H. Takanashi, et al. (2002). "Denitrification of industrial wastewater with sulfur and limestone packed column." Water Science and Technology **46**(11-12): 99-104.

Oakley, S. (2005). Design and Operation Issues for Onsite Nitrogen Removal.

Osesek, S., B. Shaw, et al. (1994). Design and optimization of two recirculating sand filter systems for nitrogen removal, Atlanta, Georgia, American Society of Agricultural Engineers.

Otis, R. J. (2007). Estimates of Nitrogen Loadings to Groundwater from Onsite Wastewater Treatment Systems in the Wekiva Study Area. Nitrogen Impact on Onsite Sewage Treatment and Disposal Systems in the Wekiva Study Area. G. R. Briggs, E. Roeder and E. Ursin, Florida Department of Health, Bureau of Onsite Sewage Systems.

Ottoson, J. and T. A. Stenström (2003). "Faecal contamination of greywater and associated microbial risks." Water Research **37**(3): 645-655.

Ovez, B. (2006a). "Batch biological denitrification using *Arundo donax*, *Glycyrrhiza glabra*, and *Gracilaria verrucosa* as carbon source." Process Biochemistry **41**(6): 1289-1295.

Ovez, B., S. Ozgen, et al. (2006b). "Biological denitrification in drinking water using *Glycyrrhiza glabra* and *Arundo donax* as the carbon source." Process Biochemistry **41**: 1539-1544.

Palmquist, H. and J. Hanæus (2005). "Hazardous substances in separately collected grey- and blackwater from ordinary Swedish households." Science of The Total Environment **348**(1-3): 151-163.

Parkin, T. B. and J. J. Meisinger (1989). "Denitrification below the crop rooting zone as influenced by surface tillage." Journal of Environmental Quality **18**.

Pastor, L., D. Mangin, et al. (2008). "A pilot-scale study of struvite precipitation in a stirred tank reactor: Conditions influencing the process." Bioresource Technology **99**(14): 6285-6291.

Patterson, R. and M. Brennan (2006). Economics of Using Local Peat for Biofiltration of Domestic Wastewater in New Zealand. New Zealand Land Treatment Collective 2006 Annual Conference Proceedings. Nelson, New Zealand.

Patterson, R., K. Davey, et al. (2001). Peat Bed for On-Site Treatment of Septic Tank Effluent. Conference On-site '01. Advancing On-site Wastewater Systems. University of New England, Armidale.

Patterson, R. A. (2004). Effective Treatment of Domestic Effluent with a Peat Biofilter - A Case Study at Tingha On-Site Wastewater Treatment X. R. Cooke. Sacramento, California, American Society of Agricultural and Biological Engineers. **X**: 526-536.

Paul, J. W. and B. J. Zebarth (1997). "Denitrification and nitrate leaching during the fall and winter following dairy cattle slurry application." Canada Journal of Soil Science **77**: 2313-2340.

Philip, H. and J. Vasel (2006). Filtre compact Eparco pour l'Assainissement non Collectif.

Pidou, M., L. Avery, et al. (2008). "Chemical solutions for greywater recycling." Chemosphere **71**(1): 147-155.

Pilot, L. and J. W. H. Patrick (1972). "Nitrate reduction in soils: New perspectives, new recommendations." Journal of Environmental Health **51**: 196-200

Piluk, R. J. and E. C. Peters (1994). Small recirculating sand filters for individual homes, Atlanta, Georgia, American Society of Agricultural Engineers.

Pronk, W., H. Palmquist, et al. (2006). "Nanofiltration for the separation of pharmaceuticals from nutrients in source-separated urine." Water Research **40**(7): 1405-1412.

Pronk, W., S. Zuleeg, et al. (2007). "Pilot experiments with electro dialysis and ozonation for the production of a fertilizer from urine." Water Sci. Technol. **56**(5): 219-227.

Ramona, G., M. Green, et al. (2004). "Low strength graywater characterization and treatment by direct membrane filtration." Desalination **170**(3): 241-250.

Rauch, W., D. Brockmann, et al. (2003). "Combining urine separation with waste design: an analysis using a stochastic model for urine production." Water Research **37**(3): 681-689.

Reneau, R. B., Jr. (1977). "Changes in Inorganic Nitrogenous Compounds from Septic Tank Effluent in a Soil with a Fluctuating Water Table." J Environ Qual **6**(2): 173-178.

Renman, A., L. D. Hylander, et al. (2008). "Transformation and removal of nitrogen in reactive bed filter materials designed for on-site wastewater treatment." Ecological Engineering **34**(3): 207-214.

Rich, B. (2007). "La Pine National Demonstration Project." La Pine National Demonstration Project Innovative Onsite Wastewater Treatment Systems. from

<http://www.deschutes.org/deq/>  
<http://www.deschutes.org/deq/innovative.htm>.

Richardson, E. E., A. T. Hanson, et al. (2004). Improving the nitrogen removal efficiency of recirculating sand filters. On-Site Wastewater Treatment X - Tenth National Symposium on Individual and Small Community Sewage Systems. K. R. Mankin. Sacramento, California, American Society of Agricultural Engineers. **10**: 288-297.

Ritter, W. F. and R. P. Eastburn (1988). "A Review of Denitrification in Onsite Wastewater Treatment Systems." Environmental Pollution **51**.

Robertson, W., G. Ford, et al. (2005). "Wood-Based Filter for Nitrate Removal in Septic Systems." Transactions of the ASAE **48**(1): 121-128.

Robertson, W. D., D. W. Blowes, et al. (2000). "Long-Term Performance of In Situ Reactive Barriers for Nitrate Remediation." Ground Water **38**(5): 689-695.

Robertson, W. D. and J. A. Cherry (1995). "In Situ Denitrification of Septic-System Nitrate Using Reactive Porous Media Barriers: Field Trials." Ground Water **33**(1): 99-111.

Rock, C., J. Brooks, et al. (1984). "Use of peat for on-site wastewater treatment: I. Laboratory evaluation." Journal of Environmental Quality

Rodgers, M., M. G. Healy, et al. (2005). "Organic carbon removal and nitrification of high strength wastewaters using stratified sand filters." Water Research **39**(14): 3279-3286.

Rodgers, M., A. Lambe, et al. (2006). "Carbon and nitrogen removal using a novel horizontal flow biofilm system." Process Biochemistry **41**(11): 2270-2275.

Ronayne, M. P., R. C. Paeth, et al. (1982). Oregon On-site Experimental Systems Program: Final Report to U.S. Environmental Protection Agency, Oregon Department of Environmental Quality.

Ronteltap, M., M. Maurer, et al. (2007a). "Struvite precipitation thermodynamics in source-separated urine." Water Research **41**(5): 977-984.

Ronteltap, M., M. Maurer, et al. (2007b). "The behaviour of pharmaceuticals and heavy metals during struvite precipitation in urine." Water Research **41**(9): 1859-1868.

Rossi, L., J. Lienert, et al. (2009). "Real-life efficiency of urine source separation." Journal of Environmental Management **90**(5): 1909-1917.

Roy, C. and J. P. Dube (1994). A recirculating gravel filter for cold climates, Atlanta, Georgia, American Society of Agricultural Engineers.

Saidou, H., A. Korchef, et al. (2009). "Struvite precipitation by the dissolved CO<sub>2</sub> degasification technique: Impact of the airflow rate and pH." Chemosphere **74**(2): 338-343.

Samatya, S., N. Kabay, et al. (2006). "Removal of nitrate from aqueous solution by nitrate selective ion exchange resins." Reactive and Functional Polymers **66**(11): 1206-1214.

Sandy, A. T., W. A. Sack, et al. (1987). Enhanced nitrogen removal using a modified recirculating sand filter (RSF2). On-Site Wastewater Treatment - Fifth National Symposium on Individual and Small Community Sewage Systems. K. Mancl. Chicago, Illinois, American Society of Agricultural Engineers. **5**: 161-170.

Sarioglu, M., G. Insel, et al. (2009). "Modeling Nitrogen Removal Performance of a Membrane Bioreactor under Dissolved Oxygen Dynamics." Environmental Engineering Science **26**(0): 1-13.

Schäfer, A. I., L. D. Nghiem, et al. (2006). "Bisphenol A retention in the direct ultrafiltration of greywater." Journal of Membrane Science **283**(1-2): 233-243.

Schönning, C., R. Leeming, et al. (2002). "Faecal contamination of source-separated human urine based on the content of faecal sterols." Water Research **36**(8): 1965-1972.

Sengupta, S. and S. Ergas (2006). Autotrophic Biological Denitrification with Elemental Sulfur or Hydrogen for Complete Removal of Nitrate-Nitrogen from a Septic System Wastewater.

Sengupta, S. and S. J. Ergas (2006). "Autotrophic Biological Denitrification with Elemental Sulfur or Hydrogen for Complete Removal of Nitrate-Nitrogen from a Septic System Wastewater."

Sengupta, S., S. J. Ergas, et al. (2007). "Investigation of Solid-Phase Buffers for Sulfur-Oxidizing Autotrophic Denitrification." Water Environment Research **79**: 2519-26.

Shah, T. (2007). Fate of Nitrogen and Phosphorus Species from a Black and Gold™ Nugget Mix in a Laboratory Column Simulated Septic Tank Drainfield. Department of Civil and Environmental Engineering. Orlando, University of Central Florida. **MS**: 91.

Shan, J. and T. Zhang (1998). Septic Tank Effluent Denitrification with Sulfur/Limestone Processes. Proceedings of the 1998 Conference on Hazardous Waste Research.

Sherman, K. (2006). Introducing a New Media for Fixed-film treatment in Decentralized Wastewater Treatment Systems. NOWRA's 15th Annual Technical Education Conference. Denver, Colorado, National Onsite Wastewater Recycling Association.

Sherman, K. (2007). Using Natural Media Filters in a Distributed Wastewater System Serving and Ecotourism-Oriented Developmen. NOWRA's 16th Annual Technical Education Conference and International Program, Baltimore, Maryland, National Onsite Wastewater Recycling Association.

Siegrist, R. L. and P. D. Jenssen (1989). Nitrogen Removal During Wastewater Infiltration as Affected by Design and Environmental Factors. 6th Northwest On-Site Wastewater Treatment Short Course. R. W. Seabloom. University of Washington, Seattle, WA, College of Engineering, University of Washington.

Sierra-Alvarez, R., R. Beristain-Cardoso, et al. (2007). "Chemolithotrophic denitrification with elemental sulfur for groundwater treatment." Water Research **41**(6): 1253-1262.

Simmons, R. C., A. J. Gold, et al. (1992). "Nitrate Dynamics in Riparian Forests: Groundwater Studies." J Environ Qual **21**(4): 659-665.

Singer, M. J. and D. N. Munns (1991). Soils: An Introduction. New York, NY, MacMillan Publishing Co.

Smith, D. (2006). Hillsborough Filter Pilot Demnstration Final Report, Hillsborough County, Florida.

Smith, D. (2008). Florida Passive Nitrogen Removal Study Additional Monitoring. Thonotosassa, Florida, Applied Environmental Technology.

Smith, D. P., R. J. Otis, et al. (2008). Florida Passive Nitrogen Removal Study - Final Report to the Florida Department of Health, Applied Environmental Technology, Thonotosassa, Florida.

St. Marseille, J. and B. Anderson (2002). "Use of leaching chambers for on-site sewage treatment." Environmental Technology **23**(3): 261-272.

Starr, J. L. and R. W. Gillham (1993). "Denitrification and organic carbon availability in two aquifers." Ground Water **31**(6).  
denitrification aquifer

Stewart, L. W. and R. B. Reneau (1988). "Shallowly Placed Low Pressure Distribution System to Treat Domestic Wastewater in Soils with Fluctuating High Water Tables." Journal of Environmental Quality **17**(3).

Stolt, M. H. and J. R. B. Reneau (1991). Potential for contamination of ground and surface waters from on-site wastewater disposal systems. Blacksburg, VA, Virginia Polytechnic Institute and State University, Blacksburg, VA.

Stratful, I., M. D. Scrimshaw, et al. (2001). "Conditions influencing the precipitation of magnesium ammonium phosphate." Water Research **35**(17): 4191-4199.

Talbot, P., D. Pettigrew, et al. (2006). Coconut mesocarp-based biofilter material and its use in wastewater treatment. U. S. P. Office, Premier Tech. **7,097,768**.

Tchobanoglous, G., F. Burton, et al. (2003). Wastewater Engineering, Treatment and Reuse. Boston, McGraw-Hill Higher Education.

Tidåker, P., C. Sjöberg, et al. (2007). "Local recycling of plant nutrients from small-scale wastewater systems to farmland--A Swedish scenario study." Resources, Conservation and Recycling **49**(4): 388-405.

Tsukuda, S., J. Ebeling, et al. (2004). Real-time Monitoring of Recirculating Sand and Peat Filters. Tenth National Symposium on Individual and Small Community Sewage Systems, American Society of Agricultural and Biological Engineers.

Tucholke, M. B., J. E. McCray, et al. (2007). Variability in Denitrification Rates: Literature Review and Analysis. NOWRA's 16th Annual Technical Education

Conference and International Program. Baltimore, Maryland, National Onsite Wastewater Recycling Association.

Udert, K. M., T. A. Larsen, et al. (2003a). "Urea hydrolysis and precipitation dynamics in a urine-collecting system." Water Research **37**(11): 2571-2582.

Udert, K. M., T. A. Larsen, et al. (2003b). "Urea hydrolysis and precipitation dynamics in a urine-collecting system." Water Research **37**(11): 2571-2582.

Udert, K. M., T. A. Larsen, et al. (2003a). "Estimating the precipitation potential in urine-collecting systems." Water Research **37**(11): 2667-2677.

Urynowicz, M. A., W. C. Boyle, et al. (2007). "Nitrogen Removal in Recirculating Sand Filter Systems with Upflow Anaerobic Components." Journal of Environmental Engineering **133**(5).

USDA (1962). Soil Survey Manual, USDA Handbook No. 18, USDA, Washington, D.C.

soil denitrification

USEPA (1993). Nitrogen Control. Washington, D.C., U.S. Environmental Protection Agency, Office of Research and Development and Office of Water.

USEPA (2002). Onsite Wastewater Treatment Systems Manual, EPA/625/R-00/008. Washington, D.C., US Environmental Protection Agency, Office of Water, Office of Research and Development.

Vallino, J. and K. Foreman (2007). "Effectiveness of Reactive Barriers for Reducing N-Loading to the Coastal Zone CICEET Progress Reports." from [http://ciceet.unh.edu/progressreports/2006/9\\_2006/vallino04/](http://ciceet.unh.edu/progressreports/2006/9_2006/vallino04/) [http://ciceet.unh.edu/progressreports/2007/3\\_2007/vallino04/](http://ciceet.unh.edu/progressreports/2007/3_2007/vallino04/)

van Driel, P., W. Robertson, et al. (2006). "Denitrification of Agricultural Drainage Using Wood-Based Reactors." Transactions of the ASABE **49**(2): 565-573.

van Voorthuizen, E., A. Zwijnenburg, et al. (2008). "Biological black water treatment combined with membrane separation." Water Research **42**(16): 4334-4340.

van Voorthuizen, E. M., A. Zwijnenburg, et al. (2005). "Nutrient removal by NF and RO membranes in a decentralized sanitation system." Water Research **39**(15): 3657-3667.

Venhuizen, D., J. H. Wiersma, et al. (1998). Washington Island Project: Evolution of the denitrifying sand filter concept, Orlando, Florida, American Society of Agricultural Engineers.

Vinnerås, B. and H. Jönsson (2002a). "Faecal separation for nutrient management--evaluation of different separation techniques." Urban Water **4**(4): 321-329.

Wakatsuki, T., H. Esumi, et al. (1993). "High performance and N & P-removable on-site domestic waste water treatment system by multi-soil-layering method." Water Science & Technology **27**(1): 31-40.

Walker, W. G., J. Bouma, et al. (1973). "Nitrogen Transformations During Subsurface Disposal of Septic Tank Effluent in Sands: II. Ground Water Quality." J Environ Qual **2**(4): 521-525.

Wallace, S. and D. Austin (2008). "Emerging Models for Nitrogen Removal in Treatment Wetlands." Journal of Environmental Health **71**(4): 10-16.

WDOH (2005). Nitrogen Reducing Technologies for Onsite Wastewater Treatment Systems - Report to the Puget Sound Action Team. Olympia, Washington, Wastewater Management Program, Washington State Department of Health.

WERF (2008). "Nutrient Removal "Limit of Technology"."

Wert, S. and R. Path (1985). Performance of Drainfield Trenches Charged with Recirculating Sand Filter Effluent. %th Northwest On-Site Wastewater Treatment Short Course. D. Lenning and B. Seabloom. University of Washington, Seattle, WA, Department of Civil Engineering and Environmental Health, University of Washington: 166-181.

WHO, Ed. (2006). Guidelines for the safe use of wastewater, excreta and greywater Geneva, Switzerland, World Health Organization, 20, Avenue Appia, 1211, Geneva, 27 Switzerland, 92-4-154684-0.

Widiastuti, N., H. Wu, et al. (2008). "The potential application of natural zeolite for greywater treatment." Desalination **218**(1-3): 271-280.

Wilsenach, J. and M. C. M. v. Loosdrecht (2006). "Integration of Processes to Treat Wastewater and Source-Separated Urine." Journal of Environmental Engineering **132**(3).

Wilsenach, J. A., C. A. H. Schuurbijs, et al. (2007). "Phosphate and potassium recovery from source separated urine through struvite precipitation." Water Research **41**(2): 458-466.

Winker, M., F. Tettenborn, et al. (2008b). "Comparison of analytical and theoretical pharmaceutical concentrations in human urine in Germany." Water Research **42**(14): 3633-3640.

Winkler, E. S. and P. L. M. Veneman (1991). A denitrification system for septic tank effluent using sphagnum peat moss. On-Site Wastewater Treatment - Sixth National Symposium on Individual and Small Community Sewage Systems. J. C. Converse. Chicago, Illinois, American Society of Agricultural Engineers. **6**: 155-164.

Winward, G. P., L. M. Avery, et al. (2008a). "A study of the microbial quality of grey water and an evaluation of treatment technologies for reuse." Ecological Engineering **32**(2): 187-197.

Winward, G. P., L. M. Avery, et al. (2008b). "Chlorine disinfection of grey water for reuse: Effect of organics and particles." Water Research **42**(1-2): 483-491.

Wisconsin, U. o. (1978). Management of Small Waste Flows, University of Wisconsin-Madison.

Woods, C., H. Bouwer, et al. (1999). Study finds biological nitrogen removal in soil aquifer treatment system offers substantial advantages. WEFTEC'99. New Orleans, LA, Water Environment Federation.

Wren, A. L., R. L. Siegrist, et al. (2004). Field performance evaluation of textile filter units employed in onsite wastewater treatment systems, Sacramento, California, American Society of Agricultural Engineers.

Xu, Z.-x., L. Shao, et al. (2009). "Biological Denitrification Using Corncobs as a Carbon Source and Biofilm Carrier " Water Environment Research **81**(3): 242-247.

Yetilmezsoy, K. and Z. Sapci-Zengin (2009). "Recovery of ammonium nitrogen from the effluent of UASB treating poultry manure wastewater by MAP precipitation as a slow release fertilizer." Journal of Hazardous Materials **166**: 260-269.

Zeng, H. and T. Zhang (2005). "Evaluation of kinetic parameters of a sulfur–limestone autotrophic denitrification biofilm process." Water Research **39**(20): 4941-4952.

Zhang, B.-h., D.-y. Wu, et al. (2007). "Simultaneous removal of ammonium and phosphate by zeolite synthesized from coal fly ash as influenced by acid treatment." Journal of Environmental Sciences **19**(5): 540-545.

Zhang, T. (2002). "Nitrate Removal in Sulfur: Limestone Pond Reactors " Journal of Environmental Engineering **128**(1): 73-84.

Zhang, T. (2004). Development of Sulfur-Limestone Autotrophic Denitrification Processes for Treatment of Nitrate-Contaminated Groundwater in Small Communities. Champaign, Illinois, Midwest Technology Assistance Center (MTAC), Illinois State Water Survey: 46.

Zhang, T. C. and H. Zeng (2006). "Development of a Response Surface for Prediction of Nitrate Removal in Sulfur--Limestone Autotrophic Denitrification Fixed-Bed Reactors." Journal of Environmental Engineering **132**(9): 1068-1072.

## Appendix A: Glossary

---

**Active nitrogen removal system:** An onsite treatment system effecting nitrogen reduction in the effluent that is not considered passive because it contains aerator pumps, more than one effluent pump, or no reactive media

**ATU:** Aerobic treatment unit, as specified in 64E-6.012 FAC

**Conventional drainfield material:** Gravel as specified in 64E-6.014(5) FAC

**Conventional System:** Standard septic tank and drainfield to treat wastewater on-site that does not perform advanced treatment.

**DOH:** Florida Department of Health or the department

**FAC:** Florida Administrative Code

**Media:** Material that effluent from a septic tank or pretreatment device passes through prior to reaching the groundwater. This may include soil, sawdust, zeolites, tire crumbs, vegetative removal, sulfur, spodosols, or other media.

**OSTDS:** Onsite Sewage Treatment and Disposal System

**Passive:** A type of onsite sewage treatment and disposal system that excludes the use of aerator pumps and includes no more than one effluent dosing pump with mechanical and moving parts and uses a reactive media to assist in nitrogen removal.

**PBTS:** Performance Based Treatment System, a type of OSTDS that has been designed to meet specific performance criteria for certain wastewater constituents as defined by 64E-6.025(10) FAC

**Reactive media:** Media that reacts with wastewater to reduce nitrogen concentrations.

**TN:** Total Nitrogen concentration in a water sample (mg/L).



# Florida Onsite Sewage Nitrogen Reduction Strategies Study

Task A

**Prioritization of Nitrogen Reduction Technologies**

**Draft Report**

June 2009

44237-001

**HAZEN AND SAWYER**  
Environmental Engineers & Scientists

In association with



**AET**  
Applied Environmental Technology

**OTIS  
ENVIRONMENTAL  
CONSULTANTS, LLC**

# Florida Onsite Sewage Nitrogen Reduction Strategies Study

## TASK A DRAFT REPORT

### Prioritization of Nitrogen Reduction Technologies

#### Prepared for:

Florida Department of Health  
Division of Environmental Health  
Bureau of Onsite Sewage Programs  
4042 Bald Cypress Way Bin #A-08  
Tallahassee, FL 32399-1713

FDOH Contract CORCL

**June 2009**

#### Prepared by:

**HAZEN AND SAWYER**  
Environmental Engineers & Scientists

In Association With:

**AET**  
Applied Environmental Technology

**Otis Environmental  
Consultants, LLC**



## **Prioritization of Nitrogen Reduction Technologies**

---

Prioritization of nitrogen reduction technologies was based on systematic application of the ranking criteria to the technologies identified in the literature review conducted in Task A.1. Technologies were grouped according to the classification scheme developed in Task A.3. Each technology classification received individual scores for the separate evaluation criterion, and the weighting criteria were used to generate the total score for the technology classification. The technologies within each classification were prioritized according to their total score.

### **List of Technologies**

The literature review and survey of manufacturers indicated that many processes and commercial systems are available for onsite wastewater treatment. The technology database is comprised of available onsite nitrogen reduction technologies from manufacturers and the literature review. The identified technologies were sorted according to the major classifications developed in Task A.3: source separation, biological treatment, physical/chemical treatment and natural systems. The basis for assignment of classification was the principal nitrogen reduction process of the technology. The systems within the major groupings were then further grouped into the process variations within each major classification.

### **Technology Evaluation Criteria**

The technology evaluation criteria were individually discussed and edited, and a final consensus list of criteria was agreed to and adopted during the Technology Classification, Ranking and Prioritization Workshop held with the Research Review and Advisory Committee on May 28, 2009. Also agreed to and adopted at that meeting were the weighting factors for each individual criterion. The finalized criteria and weighting factors are listed in Table 1.1.

o:\44237-001R006\Wpdocs\Report\Draft

**Table 1.1  
Technology Criteria and Weighting Factor**

<b>Criteria</b>	<b>Weighting Factor</b>
Effluent Nitrogen Concentration	11
Performance Reliability	10
Performance Consistency	9
Construction Cost	7.5
Operation and Maintenance Cost	7
Energy Requirement	7
Construction Complexity	5
Operation Complexity	5
Land Area Required	4.5
BOD/TSS Effluent Concentration	3.5
Restoration of Performance	3.5
System Aesthetics	2
Stage of Technology Development	0.5

For each of the individual technologies identified within the literature review (Task A.1), data were acquired from a wide variety of sources focusing on the ranking criteria. Manufacturer's information and third party test results such as the NSF International (NSF) Standard 40 Protocol, EPA Environmental Technology Verification Program (ETV), or field and/or laboratory evaluations reported in the technical literature were utilized to develop the technology database. Some performance data were available only as manufacturer's claims, other data as a range of removal percentages from field installations, and some data included detailed analytical results with statistical ranges. Nitrogen effluent data were generally available while nitrogen influent data were not. The attributes of the performance consistency and performance reliability criteria were based on the type of treatment process used. Construction cost was estimated for a newly installed, complete treatment system for a three-bedroom home in Florida, and included primary treatment (i.e. septic tank) and a conventional drainfield. Performance reliability data were available for a few systems for which frequency of maintenance visits recorded were available, and estimated for the remainder. Energy use data (kW-h/day or kW-h/year) were available for a few systems that detailed a cost per month or cost per year, and estimated for the others. For energy use, a conversion to uniform data values was obtained by using an assumption of \$0.10 per kW-h. Operation and maintenance cost estimates, land area required, constructional complexity, operational complexity, and system aesthetics data were very limited, so professional judgment were used to assign scores for individual criteria to the technology classifications.

o:\4237-001R006\Wpdocs\Report\Draft

## Criteria Scores

For each of the thirteen criteria, scores were established based on cost and/or non-cost attributes. Table 1.2 presents a summary of score assignments for each criterion. The criterion assignments were the basis for scoring and ranking of the technology classifications.

**Table 1.2  
Criteria Scores**

Criteria Number	Criteria	Score				
		1	2	3	4	5
1	Effluent Nitrogen Concentration (mg-N/L)	> 30	16 – 30	11 – 15	3 – 10	< 3
2	Performance Reliability	Monthly		Quarterly	Semi-Annually	Annually
3	Performance Consistency	Activated Sludge Nite/Denite	IFAS	MBR/IMB	Fixed Film	Physical /Chemical & Source Separation
4	Construction Cost <sup>1</sup> (\$1,000's)	>20	15-20	10-15	5-10	<5
5	Operation and Maintenance Cost <sup>2</sup> (\$/year)	>500	400-500	300-400	200-300	<200
6	Energy Requirement (kW-h/year)	>2500	1500-2500	1000-1500	500-1000	<500
7	Construction Complexity	Complex installation, specialized training, sophisticated electrical and controls knowledge req., master septic tank contractor		Some specialized knowledge and training required		Simple to install by any Contractor

o:\14237-001R006\Wpdocs\Report\Draft

**Table 1.2  
Criteria Scores**

Criteria Number	Criteria	Score				
		1	2	3	4	5
8	Operation Complexity	Complex operation with operator training required; Scheduled visits by manufacturer's representative required > quarterly		Some specialized operator training required; Scheduled visits by manufacturer's representative required twice per year		Simple operation with limited operator requirements annual scheduled visit
9	Land Area Required <sup>3</sup> (ft <sup>2</sup> )	>2000	1001-2000	501-1000	251-500	<250
10	BOD/TSS Effluent Concentration (mg/L)	>50	30/30		20/20	10/10
11	Restoration of Performance	Activated Sludge Nite/Denite	IFAS	MBR/IMB	Fixed Film	Physical /Chemical & Source Separation
12	System Aesthetics	Not Acceptable		Perceived Nuisance/ Displeasing		Acceptable
13	Stage of Technology Development	Conceptual	Experimental	Demonstration	State Use	National Use

1. Construction cost assumes a standard septic tank cost of \$2000 and drainfield cost of \$4500 installed.
2. Operation and maintenance cost includes inspections, annual operating permit fee (\$100), and maintenance entity, but it does not include power costs.
3. Land area is for a new entire system, and assumed standard septic tank 50 SF and drainfield 400 SF.

o:\4237-001R006\Wpdocs\Report\Draft

The criteria were developed with the full knowledge that data for many of the criteria would be sparse and difficult to attain. Good engineering judgment and experience with various types of systems were used to develop technology ranking scores when data were not available. A summary of the individual criterion scores for technology classifications is presented in Table 1.3. While the table encompasses the full range of possible systems contained in our classification, types of systems lacking available data are left blank. Natural systems need to be considered separately and are therefore summarized in Table 1.5.

**Table 1.3  
Criteria Scores for Physical/Chemical  
and Biological Technology Classifications**

Technology Classification	Criteria													Total Scores (Weighting Factor *Score)
	1	2	3	4	5	6	7	8	9	10	11	12	13	
	Effluent of TN Conc. (mg/L)	Performance Reliability	Performance Consistency	Construction Costs (\$1000)	O&M Cost	Energy Req. (kW-h/yr)	Construction Complexity	Operation Complexity	Land Area Req. (ft <sup>2</sup> )	BOD/TSS Effluent Conc (mg/L)	Restoration of Performance	System Aesthetics	Stage of Technology Development	
<b>Weighting Factor</b>	11.0	10.0	9.0	7.5	7.0	7.0	5.0	5.0	4.5	3.5	3.5	2.0	0.5	
<b>Physical/Chemical</b>														
Membrane Processes														
Ion Exchange														
Evapotranspiration														
<b>Biological</b>														
Mixed Biomass														
Suspended Growth: w/, w/out recycle	3	3	1	2	2	2	3	3	3	4	1	5	5	188.5
Fixed Film														
Fixed Film with recycle	2	4	4	2	3	4	3	3	3	5	4	5	5	235.5

**Table 1.3  
Criteria Scores for Physical/Chemical  
and Biological Technology Classifications**

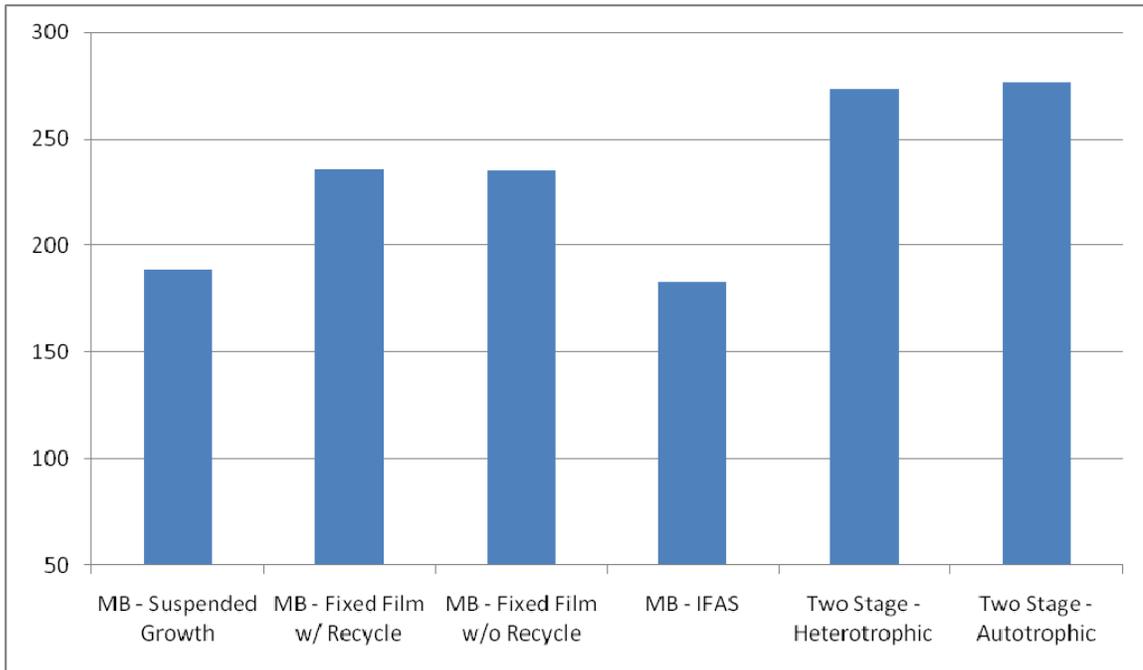
Technology Classification	Criteria													Total Scores (Weighting Factor *Score)
	1	2	3	4	5	6	7	8	9	10	11	12	13	
	Effluent of TN Conc. (mg/L)	Performance Reliability	Performance Consistency	Construction Costs (\$1000)	O&M Cost	Energy Req. (kW-h/yr)	Construction Complexity	Operation Complexity	Land Area Req. (ft <sup>2</sup> )	BOD/TSS Effluent Conc (mg/L)	Restoration of Performance	System Aesthetics	Stage of Technology Development	
Fixed Film without recycle	1	4	4	2	4	3	3	3	3	4	4	5	5	235
Integrated Fixed Film Activated Sludge	2	3	2	2	2	1	3	3	3	4	2	5	5	183
Two Stage (Segregated Biomass)														
Heterotrophic Denitrification	4	4	4	2	3	4	3	3	3	4	4	5	3	273
Autotrophic Denitrification	4	5	4	2	3	2	3	5	3	5	4	5	3	276.5

For each technology classification, the criterion scores (Table 1.3) were multiplied by the weighting factor (Table 1.1) and summed to generate a total score. The total score was used to rank technology classifications. Total scores for physical/chemical and biological technology classifications are listed in Table 1.4 and plotted in Figure 1-1.

o:\14237-001R006\Wpdocs\Report\Draft

**Table 1.4**  
**Physical/Chemical and Biological Technology Classification Overall Ranking**

<b>Technology Classification</b>	<b>Total Score</b>	<b>Overall Ranking</b>
Two Stage (Segregated Biomass) – Autotrophic Denitrification	276.5	1
Two Stage (Segregated Biomass) – Heterotrophic Denitrification	273.0	2
Mixed Biomass – Fixed Film with Recycle	235.5	3
Mixed Biomass – Fixed Film without Recycle	235.0	4
Mixed Biomass – Suspended Growth	188.5	5
Mixed Biomass – Integrated Fixed Film Activated Sludge	183.0	6



**Figure 1-1: Overall Ranking of Physical/Chemical and Biological Technology Classifications**

The top ranked technology classifications (1 & 2) were biological systems with two stage segregated biomass employing autotrophic and heterotrophic denitrification. These systems are passive, require little operator attention, and provide high reliability. The total scores for autotrophic and heterotrophic denitrification technologies in two stage segre-

o:\4237-001R006\Wpdocs\Report\Draft

gated biomass systems were sufficiently close that they were considered essentially equal.

The third and fourth ranked technology classifications were mixed biomass fixed film biological systems with recycle and without recycle, respectively. The total scores for these systems were sufficiently close that they were considered essentially equal. These technology classifications have the stability advantages that are inherent in fixed film processes.

Mixed biomass suspended growth systems were the fifth ranked technology classification and mixed biomass integrated fixed film systems were the sixth. These systems employ suspended growth basins and exhibit higher effluent nitrogen concentrations and lower performance consistency and reliability.

**Table 1.5  
Criteria Scores for Natural  
System Technology Classifications**

Technology Classification	Criteria													Total Scores (Weighting Factor *Score)
	1	2	3	4	5	6	7	8	9	10	11	12	13	
	Effluent of TN Conc. (mg/L)	Performance Reliability	Performance Consistency	Construction Costs (\$1000)	O&M Cost	Energy Req. (kW-h/yr)	Construction Complexity	Operation Complexity	Land Area Req. (ft <sup>2</sup> )	BOD/TSS Effluent Conc (mg/L)	Restoration of Performance	System Aesthetics	Stage of Technology Development	
<b>Weighting Factor</b>	11.0	10.0	9.0	7.5	7.0	7.0	5.0	5.0	4.5	3.5	3.5	2.0	0.5	
<b>Natural</b>														
Soil Infiltration														
With dosing	1	5	4	5	4	5	5	5	5	5	4	5	5	305
With reactive barriers	5	5	4	3	3	5	3	4	5	5	3	5	3	316.5

o:\4237-001R006\Wpdocs\Report\Draft

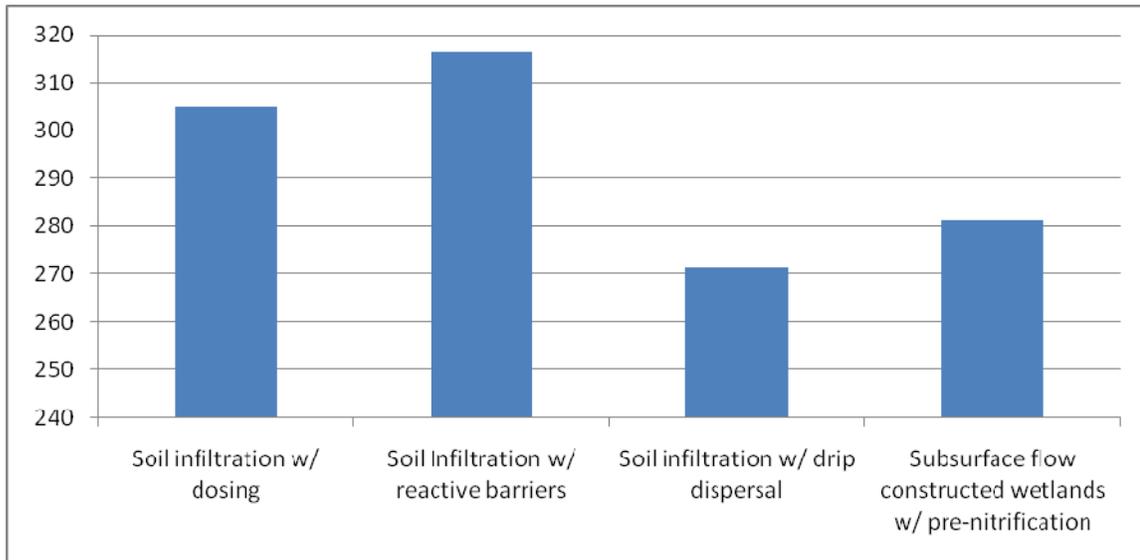
**Table 1.5  
Criteria Scores for Natural  
System Technology Classifications**

Technology Classification	Criteria													Total Scores (Weighting Factor *Score)
	1	2	3	4	5	6	7	8	9	10	11	12	13	
	Effluent of TN Conc. (mg/L)	Performance Reliability	Performance Consistency	Construction Costs (\$1000)	O&M Cost	Energy Req. (kW-h/yr)	Construction Complexity	Operation Complexity	Land Area Req. (ft <sup>2</sup> )	BOD/TSS Effluent Conc (mg/L)	Restoration of Performance	System Aesthetics	Stage of Technology Development	
With drip dispersal	3	4	4	4	3	5	3	3	5	5	4	5	5	271.5
Heterotrophic Nitrification / Denitrification														
Annamox														
Constructed Wetlands														
Subsurface flow with pre-denitrification	3	5	4	3	4	5	3	3	4	3	3	5	5	281.5
<b>Source Separation</b>														
Urine Recovery														
Wastes Segregation														

o:\14237-001R006\Wpdocs\Report\Draft

**Table 1.6  
Natural System Technology Classification Overall Ranking**

<b>Technology Classification</b>	<b>Total Score</b>	<b>Overall Ranking</b>
Soil Infiltration with reactive barriers	316.5	1
Soil Infiltration with dosing	305.0	2
Constructed Wetlands subsurface flow with pre-denitrification	281.5	3
Soil Infiltration with drip dispersal	271.5	4



**Figure 1-2: Overall Ranking of Natural System Technology Classifications**

The top ranked natural system was soil infiltration with reactive barriers. The second ranked natural system is traditional trench drainfield with timed dosing of septic tank effluent. However, this system received the lowest treatment score. Application of our ranking system to certain kinds of natural systems can be misleading from a purely quantitative perspective: in this instance, the score is high because of its passive characteristics and low operating costs, but does not address the difficulty of performance monitoring and the costs associated with correcting poor performance.

Subsurface-flow constructed wetlands with pre-nitrification and drip dispersal of septic tank effluent to soil infiltration technologies ranked within 3.5% of each other. The constructed wetlands can achieve more complete nitrification and denitrification than soil

infiltration with drip dispersal, but drip dispersal offers much greater control of performance and repairs of malfunctions are less costly and easier to perform. Aesthetically, the systems scored the same, but the acceptance could be quite different among property owners.

It is important to note that the natural systems should not be quantitatively compared, using these ranking criteria, to the groups of proprietary and non-proprietary biological systems detailed in Tables 1.3 and 1.4, and Figure 1-1. Primary among considerations supporting this division of technologies is the need to consider separately the elements of each system that performs treatment. The soil infiltration units utilize the soil's ecology and physical characteristics to perform treatment and all relevant data measures the treatment capacity within the soil pedon to reduce nitrogen. However, it must be kept in mind that the vast majority of proprietary systems also discharge to the soil. In order to be able to rank each technology fairly, only the nitrogen reduction components were considered. Moreover, management of non-soil based technologies, though more expensive, is simplified because the units can be operated effectively to adjust to varying conditions and serviced easily, which may not be the case with soil-based nitrogen reduction technologies. When malfunctions occur with soil-based technologies, repairs may be necessary and could lead to expensive reconstruction. When the latter is necessary, available land area can become a severe constraint. Finally, while soils provide good treatment over a broad range of conditions, variability of characteristics among soil units can be large creating significant uncertainty in predicting a soil's nitrogen reduction capacity.

### **Recommendations for Testing**

The technology classification ranking provides the basis from which to formulate recommendations for the field testing to be conducted in Task B of the Florida Onsite Sewage Nitrogen Reduction Strategies Study. The criteria used to consider in establishing priorities for testing include representation of several technology classifications, nitrogen effluent performance data, similarity of technologies, and maturity level of technologies. The purpose of prioritization is to select the more promising technologies that may not have sufficient prior testing or may be differently configured to improve performance, and to avoid duplicate testing where substantial experience already exists. The priority list for Task B testing is listed in Table 1.7. The recommended technologies include mixed biomass, two stage segregated biomass biofiltration systems, natural systems with and without external sources of electron donors for denitrification, fixed film and integrated fixed film activated sludge processes, denitrification filters with reactive media as post-treatment to commercial aerobic treatment processes, onsite elimination of urine effluent, and urine separation and recovery.

**Table 1.7  
Technologies Recommended for Testing in Task B**

<b>System</b>	<b>Technology</b>	<b>Comment</b>
<b>1</b>	Two stage (segregated biomass) system: Stage 1: Biofiltration with recycle (nitrification) Stage 2: Autotrophic denitrification with reactive media biofilter	<ul style="list-style-type: none"> <li>• Top ranked system capable of meeting the lowest TN concentration standard</li> <li>• Suitable for new systems or retrofit</li> </ul>
<b>2</b>	Two stage (segregated biomass) system: Stage 1: Biofiltration with recycle (nitrification) Stage 2: Heterotrophic denitrification with reactive media biofilter	<ul style="list-style-type: none"> <li>• Top ranked system capable of meeting the lowest TN concentration standard</li> <li>• Suitable for new systems or retrofit</li> </ul>
<b>3</b>	Natural system: Septic tank/Mound with in-situ reactive media layer	<ul style="list-style-type: none"> <li>• Lower cost natural system that is untested but appears capable of achieving 75-78% TN removal before reaching groundwater</li> <li>• Suitable for new systems or replacing existing systems at end of useful life</li> </ul>
<b>4</b>	Natural system: Settled or secondary effluent with drip dispersal	<ul style="list-style-type: none"> <li>• Suitable for reducing TN impacts on groundwater through enhanced TN removal and reduced TN loading on soil</li> <li>• Suitable for new systems or retrofit</li> </ul>
<b>5</b>	Two stage (segregated biomass) system: Stage 1: Mixed biomass fixed film with recycle Stage 2: Heterotrophic denitrification with reactive media biofilter	<ul style="list-style-type: none"> <li>• High performance aerobic treatment with anoxia for enhanced TN removal followed by second stage heterotrophic denitrification for high nitrogen removal</li> <li>• Suitable for new systems or nitrogen reduction upgrades</li> </ul>
<b>6</b>	Two stage (segregated biomass) system: Stage 1: Mixed biomass fixed film with recycle Stage 2: Autotrophic denitrification with reactive media biofilter	<ul style="list-style-type: none"> <li>• High performance aerobic treatment with anoxia for enhanced TN removal followed by second stage autotrophic denitrification for meeting low TN concentration standard</li> <li>• Suitable for new systems or nitrogen reduction upgrades</li> </ul>
<b>7</b>	Mixed biomass integrated fixed film activated sludge system: Suspended growth with recycle	<ul style="list-style-type: none"> <li>• High performance aerobic treatment</li> <li>• Suitable for new systems or nitrogen reduction upgrades</li> </ul>

o:\14237-001R006\Wpdocs\Report\Draft

**Table 1.7  
Technologies Recommended for Testing in Task B**

<b>System</b>	<b>Technology</b>	<b>Comment</b>
<b>8</b>	Mixed biomass integrated fixed film activated sludge system: Moving bed bioreactor	<ul style="list-style-type: none"> <li>• High performance aerobic treatment with simultaneous denitrification</li> <li>• Suitable for new systems or nitrogen reduction upgrades</li> </ul>
<b>9</b>	Mixed biomass suspended growth system: Suspended growth sequencing batch reactor	<ul style="list-style-type: none"> <li>• Aerobic treatment</li> <li>• Suitable for new systems or nitrogen reduction upgrades</li> </ul>
<b>10</b>	Membrane process system: Membrane bioreactor (MBR)	<ul style="list-style-type: none"> <li>• Suitable for new systems or nitrogen reduction upgrades</li> </ul>
<b>11</b>	Source separation system: Dry toilet (evaporative or composting)	<ul style="list-style-type: none"> <li>• Eliminates liquid disposal or wastes</li> </ul>
<b>12</b>	Source separation system: Urine separating (recovery) toilet	<ul style="list-style-type: none"> <li>• Innovative system that is capable of removing 70-80% of the household TN at little capital cost</li> <li>• Provides potential for sustainable recovery of nutrients</li> </ul>

The first two technologies listed in Table 1.7 are two stage segregated biomass. The first stage of each is a recirculating biofilter through which nitrification occurs. Significant denitrification also occurs due to the recirculation. The biofilters can employ a variety of fixed film media, many of which are in current use and are described in the literature review in Task A.1. PNRS II testing will provide additional data for biofiltration with recycle using clinoptilolite, expanded clay, and polystyrene. The best performing media from PNRS II testing will be recommended for Task B testing. Stage 2 of these segregated biomass systems will employ autotrophic denitrification (System 1) and heterotrophic denitrification (System using reactive media biofilters). The hybrid Systems 1 & 2 can be employed for new installations or inserted between primary treatment (i.e. septic tank) and soil dispersal in existing systems.

System 3 is a natural system that uses drip dispersal into the soil of settled or secondary effluent. To enhance denitrification, an in-situ reactive media barrier will be constructed below the drip dispersal tubing. Effluent is dispersed within the root zone and percolates downward through the reactive media barrier containing high water retention materials such as expanded clay and lignocellulosic or elemental sulfur electron donors to support heterotrophic or autotrophic denitrification. This system would meet the FDOH definition of passive technology and has the potential to be a low cost in-situ system that can be applied for new installations or retrofits.

o:\4237-001R006\wpdocs\Report\Draft

System 4 is a natural system using drip dispersal of settled or secondary effluent into the soil. By dosing septic tank effluent into the soil on timed cycles alternating aerobic and anoxic conditions are created in the soil near each emitter, which creates the necessary conditions for nitrification/denitrification to occur. This intermittent dosing of septic tank effluent has been shown by several studies to reduce the total nitrogen applied.

Systems 5 and 6 are similar to Systems 1 and 2, in that it is a mixed biomass fixed film system with recycle, followed by a heterotrophic or autotrophic denitrification filter. While Systems 1 and 2 utilize various widely available media, System 5 and 6 consist of a combination of different proprietary and non-proprietary media systems. As with most systems intended for nitrogen removal, recycling is used to treat effluent more than once before discharge.

Systems 7 and 8 are IFAS (Integrated Fixed-Film Activated Sludge) systems. They combine elements of both fixed film and suspended growth microbial communities, resulting in highly stable treatment processes that achieve more reliable and consistent performance than other mixed biomass processes.

System 9 is a suspended growth system, specifically Sequencing Batch Reactors (SBR). Theoretically, SBR's should be able to control the loss of carbon better than other mixed biomass systems.

System 10 is a membrane bioreactor (MBR) which combines suspended growth with a membrane filtration unit. MBR is an emerging treatment option for single family home systems.

Systems 11 and 12 are source separation systems. Source separation is an emerging option for treatment, likely to become increasingly prevalent in keeping with trends towards sustainability and resource recovery. With regard to nitrogen removal, source separation has the potential to be a particularly efficient option since 50 to 75% of household waste nitrogen is from urine. Accordingly, separating the waste streams allows for more efficient, dedicated treatment options.



# Florida Onsite Sewage Nitrogen Reduction Strategies Study

## Task C

### Literature Review of Nitrogen Reduction by Soils and Shallow Groundwater

#### Draft Report

June 2009

44237-001

**HAZEN AND SAWYER**  
Environmental Engineers & Scientists

In association with



**AET**  
Applied Environmental Technology

**OTIS  
ENVIRONMENTAL  
CONSULTANTS, LLC**

# Florida Onsite Sewage Nitrogen Reduction Strategies Study

## TASK C DRAFT REPORT

### Literature Review of Nitrogen Reduction by Soils and Shallow Groundwater

#### Prepared for:

Florida Department of Health  
Division of Environmental Health  
Bureau of Onsite Sewage Programs  
4042 Bald Cypress Way Bin #A-08  
Tallahassee, FL 32399-1713

FDOH Contract CORCL

**June 2009**

#### Prepared by:

**HAZEN AND SAWYER**  
Environmental Engineers & Scientists

In Association With:





**List of Figures**

Figure 1-1 Nitrogen Processes Occurring in a Typical OWTS..... 1-4  
(Heatwole and McCray 2007)

Figure 2-1 Isotopic Fingerprinting of Nitrate Sources..... 2-7  
(McQuillan 2004)

Figure 2-2 Source Contribution to Nitrate Impacts ..... 2-9  
(Briggs, Roeder et al. 2007)

o:\44237-001\F008\Wpdocs\Report\Draft



## Section 1.0 Introduction

---

### 1.1 Project Background

As a result of the widespread impacts of nitrogen on groundwater and surface waters in Florida, the management of nitrogen sources, including onsite wastewater treatment systems (OWTS), is of paramount concern for the protection of the environment. As part of Task C of the Florida Onsite Sewage Nitrogen Reduction Strategies (FOSNRS) Study, a review of available research related to the fate and transport of nitrogen is being developed. The primary objectives of this review are to:

- Assess the current available information on nitrogen treatment in soils and the effects to the receiving groundwater;
- Develop a searchable database of available literature concerning nitrogen groundwater contamination and OWTS;
- Assist in the conceptual understanding of the fate and transport processes that influence distribution of nitrogen in groundwater; and
- Guide future field evaluation efforts and provide additional information to the development of a modeling tool for simulation of nitrogen in groundwater (Task D).

The following presents a literature review to assess the current state-of-knowledge regarding the fate and transport of nitrogen and its movement and distribution in groundwater related to OWTS. The review will identify existing studies and reports that examine the influence of OWTS-derived nitrogen inputs, the transformative processes that impact nitrate distribution, and the key factors that result in a significant effect to groundwater quality from OWTSs. As part of the literature review, a database of the references was developed in conjunction with this summary report. This database (see separate Excel file “CSM\_C-1 Nitrogen Soil-GW Studies”) includes a summary table of the relevant features and parameters of each modeling study. As a result of the large number of identified sources, some studies that were deemed as not valuable to this effort and are mentioned in this report, but are not described in detail and the reader is directed to the database for further information.

o:\44237-001R008\Wpdocs\Report\Draft

## 1.2 Nitrogen in Ground Water; Conceptual Considerations

Nitrogen is an important concern for water quality and nitrates represent perhaps the most common groundwater pollutant. Animals, crops, ecosystems, and human health can be adversely impacted by the presence of nitrogen in water supplies. Of these concerns, nitrate impacts to human health have been a primary consideration. The consumption of nitrates has been linked to various illnesses, including cyanosis in infants and some forms of cancer. As a result, in the United States, a maximum allowable nitrate concentration of 10 mg/L as N has been established as protective of human health (Canter 1996). Other agencies around the world have also established such standards for nitrates in groundwater.

Also of concern are the environmental effects on groundwater and surface water that can result from nitrogen impacts. The degradation of groundwater quality can ultimately lead to the degradation of surface waters in watershed systems that have strong groundwater/surface water interactions. Nitrogen that enters surface water bodies via these interactions can lead to algal blooms and eutrophication. These processes lead to oxygen depletion in surface waters which can be harmful to natural aquatic life. In Florida, the protection of watersheds, in particular surface water bodies, has led to the legislation of protection of these areas (i.e., the Wekiva River Protection Act).

A survey of community service wells and private domestic wells performed by the U.S. Environmental Protection Agency (EPA) indicated that over half of these water supply wells contained detectable levels of nitrate (Canter 1996). The sources of this contamination are various, and include agricultural and domestic fertilizer applications, natural sources, wastewater treatment applications, and the use of OWTS. The last category is often of concern, as nearly 25% of the population in the U.S. and 30% of all new development utilize OWTS (Lowe et al., 2007). In Florida, nearly a third of all house-holds are serviced by OWTS and 92% of water supplies come from groundwater (Briggs et al. 2007, Lowe et al. 2007).

Due to the unique features of the geology and hydrogeology, the groundwater systems and ultimately ecological systems and human health may be adversely impacted by nitrogen contamination of groundwater. The geology in Florida is characterized by the presence of sinkholes and fractures that develop in the karst limestone prevalent in many areas (Briggs, Roeder et al. 2007). These features tend to act as preferential flow-paths that can contribute to widespread groundwater contamination and potentially can impact protected surface waters.

Nitrogen transport in the subsurface is a complex process, especially when considering the nitrogen inputs from OWTS. Meeting the objectives of the FOSNRS project therefore

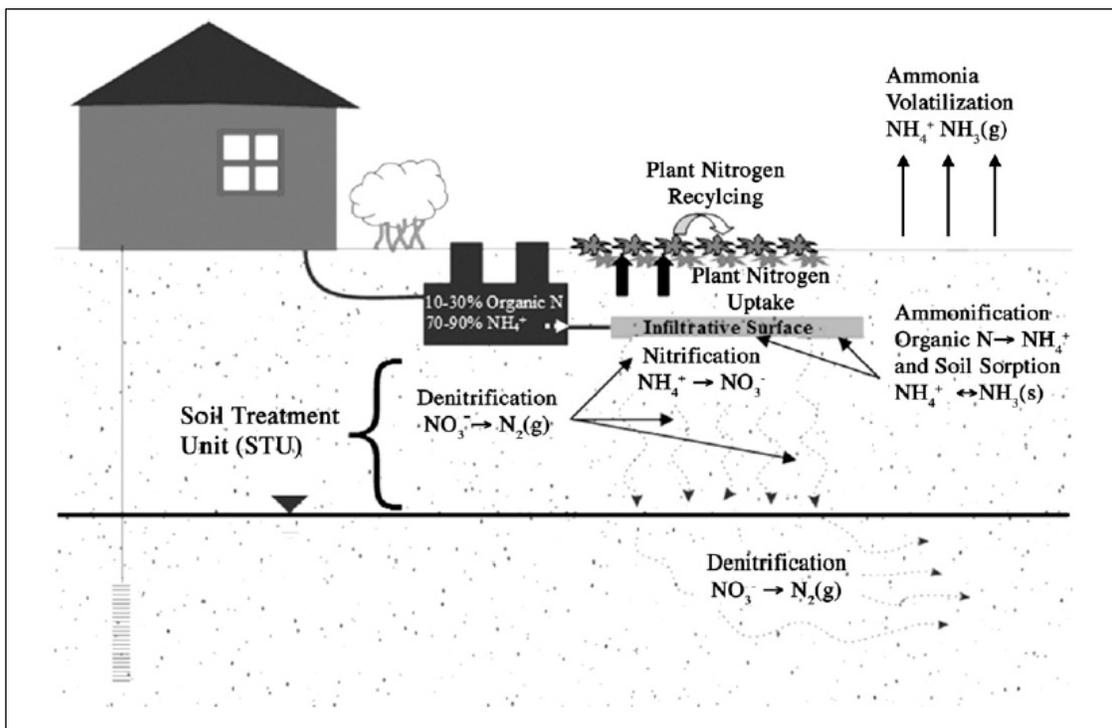
o:\44237-001R008\wpdocs\Report\Draft

requires the development of a conceptual understanding that includes the relevant fate and transport processes, parameters, and simulation approaches that will appropriately achieve the goals of the project. Figure 1-1 summarizes the conceptual understanding of the inputs of nitrogen and the transformative and advective processes that lead to nitrogen contamination of groundwater. The FOSNRS project should result in tools that will consist of the adequate level of complexity to represent these processes to accurately simulate the fate and transport of nitrogen species.

Proper OWTS design, installation, operation, and management are essential to ensure protection of the water quality and the public served by that water source. Assuming soils and site conditions are judged suitable, a wide variety of OWTS are designed and implemented (U.S. EPA, 1997, 2002; Crites and Tchobanoglous, 1998; Siegrist, 2001). Conventional OWTS rely on septic tanks for the primary digestion of raw wastewater followed by discharge of septic tank effluent (STE) to the subsurface soils for eventual recharge to underlying groundwater (Crites and Tchobanoglous, 1998; Metcalf and Eddy, 1991; U.S. EPA, 2002). However, increasing uses of alternative OWTS rely on additional treatment of the STE prior to discharge to the environment in sensitive areas or may eliminate use of a septic tank altogether.

Septic tanks are anaerobic and have long solids retention times (e.g., years) that can enable digestion resulting in a reduction of sludge volume (40%), biochemical oxygen demand (60%), suspended solids (70%) and conversion of much of the organic nitrogen to ammonium (Reneau et al. 2001). Septic tanks are also important as they attenuate instantaneous peak flows from the dwelling unit or establishment. The effluent discharged from the septic tank (i.e., septic tank effluent or STE) then flows to subsequent engineered treatment or the directly to the soil treatment unit where the processes of soil adsorption, filtration, and transformation (biological and chemical) occur.

o:\44237-001R008\Wpdocs\Report\Draft

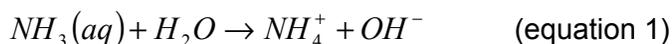


**Figure 1-1: Nitrogen Processes Occurring in a Typical OWTS (Heatwole and McCray 2007)**

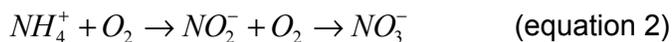
Nitrogen waste products are a considerable component of septic tank effluent. Total nitrogen, composed primarily of organic nitrogen products and ammonium-nitrogen, is typically assumed to range between 20-190 mg-N/L in untreated waste water, and 26-125 mg-N/L in STE (Canter 1996, Crites and Tchobanoglous, 1998, Lowe et al., 2009). Furthermore, in a recent study that evaluated the composition of raw wastewater and STE, the median total nitrogen concentration in STE specific to Florida was determined to be 65 mg-N/L (average = 61 mg-N/L) (Lowe et al., 2009). In terms of mass loading to the subsurface, the median loading rate was determined to be 10 g-N/capita/d (average = 13.3 g-N/capita/d) (Lowe et al., 2009). McCray et al. (2005) suggested that an average subdivision can generate up to 2880 kg/km<sup>2</sup> annually. While this value is significantly higher than estimates of naturally generated deposition (600-1,200 kg/km<sup>2</sup> annually), it is much lower than the loading that results from fertilizer application (10,000-20,000 kg/km<sup>2</sup> annually). Nonetheless, OWTS should be considered a potential contributor to groundwater nitrogen concentrations.

o:\144237-001R008\wpdocs\Report\Draft

The first stages of nitrogen transformation related to OWTS occur in the septic tank. Organic nitrogen is mineralized to the inorganic form (ammonia) via the process of ammonification, followed by volatilization to ammonium ions.

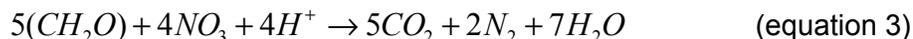


Once the liquid portion of the wastewater enters the drainfield through the subsurface infiltration system, nitrogen species (specifically ammonium and nitrate) are further transformed in the soil by nitrification and denitrification. Nitrification is a two step process by which ammonium is converted first to nitrite than to nitrate via biological oxidation.



Although a two step process, it can be assumed to be a one step process since the conversion of ammonium to nitrite is relatively rapid. Nitrification is either described as a zero-order or first-order reaction or via Monod kinetics. This particular reaction is of importance, as it represents the transformation from the relatively immobile nitrogen form (ammonium) to the highly mobile form (nitrate). Most studies of OWTS with suitable unsaturated soil have indicated that little ammonium reaches the underlying groundwater and that most impacts to groundwater from nitrogen are in the nitrate form. Nitrate behaves essentially as a conservative solute, with virtually no sorption or retardation processes affecting its movement in the aquifer. It is, however, subject to transformative processes.

Denitrification is the transformation of nitrate to N<sub>2</sub> gas.



Denitrification occurs in oxygen-free conditions, and is therefore seen in anoxic zones in the soil and groundwater. This reaction is typically described as first-order. However, nitrogen transformations are probably best modeled using Monod kinetics, which result in zero-order rate constants for concentrations typical of nitrate-impacted groundwater. The process, while studied extensively, is not well understood or well quantified.

Understanding denitrification in the saturated zone, while receiving much less focus in the literature, is nonetheless a potentially valuable topic. Korom (1982) provides a thorough review of denitrification in the saturated zone. Although not specific to OWTS impacts, aquifer denitrification can naturally reduce nitrate concentrations, and can be potentially enhanced via the addition of *in situ* amendments such as sucrose or methanol.

o:\144237-001R008\pdocs\Report\Draft

This review goes on to include data and estimated denitrification rates found in both laboratory and field studies. In order to assess the contribution denitrification makes to nitrate reductions, researchers will often use the ratio of non-reactive solute (typically chloride) to nitrate along the plume flowpath. Any relative reduction in nitrate can be attributed to denitrification, since a reduction due to mixing with ambient groundwater would not change the ratio. Depending on the aquifer conditions, previous studies concerning the reduction of nitrate concentrations specifically from OWTS identify denitrification rates as relatively small, and that most reductions occur as a result of mixing with ambient groundwater (see Reneau et al. 1989). A small number of studies however indicate that denitrification may be the dominant process, perhaps characterizing aquifers with low groundwater flux (see Hantzche and Finnemore 1992).

o:\44237-001R008\wpdocs\Report\Draft



## Section 2.0

### Literature Review

---

The following presents a summary of available research related to the treatment of nitrogen in soils and the subsequent fate and transport of nitrogen in groundwater.

#### 2.1 OWTS Performance – Laboratory and Field Studies

A number of studies looked at performance of either experimental or conventional OWTS in terms of the treatment of nitrogen wastes from effluent. The ability of a system to adequately treat nitrogen wastes will have a significant influence into the resulting impacts to groundwater. These types of studies are valuable in that they can indicate which factors influence the transformative processes and how various loading rates, soil types, and geochemical parameters may lead to excessive nitrogen concentrations. Furthermore, these studies suggest ways of improving performance of older or failing OWTS. A large body of research has been dedicated to this topic and is important for assessing nitrogen in groundwater; however, a full discussion on this topic is beyond the scope of this review, and therefore only a few relevant studies are indicated below.

An in-depth review of the fate and transport of contaminants from on-site systems is provided by Reneau et al., (1989). This study considers multiple factors, including soil type, loading rates, effluent quality, and carbon content. In this review the author describes the important mechanisms related to OWTS performance. Firstly, he describes the importance of conditions conducive to nitrification, namely coarse-textured soils in which aerobic conditions are dominant. This is even true in fine-grained clay soils as long as unsaturated conditions are present. Denitrification in soils utilized for OWTS is expected to be minimal except in anaerobic microsites. However, soils that are influenced by fluctuating water tables in which saturated conditions can occur will see increases in denitrification rates. For groundwater, sites which are ideal for OWTS are often the most vulnerable to nitrate impacts, since they are often well drained soils with limited capacity for denitrification. In this case, often the most important mechanism for nitrate reduction is dilution by ambient groundwater.

Cogger and Carlile (1984) looked at the performance of 15 conventional and alternative OWTSs to determine their performance in soils with high water tables in North Carolina. The alternative methods included low-pressure pipe systems, soil replacement systems, and pressure-dosed mounds. At the study site, shallow groundwater wells were installed

o:\44237-001R008\Wpdocs\Report\Draft

around the systems and monitored monthly for nitrogen species. The study found in general that nitrogen species concentrations were markedly influenced by seasonal variations in the water table, although some systems experienced continuous saturation. Those systems that were continually saturated had the poorest performance, as well as those with the heaviest effluent loadings. Additionally, transport of nitrogen products was facilitated by those systems located in areas with high gradients and continuous soil saturation. The low-pressure pipe systems, designed to distribute the effluent of the entire adsorption field and provide occasional dosing rest periods, performed the best in spite of any level of saturation from the water table. The mound system did not perform well, however the authors indicate the pumps feeding the system were not operating correctly and the dosing recommendations were being exceeded. The soil replacement systems showed no improved performance over the conventional systems.

Similarly, Costa (2002) conducted a series of experiments comparing the nitrogen removal capabilities of a conventional system, two proprietary nitrogen removal systems (the Waterloo Biofilter and the MicroFAST Model), and a recirculating sand filter (RSF) system. “Nitrogen losses” are described as reduction in nitrogen from the septic tank effluent to the groundwater. Measurements were conducted over an 18 month period. Results indicate that the conventional system removed 21-25%, the Waterloo 60%, the MicroFAST removed 55%, and the RSF removed 41%.

Cogger et al. (1988) examined the performance of an OWTS on a coastal barrier island. The study considered loading rate and water table as the primary influences on OWTS performance. Two absorption fields were constructed and sampled biweekly for a period of 18 months. Three loading rates (one, four, and six cm/day) were applied in a random fashion. Loading rate was identified as significant. Additionally, periods with a high water table in the early part of the year resulted in anaerobic conditions which inhibited nitrification. Redox conditions were generally considered low. However, in drier conditions, aerobic conditions dominated and more nitrification resulted with corresponding increases in redox parameters. The authors concluded that although loading was a factor, the fluctuations in the water table were more influential in determining the rates of transformation.

Various loading rates were applied and the resulting leaching of nitrogen compounds in an OWTS were measured (Uebler 1984). Loading rates of 7.5, 11.3 and 15 L m<sup>-2</sup> d<sup>-1</sup> were tested. Additionally, soil amendments (cement and lime) were also part of the experiment. Transformation of ammonium to nitrate was enhanced by the soil amendments, particularly the cement amendment when water levels were higher. Interestingly, the nitrate concentrations were highest with the lowest loading rate, particularly during high water table conditions. This observation suggests that water table level influences the production of nitrate more than the loading rate.

Lowe and Seigrist (2008) describe a pilot-scale study to evaluate the effects of infiltrative surface architectures (ISA) and hydraulic loading rates (HLR) on soil treatment of septic tank effluent. A test site was established in Golden, Colorado with three different ISAs (open, stone, and synthetic) and with two different HLRs (four and eight cm/day). Monitoring was done over a two-year period to evaluate the infiltration capacity and purification performance of the different conditions. Results indicate improved infiltration using the higher HLR and using the open ISA. The higher HLR resulted in increased nitrogen mass removal (42%) compared to the lower HLR. No significant difference was reported for the different ISAs. The data suggests that improved purification can be achieved by applying higher HLRs to a portion of the soil treatment area rather than a low HLR over the entire area.

In another study, Lowe et al. (2007 and 2008) describes a large field-scale study examining the purification performance of three different treatment units: a septic tank, a septic tank with a textile filter unit (TFU) and a septic tank with a membrane bioreactor (MBR). The different units were operated over a period of 16 to 28 months, with water quality monitoring for different parameters including nitrogen. Results showed an improved performance for both the MBR and TFU over the conventional septic tank, with a 30% and 61% nitrogen removal rate for the TFU and MBR, respectively (compared to the conventional septic tank only). The use of a treatment unit such as a TFU or MBR enables the application of a higher quality effluent at a higher HLR without subsequent soil clogging, although this can be dependent on the native soil characteristics.

## 2.2 Vadose Zone Processes and Impacts to Groundwater

Soil treatment of nitrogen from OWTS in the vadose zone can also have a significant influence on the resulting nitrogen concentrations in the aquifer. The transformations and reactions of sorption, nitrification, and denitrification described earlier are present in this zone. Nitrogen that is present as ammonium is subject to adsorption to negatively charged soil particles, plant uptake or microbial bioaccumulation. Nitrate, on the other hand, is mobile in the vadose zone but can be subject to denitrification. It is therefore important to quantify the vadose zone processes to assess nitrogen attenuation prior to entering the saturated zone.

In another study conducted by the project team members, a summary of the available literature related to nitrogen attenuation in the soil treatment unit (STU) was done to identify the parameters that influenced transformations and reactions (McCray, et al 2008). Data from available literature was collected and tabulated for nitrogen concentration vs. depth, vadose zone characteristics, and soil type. Additional data was collected considering wastewater type, hydraulic loading rates, and source type characterization. Data analysis was performed to indicate the correlation between nitrogen attenuation

and the various parameters. Initial analysis indicated no significant relationship existed between expected nitrogen concentrations and depth, soil type, or HLR. A more in-depth analysis found that the data variability was most related to HLR, suggesting that this parameter may be more influential than soil type when considering nitrogen attenuation. However, the study also indicates that different soil types will have different hydraulic properties and this can influence nitrogen attenuation.

Ammonium that is not immobilized can be converted to nitrate via nitrification. This form of nitrogen, as mentioned before, is highly mobile and can impact aquifers under OWTS. Within the vadose zone, the pathway of nitrate reduction is denitrification. In the vadose zone, denitrification is the dominant process affecting nitrate concentrations below the absorption field (Wilhelm, Schiff et al. 1998) and is therefore a key process in estimating the resultant nitrate loading to the aquifer. A body of research has been involved with understanding and quantifying denitrification in the vadose zone.

Ritter and Eastburn (1988) provide a summary of available literature related to denitrification and OWTS. Based on their review of available literature, several factors which may influence nitrogen attenuation are:

- adequate supply of a carbon source;
- infiltrative surface biozones (the biozone has been shown to improve denitrification);
- OWTS with high water tables (potentially insignificant denitrification due to lack of conditions conducive to nitrification);
- dosing (likely to improve denitrification); and
- recirculating sand filters (and other aerobic treatment units may improve denitrification).

Degen (1991) conducted a study that considered multiple factors that could potentially influence denitrification processes including effluent loading rates, effluent type, dosing rates, and temperature. This study included both experiments on soil cores in the laboratory and field sampling and measurements on sites in Virginia. The predominant soil types consisted mainly of silt loams collected in Blacksburg, Virginia. Soil cores collected for the laboratory experiments were subjected to a variety of effluent dosing rates and effluent types in order to quantify the response in a more controlled environment. The study attempted to quantify the denitrification via a number of methods, including nitrate/chloride ratios, soil chemical analyses, and microbial activity analyses. Field studies

o:\44237-001R008\Wpdocs\Report\Draft

used similar analyses. Additionally, an attempt was made to model the expected denitrification in the field based on the lab results. The study made several key conclusions as follows:

- Carbon content was the limiting factor for denitrification.
- Applications every 48 hours doubled the denitrification rates compared to applications every 24 hours.
- The model was not useful for predicting denitrification in the field, likely due to the more favorable anaerobic conditions present in the field study.

Tucholke (2006) provides an analysis for relating denitrification rates in the vadose zone with soil type. The study consisted primarily of identifying studies that measured denitrification rates and described the soil characteristics of the study site with the hypothesis that predictions of denitrification could be made based on soil type. While the data did not support the hypothesis, it did show that denitrification varied significantly with soil type. However, the study concluded that denitrification is a process dependent on many variables, such as organic carbon content, soil temperature, water content, and soil pH. This conclusion was verified by statistical analysis that demonstrated that data variability was dependent on the variability in the various parameters.

One of the major research concerns with quantifying denitrification is the wide variation in measured rates in different studies. This issue makes correlation of site characteristics and denitrification difficult. Tucholke et al (2007) provides a review discussing the variability seen in the literature. This variation is attributed to variations in measurement method and wide variations observed spatially and temporally in the field. For example, rates determined in the laboratory as compared to the field varied widely, as did rates determined by isotope analysis as compared to other methods. Also, site heterogeneities in limiting factors such as water content and pH also impacted the rate determination.

Nitrogen in the vadose zone that results from OWTS is subject to various transformations and reactions which are dependent on numerous factors within the soil and from the source. Attenuation of nitrogen is accomplished via sorption, plant uptake, bioaccumulation, or conversion of nitrate to nitrogen gas (denitrification). No single dominant process or parameter can be identified; rather, an interconnected complex of factors will ultimately influence the nitrogen attenuation. Due to the complexity of the issue, more research is required in the future to relate all of the processes and variables to observed changes in nitrogen concentration from the source to the groundwater.

o:\44237-001R008\wpdocs\Report\Draft

### **2.3 Land Planning and OWTS Density**

While a large number of studies consider lot size or OWTS density to be important factors, two studies were identified that examined these as primary characteristics for estimating potential groundwater impacts from OWTS. Ultimately consideration of lot size or septic tank density will play a key role in land planning and developments considering OWTS as the primary method of wastewater disposal.

A method of determining lot size and density related to land development in Pennsylvania was developed by Taylor that assumes the reduction of nitrate is primarily via groundwater dilution (Taylor 2003). The author reiterates the discussion of whether or not denitrification is a significant process in groundwater, and ultimately concludes that land planning must consider dilution as the primary factor in nitrate reduction, since this approach is both conservative and simple. Also, the author indicates denitrification is a poorly understood process and should not be relied on for nitrate reductions.

Similarly, Yates concludes in her study of OWTS distribution in various watersheds in the United States that the most important factor in limiting OWTS impacts is restricting system density (Yates 1985). The author looks at nitrate impacted areas in New Mexico, Colorado, New York, Massachusetts, Delaware, and North Carolina. The study cites other research in these areas that quantifies the number of septic tanks in a particular watershed and the level of nitrate impacts. However, little quantitative analysis is provided and significant conclusions that specify lot size or density of septic tanks and how that relates to high nitrate concentrations in groundwater is given.

### **2.4 Groundwater Monitoring Studies and Reports**

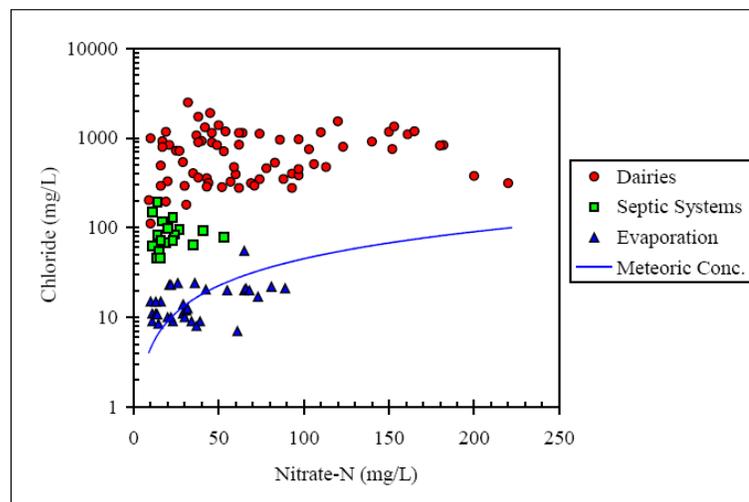
A relatively large number of studies and reports were found that considered nitrate distribution, plume delineation, and estimates of the source contribution of OWTS. Generally these are characterized by various levels of groundwater sampling, and usually some effort to make conclusions as to the nature of the nitrogen impacts based on the results of the sampling. In some cases the studies or reports are quite simple, considering only nitrate concentrations. Others are highly detailed, considering not only nitrogen species concentrations, but a variety of hydraulic and geochemical parameters. Typically the more complex studies draw more conclusions as to the transport and transformative processes at work at the various sites. However, this level of complexity does not always correspond with superior results; in some cases, the simple study addresses the objectives and can make some significant conclusions related to nitrogen impacts.

A study in Helena, Montana, examined the change in groundwater nitrate distribution as correlated with the increase in population in the area (Drake and Bauder 2005). The study indicates a potential relationship between the increase in observed nitrate concen-

o:\44237-001R008\wpdocs\Report\Draft

trations and the increased use of OWTS between 1971 and 2003. The study compiled data for aquifer nitrate concentrations from 10 publicly funded investigations in the defined time period. From this data, trend analysis with statistical significance methods was applied to identify any trend between the increasing population and nitrate concentration trends. Additionally, the data was plotted geographically for spatial trend analysis. The area surrounding Helena experienced a 17% increase in population and a 68% increase in septic tank use in the decade between 1990 and 2000. The statistical analysis confirmed a correlation between nitrate concentrations and increasing population. The geographical analysis also indicated a spatial trend, showing the highest increases occurred in rural areas. This was especially the case in areas overlying bedrock aquifers and areas with high density and unpermitted OWTS.

A similar study summarized the overall impacts due to OWTS in New Mexico that also considered nitrate distribution (McQuillan 2004). In this study, data was compiled in a similar fashion the study described above. The study compared the level of nitrate impacts of aquifers with largely oxic conditions to aquifers with anoxic conditions. Also, data results from geochemical isotopic fingerprinting are provided, to identify the source of nitrate contamination. Figure 2-1 shows the results using isotopic fingerprinting. This study indicated that areas with more significant nitrate occur in aquifers with oxic conditions, whereas aquifers with anoxic conditions have lower impacts due to conditions not being favorable for the transformation of ammonium to nitrate. The results of the study also indicated that isotopic fingerprinting can be a useful tool for identifying nitrate sources, which can be useful for targeting primary nitrate sources.



**Figure 2-1: Isotopic Fingerprinting of Nitrate Sources (McQuillan 2004)**

o:\44237-001R008\Wpdocs\Report\Draft

A study of the Darling Plateau region near Perth, West Australia area also examined the nitrate contributions from OWTS in a populated area served almost exclusively by individual OWTS (Gerritse, Adeney et al. 1995). It was estimated in this study that nearly 80% of the nitrogen in the subsurface could be attributed to OWTS source contributions. This study specifically looked at impacts of a neighboring surface water body approximately 70 meters downgradient. Monitoring of nitrogen species and bromide tracers showed significant decreases in inorganic nitrogen as the groundwater approached the creek. Interestingly, the surface water body had relatively high background concentrations of nitrate, but the study showed no significant contribution from this soil treatment unit.

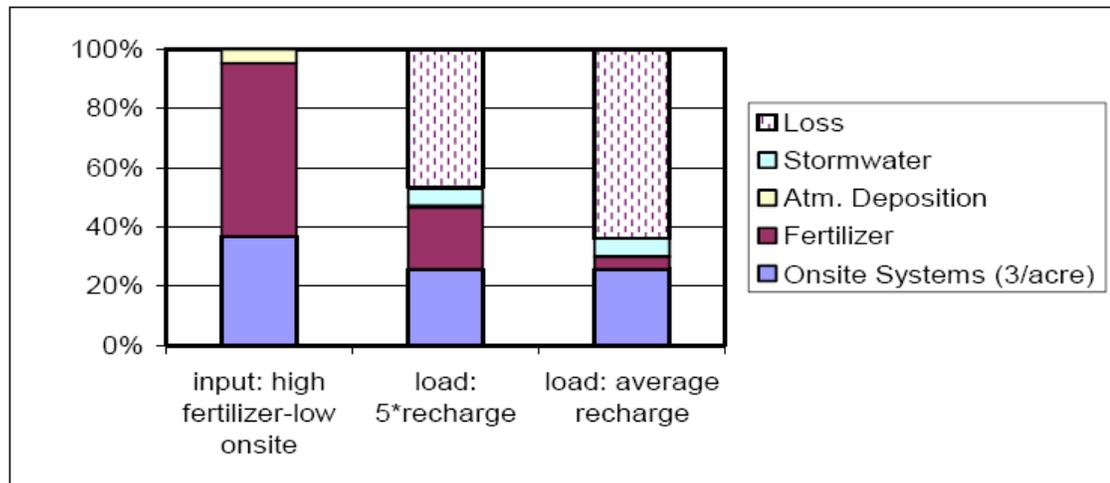
Lapointe et al. (1990) conducted a study to relate groundwater impacts to nearby marine surface waters via tidally-influenced groundwater recharge in the Florida Keys. The area in the study is characterized by typical tropical wet and dry seasons, with most of the precipitation falling between June and October. The subsurface is a highly porous and permeable limestone that allows for rapid lateral groundwater flow. For sampling, networks of monitoring wells were established on seven residences using OWTS and one control site in a neighboring wildlife refuge. Wells were sampled monthly for approximately one year for nitrogen species and other biogeochemical factors. Groundwater flow was measured directly using an *in situ* flow meter. Surface water was also sampled. The results indicated that the contribution of nitrogen to the groundwater by OWTS was significant in this area, in some cases as much as two orders of magnitude higher than when compared to control groundwater. Ammonium was the dominant species, the result of the largely unfavorable conditions for nitrification. Surface water showed a seasonal variation, with the highest concentrations occurring in the summer months. The study also concluded that seasonal variations in tides and groundwater levels result in significant contributions from OWTS to surface waters in the Florida Keys.

A series of reports have been previously completed for assessment of OWTS contributions to nitrate contamination of groundwater and surrounding surface waters in the Wekiva watershed in Florida. This includes reports prepared by: Anderson (2006); Briggs, Roeder et al. (2007); MACTEC (2007); Otis (2007); and Young (2007). The study was initiated to protect the Wekiva river system which had been assigned protection under the Wekiva River Protection Act. The watershed occupies roughly 304,000 acres and includes parts of Lake, Orange, and Seminole counties in central Florida. The project has been performed over a number of years and includes a series of tasks in order to assess the contribution to groundwater impacts from OWTS and ultimately strategies to reduce these impacts. The tasks included:

- Field sampling for watershed characteristics, nitrogen concentrations and OWTS loading estimates.

- A literature review for refining estimates of OWTS loading.
- Integrating these estimates with estimates of other source contributions.
- Development and discussion of alternatives for reducing OTWS contributions.

Three sites were selected for sampling that met the criteria and were deemed suitable for assessment of the desired data. After completion of the tasks, a number of conclusions were reached. For example, an average home with 2.6 people on average contributes 18 pounds of nitrogen to the groundwater with the main nitrogen contributor attributed to fertilizer use. This is slightly higher than reported by Lowe et al (2008) of approximately 14 pounds of nitrogen annually (excluding outdoor residential nitrogen sources). The studies also concluded that OWTS contribution to shallow groundwater contamination was similar in terms of intensity to atmospheric deposition, however due to the areal distribution, nitrate impacts from OWTS were approximately an order of magnitude higher and distinct plumes could be delineated. Furthermore, OWTS tended to be in high-vulnerability areas and did not have effective nitrogen removal as compared to centralized wastewater methods. Figure 2-2 shows the estimated distribution when comparing the various sources.



**Figure 2-2: Source Contribution to Nitrate Impacts (Briggs, Roeder et al. 2007)**

Other aspects of the study considered transport and transformation of nitrogen. Two factors were identified that influenced nitrogen entering the drainfield; the amount of nitrogen present in the effluent, and the level of pre-treatment prior to discharge. In the event pre-treatment was present then ammonia is converted to nitrate. However, nitrification

o:\44237-001R008\wpdocs\Report\Draft

will be limited in soils with high water tables. After discharge, if there is adequate organic carbon present, the nitrate can be denitrified to nitrogen gas.

Soils with moderate to poor drainage, fine loamy texture with clay, shallow water tables and some organic matter have the highest potential for denitrification.

Ultimately the study found that contributions from OWTS could be effectively minimized by reduced loading and improving OWTS performance with pre-treatment methods and improvement of subsurface characteristics, especially considering high water table areas.

Andreadakis (1987) performed laboratory simulations of an alternative OWTS in Greece to estimate the effectiveness of the system for nitrogen removal. The system consisted of a septic tank, gravel filter, two sand filters operated alternatively and two soil absorption trenches operated alternatively. The study found the system could achieve approximately 70% nitrogen removal. The factors that influenced the effectiveness were the compaction characteristics of the filters and soil, loading rates, and variability in saturated/unsaturated conditions.

Reneau (1977) conducted a study of changes in inorganic nitrogen compound concentrations from a septic tank in a soil with a fluctuating water table in Virginia coastal plain area. Samples were collected and analyzed for nitrate, nitrite and ammonium ions over a three year period. The relationship between nitrate and ammonium and distance is demonstrated by the ratio of these constituents to chloride ( $Cl^-$ ). Assuming that chloride undergoes no significant transformations or adsorption, any variation in the ratio can be the result of either adsorption or transformation. In this case, the ratio of ammonium to chloride decreased with depth, indicating that at higher points anaerobic conditions dominated and nitrification could not take place. Following this trend, decreases in the nitrate to chloride ratio suggested that in some areas denitrification could take place due to the rising water table.

Arnade (1999) examined the relationship between nitrate well contamination and distance from OWTS as related to seasonal variations in water level in Palm Bay, Florida. The study area experiences high precipitation during the summer months and results in high water tables in sandy soils that cause septic tank overflows and ultimately groundwater contamination. Results indicated that during the wet season, nitrate concentrations tended to be higher as distance increased as compared to the dry season, although the opposite was true closer to the OWTS. The reasons provided for this observation were perhaps dilution, plant uptake or enhanced transformation. This reasoning seems suspect, as if dilution is a factor in reduced concentrations, then concentrations should follow the same pattern throughout the flow path.

o:\44237-001R008\wpdocs\Report\Draft

Walker et al. (1973a and b) describes two studies that look at nitrogen transformations of septic tank effluent in sands. The first study focused on transformations in sand while the second study examined transformations related to groundwater quality. Research was done at the field scale at five separate locations in Wisconsin. In all cases, effluent was ponded near the surface due to the formation of a “crust” (aka, the biozone) which was the result of biological processes. As a result, unsaturated flow rates were extremely low (8 cm/day). The biozone conditions were favorable for nitrification where groundwater was not present. Most of the sites showed complete nitrification was possible at six centimeters below the biozone. One site had a high water table and as a result nitrification did not occur unless seasonal variations resulted in a lowering of the water table. Denitrification was identified in an underlying clay layer at some of the sites, although this was not the case if the site had an underlying sandy layer. In the groundwater, the dominant process reducing the nitrate concentrations were dilution with ambient groundwater and not denitrification due to the nature of the well-aerated sandy soils and the low carbon content of the groundwater. The authors concluded that in order to minimize impacts from OWTS in such aquifers, considerable land size is necessary in order to maximize the effects of dilution from clean water.

Harman et al. (1996) looked at the groundwater impacts resulting from an OWTS at a school in Langton, Ontario, Canada. In this community, over 30% of the water supply wells exceeded the standard for nitrate. Multiple sources, primarily from OWTS use and agricultural practices contributed to the high nitrate concentrations. The study aquifer in question was characterized by fine to medium sands and has a relatively high groundwater velocity (170 meters/year). The wastewater from the facility was largely from washrooms, as the site had no laundry facilities present. The effluent was primarily ammonium. At the site over 400 samples were collected at 45 multilevel monitoring points at various locations downgradient of the OWTS. Samples were collected for all major ions, DOC, alkalinity, pH, and dissolved oxygen. The results found high nitrate concentrations were observed (20-120 mg/L) and extended over 100 meters downgradient owing to the high groundwater velocity. Vadose zone residence time was one to two weeks but did not appear to allow for complete conversion of ammonium to nitrate. However, geochemical analyses indicated reduced ammonium and organic carbon concentrations coinciding with increases in nitrate which suggest that nitrification was occurring. Denitrification was limited and isolated due to low levels of organic carbon and aerobic conditions. It appeared that most of the reduction of nitrate along the plume extent was likely due to natural dilution; denitrification was limited by low levels of organic carbon and aerobic conditions.

Robertson et al. (1991) studied the OWTS impact to sand aquifer from two single-family homes in Ontario, Canada. The first site was a home in Cambridge, Ontario. The surficial aquifer was characterized as a coarse sand overlying a low permeability silt. The

home was occupied by four people. The second site in Bracebridge, Ontario was situated on a fine sand aquifer with a household occupied by two people. Major ion geochemistry and typical septic tank nutrients were sampled. Bromide tracer tests were also performed. Both sites showed evidence of nitrification due to high concentrations of nitrate, and low concentrations of dissolved organic carbon and ammonium. High concentrations of nitrate were observed more than 130 meters downgradient from the sources which suggested little or no denitrification was occurring and that aquifer conditions were favorable for considerable nitrate migration. However, almost complete denitrification was observed in the carbon rich river sediments downgradient. In this aquifer, it was concluded that due to the low dispersive nature of this type of aquifer, current minimum distance to well regulations may not be protective. This was verified by natural-gradient bromide tracer tests.

Another study in the literature conducted water sampling from domestic supply wells in five unsewered subdivisions in Wisconsin (Tinker 1991). The objective of the study was to identify the sources of nitrate impacts to drinking water. Water samples were collected on two separate occasions from supply wells in five subdivisions and tested for nitrate concentrations. Sources of nitrate impacts were assessed by the location of the OWTS and the water supply well in relation to the groundwater flow direction and comparison of the results of three mass-balance models. The combination of methods resulted in a good correlation between the locations and the groundwater flow, as well as the results of the mass-balance modeling. The author concluded that elevated nitrate concentrations could be attributed to lot size (from the mass-balance modeling) and locations of water supply wells and OWTS.

Reay (2004) examined the impacts from OWTS to near shore areas along Chesapeake Bay. Due to the sandy characteristics of the aquifer and the shallow water table, significant nitrate impacts to near shore sediments were observed. Multiple characteristics were analyzed at three separate sites in Virginia considered representative of the Virginia coastal plains. Among the characteristics were depth to water, aquifer thickness, soil characteristics, lot size and persons per household. Groundwater was sampled for nitrogen species and phosphorus as was neighboring surface waters. The author noted the lot size and relatively high loading rates contributed to the observed concentrations. Furthermore, the sites showed potentially high nitrification rates are likely present due to the observed concentrations of nitrate versus ammonium, and that very little denitrification was occurring, which led to significant nitrate impacts to nearby surface waters.

A sampling study to quantify the nitrogen impacts from OWTSs was performed for a community in Nevada (Rosen et al. 2006). This study combined field data and a mass-balance approach to assess the nitrogen impacts attributed to OWTS. The area under study was a densely populated area north of Reno, Nevada. In this area, 2,070 septic

o:\44237-001R008\wpdocs\Report\Draft

tanks were in use. Annual precipitation was low (20-25 cm/year) and recharge water to the aquifer also came from irrigation ditches (54%) and septic tank effluent (17%). Four separate sites were sampled monthly for one year. No geochemical or hydraulic parameters were collected. The final results of the estimates indicated that 25-30 metric tons of nitrogen in the groundwater could be the result of OWTS use, although the authors concede that considerable error is possible and that future studies considering more parameters will be needed.

## 2.5 OWTS Plume Geochemistry

A number of researchers went beyond the approach of considering nitrate concentrations only and considered numerous factors of OWTS-generated nitrate plumes to delineate the important parameters that may affect nitrate transport and transformation. In most cases, the study collected samples related to all major ions present in groundwater ( $K^+$ ,  $Cl^-$ ,  $NO_3^-$ ,  $SO_4^{2-}$ ,  $Ca^{+2}$ ,  $Na^+$ ,  $Mg^{+2}$ ,  $PO_4^{-3}$ ), field parameters such as pH, conductivity, alkalinity, dissolved oxygen, and other factors such as dissolved organic carbon. Additionally, complete characterization of the aquifer parameters were collected, such as those related to soil type and groundwater flow and velocity. These studies were often performed at the field scale, although some laboratory experiments were done as well. The value of these studies is the opportunity to understand how the aquifer responds to transformative processes in terms of changes to other constituents and physical characteristics, and provide a rationale for the extent of impacts observed.

Wilhelm et al. (1998) looked at changes in geochemistry for two operating OWTS in a sandy aquifer in Ontario for evidence of nitrate transformation. The objective of the study was to confirm a conceptual model that indicated the transformative processes related to nitrogen would result in the creation of redox zones. Changes in geochemical parameters could be measured to confirm the presences of these zones. Sampling was performed along the wastewater flow path at two sites from 1987 to 1990. In the septic tanks themselves a primarily anaerobic environment existed, with low concentrations of nitrate and high concentrations of ammonium and carbon. Aerobic conditions dominated below the discharge pipes. The research indicated that nitrification zones could be identified in areas with decreases in pH and alkalinity, whereas zones of denitrification were characterized by increases in both parameters. Differences in the sediment composition led to different behaviors of nitrate in the groundwater. For example, the plume at the second site entered carbon-rich sediments near a river bed, ultimately leading to complete denitrification and an increase in alkalinity.

Another study looked primarily at changes in inorganic nitrogen compounds related to septic tank effluents, but also looked at subsequent changes in pH and Eh (redox potentials) in a groundwater system in Virginia (Reneau 1979a). The objective of the study

was to relate changes in concentrations as related to distance traveled, soil properties, and seasonal variation. At three different sites, rows of sampling wells were established at 1.5, five, 10, and 13.5 meters downgradient and sampled semi-monthly for phosphate, nitrogen species, Eh, and pH. Sampling occurred over a two-year period. For nitrate, concentrations reached a maximum (average values ranging from 2.7 to 3.9 mg/L) at the five meter sampling points then decreased with distance. This was attributed to nitrification of ammonium and the subsequent denitrification of nitrate to a relatively large degree. This was accompanied by a drop in pH and a slight increase in Eh values.

A study conducted in Ontario, Canada examined multiple geochemical factors which can be related to OWTS impacts (Ptacek 1998). Temperature, pH, dissolved organic carbon redox conditions and nitrogen species concentrations were all sampled. The original OWTS effluent contained 98 mg/L of nitrogen as ammonium. Nitrate concentrations were high in the shallow portions of the aquifer, along with diminishing concentrations of DOC downgradient. pH stayed near neutral which was attributed to the buffering capacity of the aquifer due to carbonate content. Nitrate concentrations were low, which may suggest low rates of denitrification.

Robertson and Blowes (1995) observed nitrate concentrations in an acidic OWTS plume. The study site was again located in Ontario, Canada at a location using an OWTS for wastewater at a seasonal-use cottage. Sampling was performed at 38 piezometers adjacent to and underneath the infiltration bed. Major ion geochemistry samples were collected. Subsurface soil characteristics were various, from clays to silts to sands. The water table was generally consistent (1.5 meters below the field tiles), but became much shallower during the off-season winter months. In this system, background pH was naturally low; however, more acidic conditions existed within the plume core. Ammonium levels dropped substantially suggesting nitrification was occurring. The authors suggest that changes in nitrate concentrations downgradient were due to denitrification that was facilitated by relatively high levels of dissolved organic carbon and anaerobic conditions. Furthermore, at greater depths in the groundwater, high levels of sulfate coinciding with drops in nitrate concentrations suggested an alternative pathway for consumption of nitrate via sulphur oxidation.

o:\44237-001R008\Wpdocs\Report\Draft



## Section 3.0

### Discussion and Analysis

---

The literature review revealed important conceptual information for the assessment of nitrogen impacts in groundwater due to OWTS. One of the primary objectives of the review was to examine the current state-of-knowledge related to the primary influences on the fate and transport of nitrogen following the initial loading into the soil from the use of OWTS. A cascade of processes and factors contribute to nitrogen contamination. These include loading rate, OWTS density, soil characteristics, oxygen content, and aquifer recharge and water table elevation and fluctuation. Primary factors that can lead to significant nitrogen concentrations are found in both the septic tank and the vadose zone and an understanding of the processes within these is important rather than just considering processes in the aquifer.

The transformative processes of nitrification and denitrification require further study and quantification, especially when considering septic tank performance and processes within the vadose zone. Additionally, an understanding of the aquifer characteristics, such as groundwater velocity and flux estimates can greatly improve the quantification of dilution for reduction of nitrate. Nitrification can be inhibited by high water tables and overloading of OWTS. Likewise, denitrification, a potentially important process in the reduction of nitrate in groundwater, requires anoxic conditions in the presence of adequate carbon sources.

An improved understanding and assessment of field conditions prior to septic tank design can improve performance and result in reduced impacts from OWTS. A large number of reports have been generated that are essentially monitoring reports describing nitrogen levels in observation wells. In some cases, these reports considered factors beyond nitrogen concentrations and included multiple geochemical factors as well. These studies have immense value in the light of other studies, in which the influence of important factors for nitrogen contamination can be quantified in real field-scale studies. Specifically, these studies provide quantitative data concerning:

- Downgradient and cross gradient nitrogen concentrations in groundwater which provides plume delineation spatially and in some cases temporally;
- Site-specific subsurface characterization such as soil type and distribution;

o:\44237-001R008\Wpdocs\Report\Draft

- Groundwater measurements that provide data concerning groundwater flow paths, velocities, and fluxes which can strongly influence the extent of the impacts in terms of concentrations and distance from the OWTS;
- Total nitrogen loading rates at the source, which when compared to downgradient nitrogen concentrations provide data concerning OWTS performance, and nitrogen conversion rates; and
- In some cases, surface water sampling which may indicate the level of groundwater/surface water interaction and/or transformative processes present at the groundwater /surface water interface.

The conclusions reached using the data in these studies can then be applied for nitrogen impact estimates in future studies and how to appropriately monitor and sample a site that will utilize OWTS. Furthermore, these studies can be examples for assisting in OWTS design and installation to minimize nitrogen in groundwater. Lastly, data from these studies can be applied to the further study of the OWTS and vadose zone processes affecting nitrogen transport and fate in groundwater and lead to better predictive methods for estimating nitrogen impacts.

o:\44237-001R008\pdocs\Report\Draft

## Section 4.0

### Conclusions

---

The literature review revealed numerous factors that may influence nitrogen impacts to groundwater resulting from the use of OWTS. Transport and fate processes that are present in the OWTS, vadose zone and saturated zone all will influence the extent of nitrogen impacts to groundwater. Furthermore, these factors, along with factors related to groundwater/surface water interactions will also determine if nearby surface water bodies are adversely affected. In doing site assessments, it is therefore important to develop sampling plans that can collect data for a majority of the factors described in the literature. Also, predictive efforts and efforts aimed at reduction of impacts should also consider the findings of the literature review. A brief summary of important points is as follows:

- Some studies identified lot size and location of water supply wells in relation to OWTS as important factors in determining nitrate contamination to groundwater.
- OWTS loading rate can significantly impact the performance of the soil and ultimately nitrogen concentrations in the aquifer.
- In certain cases, water table fluctuations may be a larger factor than loading rate of nitrogen on the overall OWTS performance.
- Nitrogen reduction in the vadose zone is an important determining factor for nitrate concentrations in the groundwater. This is a complex process dependent on numerous factors that need to be studied in depth.
- Nitrification can be influenced by soil type and appropriate loading of an OWTS. Sikora and Corey (1976) indicate that coarse-textured strongly-aggregated soils favor nitrification while finer textured soils lead to the development of anaerobic conditions and inhibit the process.
- Sandy soil aquifers are particularly susceptible to nitrate contamination, particularly in the case of low carbon content aquifers with relatively high groundwater velocities. In these cases, high concentrations and large areas of impact may be expected due to the lack of transformation and the distance nitrate can travel in a short time period.

- Denitrification occurs largely in anoxic soils and groundwaters with adequate carbon sources. In the soil column, denitrification may occur in systems with high or fluctuating water tables that allow the creation of anoxic conditions, providing the organic carbon content of the soil is adequate. In groundwater, dilution is often seen as the dominant mechanism for the reduction of nitrate, although some studies identify denitrification as the dominant factor. This is highly dependent on site-specific characteristics.
- Denitrification, while being a well-understood process is poorly quantified and not correlated with other site characteristics especially when considering the saturated zone. This should be a significant topic of further study.
- Some studies identified the relatively high denitrification capacity of river bed sediments, particularly if they contained high levels of organic carbon. This is especially relevant if the protection of adjacent surface water bodies is a key concern.

The literature review suggests reductions in groundwater nitrogen impacts associated with OWTS are achievable with a few steps. Nitrate is highly mobile in groundwater and the only significant methods of natural attenuation is denitrification, a process that the review indicates is not always present in natural aquifers (however, it should be noted that saturated zone denitrification can be enhanced with amendments as a potential treatment process, see Korom (1992)). Therefore, reduction of nitrate contamination may be most efficiently approached in the design and installation processes when considering OWTS as a treatment alternative. Appropriate land planning and density of OWTS in new developments is a first step. OWTS should be placed within protective distance of downgradient groundwater and surface water resources. Additionally, recognizing the importance of dilution for nitrate concentration reductions, appropriate lot size should be in the design to allow adequate dilution from recharge water. Within the design of OWTS, appropriate loading rates and an understanding of OWTS effluent can achieve lower levels of nitrogen entering the subsurface. Lastly, the review indicates the performance value of appropriate treatment units can improve effluent quality by reducing nitrogen prior to infiltration.

Additional optimization can be achieved by a thorough understanding of site characteristics and how these may influence OWTS performance and ultimately nitrogen concentrations in groundwater. Numerous studies were identified that have data related to existing systems and their performance within the framework of the characteristics of the site. Certain water table conditions, soil types, and other subsurface characteristics such as pH or temperature can have an effect on the treatment ability of OWTS by varying oxygen content and redox conditions. If detrimental conditions are seen at a site being con-

o:\44237-001R008\wpdocs\Report\Draft

sidered for OWTS, other methods of wastewater treatment may be appropriate. This can also be true for areas identified as “high-risk,” such as areas adjacent to a protected water body. Alternatively, it may be possible to amend the site conditions or use an effluent pre-treatment method to improve OWTS performance. Future work may be needed to examine the data in such studies and make attempts to correlate hydraulic and reactive parameters to observed nitrogen impacts.

o :144237-001R008\pdocs\Report\Draft



## Section 5.0 References

---

Anderson, D.L. (2006). A Review of Nitrogen Loading and Treatment Performance Recommendations for Onsite Wastewater Treatment Systems (OWTS) in the Wekiva Study Area. Florida Department of Health Report.

Andreadakis, A.D. (1987). Organic Matter and Nitrogen Removal by an On-site Sewage Treatment and Disposal System. *Water Research*, **21**: 559-565.

Arnade, L.J. (1999). Seasonal Correlation of Well Contamination and Septic Tank Distance. *Ground Water*, **37**(6): 920-923.

Briggs, G.R., E. Roeder, E. Ursin (2007). Nitrogen Impact of Onsite Sewage Treatment and Disposal Systems in the Wekiva Study Area. Florida Department of Health Report.

Canter, L.W. (1996). Nitrates in Groundwater.

Cogger, C.G. and B.L. Carlile (1984). Field Performance of Conventional and Alternative Systems in Wet Soil. *Jour. Environ. Quality* **13**: 137-142.

Cogger, C.G., L.M. Hajjar, C.L. Moe, M.D. Sobsey (1988). Septic System Performances on a Coastal Barrier Island. *Jour. Environ. Quality* **17**(3): 401-407.

Costa, J.E., G. Heufelder, S. Foss, N.P. Milham, and B. Howes (2002). Nitrogen Removal Efficiencies of Three Alternative Septic System Technologies and a Conventional Septic System. *Environment Cape Code* **5**(1): 15-24.

Degen, M.J., C. Hagedorn, D.C. Martens (1991). Denitrification in Onsite Wastewater Treatment and Disposal Systems. *Virginia Water Resources Research Center Bulletin #171*. V. W.R.R. Center.

Drake, V.M. and J.W. Bauder (2005). Ground Water Nitrate-Nitrogen Trends in Relation to Urban Development, Helena, Montana, 1971-2003. *Ground Water Monit. & Remed.* **25**(2): 118-130.

o:\44237-001R008\Wpdocs\Report\Draft

Gerritse, R.G., J.A. Adeney, J. Hosking (1995). Nitrogen Losses from a Domestic Septic Tank System on the Darling Plateau in Western Australia. *Water Research* **29**(9): 2055-2058.

Hantzche, N.N. and E.J. Finnemore (1992). Predicting Ground-Water Nitrate-Nitrogen Impacts. *Ground Water* **30**(4): 490-499.

Harman, J., W.D. Robertson, J.A. Cherry, L. Zanini (1996). Impacts on a Sand Aquifer from an Old Septic System: Nitrate and Phosphate. *Ground Water* **34**(6): 1105-1114.

Heatwole, K.K. and J.E. McCray (2007). Modeling Potential Vadose-zone Transport of Nitrogen from Onsite Wastewater Systems at the Development Scale. *Jour. Of Contam. Hydrology* **91**: 184-201.

Korom, S. (1992). Natural Denitrification in the Saturated Zone: A Review. *Water Resources Research* **28**(6): 1657-1668.

Lapointe, B.E., J.D. O'Connell, G.S. Garrett (1990). Nutrient Couplings Between On-site Sewage Disposal Systems, Groundwater, and Nearshore Surface Waters of the Florida Keys. *Biogeochemistry* **10**: 289-307.

Lowe, K.S., M.B. Tucholke, J.M.B. Tomaras, K. Conn, C. Hoppe, J.E. Drewes, J.E. McCray, J. Munukata-Marr (2007). Influent Constituent Characteristics of the Modern Waste Stream from Single Sources: Final Report. WERF Report #04-DEC-01. 112 pp.

Lowe, K.S., S.M. Van Cuyk, R.L. and Siegrist (2007). Soil Treatment Unit Performance as Affected by Hydraulic Loading Rate and Applied Effluent Quality. *11<sup>th</sup> Individual and Small Community Sewage Systems Conference Proceedings*, Warwick, RI. 10 pp.

Lowe, K.S., S.M. Van Cuyk, R.L. Siegrist, and J.E. Drewes (2008). Field Evaluation of the Performance of Engineered On-Site Wastewater Treatment Units. *Journal of Hydrologic Engineering* August 2008: 735-743.

Lowe, K.S. and R.L. Siegrist (2008). Controlled Field Experiment for Performance Evaluation of Septic Tank Effluent Treatment during Soil Infiltration. *Journal of Environmental Engineering* February 2008: 93-101.

MACTEC (2007). Phase I Report Wekiva River Basin Nitrate Sourcing Study. Florida Department of Health Report.

o:\44237-001R008\wpdocs\Report\Draft

McCray, J.E., K.S. Lowe, M. Geza, J. Drewes, S. Roberts, A. Wunsch, D. Radcliffe, J. Amadore, J. Atoyán, T. Boving, D. Kalen, and G. Loomis (2008). Development of Quantitative Tools to Determine the Expected Performance of Unit Processes in Wastewater Soil Treatment Units: Literature Review. WERF Report # DEC1R06. 182 pp.

McCray, J.E., S.L. Kirkland, R.L. Siegrist, G.D. Thyne (2005). Model Parameters for Simulating Fate and Transport of On-Site Wastewater Nutrients. *Ground Water* **43**(4): 628-639.

McQuillan, D. (2004). Ground-water Quality Impacts from On-Site Septic Systems. National Onsite Wastewater Recycling Association (NOWRA), 13th Annual Conference, Albuquerque, N.M.

Otis, R.J. (2007). Estimates of Nitrogen Loadings to Groundwater from Onsite Wastewater Systems in the Wekiva Study Area: Task 2 Report. Florida Department of Health Report.

Ptacek, C.J. (1998). Geochemistry of a Septic-System Plume in a Coastal Barrier Bar, Point Pelee, Ontario, Canada. *Jour. Contam. Hydro.* **33**: 293-312.

Reay, W.G. (2004). Septic Tank Impacts on Ground Water and Nearshore Sediment Nutrient Flux. *Ground Water* **42**(7): 1079-1089.

Reneau, R.B.J. (1977). Changes in Inorganic Nitrogenous Compounds from Septic Tank Effluent in a Soil with a Fluctuating Water Table. *Jour. Environ. Quality* **6**: 173-178.

Reneau, R.B.J. (1979a). Changes in Concentrations of Selected Chemical Pollutants in Wet, Tile-drained Soil Systems as Influenced by Disposal of Septic Tank Effluents. *Jour. Environ. Quality* **8**: 189-196.

Reneau, R.B.J., C. Hagedorn, M.J. Degen (1989). Fate and Transport of Biological And Inorganic Contaminants from On-site Disposal of Domestic Wastewater. *Jour. Environ. Quality* **18**(2): 135-144.

Ritter, W.F. and R.P. Eastburn (1988). A Review of Denitrification in On-site Wastewater Treatment Systems. *Environ. Pollution* **51**: 49-61.

Robertson, W.D. and D.W. Blowes (1995). Major Ion and Trace Metal Geochemistry of an Acidic Septic-System Plume in Silt. *Ground Water* **33**(2): 275-283.

o:\44237-001R008\Wpdocs\Report\Draft

Robertson, W.D., J.A. Cherry, E.A. Sudicky (1991). Ground-Water Contamination from Two Small Septic Systems on Sand Aquifers. *Ground Water* **29**(1): 82-92.

Rosen, M.R., C. Kropf, K.A. Thomas (2006). Quantification of the Contribution of Nitrogen from Septic Tanks to Groundwater in Spanish Springs Valley, Nevada. USGS Investigations Report # 2006-5206. United States Geological Society.

Sikora, L.J. and R.B. Corey (1976). Fate of Nitrogen and Phosphorus in Soils Under Septic Tank Waste Disposal Fields. *Transactions of ASAE* **19**: 866-870.

Taylor, J.R. (2003). Evaluating Groundwater Nitrates from On-Lot Septic Systems, A Guidance Model for Land Planning in Pennsylvania.

Tinker, J.R.J. (1991). An Analysis of Nitrate-Nitrogen in Ground Water Beneath Unsewered Subdivisions. *Ground Water Monitoring Review Winter*: 141-150.

Tucholke, M.B. (2006). Statistical Assessment of Relationships between Denitrification and Easily Measured Soil Properties: A Simple Predictive Tool for Watershed Modeling. M.S. Thesis, Colorado School of Mines. 187 pp.

Tucholke, M.B., J.E. McCray, G.D. Thyne, and R.M. Waskom (2007). Variability in Denitrification Rates: Literature Review and Analysis. NOWRA 16<sup>th</sup> Annual Technical Education and Exposition Conference, Baltimore, MD. 18 pp.

Uebler, R.L. (1984). Effect of Loading Rate and Soil Amendments on Inorganic Nitrogen and Phosphorus Leached from a Wastewater Soil Absorption System. *Jour. Environ. Quality* **13**: 475-479.

Walker, W.G., J. Bouma, D.R. Keeney, F.R. Magdoff (1973). Nitrogen Transformations During Subsurface Disposal of Septic Tank Effluent in Sands: I. Soil Transformations. *Jour. Environ. Quality* **2**(4): 475-480.

Walker, W.G., J. Bouma, D.R. Keeney, F.R. Magdoff (1973). Nitrogen Transformations During Subsurface Disposal of Septic Tank Effluent in Sands: II. Ground Water Quality. *Jour. Environ. Quality* **2**(4): 521-525.

Wilhelm, S.R., S.L. Schiff, W.D. Robertson (1998). Biochemical Evolution of Domestic Waste Water in Septic Systems: 2. Application of a Conceptual Model in Sandy Aquifers. *Ground Water* **34**: 853-864.

o:\44237-001R008\wpdocs\Report\Draft

Yates, M.V. (1985). Septic Tank Density and Ground-Water Contamination. *Ground Water* **23**(5).

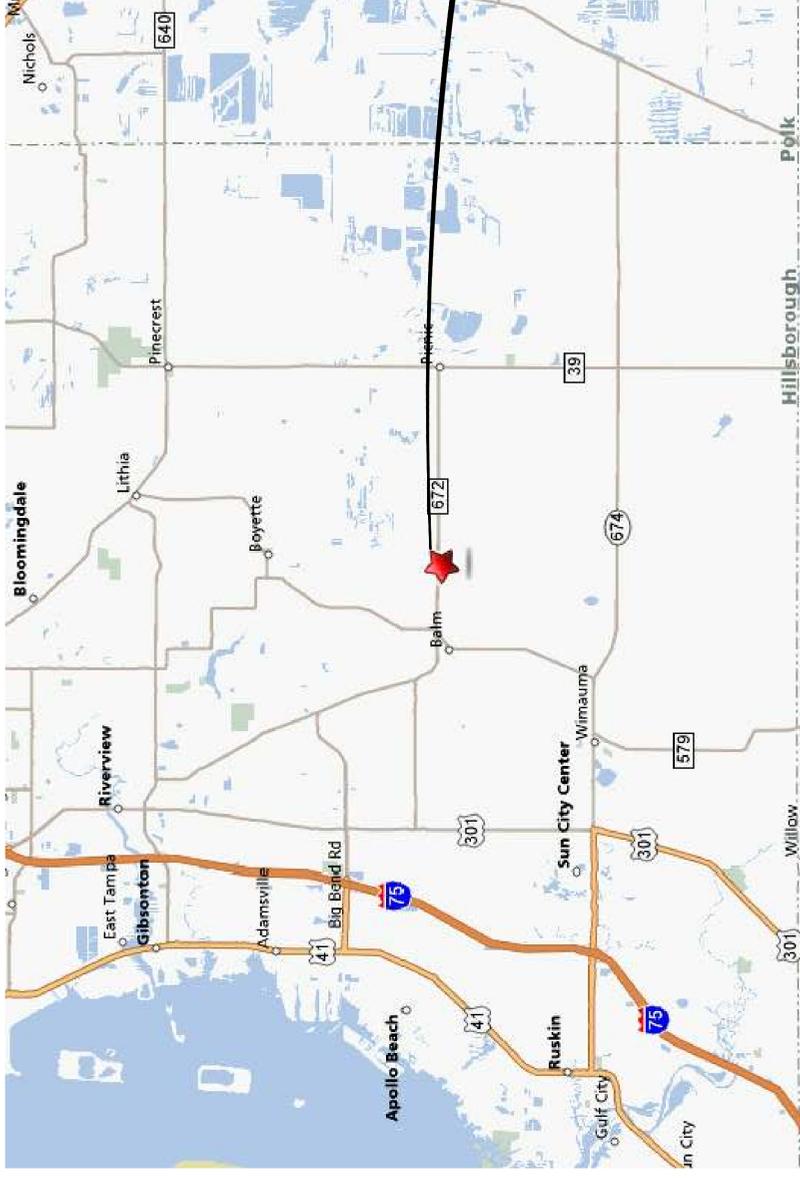
Young, L.J. (2007). Final Report, Task 3: Assess Contributions of Onsite Wastewater Treatment Systems Relative to Other Sources; Wekiva Onsite Nitrogen Contribution Study June 4, 2007. Florida Department of Health Report.

o:\44237-001R008\Wpdocs\Report\Draft

# FLORIDA ONSITE SEWAGE NITROGEN REDUCTION STRATEGIES STUDY 50% DESIGN DOCUMENTS

## LIST OF DRAWINGS

SHEET COUNT	SHEET NUMBER	SHEET TITLE
GENERAL		
1	G-1	COVER SHEET AND INDEX OF DRAWINGS
CIVIL		
2	C-1	EXISTING ONSITE SEPTIC TANK SYSTEM
3	C-2	PROPOSED OVERALL SITE PLAN
4	C-3	PNRS II DETAILS
5	C-4	PNRS II DETAILS
6	C-5	TASK C NITROGEN FATE OF TRANSPORT STUDY DETAILS
7	C-6	WASTEWATER SOURCE COMPONENTS DETAILS
8	C-7	MONITORING PLAN
MECHANICAL		
9	M-1	YARD PIPING PLAN
ELECTRICAL		
10	E-1	ELECTRICAL SITE PLAN



LOCATION MAP

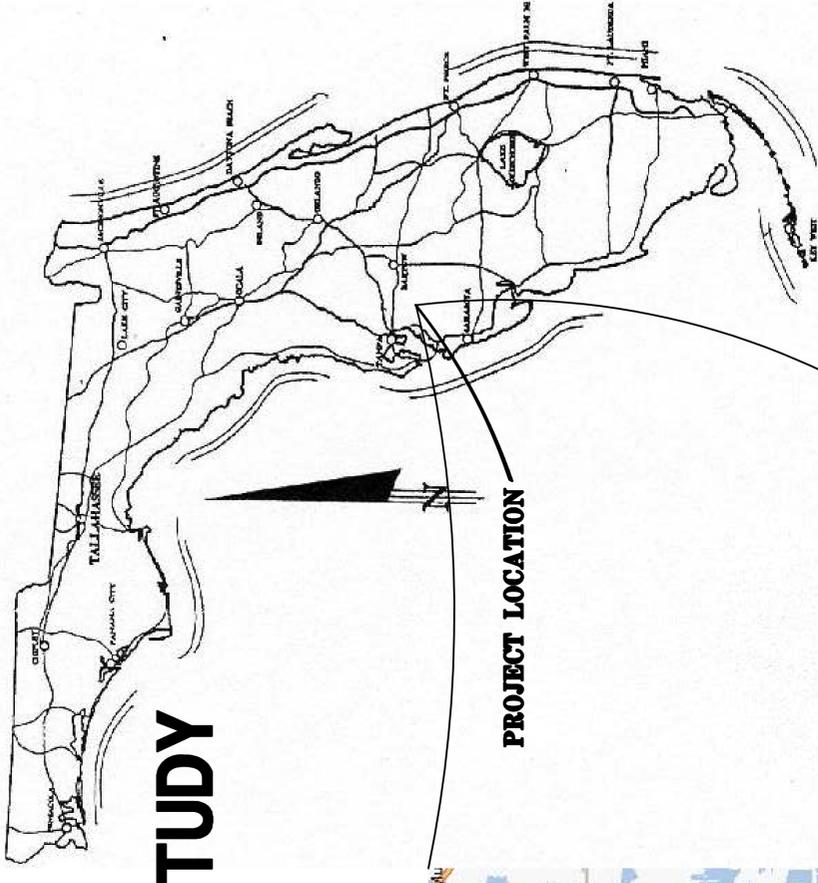
N.T.S.

**HAZEN AND SAWYER**  
Environmental Engineers & Scientists

10002 Princess Palm Ave., Suite 200  
Tampa, Florida 33619  
Certificate of Authorization Number: 2771



FLORIDA DEPARTMENT OF HEALTH  
4052 BALD CYPRESS WAY, BIN A08  
TALLAHASSEE, FLORIDA 32399-1713  
(850) 245-4070



PROJECT LOCATION

PROJECT LOCATION  
UNIVERSITY OF FLORIDA  
GULF COAST RESEARCH AND  
EDUCATION CENTER  
WIMAUMA, FL.

**EXISTING DRAIN FIELD CALCULATION**

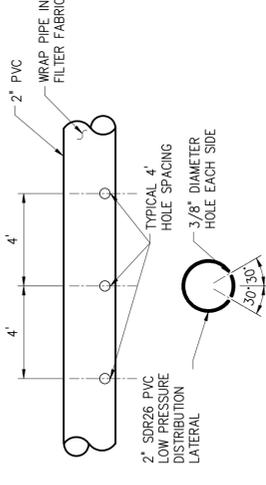
BASED ON UNSUITABLE SUBSURFACE CONDITIONS

LOAD RATE = 0.65  
 TRENCH AREA = 2828 GPD/0.65 = 4351 SF  
 OPEN AREA = 2 x 4351 SF = 8702 SF

**EXISTING SYSTEM FLOW CALCULATION**

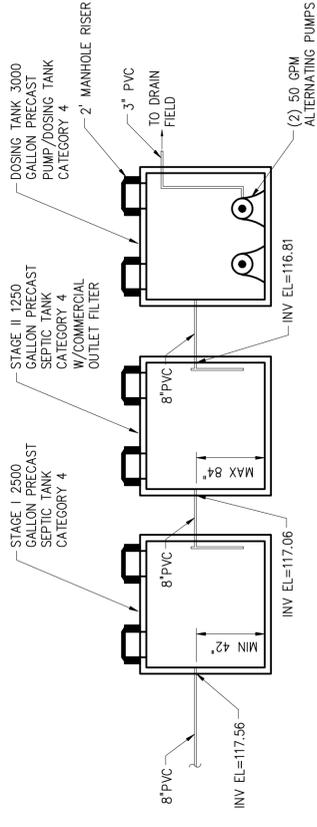
DESIGN FLOW (PER 64E-6.008(1b) TABLE 1)

ADMIN/STAFF (100x15) = 1500 GPD  
 SCHOOL [STUDENT BOARDING TYPE] = 1200 GPD  
 (16 x 75 GPD)  
 ADD FOR SHOWERS (16 x 4) = 64 GPD  
 ADD FOR CAFETERIA (16 x 4) = 64 GPD  
**TOTAL = 2828 GPD**



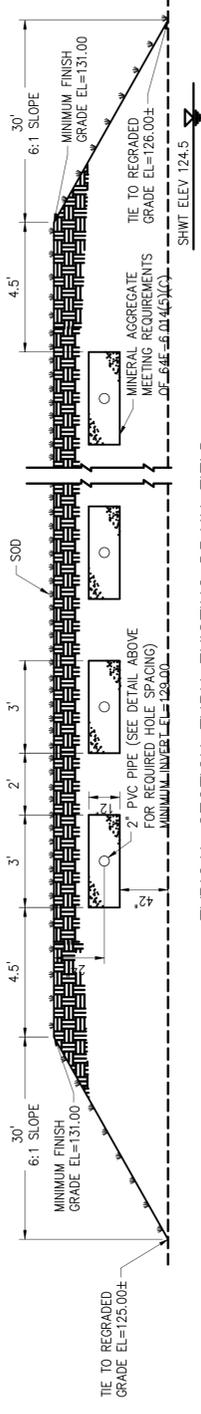
**EXISTING HOLE SPACING DETAIL**

N.T.S.



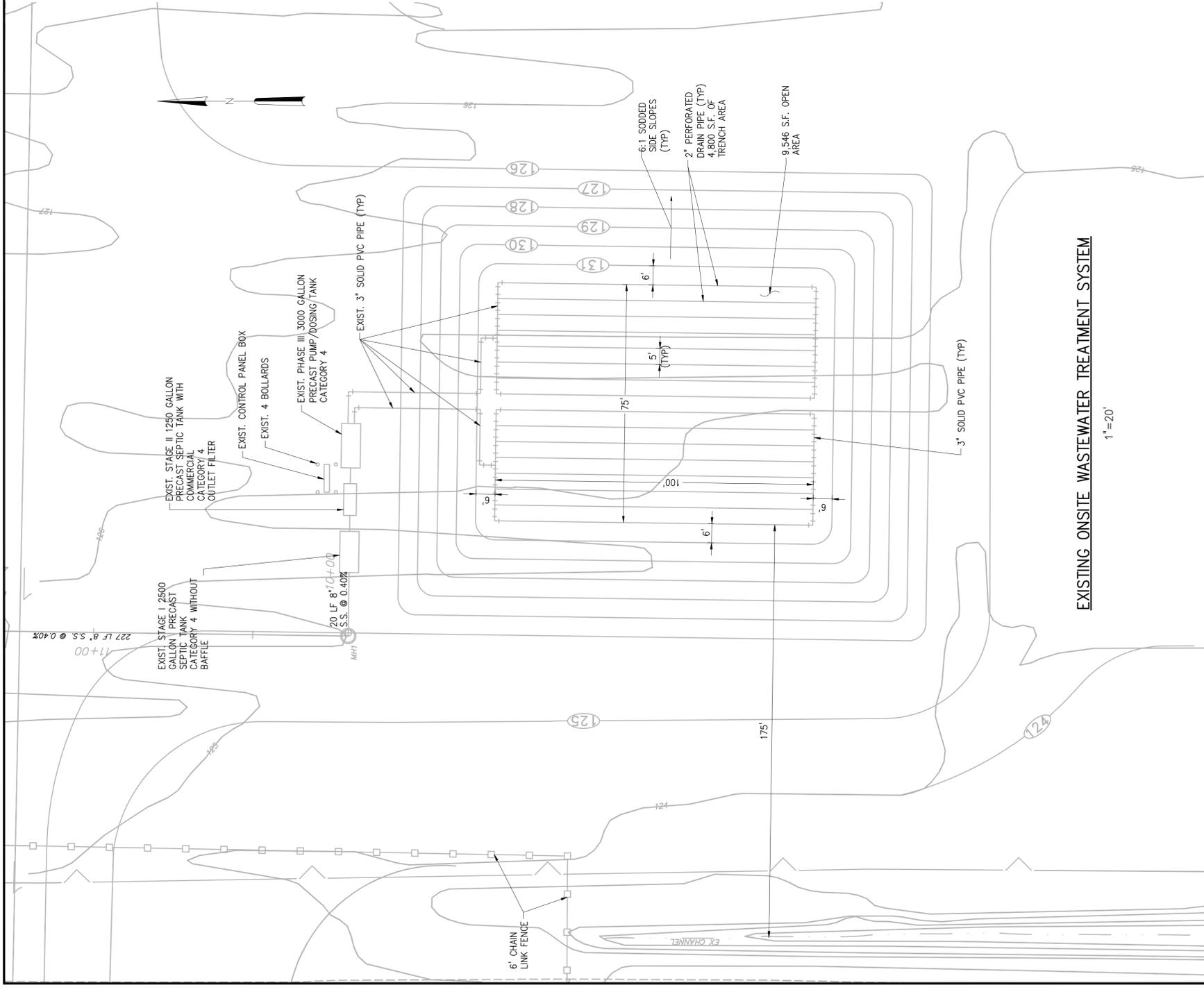
**EXISTING SEPTIC TANK CROSS-SECTION**

N.T.S.



**TYPICAL SECTION THRU EXISTING DRAIN FIELD**

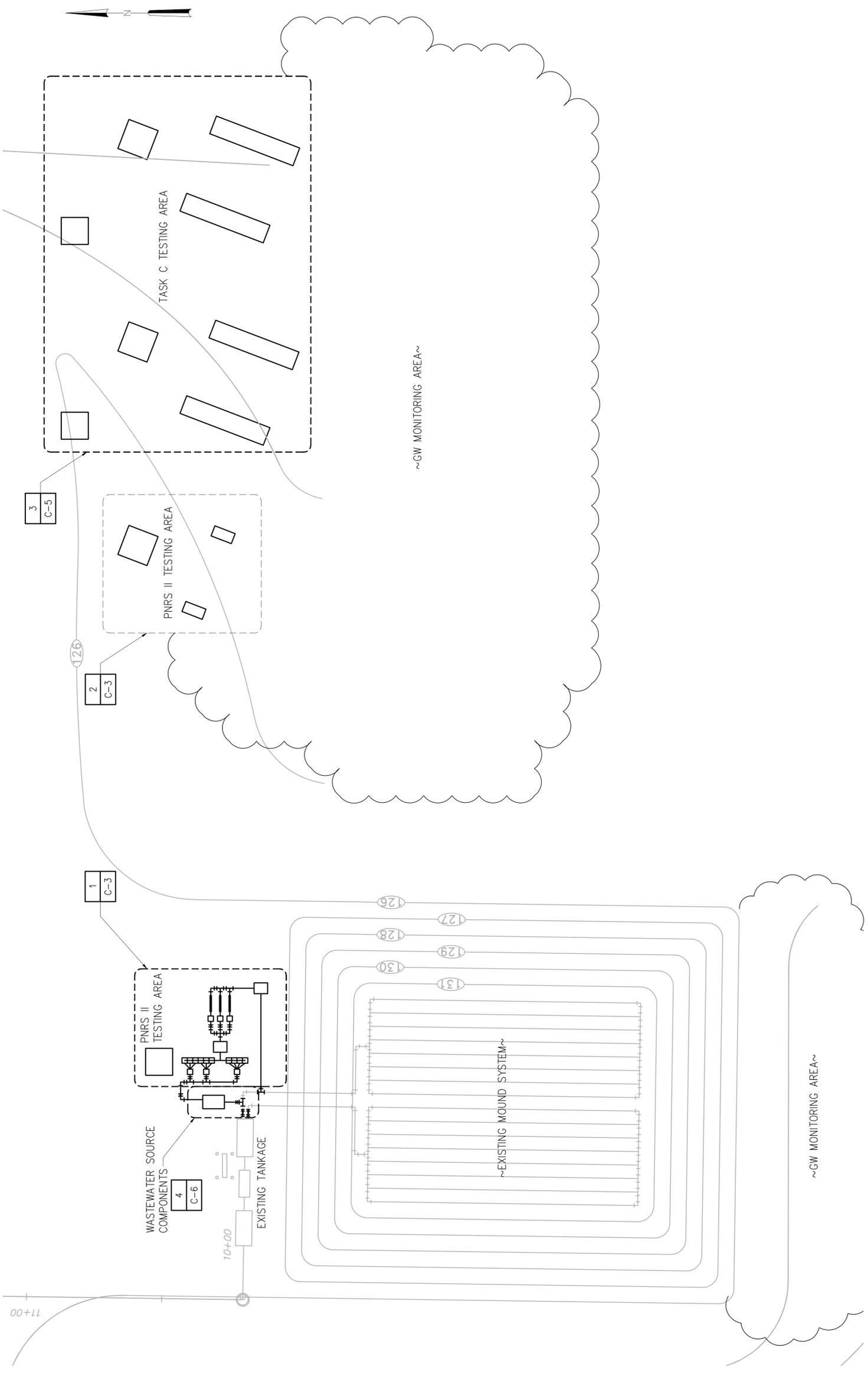
N.T.S.



**EXISTING ONSITE WASTEWATER TREATMENT SYSTEM**

1" = 20'

NO.	ISSUED FOR	DATE	BY	APPROVED	DESIGNED	—
					DRAWN	—
					CHECKED	—
					PROJ. ENGR.	—
					Name: _____ Date: _____ Florida Professional Engineer's Registration Number: _____	
					FLORIDA DEPARTMENT OF HEALTH FLORIDA ONSITE SEWAGE NITROGEN REDUCTION STRATEGIES STUDY EXISTING ONSITE WASTEWATER TREATMENT SYSTEM	
					THE SCALE BAR SHOWN BELOW MEASURES ONE INCH LONG ON THE ORIGINAL DRAWING.	
					DATE	JUNE 2009
					H & S JOB NUMBER	44237-001
					CONTRACT NUMBER	
					DRAWING NUMBER	C-1



PROPOSED SITE PLAN  
1" = 20'

NO.	ISSUED FOR	DATE	BY	APPROVED

DESIGNED \_\_\_\_\_  
 DRAWN \_\_\_\_\_  
 CHECKED \_\_\_\_\_  
 PROJ. ENGR. \_\_\_\_\_  
 APPROVED \_\_\_\_\_  
 Name: \_\_\_\_\_  
 Florida Professional Engineer's Registration Number: \_\_\_\_\_  
 Date: \_\_\_\_\_

**HAZEN AND SAWYER**  
 Environmental Engineers & Scientists  
 10002 Princess Palm Avenue  
 Registry One Building, Suite 200  
 Tampa, Florida 33619  
 Certificate of Authorization Number: 2771

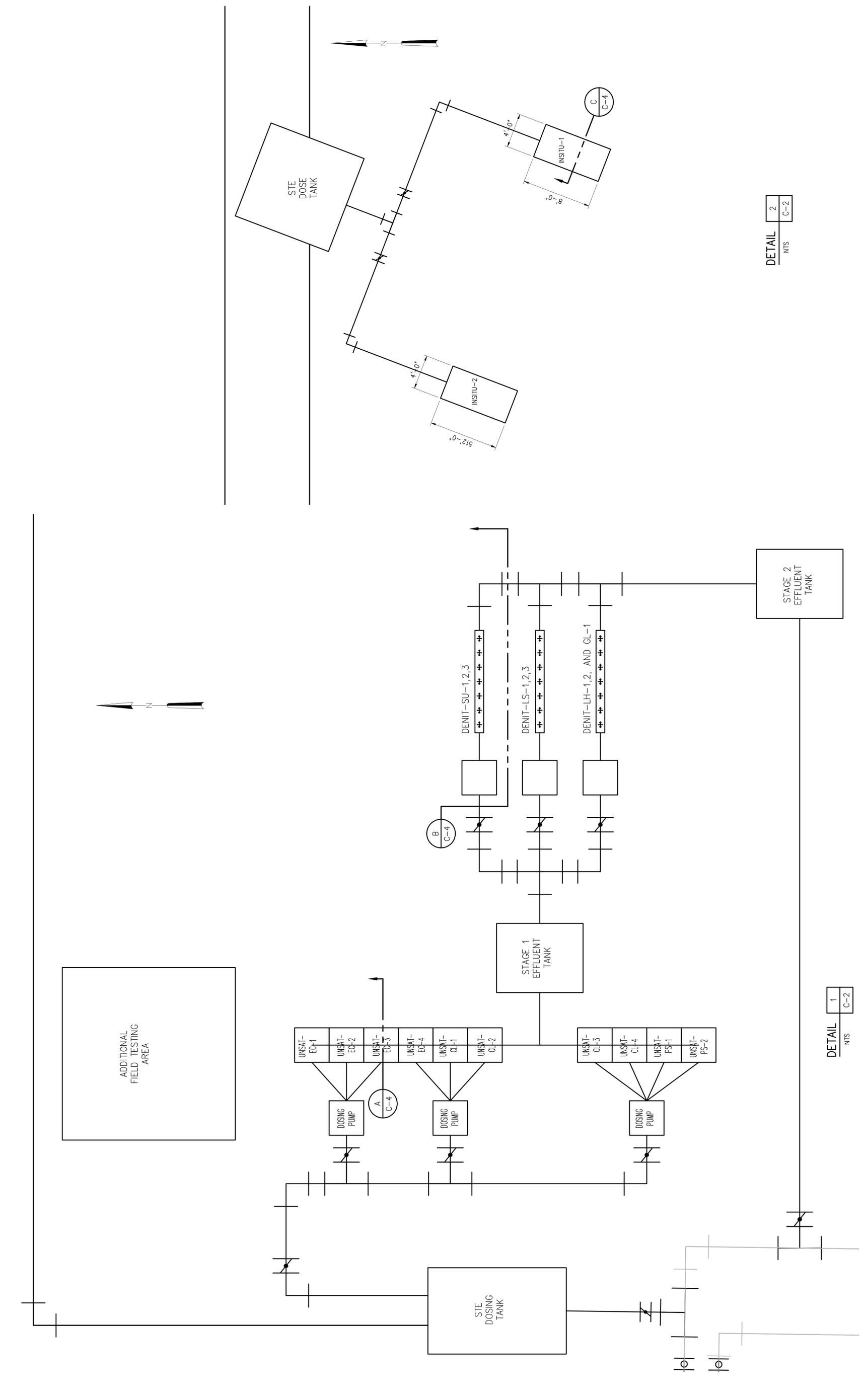
**FLORIDA DEPARTMENT OF HEALTH**  
 4692 BALD CYPRESS WAY, BIN A08  
 TALLAHASSEE, FL 32399-1713  
 (850)-245-4070

**FLORIDA DEPARTMENT OF HEALTH**  
 FLORIDA ONSITE SEWAGE NITROGEN REDUCTION STRATEGIES STUDY

**OVERALL PROPOSED SITE PLAN**

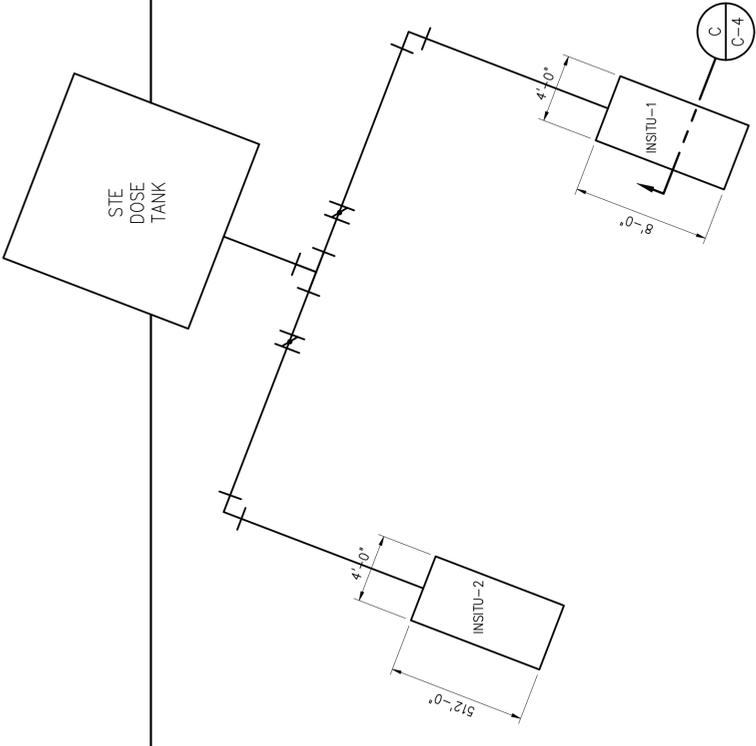
THE SCALE BAR SHOWN BELOW MEASURES ONE INCH LONG ON THE ORIGINAL DRAWING.

DATE: JUNE 2009  
 H & S JOB NUMBER: 44237-001  
 CONTRACT NUMBER: \_\_\_\_\_  
 DRAWING NUMBER: C-2

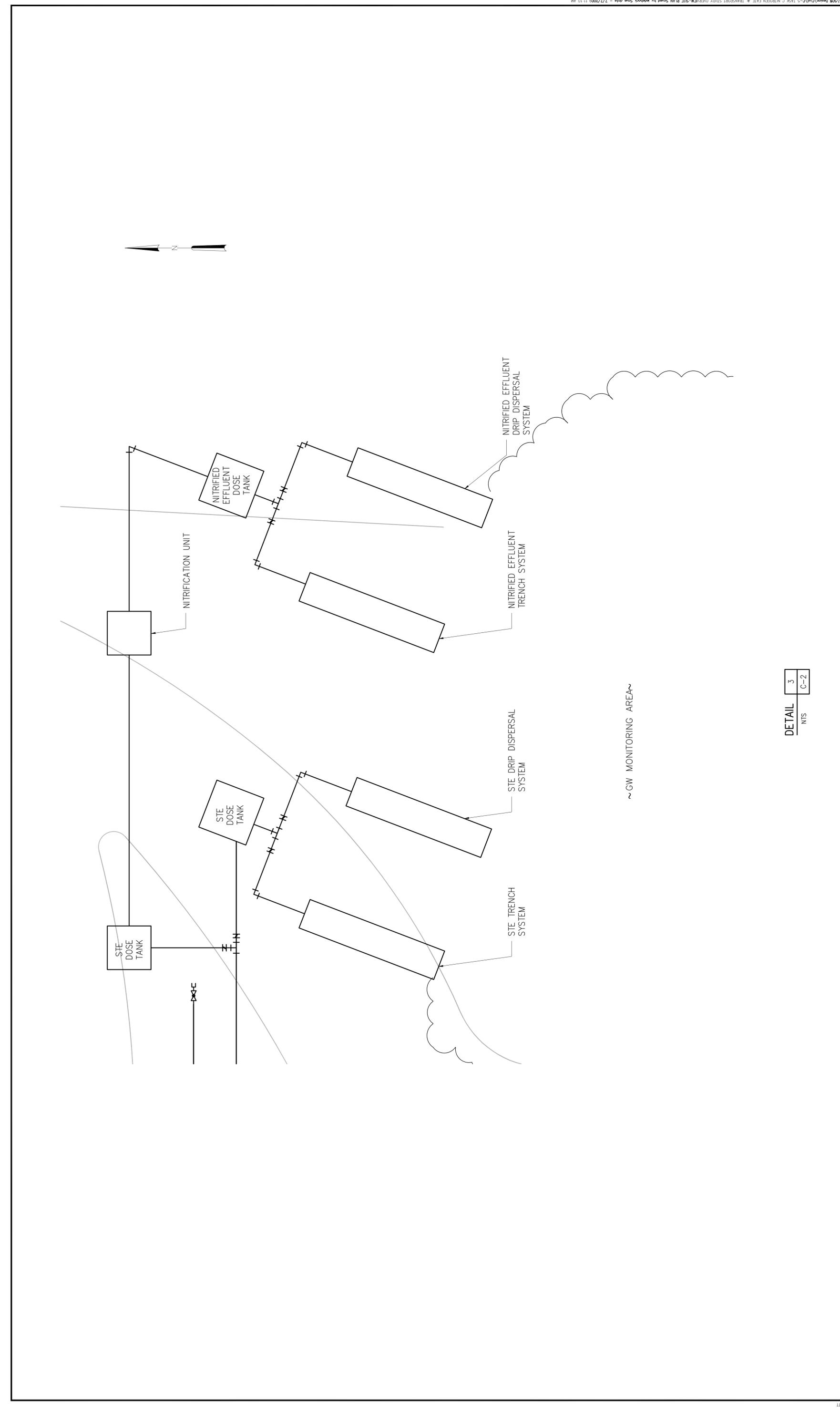


DETAIL 2  
 NTS C-2

DETAIL 1  
 NTS C-2



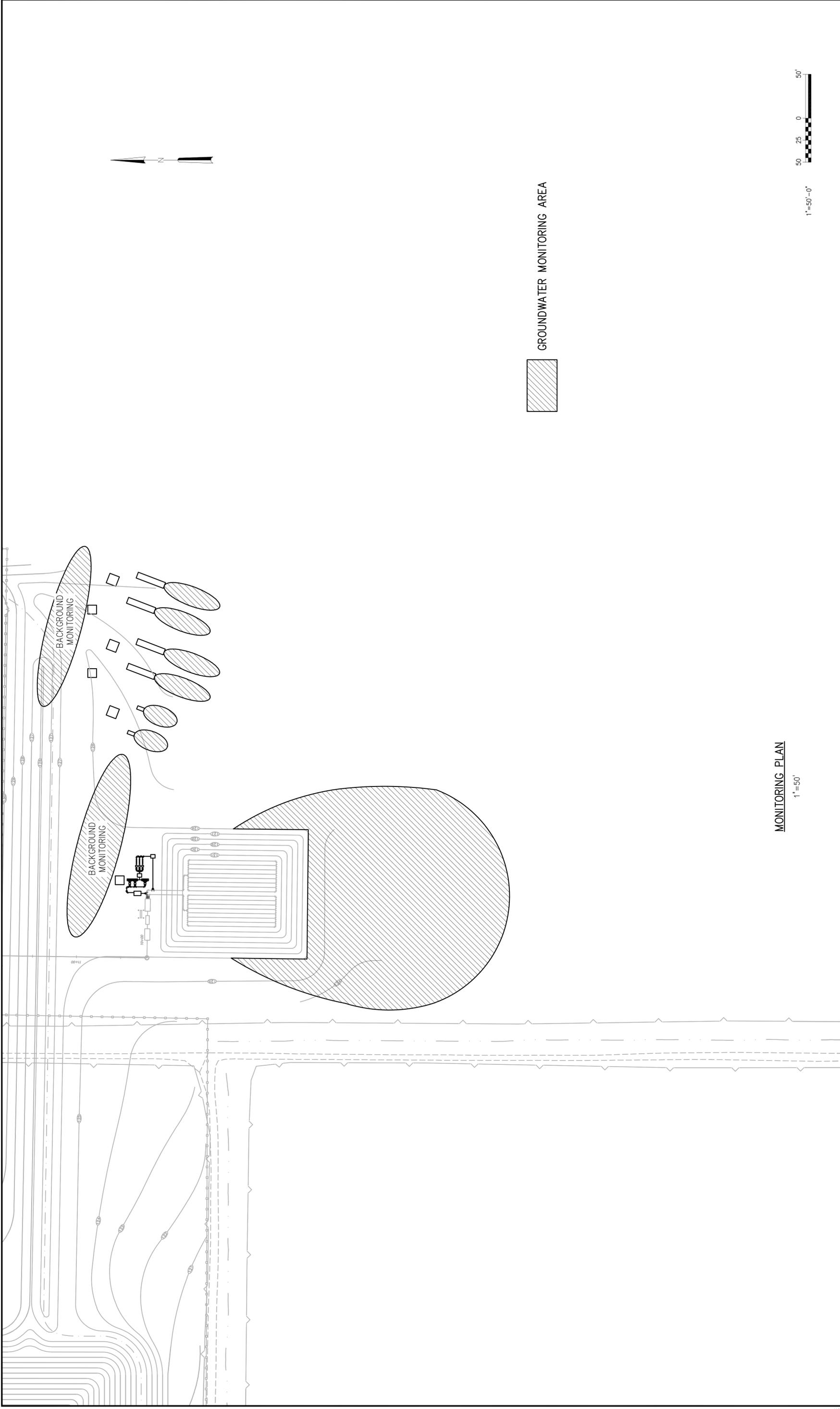




DETAIL 3  
NTS C-2

DESIGNED	---
DRAWN	---
CHECKED	---
PROJ. ENGR.	---
APPROVED	---
NO.	ISSUED FOR
BY	DATE
Name: _____ Date: _____ Florida Professional Engineer's Registration Number: _____	
<b>HAZEN AND SAWYER</b> Environmental Engineers & Scientists 10002 Princess Palm Avenue Registry One Building, Suite 200 Tampa, Florida 33619 Certificate of Authorization Number: 2771	
FLORIDA DEPARTMENT OF HEALTH 4062 BALD CYPRESS WAY, BIN A08 TALLAHASSEE, FL 32399-1713 (850)-245-4070	
FLORIDA DEPARTMENT OF HEALTH FLORIDA ONSITE SEWAGE NITROGEN REDUCTION STRATEGIES STUDY	
TASK C NITROGEN FATE & TRANSPORT STUDY DETAILS	
THE SCALE BAR SHOWN BELOW MEASURES ONE INCH LONG ON THE ORIGINAL DRAWING.	
DATE: JUNE 2009 H & S JOB NUMBER: 44237-001 CONTRACT NUMBER: DRAWING NUMBER: C-5	





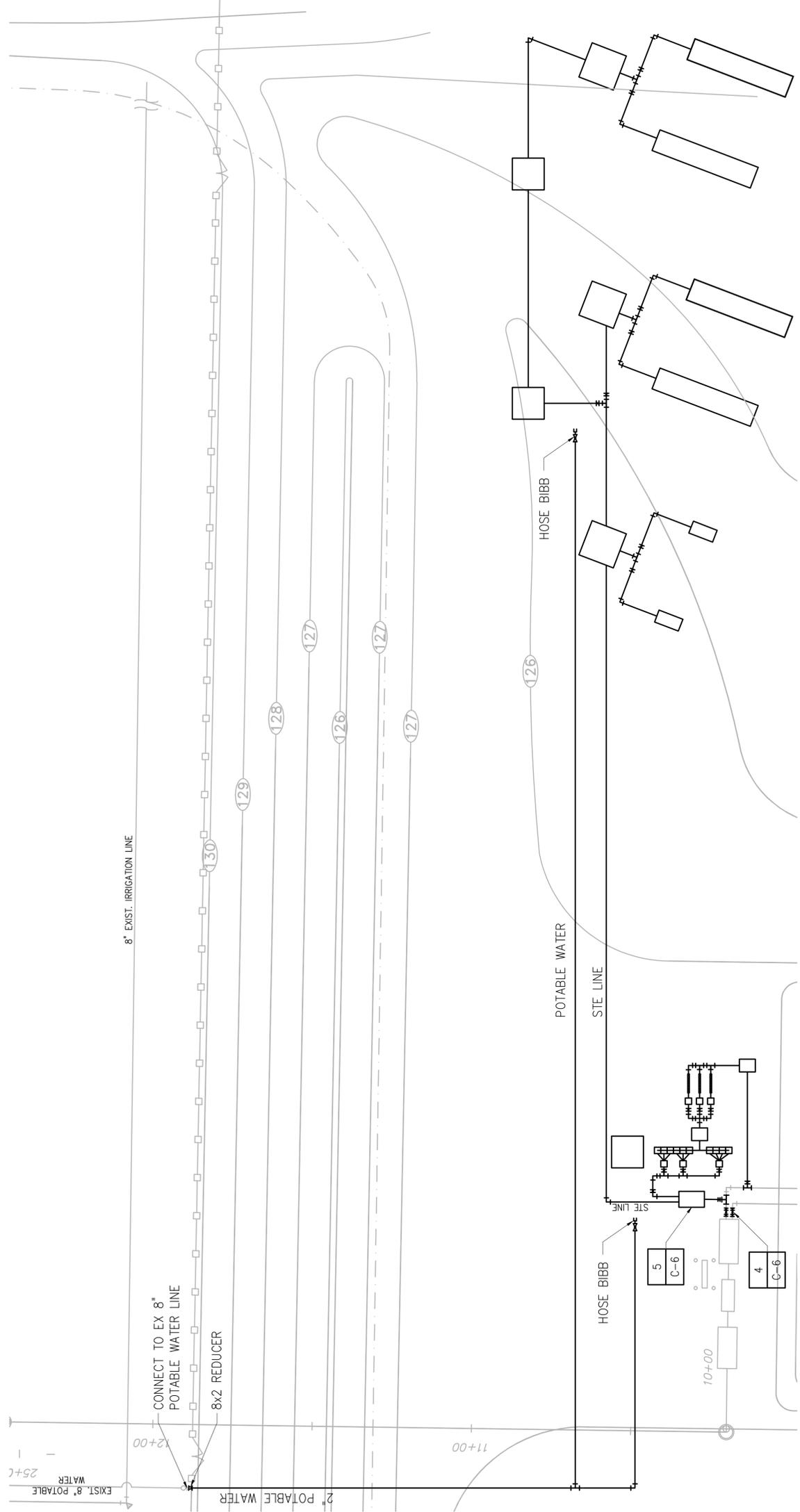
GROUNDWATER MONITORING AREA

MONITORING PLAN

1" = 50'



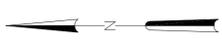
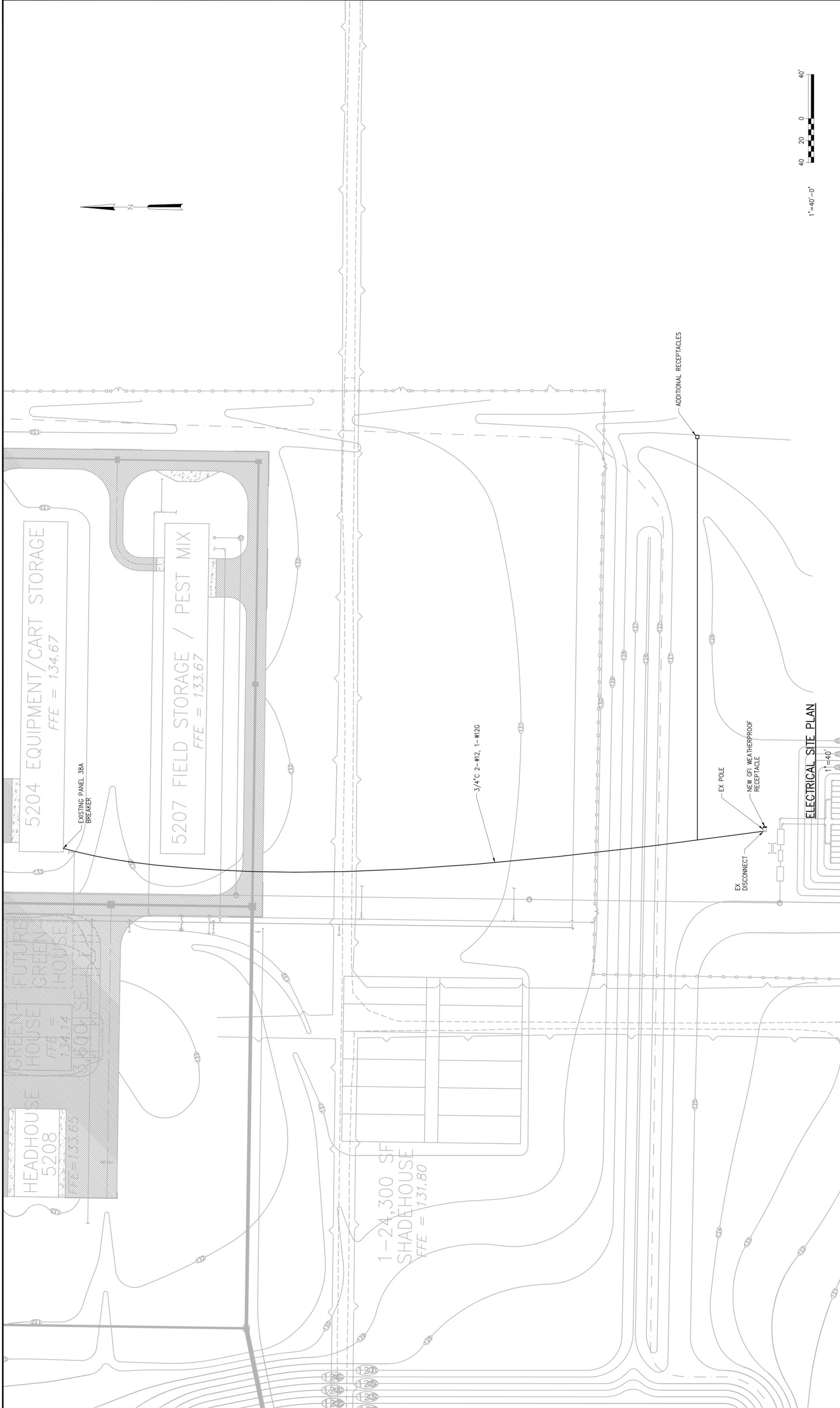
NO. _____ ISSUED FOR _____ DATE _____ BY _____	DESIGNED _____ DRAWN _____ CHECKED _____ PROJ. ENGR. _____ APPROVED _____ Name: _____ Florida Professional Engineer's Registration Number: _____	 <b>HAZEN AND SAWYER</b> Environmental Engineers & Scientists 10002 Princess Palm Avenue Registry One Building, Suite 200 Tampa, Florida 33619 Certificate of Authorization Number: 2771	 <b>FLORIDA DEPARTMENT OF HEALTH</b> 4682 BALD CYPRESS WAY, BIN A08 TALLAHASSEE, FL 32399-1713 (850)-245-4070	FLORIDA DEPARTMENT OF HEALTH FLORIDA ONSITE SEWAGE NITROGEN REDUCTION STRATEGIES STUDY	THE SCALE BAR SHOWN BELOW MEASURES ONE INCH LONG ON THE ORIGINAL DRAWING.	DATE: JUNE 2009 H & S JOB NUMBER: 44237-001 CONTRACT NUMBER: _____ DRAWING NUMBER: C-7
				MONITORING PLAN		



**YARD PIPING PLAN**  
1"=20'



NO.	ISSUED FOR	DATE	BY	APPROVED	DESIGNED _____	DRAWN _____	CHECKED _____	PROJ. ENGR. _____	Name: _____ Date: _____ Florida Professional Engineer's Registration Number: _____
					<p><b>HAZEN AND SAWYER</b> Environmental Engineers &amp; Scientists 10002 Princess Palm Avenue Registry One Building, Suite 200 Tampa, Florida 33619 Certificate of Authorization Number: 2771</p>				



NO.	ISSUED FOR	DATE	BY	DESIGNED	—
				DRAWN	—
NO.	ISSUED FOR	DATE	BY	CHECKED	—
				PROJ. ENGR.	—
Name: _____ Date: _____				Florida Professional Engineer's Registration Number: _____	
<b>Hazen and Sawyer</b> Environmental Engineers & Scientists 10002 Princess Palm Avenue Registry One Building, Suite 200 Tampa, Florida 33619 Certificate of Authorization Number: 2771				<b>FLORIDA DEPARTMENT OF HEALTH</b> 4862 BALD CYPRESS WAY, BIN A08 TALLAHASSEE, FL 32399-1713 (850)-245-4070	
<b>FLORIDA DEPARTMENT OF HEALTH</b> FLORIDA ONSITE SEWAGE NITROGEN REDUCTION STRATEGIES STUDY				THE SCALE BAR SHOWN BELOW MEASURES ONE INCH LONG ON THE ORIGINAL DRAWING.	
<b>FLORIDA DEPARTMENT OF HEALTH</b> FLORIDA ONSITE SEWAGE NITROGEN REDUCTION STRATEGIES STUDY				DATE: JUNE 2009 H & S JOB NUMBER: 44237-001 CONTRACT NUMBER: DRAWING NUMBER: E-1	

ELECTRICAL SITE PLAN



# Florida Onsite Sewage Nitrogen Reduction Strategies Study

Task D.3

**Selection of Existing Data Sets for Model Calibration**

**Final Report**

June 2009

44237-001

**HAZEN AND SAWYER**  
Environmental Engineers & Scientists

In association with



**AET**  
Applied Environmental Technology

**OTIS  
ENVIRONMENTAL  
CONSULTANTS, LLC**

# Florida Onsite Sewage Nitrogen Reduction Strategies Study

## TASK D.3 FINAL REPORT

### Selection of Existing Data Sets for Model Calibration

#### Prepared for:

Florida Department of Health  
Division of Environmental Health  
Bureau of Onsite Sewage Programs  
4042 Bald Cypress Way Bin #A-08  
Tallahassee, FL 32399-1713

FDOH Contract CORCL

**June 2009**

#### Prepared by:

**HAZEN AND SAWYER**  
Environmental Engineers & Scientists

In Association With:



## Selection of Existing Data Sets for Model Calibration

---

In support of the Florida Onsite Sewage Nitrogen Reducing Strategies (FOSNRS) Study, a simple tool evaluating flow and transport in groundwater to produce output predictions for nitrogen concentration or mass flux is being developed. The overall purpose of this tool is to simulate aquifer nitrate concentrations down gradient from onsite sewage treatment and disposal systems (OSTDS). The modeling tool will be a spreadsheet-implemented, multidimensional, transient analytical solution to the advection dispersion equation with reactions. While the aquifer model to be used has not yet been formally determined, it will likely be a modification or combination of the models presented by Domenico (1986) and Galva (1987). These models assume a planar source of nitrate below or at the water table, and account for relevant advection-dispersion-reaction processes. This initial version of the model will not contain a detailed expression for vadose zone transport processes. Rather, the model will use a simple term for vadose zone attenuation that will be determined by comparing source loading rates (the OSTDS source term) to concentrations at the water table (the aquifer source term). The development of a detailed vadose zone soil treatment model will be completed in the future.

The output of the aquifer model will be concentration as a function of  $x$ ,  $y$ ,  $z$ , and time,  $C(x, y, z, t)$ , where  $C(x, y, z, t)$  is the computed concentration at a given point defined spatially by  $(x, y, z)$  and temporally by time  $t$ . For this model, the required input is the initial source concentration ( $C_0$ ), groundwater velocity ( $v$ ), dispersivity ( $\alpha$ ) terms that are multiplied by velocity to calculate dispersion coefficients, first-order nitrogen transformation rate constants ( $\lambda$ ) and rectangular source dimensions. Measured concentrations at the water table below the OSTDS surface footprint will be the primary means to determine  $C_0$  (initial concentration) for model input. However, we will also use loading information from the surface OSTDS sources, along with simple formulations to account for unsaturated soil treatment, because this is information that is more typically available. It is a goal of future work associated with this project to develop a robust soil treatment model that can be linked to an aquifer model. However, performance evaluation and calibration are not being conducted for that linked model as part of this task.

To evaluate the aquifer model performance, we endeavor to conduct simulations of flow and transport at actual sites, and compare the model results to existing field data. This model evaluation process can be useful for understanding treatment processes in the

vadose zone, but calibration of a vadose-zone treatment model is not an objective of this task. Candidate data sets should provide:

- groundwater velocity, or at a minimum, parameters that allow for calculation of velocity, such as hydraulic conductivity, porosity, and measured head values for determining site-specific horizontal gradient (porosity values could be estimated and this would not introduce unacceptable error);
- temporal and spatial concentrations at monitoring points downgradient from the source; and
- source information, (i.e., number of septic tanks and location, surface-expression shape of the source zone, and loading rates for the septic systems. Loading rates can be estimated from statistical distributions provided by McCray et al. (2005), but would add additional uncertainty to the model-performance evaluation).

The process of identifying candidate data sets for the model evaluation involved analyzing over 25 studies and reports to ascertain if the desired data listed above were available in the study. In some cases, studies were identified that at the very least contained appropriate nitrate concentration data and information concerning the site characteristics. In these cases, it would be possible to use literature values for other parameters not specifically measured. In choosing primary data sets, preference was given to studies with the most complete characterization and monitoring.

Parameters for which no data are available can be estimated via calibration. However, only a certain number of parameters can be uniquely calibrated (the exact number depends on the data set). If all model parameter values cannot be confidently obtained via available data, unique calibration, or independent estimation, then a model uncertainty analysis can be implemented to evaluate the model performance.

The literature review completed in support of model development yielded a number of studies and reports that contain data sets that likely satisfy the above criteria (although, one cannot be certain until the model-performance evaluation is completed). The top candidate study is described first, followed by several alternates. While the primary study data are adequate for model calibration, the model can be verified against another data set for a site with different characteristics. In this way, the ability of the model to simulate flow and transport for various conditions can be demonstrated.

## Primary Candidate Studies

### 1. Indian River Lagoon Study

This study was initiated to assess the impacts to water quality from OSTDS in the Turkey Creek sub-basin in the Indian River Lagoon. Specifically, the study sought to determine the impacts from wastewater practices to nearby canals. Three sites that consisted of residences that used OSTDS were used for sampling. This included the sampling of 25 monitoring wells, 12 piezometers and surface water quality points performed over 14 different events between February 1990 and March 1992. Additionally, two separate tracer tests using bromide were done, and the septic tank effluent was measured for quality and quantity. The resulting data set includes data for nitrate concentrations in groundwater and surface water, STE data and hydraulic parameters. Additional data is available for precipitation and seepage, which is useful for estimating source loading.

### 2. St. George Island

In the St. George study (Corbett and Iverson, 1999; Corbett et al., 2002), groundwater flow was monitored down gradient of three residences served by OSTDS. Conservative tracers were used to determine groundwater flow velocity, hydraulic conductivity, and dispersivity at the three sites (Corbett et al., 2000). Twelve monitoring wells and 13 multi-level samplers were installed at Site 1, seven wells and eight multi-level samplers at Site 2, and seven wells and seven samplers at Site 3. The total groundwater flux into the adjacent bay was estimated using two different techniques, which agreed well with each other. Nutrient concentrations (total nitrogen, ammonia, nitrite and nitrate) were measured in all samples collected. Nutrient samples were collected monthly for over one year at all sites. In addition, rainfall data was collected over the sampling period (Corbett and Iverson, 1999; Corbett et al., 2002).

### 3. St. Johns County, Florida

The objective of this study was to delineate the impacts to groundwater below and downgradient of an OSTDS in Florida. The primary constituents of concern were household cleaning products, but data was also collected for nitrate, chloride, and subsurface characterization. The site consisted of a single family home in St. Johns County, Florida. Subsurface characterization data is available through well logs and completion data, and grain size analysis. Groundwater was monitored at multiple monitoring points in three separate events and consisted of groundwater elevation data (providing gradient and flow direction), and constituent analysis for nitrate, chloride, and phosphorus among others. Additional data was collected for STE quality and quantity over several events as well.

#### **4. Wekiva Nitrogen Source Study**

A study designed to estimate the nitrogen loading contribution from OSTDS in the Wekiva watershed in central Florida was performed by multiple researchers for the Florida Department of Health (FDOH) (Briggs et al., 2007; Roeder, 2008; Aley IV et al., 2007). The overall objectives of the study is to assess the significance of the nitrogen impacts to groundwater and surface water as part of the Wekiva Parkway and Protection Act and recommend strategies to reduce these impacts. Groundwater field data were collected from three sites in the study area. Nutrient loading from OSTDS into the groundwater were calculated, after which it was determined whether OSTDS were a significant source of nitrogen to groundwater relative to other sources (Briggs et al., 2007; Roeder, 2008). Three nitrogen plumes were also investigated in great detail. Several monitoring wells/piezometers were installed and samples for nitrogen species and other parameters were collected (Aley IV et al., 2007). However, sampling was limited to one event per site.

#### **Alternate Candidate Studies**

##### **1. Lake Okeechobee**

The Lake Okeechobee study investigated the impact of high-density OSTDS installations on water quality along the northern periphery of Lake Okeechobee in central Florida. Monitoring wells, lysimeters and piezometers were installed at several sites for the investigation of the groundwater flow, height of water table, and quality of leachate from OSTDS as it entered the aquifer and down gradient of the soil treatment unit. Both nitrogen and phosphorous data were collected over the course of one year (ESE, 1993).

##### **2. Other Studies**

Additional studies were identified as potential candidates for verification and calibration data. Other studies described above were chosen as primary candidates, mostly because the research was conducted in Florida. However, other available studies contain data that could be used for model verification. These include studies by LaPointe et al (1990), Robertson et al (1991) and Morgan et al. (2008). An additional study by Katz et al (publication pending) could provide a useful database, however the study is currently under USGS review and the complete dataset is not available at this time.

#### **Recommendation**

In identifying studies for calibration data sets, preference was given to studies performed in Florida. Additionally, most of the data sets contain data that has been collected over a period of time acceptable for verification of the modeling tool's ability to simulate tran-

sient conditions (in most cases, at least one year). Table 1.1 summarizes the types of data available in the primary and alternate data sets.

The Indian River Lagoon study and the St. George Island studies are the leading candidates as they both contain extensive data related to the input and calibration parameters needed to test and validate the modeling tool being developed. Temporal data is available for hydraulic parameters, source orientation and loading, and source area and down gradient nitrogen concentrations in *x*, *y*, *z* orientations. Both studies have multiple sampling points both downgradient and in the source areas.

Our goal is to use the primary data sets for model performance evaluation. However, one cannot predict whether model calibration will be successful for any particular model or data set until the process is complete. In this case, another data set may be applied to the model for further verification and testing. For this reason, several data sets were chosen as alternatives in the event more data is required to assess model performance. This memo satisfies the deliverable D.3 to choose a data set for preliminary model-performance evaluation of a simple aquifer-modeling tool.

**Table 1.1**  
**Summary of type of data available for previous Florida Nitrogen Studies**

Study	Hydraulic Parameters				Nitrogen Conc. (x,y,z,t)	Source Data			Additional Data
	GW Velocity	K	Gradient	Porosity		OSTDS Location	Footprint	Loading Rates	
Indian River Lagoon Study (Aley, et al., 2007)	X	X	X	X	X	X	X	X	climate data, surface water data, tracer test data
St. George Island (Corbett, et al 1999)	X	X	X	X	X	X	X	X	precipitation, tracer study results
St. Johns County, Florida	X	X	X	X	X	X	X	X	tracer test results, grain size analysis
Wekiva Nitrogen Study Area (Aley, et al 2007)	X	X	X	X		X	X	X	climate data
Lake Okeechobee (ESE 1993)	X	X	X	LV	X	X	X	X	surface water data
Florida Keys (LaPointe et al 1993)	n/a	n/a	n/a	LV			X	n/a	gw flow*, surface water data
La Pine, Oregon (Morgan, et al 2007)	X	X	X	X	X	X	X	X	
Ontario, Canada (Robertson et al, 1991)	n/a	X	X	LV	X	X	X	X	tracer study, model-generated dispersivity values

*n/a = not available*

*K = hydraulic conductivity*

*LV = literature values based on site characteristics and/or CFD*

*\* measured with in situ flow meter*

## References

Aley IV, W.C., M. Mark Mechling, G.S. Pastrana, and E.B. Fuller. 2007. Multiple nitrogen loading assessments from onsite wastewater treatment and disposal systems within the Wekiva river basin; Wekiva study area. Prepared for State of Florida, Department of Health, Tallahassee, FL, by Ellis & Associates, Inc., Jacksonville, FL.

Ayers Associates. 1991. Definition of Contaminant Plumes from an Onsite Sewage Disposal System Serving an Individual Home in Florida; Phase 1 Report. Prepared for the Soap and Detergent Association.

Ayers Associates. 1993. An Investigation of Groundwater Contamination from an Onsite Sewage Disposal System (OSDS) at a Single Family Home in Florida; Phase 2 Report. Prepared for the Soap and Detergent Association.

Ayers Associates. 1993. *An Investigation of the Surface Water Contamination Potential from On-site Sewage Disposal Systems (OSDS) in the Turkey Creek Sub-basin of the Indian River Lagoon Basin; Final Report*. Prepared for St. Johns River Water Management District SWIM Project IR-1-110.1-D under Florida Department of Health and Rehabilitative Services (HRS) contracts No. LP114 and LP596.

Ayers Associates. 1994. *An Investigation of Groundwater Contamination from an Onsite Sewage Disposal System (OSDS) at a Single Family Home in Florida; Phase 3 Status Report*. Prepared for the Soap and Detergent Association.

Briggs, G.R., E. Roeder, and E. Ursin. 2007. *Nitrogen impact of onsite sewage treatment and disposal systems in the Wekiva study area*. Prepared for Florida Department of Health, Tallahassee, FL, by Bureau of Onsite Sewage Programs, Division of Environmental Health.

Corbett, D.R., K. Dillon, and W. Burnett. 2000. Tracing groundwater flow on a barrier island in the North-east Gulf of Mexico. *Estuarine, Coastal and Shelf Science* 51 (2):227-242.

Corbett, D.R., K. Dillon, W. Burnett, and G. Schaefer. 2002. The spatial variability of nitrogen and phosphorus concentration in a sand aquifer influenced by onsite sewage treatment and disposal systems: A case study on St. George Island, Florida. *Environmental Pollution* 117 (2):337-345.

Corbett, D.R., and R. Iverson. 1999. Groundwater and nutrient dynamics on a strip barrier island served by on-site sewage treatment and disposal systems in the northern

Gulf of Mexico. Contract No C0005. Prepared for Florida Department of Health, Tallahassee, FL, by Florida State University, Department of Oceanography.

ESE. 1993. *Onsite sewage disposal system research on the northern periphery of Lake Okeechobee*. Contract No. LP555. Prepared for State of Florida, Department of Health and Rehabilitative Services, Tallahassee, FL, by Environmental Science & Engineering, Inc, Gainesville, FL.

LaPointe, B.E., J.D. O'Connell, and G.S. Garrett. 1990. Nutrient couplings between on-site sewage disposal systems, groundwater, and nearshore surface waters of the Florida Keys. *Biogeochemistry* 10: 289-307.

Lindberg, J.W. and R.P. Elmore. 2005. Interim-status RCRA groundwater monitoring plan for the 216-A-10, 216-A-36B, and 216-A-37-1 PUREX cribs. DOE Report PNNL-11523 Rev. 1.

McCray, J.E., S.L. Kirkland, R.L. Siegrist, and G.D. Thyne. 2005. Model parameters for simulating fate and transport of on-site wastewater nutrients. *Ground Water* 43(4): 628-639.

Morgan, D.S., S.R. Hinkle, and R.J. Weick. 2007. Evaluation of approaches for managing nitrate loading from on-site wastewater systems near La Pine, Oregon: U.S. Geological Survey Scientific Investigations Report 2007-5237, 64 p.

Robertson, W.D., J.A. Cherry, and E.A. Sudicky. 1991. Ground-water contamination from two small septic systems on sand aquifers. *Ground Water* 29(1): 82-92.

Roeder, E. 2008. *Revised estimates of nitrogen inputs and nitrogen loads in the Wekiva study area*. Prepared for Florida Department of Health, Tallahassee, FL, by Bureau of Onsite Sewage Programs, Division of Environmental Health.



# Florida Onsite Sewage Nitrogen Reduction Strategies Study

## Task D

## Selection of Existing Data Sets for Calibration

### Draft Report

June 2009

44237-001

**HAZEN AND SAWYER**  
Environmental Engineers & Scientists

In association with



**AET**  
Applied Environmental Technology

**OTIS  
ENVIRONMENTAL  
CONSULTANTS, LLC**

# Florida Onsite Sewage Nitrogen Reduction Strategies Study

## TASK D DRAFT REPORT

### Literature Review of Nitrogen Fate and Transport Modeling

#### Prepared for:

Florida Department of Health  
Division of Environmental Health  
Bureau of Onsite Sewage Programs  
4042 Bald Cypress Way Bin #A-08  
Tallahassee, FL 32399-1713

FDOH Contract CORCL

**June 2009**

#### Prepared by:

**HAZEN AND SAWYER**  
Environmental Engineers & Scientists

In Association With:





## Table of Contents

---

Section 1.0	Introduction .....	1-1
1.1	Project Background.....	1-1
1.2	Nitrogen in Ground Water; Conceptual Considerations..	1-2
Section 2.0	Literature Review .....	2-1
2.1	Modeling Research Summary.....	2-1
2.2	Vadose Zone Models .....	2-2
2.3	Saturated Zone Flow and Transport Models.....	2-6
2.4	Combined Vadose Zone and Saturated Zone Models..	2-14
2.5	Nitrogen Modeling at the Watershed Scale .....	2-17
2.6	General Fate and Transport Modeling .....	2-20
2.7	Parameter Estimation .....	2-24
2.8	Nitrification and Denitrification Modeling.....	2-26
Section 3.0	Discussion and Analysis .....	3-1
Section 4.0	Conclusions .....	4-1
Section 5.0	References .....	5-1

O:\44237-001R005\Wpdocs\Report\Draft

**List of Figures**

Figure 1-1: Nitrogen Processes Occurring in a Typical OWTS..... 1-3  
(Heatwole and McCray 2007)

Figure 2-1: Modeling Results from Mironenko and Pachepsky (1984) .....2-4

Figure 2-2: Results of Modeling with Comparison to Field Data .....2-8  
(Hantzche and Finnemore 1992)

Figure 2-3: Conceptual Model for Mass-balance Approach.....2-10  
(Taylor, 2003)

Figure 2-4: Conceptual Schematic of the Model .....2-15  
(Mehran et al. 1983-1984)

Figure 2-5: Hypothetical Aquifer for Model Example Simulation .....2-16  
(Mehran et al. 1983-1984)

Figure 2-6: Example Output for Unsaturated Zone .....2-16  
(Mehran et al. 1983-1984)

Figure 2-7: Conceptual Schematic of MIKE SHE Model.....2-17  
(Refsgaard, 1999)

Figure 2-8: Model Output for HPS Model Simulation Two .....2-22  
(Galya 1987)

Figure 2-9: Source Orientations .....2-23  
(Leij et al. 1991)

Figure 2-10: Example Cumulative Frequency Distribution.....2-25  
(McCray et al. 2005)

o:\44237-003\Wpdocs\Report\Draft



## Section 1.0 Introduction

---

### 1.1 Project Background

As a result of the widespread impacts of nitrogen on groundwater, the management of nitrogen sources, particularly onsite wastewater treatment systems (OWTS), is of paramount concern for the protection of human health. Mathematical models of groundwater flow and solute transport historically have been utilized for simulating concentration and plume distribution of contaminants and assisting in management practices by providing representations of groundwater behavior. An appropriate model can provide guidance for land-use planning and remedial approaches. As part of the Florida Onsite Sewage Nitrogen Reduction Strategies (FOSNRS) Study, a groundwater flow and transport modeling tool is being developed to provide a management tool for potential impacts of nitrogen from OWTS. The primary objectives of the model development are to:

- create a user-friendly flow and transport model (i.e., a programmed Microsoft Excel spreadsheet), and
- develop a model that can be used to predict nitrogen concentrations and mass flux/loading at a point or plane down-gradient of an OWTS or systems assuming the model:
  - adequately represents the identified processes that govern the fate and transport of OWTS-generated nitrogen in groundwater, and
  - should also be capable of simulating temporally variable source input and account for non-uniform spatial distribution of OWTS sources.

The following presents a literature review to assess the current state-of-knowledge regarding the mathematical modeling of nitrogen and nitrate movement and distribution in groundwater related to OWTS. The review will attempt to identify existing models that may satisfy the above-stated objectives, modeling approaches that can be useful, relevant input and calibration parameters and the level of effort required in developing a modeling tool. As part of the literature review, a database of the references was developed in conjunction with this summary report. This database (see separate Excel file “CSM\_D-1 Nitrogen Modeling Studies”) includes a summary table of the relevant fea-

C:\4237-001R005\pdocs\Report\Draft

tures and parameters of each modeling study. As a result of the large number of identified sources, some modeling studies not deemed valuable to this effort are mentioned in this report but are not described in detail and the reader is directed to the database for further information.

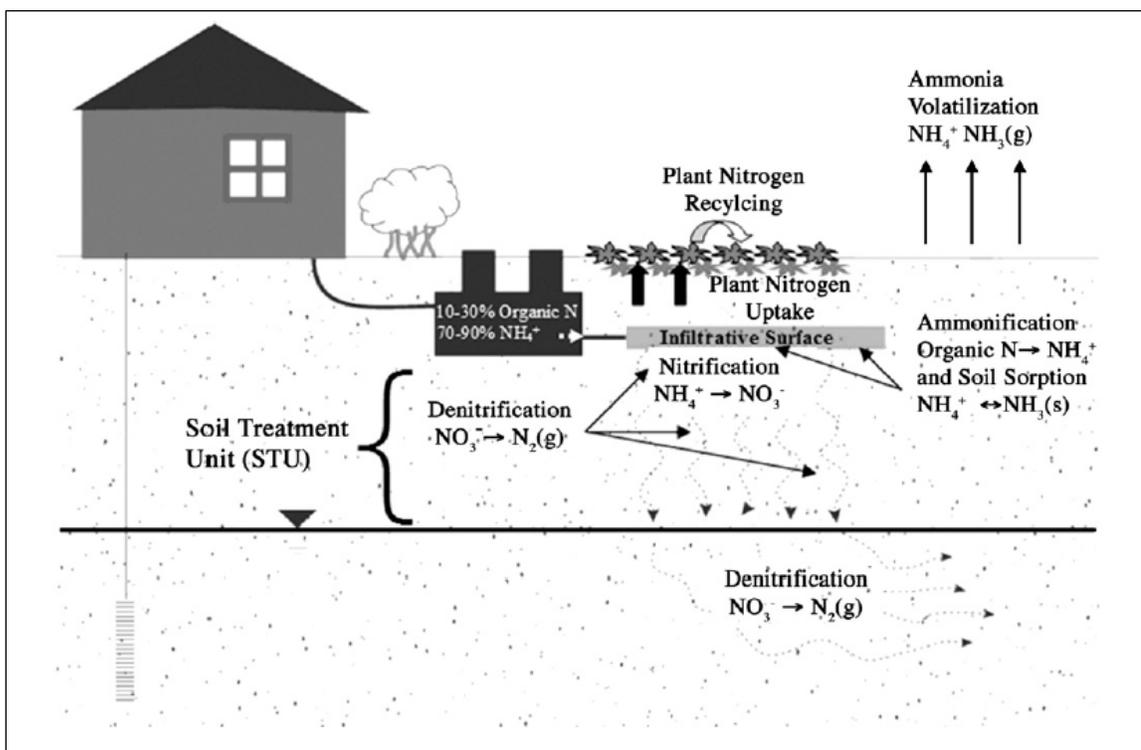
## 1.2 Nitrogen in Ground Water; Conceptual Considerations

Nitrogen is an important concern for water quality and nitrates represent perhaps the most common groundwater pollutant. Animals, crops, ecosystems, and human health can be adversely impacted by the presence of nitrogen in water supplies. Of these concerns, nitrate impacts to human health are a primary consideration. The consumption of nitrates has been linked to various illnesses, including cyanosis in infants and some forms of cancer. As a result, in the United States, a maximum allowable nitrate concentration of 10 mg/L as N has been established as protective of human health (Canter 1996). Other agencies around the world have also established such standards for nitrates in groundwater.

A survey of community service wells and private domestic wells performed by the U.S. Environmental Protection Agency (EPA) indicated that over half of these water supply wells contained detectable levels of nitrate (Canter 1996). The sources of this contamination are various, and include agricultural and domestic fertilizer applications, natural sources, wastewater treatment applications, and the use of OWTS. The last category is often of concern, as nearly 25% of the population in the U.S. and 30% of all new development utilize OWTS (Lowe et al., 2007). In Florida, nearly a third of all households are serviced by OWTS and 92% of water supplies come from groundwater (Briggs et al. 2007, Lowe et al. 2007).

Nitrogen transport in the subsurface is a complex process, especially when considering the nitrogen inputs from OWTS. The objectives of model development therefore requires the development of a conceptual understanding that includes the relevant fate and transport processes, parameters, and simulation approaches that will appropriately achieve the goals of the model. Figure 1-1 summarizes the conceptual understanding of the inputs of nitrogen and the transformative and advective processes that lead to nitrogen contamination of groundwater. The model development should result in a tool that will consist of the adequate level of complexity to represent these processes to accurately simulate the fate and transport of nitrogen species.

C:\44237-001R005\Wpdocs\Report\Draft



**Figure 1-1: Nitrogen Processes Occurring in a Typical OWTS (Heatwole and McCray 2007)**

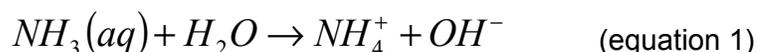
Proper OWTS design, installation, operation, and management are essential to ensure protection of the water quality and the public served by that water source. Assuming soils and site conditions are judged suitable, a wide variety of OWTS are designed and implemented (U.S. EPA, 1997, 2002; Crites and Tchobanoglous, 1998; Siegrist, 2001). Conventional OWTS rely on septic tanks for the primary digestion of raw wastewater followed by discharge of septic tank effluent (STE) to the subsurface soils for eventual recharge to underlying groundwater (Crites and Tchobanoglous, 1998; Metcalf and Eddy, 1991; U.S. EPA, 2002). However, increasing uses of alternative OWTS rely on additional treatment of the STE prior to discharge to the environment in sensitive areas or may eliminate use of a septic tank altogether.

Septic tanks are anaerobic and have long solids retention times (e.g., years) that can enable digestion resulting in a reduction of sludge volume (40%), biochemical oxygen demand (60%), suspended solids (70%) and conversion of much of the organic nitrogen to ammonium (Reneau et al. 2001). Septic tanks are also important as they attenuate instantaneous peak flows from the dwelling unit or establishment. The effluent discharged from the septic tank (i.e., septic tank effluent or STE) then flows to subsequent engineered treatment or the directly to the soil treatment unit where the processes of soil adsorption, filtration, and transformation (biological and chemical) occur.

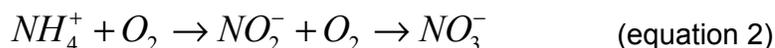
O:\4237-001R005\Wpdocs\Report\Draft

Nitrogen waste products are a considerable component of septic tank effluent. Total nitrogen, composed primarily of organic nitrogen products and ammonium-nitrogen, is typically assumed to range between 20-190 mg-N/L in untreated waste water, and 26-125 mg-N/L in STE (Canter 1996, Crites and Tchobanoglous, 1998, Lowe et al., 2009). Furthermore, in a recent study that evaluated the composition of raw wastewater and STE, the median total nitrogen concentration in STE specific to Florida was determined to be 65 mg-N/L (average = 61 mg-N/L) (Lowe et al., 2009). In terms of mass loading to the subsurface, the median loading rate was determined to be 10 g-N/capita/d (average = 13.3 g-N/capita/d) (Lowe et al., 2009). McCray et al. (2005) suggested that an average subdivision can generate up to 2880 kg/km<sup>2</sup> annually. While this value is significantly higher than estimates of naturally generated deposition (600-1,200 kg/km<sup>2</sup> annually), it is much lower than the loading that results from fertilizer application (10,000-20,000 kg/km<sup>2</sup> annually). Nonetheless, OWTS should be considered a potential contributor to groundwater nitrogen concentrations.

The first stages of nitrogen transformation related to OWTS occur in the septic tank. Organic nitrogen is mineralized to the inorganic form (ammonia) via the process of ammonification, followed by volatilization to ammonium ions.

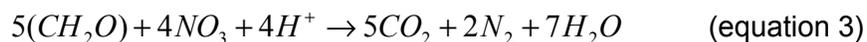


Once the liquid portion of the wastewater enters the drainfield through the subsurface infiltration system, nitrogen species (specifically ammonium and nitrate) are further transformed in the soil by nitrification and denitrification. Nitrification is a two step process by which ammonium is converted first to nitrite than to nitrate via biological oxidation.



Although a two step process, it can be assumed to be a one step process since the conversion of ammonium to nitrite is relatively rapid. Nitrification is either described as a zero-order or first-order reaction or via Monod kinetics. This particular reaction is of importance, as it represents the transformation from the relatively immobile nitrogen form (ammonium) to the highly mobile form (nitrate). Most studies of OWTS with suitable unsaturated soil have indicated that little ammonium reaches the underlying groundwater and that most impacts to groundwater from nitrogen are in the nitrate form. Nitrate behaves essentially as a conservative solute, with virtually no sorption or retardation processes affecting its movement in the aquifer. It is, however, subject to transformative processes.

Denitrification is the transformation of nitrate to N<sub>2</sub> gas.



Denitrification occurs in oxygen-free conditions, and is therefore seen in anoxic zones in the soil and groundwater. This reaction is typically described as first-order. However, nitrogen transformations are probably best modeled using Monod kinetics, which result in zero-order rate constants for concentrations typical of nitrate-impacted groundwater. The process, while studied extensively, is not well understood or well quantified. Previous studies identifying significant processes that lead to the reduction of nitrate concentrations identify denitrification rates as relatively small, and that most reductions occur as a result of mixing with ambient groundwater (to be discussed in more detail later in this review).

The development of a conceptual understanding of nitrogen fate and transport from source to receptor indicates that there are potentially a large number of processes that can be simulated depending on the objectives of the model. In the literature review that follows, researchers have in some cases used simplifying assumptions to account for certain processes if data is not available or the model does not need to simulate the process to achieve desired outputs. In other cases, researchers use relatively complex mathematical models in attempt to model multiple transformation or transport processes as accurately as possible. For example, the development of a model that considers all of the sequential steps of denitrification. The approach chosen is highly dependent on the goals of the modeling and the data available, as well as the scale that is to be represented.



## Section 2.0

### Literature Review

---

The following presents a summary of available research related to the modeling of fate and transport of nitrogen in groundwater. Modeling research directly related to nitrogen is presented, as well as modeling for general solute transport. The purpose of the summary is to:

- assess the state-of-knowledge of modeling nitrogen fate and transport in the vadose zone and in groundwater,
- identify the relevant processes, parameters and data used in the simulation of nitrogen transport,
- identify the modeling methods that enable quality simulation with an appropriate level of complexity in the context of the important processes that govern nitrogen fate and transport in the subsurface, and
- identify the merits and drawbacks of the various modeling studies and develop a guidance in designing the mathematical approach to address the project objectives.

#### 2.1 Modeling Research Summary

The literature review discovered over 70 reports or articles related to the modeling of nitrogen fate and transport. Additionally, the review discovered more than 20 modeling codes or solutions not specific to a particular contaminant that could potentially be applied to the simulation of nitrogen in the subsurface, based on the conceptual understanding of the processes governing nitrogen movement and transformation. Only a very small number of models specific to OWTS were discovered, and generally were concerned with land-use planning related to septic tank density.

A relatively large number of studies investigated the behavior of nitrogen in the vadose zone. These models were typically physically-based deterministic solutions of the Richards' equation for groundwater flow with a variation of the advective-dispersive equation (ADE) to simulate solute transport. Some researchers used a stochastic solution approach; this approach assumes that vadose zone parameters are too heterogeneous

C:\4237-001R005\wpdocs\Report\Draft

to be captured by a physical model, and that transport through the unsaturated zone is better modeled by using probabilistic functions for model input parameters.

The review identified fewer models considering nitrogen in the saturated zone. This may be in large part due to the fact that nitrate acts as a conservative solute in groundwater and therefore the development of complex models to describe this movement are not necessarily valuable or appropriate. A number of mass-balance models were created for nitrates in the saturated zone, because such a model could satisfy the objectives of the study. The models in this category consist of land-use planning models, studies identifying nitrate sources, and studies of specific groundwater systems. Modeling efforts that were not specific to nitrogen also tended to fall in this category, as researchers were concerned with developing methods that provided appropriate approximations of the ADE. Because solutions to this equation are approximate, many researchers were developing or comparing solution methods in order to identify the method that provided the most accurate solution.

Fewer still have considered the combined simulation of nitrogen in the vadose and saturated zone, and among these only a handful simulated flow and transport processes at the field scale. The latter category of models is often developed at the watershed scale and included impacts to surface water bodies. The inputs and the models themselves are often fairly large and complex, and include data and simulations for climate, stream-flow characteristics and fluxes, and land-use and vegetative patterns. Simulations and calibration procedures are usually time-consuming and complex, and require a considerable amount of input data.

Additionally, a large body of research has exclusively modeled the denitrification processes. In fact, numerous simple models have been developed that generate empirical expressions for denitrification at particular sites of interest. As a result, broad applicability and transferability of the models described to other sites is questionable.

## 2.2 Vadose Zone Models

Many modeling studies were identified that addressed solute transport in the vadose zone, of which a majority of the models selected for this review specifically simulated nitrogen transport. The studies could generally be classified as either deterministic physical models or stochastic, probabilistic models. Among the studies that examine the problem of nitrogen fate and transport, physically-based deterministic models for the vadose zone are generally solutions of the Richards' equation combined with a one-dimensional solution of the ADE for representing vertical flow and transport (in the "z" direction) and assuming horizontal flow vectors are not significant, although in some cases dispersion was also considered.

O:\44237-001R005\Wpdocs\Report\Draft

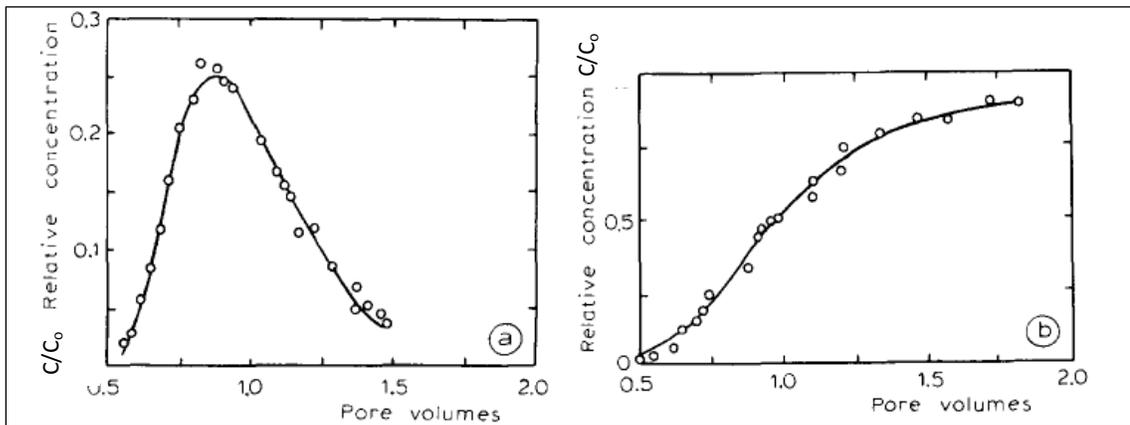
Addiscott and Wagenet (1985) provided a summary of soil leaching models and provided brief descriptions of modeling approaches and specified studies that applied the various methods. The key distinction the authors make when comparing the modeling approaches is comparing deterministic models with stochastic modeling approaches. The authors note that in most cases the selection of methods is based on the preferences of the researchers and tend to ignore the fact that models are intended for different purposes.

Among the physically-based models, most studies examine the problem of nitrogen transport in the unsaturated zone related to the impacts of fertilizer applications. These include studies by Bakhsh et al. (2004), Hansen et al. (1991), Jabro et al. (2001), Johnson et al. (1999), Moreels et al. (2003), and Johnsson et al. (1987). Generally, these modeling studies used numerical computer simulation programs designed for one-dimensional solutions of the Richards' equation coupled with the ADE or a variation of the ADE that contains provisions for partially-saturated flow and transport. In some cases, these programs are relatively complex, requiring large amounts of computing power and time, as well as complex data inputs. For example, (Bakhsh et al. 2004) used an updated version of the Root Zone Water Quality Model (RZWQM) to simulate nitrogen transport in a watershed in Iowa that is potentially impacted from corn and soybean field fertilizer applications. The RZWQM simulates solute transport using a one-dimensional solution to the Richards' equation and ADE. Input data for meteorological parameters includes daily minimum and maximum temperature, hourly wind speed, and solar radiation. Additionally, a full suite of soil and crop management inputs are required as well. Jabro et al. (2001) used the SOIL-SOILN model to simulate nitrogen transport. This model simulates fluid flow and heat transport using a coupled program that solves the ADE for fluids and the Fourier equation for heat transport. Again as with the RZWQM, inputs are complex including meteorological, soil, and crop management data. Simulations were performed for a three-year period and showed generally good model performance.

Other unsaturated zone models simulated nitrogen movement associated with the practice of wastewater treatment via land applications. Modeling studies by Reynolds and Iskandar (1995) and Beggs et al. (2005) looked at effectiveness of this practice at minimizing the impacts of effluents. Beggs et al. (2005) used HYDRUS 2D to look at the effectiveness of using subsurface drip irrigation as a means of treating STE. HYDRUS 2D, like the RZWQM, uses the Richard's equation for flow and the ADE with reaction parameters (including rate constants for nitrification and denitrification) for transformation and transport. The study showed an appropriately designed system could reduce annual nitrogen percolation through the soil column. Reynolds and Iskandar (1995) used the previously developed computer code WASTEN to simulate various scenarios of wastewater

land application at a treatment facility at Fort Dix, New Jersey. The code utilizes a subroutine for the ADE and can also simulate transformation processes such as nitrification and denitrification. Additionally, WASTEN is capable of simulating the effects of plant uptake, evapotranspiration, leaching, and rainfall.

Selim and Mansell (1976) and Mironenko and Pachepsky (1984) developed one-dimensional analytical solutions of the ADE for the simulation of solute transport through soils. Selim and Mansell develop a solution that can simulate constant source or pulse source inputs, and can also simulate reversible linear adsorption and irreversible sorption. No parameters for reactions are provided. In comparison to other solutions, the model performed more favorably at lower pore velocities, and performed similarly at higher pore velocities. Mironenko and Pachepsky developed a solution that could simulate adsorption as well as biological or chemical transformations. The heterogeneity of the soil pore scale was addressed by introducing mobile and immobile transport domains. The model was then used to simulate nitrogen transport and denitrification in a soil column. The results are presented as relative concentrations ( $C/C_0$ ) vs. pore volumes, as shown in Figure 2-1. The researchers were able to reasonably match observed data using a model calibration procedure to determine input parameter values.



**Figure 2-1: Modeling Results from Mironenko and Pachepsky (1984)**

Two modeling studies were found that simulated wastewater vadose zone transport associated with OWTS. Huntzinger and McCray (2003) used HYDRUS2D to examine the problem of soil pore clogging and its impact on the effectiveness of wastewater soil absorption systems. Results indicated the importance of understanding the influence of clogging on system design to optimize residence times and treatment of wastewater. Heatwole and McCray (2007) applied HDYRUS1D to estimations of nitrogen contamination flux from a proposed housing development in Weld County, Colorado. The modeling

O:\44237-001R005\Wpdocs\Report\Draft

used some site-specific data, and statistically-based N-transformation rate parameters to simulate nitrate impacts to the groundwater below. The model was highly sensitive to nitrogen mass-flux input and the denitrification rate coefficient. The latter sensitivity is important, because published denitrification rates are highly variable and therefore the estimates can potentially have a high degree of uncertainty. In contrast, nitrogen mass flux inputs to the subsurface are less uncertain.

In contrast to the physically-based models, a stochastic modeling approach was developed by Jury (1982) for one- or two-dimensional transport of solutes through the vadose zone. The transfer function model (TFM) considers that the spatial distribution of the physical, chemical and biological transport mechanisms are not well known especially when considering a heterogeneous media such as the soil column. Therefore, the model simulation is independent of site-measured characteristics and the behavior of a solute entering the soil matrix is based on probabilistic functions rather than physical functions; in other words, the model produces outputs based on the probability that a solute will reach a defined depth in the soil column. This is done using the probability density function (PDF), a mathematical operator that can estimate solute concentrations at a given depth based on the average and variance values of either travel time or input water flux at the surface. As a result, the model can consider uniform spatial distribution of input water flux or spatially variable inputs. Models in this category, while using agricultural problems as examples, could potentially have simulation capabilities for a variety of sources.

Studies by White (1987) and White et al. (1998) used applications of the TFM at the field scale to address the problem of nitrogen leaching in pasture lands in New Zealand. The first study developed a probability distribution for solute transport times from observed data related to numerous rainfall events and soil moisture conditions. The transfer function was then calibrated against measured quantities of nitrate leached. The TFM was capable of representing the measured data with reasonable accuracy. The researchers suggest that predictive simulations using a TFM are possible using a time and space-averaged value for solute travel times. However, this would require numerous additional studies to characterize a variety of soil types. The second study is similar, using a TFM to simulate nitrate leaching in a soil near Palmerston North, New Zealand. Results are generally good; however, there is a consistent tendency of the TFM to over-estimate the nitrate leaching in this case.

A number of studies compared modeled solute transport through the vadose zone using the TFM and an analytical solution of the ADE. Jury and Sposito (1985) used data collected from soil core and soil solution samplers to calibrate and validate results using both modeling methods. Based on the data collected, model parameters were optimized.

O:\44237-001R005\Wpdocs\Report\Draft

In the case of the ADE analytical solution, the parameters were pore-water velocity ( $V$ ) and a field-scale dispersion coefficient ( $D$ ). For the TFM, the parameters were mean and variance of travel time through the media. Parameters were estimated using three methods: a sum of squares method, a method of moments, and maximum likelihood estimation. The parameter estimations for the solution sampler data had relatively high uncertainties, owing to the deviations between the shape of the average data curve and the model estimates of the curve, and the small number of replicate measurements. Therefore, a comparison of performance could not be done. For the soil core samples, the TFM was determined to have provided a better representation of the data.

Dyson and White (1987) conducted a similar study comparing the two modeling approaches for the transport of chloride through a structured clay soil. Also considering soil cores, the researchers found that the TFM model, using an assumption of a log-normal distribution of travel times (characterized by the mean and the variance), could model the flux-averaged breakthrough curves well. Also, the ADE could model the breakthrough curves equally well when the velocity and dispersion parameters were optimized via the least squares method.

### **2.3 Saturated Zone Flow and Transport Models**

Due to the scope of the problem related to the protection of groundwater, the modeling of fate and transport of contaminants in aquifers has been a significant objective for research. A relatively large body of numerical, analytical and mass-balance models has been developed to study the movement of solutes in the saturated zone. Additional effort aims to provide accurate simulation of solute reactions and adsorption. Among many studies that develop general solutions of flow and transport, numerous models have been focused on the behavior of nitrogen (specifically nitrate) in the saturated zone. These can be either site-specific or more broadly-focused nitrate transport models that can potentially be applied to any site or problem.

Among the aquifer models, the simplest form of simulating solute fate and transport is the mass-balance model. This type of model ignores aquifer parameters that influence groundwater direction and velocities and transformative processes. The objective is to simply balance source and groundwater inputs and outputs (usually expressed as fluxes or rates) based on observed data. Typically, these models have numerous simplifying assumptions.

O:\44237-001R005\Wpdocs\Report\Draft

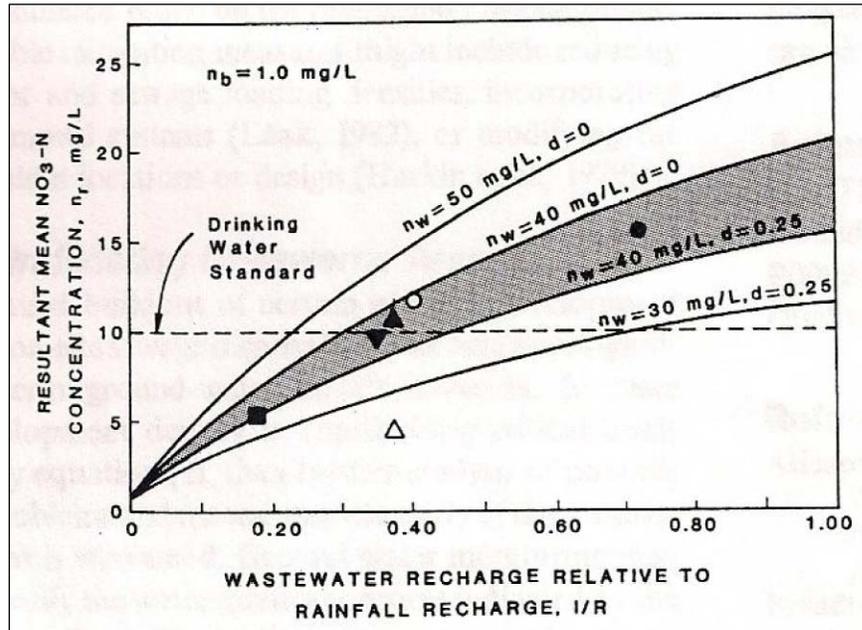
A simple mass-balance equation was developed and then compared the model-predicted results to field data for three communities in California (Hantzche and Finne-more 1992):

$$n_r = \frac{In_w(1-d) + Rn_b}{(I + R)} \quad (\text{equation 4})$$

where:  $n_r$  = net nitrate concentration in recharge groundwater,  $I$  = volume rate of waste water entering the soil averaged over the gross developed area (inches  $\text{yr}^{-1}$ ),  $n_w$  is the total nitrogen concentration of wastewater ( $\text{mg L}^{-1}$ ),  $d$  = the fraction of nitrate-nitrogen loss due to denitrification in the soil,  $R$  = average recharge rate of rainfall (inches  $\text{yr}^{-1}$ ), and  $n_b$  = background nitrate-nitrogen concentration of the rainfall ( $\text{mg L}^{-1}$ ).

Results were plotted as mean nitrogen-nitrate concentration versus wastewater recharge relative to rainfall recharge ( $I/R$ ) and include comparison to field data values from the different sites in the study (Figure 2-2). In general, model-predicted results compared favorably with the concentrations measured in the field. The authors note that the model has the following limitations:

- The equation considers only vertical components of groundwater recharge, and does not consider fluxes from upgradient areas.
- The predicted concentrations are long-term values, as loading rates may take many years to develop and may be affected by the nature and thickness of the vadose zone.
- Results cannot be applied to a single point, as in considering a specific water supply well.
- This method does not account for other sources of nitrogen, such as fertilizer or animal wastes.



**Figure 2-2: Results of Modeling with Comparison to Field Data (Hantzche and Finnemore 1992)**

DeSimone and Howes (1998) used a mass-balance solution to estimate fate and transport rate values based on observed field data. The source of nitrogen in this study was a waste treatment facility in Cape Cod, Massachusetts. The objective of the research was to use a mass-balance method to identify the key hydrogeochemical processes, estimate rate values for these processes, and ultimately estimate potential mass flux into nearby surface water bodies.

Calculated wastewater input fluxes from the treatment facility, waste loads to the aquifer, and the observed concentrations at downgradient sampling points were considered and input into the mass-balance equation. Based on the observed data, values for advective and transformative processes were estimated. The researchers determined that within the unsaturated zone, nitrification and ammonification processes were the most important to nitrogen transport, whereas in the saturated zone denitrification and sorption of ammonium had the most influence. They concluded, based on the estimated fate and transport processes, that approximately 75% of the input waste load could potentially reach the nearby surface water body.

Mass-balance models are often utilized as land-planning tools. The objective in most cases is to estimate the optimal lot size or housing density to minimize the impacts from OWTS. A few examples are summarized below.

O:\44237-001R005\Wpdocs\Report\Draft

Rogers, Golden and Halpern (1988) developed a groundwater-dilution model based on mass-balance inputs for the State of New Jersey. The ultimate goal of the model was to assess optimal numbers of households and lot sizes in new developments using septic systems to minimize the impact of nitrate groundwater contamination on surface waters. In this case, the model considers dilution of nitrate contamination by recharge fluxes alone as a way to reduce the waste mass flux into the aquifer, and does not consider upgradient groundwater inputs or transformative processes in the soil or the aquifer.

A very simple modeling approach ultimately defines carrying capacity as acres required per household to optimize nitrate dilution from precipitation recharge so as not to exceed the groundwater protection standards. The equation, from an earlier study by Trela and Douglas (1978) is as follows:

$$H = \frac{V_e C_e}{(V_i + C_i) C_q} \quad \text{(equation 5)}$$

Where: H= carrying capacity;  $V_e$ = Volume of septic effluent entering system;  $C_e$ = Nitrate concentration in septic effluent;  $V_i$ = Volume of infiltrating precipitation;  $C_i$ = Nitrate concentration in precipitation; and  $C_q$ = Water quality standard for nitrate.

Two example runs considering varying target nitrate concentrations are provided, but not verified with field data.

A similar dilution-based mass-balance model for land use planning was developed for Pennsylvania (Taylor 2003). Through a mass-balance equation, the model is intended to estimate appropriate lot sizes to allow for adequate dilution of the input nitrogen to minimize the impacts of a septic system:

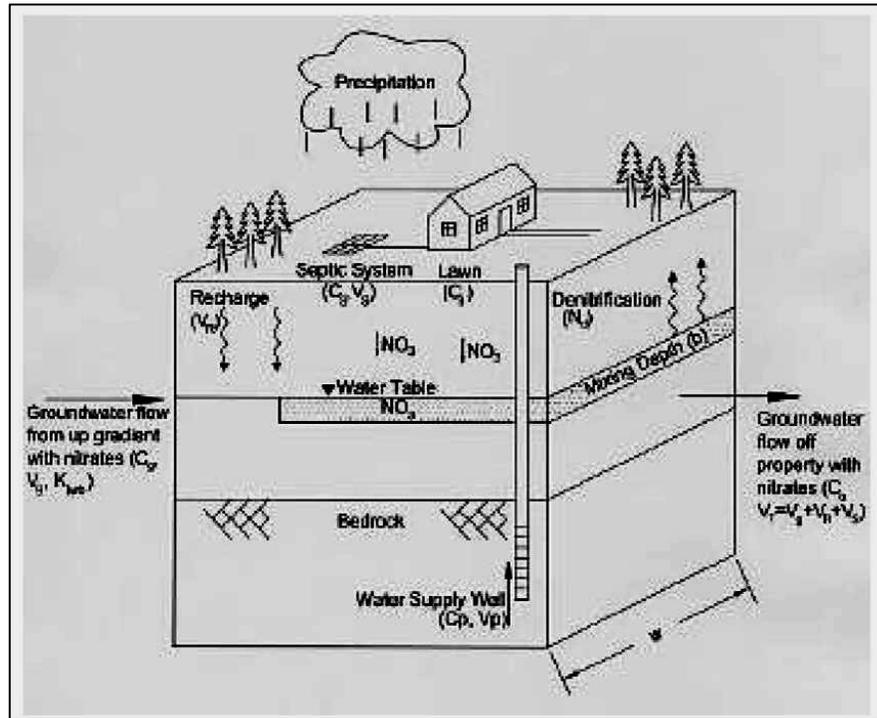
$$V_s C_s + V_r C_f + V_g C_g = (V_s + V_r + V_g) C_o + (V_s + V_r + V_g) C_d \quad \text{(equation 6)}$$

Where  $V_s$  = volume of septic tank effluent (gpd),  $C_s$  = Concentration of nitrate in septic tank effluent (mg/L),  $V_r$  = Volume of groundwater recharge/infiltration (gpd),  $C_f$  = nitrate concentration in fertilizer that reaches the groundwater (mg/L),  $V_g$  = Volume of upgradient recharge water (gpd),  $C_g$  = nitrate concentration in upgradient groundwater (mg/L),  $C_o$  = nitrate concentration of groundwater leaving the site (mg/L), and  $C_d$  = concentration of nitrate lost due to denitrification.

In order to arrive at these terms, this mass-balance approach utilizes a number of site parameters including hydraulic conductivity, gradient, average recharge rate due to precipitation, and mass of fertilizer applied as examples. These parameters are then used in empirical relationships to define the needed inputs for the mass-balance equation. Un-

O:\44237-001R005\Wpdocs\Report\Draft

like the model developed for the state of New Jersey, this model does consider upgradient groundwater flux. The model conceptualization is shown in Figure 2-3.



**Figure 2-3: Conceptual Model for Mass-Balance Approach (Taylor, 2003)**

This model, as with the other mass-balance models, has several simplifying assumptions, such as: complete mixing of wastewater and recharge water within a specified aquifer thickness; complete conversion of nitrogen to nitrate; and neglecting most chemical transport and reactive processes including diffusion, dispersion, adsorption, and denitrification. The author provides an example use of the model to calculate the optimal lot size for a hypothetical development in Pennsylvania and makes land-use recommendations based on the results. However, as with the previously described model, the model-predicted results are not verified with field data.

While examination of available research indicated mass-balance models are often used for land planning tools, some researchers have utilized them for nitrate source identification or as a predictive tool. Tinker (1991) compared the results from three mass-balance models, along with other investigative tools, to help identify possible sources of nitrate impacts on wells in Wisconsin. Tinker used a mass-balance model developed by Wehr-

O:\44237-001R005\Wpdocs\Report\Draft

mann (1984), the BURBS model and a combination of the two models to determine if the nitrate in the groundwater was primarily from fertilizer applications or wastewater treatment.

The Wehrmann mass-balance model uses estimates of water volumes and nitrate concentrations to evaluate the diluted nitrate leaving the area of the subdivision in question. In this case, the model considers upgradient groundwater and also pumped water volumes. The primary source for this model is septic effluent:

$$V_b C_b + V_i C_i + V_s C_s - V_p C_p = (V_b + V_i + V_s - V_p) C_o \quad (\text{equation 7})$$

Where  $V_b$  = Volume of upgradient groundwater,  $C_b$  = Nitrate concentration in upgradient groundwater,  $V_i$  = Volume of precipitation infiltration,  $C_i$  = Nitrate concentration in the infiltration,  $V_s$  = Volume of septic effluent introduced beneath subdivision,  $C_s$  = Nitrate concentration in the septic effluent,  $C_p$  = Nitrate concentration in the pumped groundwater and  $C_o$  = Diluted nitrate concentration leaving the subdivision. The BURBS model is similar, but also considers contributions from turf, impervious land, and natural land. A bulk nitrogen-nitrate concentration ( $C_{BURBS}$ ) is used in the water contribution from these three sources. The combined mass-balance expression is as follows:

$$V_b C_b + (V_t + V_i + V_n + V_s - V_p) C_{BURBS} = (V_b + V_t + V_i + V_n + V_s - V_p) C_o \quad (\text{equation 8})$$

With terms defined as above and also  $V_t$  = Volume of recharge water from turf,  $V_i$  = Volume of water recharged from impervious land and  $V_n$  = Volume of water recharged from natural lands.

The author predicted that a majority of the nitrate in the groundwater could be accounted for by OWTS and fertilizer applications, and the mass-balance modeling agreed with that prediction, based on sampling results from residential wells.

Frimpter et al. (1990) developed a simple mass balance equation to predict the potential nitrate impacts to municipal supply wells based on loading rates from natural and anthropogenic sources. The model can consider individual sources and therefore is a potential tool for determining septic tank density in new developments. Assumptions of the model are steady-state conditions, complete mixing, and modeling nitrate as a conservative solute. The author provides example calculations in which nitrate concentrations are predicted based on source density and flow rates of withdrawing municipal supply wells.

O:\44237-001R005\Wpdocs\Report\Draft

Beyond mass-balance models, a variety of approximations of the ADE or similar governing equations that apply to nitrates were found in the literature. Methods of simulation were variable, from simple analytical solutions to complex numerical codes.

Lerner and Papatolios (1993) developed a unique, simple analytical expression for predicting nitrate concentrations in pumped groundwater:

$$C_t = C_o + (C - C_o) \left[ 1 - \exp\left(-\frac{Rt}{bn}\right) \right] \quad (\text{equation 9})$$

Where  $C_t$  = Time variant pumped concentration,  $C_o$  = Initial groundwater nitrate concentration,  $C$  = Concentration of nitrate in groundwater in area of influence of the pumping well,  $R$  = Recharge rate,  $b$  = Aquifer thickness,  $t$  = Time, and  $n$  = Porosity.

This expression considers nitrate in the saturated zone with source inputs mostly originating from recharge waters.  $C_o$  is arrived at by estimating leaching rates of nitrate through the vadose zone. The model considers only nitrate in groundwater as an initial concentration and does not consider source input rates or flux. The model was applied to a pumping station in England, and predictive calculations were performed. In the two years following the model simulations, field data was collected and then compared to the simulations to verify if the model predictions agreed with the actual observed nitrate concentrations. The model predicted future nitrate concentrations with reasonable accuracy. Also, a sensitivity analysis was done to determine sensitive parameters in terms of conservative solute transport for this system (in this case, porosity, aquifer thickness and estimated leaching rates from the vadose zone that determine  $C_o$ ).

Young et al. (1976) also develop an analytical approach to predicting future nitrate concentrations in groundwater in a fractured sandstone aquifer in England. The investigators used an unsaturated zone flow model to forward model nitrate impacts to groundwater based on current land use practices. However, the exact analytical approach is not described, except to describe important parameters and processes.

MODFLOW and MODPATH are two commonly used numerical modeling codes that have been widely utilized by both academic and industry hydrogeologists. MODFLOW is a three dimensional, finite difference modeling codes that has a wide variety of capabilities for modeling multiple layers of an aquifer system and heterogeneous parametric distributions. The model code is often employed with a graphical pre- and post-processor to assist with the construction of the input files. MODPATH is an extension of MODFLOW that utilizes model-calculated groundwater velocities and flow vectors to give particle tracking of groundwater movement. The tracking of the groundwater movement can be time-stepped to give estimates of travel time.

Puckett and Lowderly (2002) constructed a MODFLOW groundwater flow and transport model and used it in conjunction with sample analyses and water-aging to determine the relationship between agricultural practices and nitrate concentrations in groundwater in a glacial outwash aquifer in Minnesota. The model was constructed as a three layer model with assigned boundary conditions based on the conceptual understanding of the hydro-geologic system. The model was calibrated against observed water levels at monitoring wells in the study area, and MODPATH was used to compute groundwater travel times and flow paths. The modeling was used to indicate expected groundwater travel times and correlated with the groundwater aging aspect of the study, and showed generally good agreement between the two methodologies. Although no transport modeling was done, an extension of MODFLOW, MT3DS is a full transport modeling code and can consider adsorption and chemical processes in numerous ways. This could be a potential extension of the study to further validate the conceptual understanding of the groundwater system.

A study by Molenat and Gascuel-Odeux (2002) provides an example of nitrate modeling using a combination of MODFLOW and MT3DS. In this study, these model codes were coupled to simulate different spatial distribution scenarios of nitrate inputs from agricultural practices in the Kervidy watershed in Brittany, France. The initial simulation of groundwater flow and nitrate distribution was done as a steady-state simulation, and simulation results matched field observations well. Two scenarios reduced the uniform spatial distribution of nitrate recharge from the initial 100 mg/L to 80 mg/L and 60 mg/L, respectively. The other four scenarios redistributed the nitrate recharge to the watershed hillsides, but retained the initial rate of 100 mg/L. The results of the scenarios indicate that the impacts to groundwater could be reduced by reducing the nitrate inputs on the hillsides more than reducing nitrate inputs over the entire watershed.

A simple distributed transport model was developed to simulate and predict groundwater nitrate concentrations. Based on a numerical code developed by Bear (1979), a study was completed for the Great Ouse Chalk aquifer in England by Carey and Lloyd (1985). The chalk is a fine-grained fractured limestone aquifer that has seen increasing impacts from nitrate pollution. For the modeling, a groundwater flow and transport model was constructed using numerous computation cells representing small volumes of the aquifer with the nitrate concentrations into and out of the model cells calculated using a mass-balance approach. The model assumed that the concentration of nitrate in groundwater recharge was constant, but that the rate of recharge varied. Nitrate sources in this case were natural land, plowed land, and nitrogen inputs that resulted from fertilizer applications. The downward migration of nitrate is solved via a simple equation that calculates a velocity based on porosity and recharge. The model was calibrated using observed groundwater and nitrate concentration data, and a sensitivity analysis was performed to

O:\44237-001R005\Wpdocs\Report\Draft

assess the effects of various parameters. The model was able to simulate past trends of groundwater behavior and nitrate concentration reasonably well. Further, the model predicted an increasing trend in nitrate concentrations, but the severity of the increase was dependent on changes in land practices.

Hendry et al. (1983) apply this methodology to the problem of nitrate transport in an aquifer in England with impacts resulting from agricultural practices. The researchers apply a numerical model among other investigative techniques to test two hypotheses of nitrate fate and transport; 1) that the resulting concentrations observed are a result of mixing with upgradient groundwater, and 2) that the vertical concentration distributions are due to denitrification processes. Based on the modeling and other investigations, they conclude that the denitrification processes are dominant.

## 2.4 Combined Vadose Zone and Saturated Zone Models

The research that examines the combined fate and transport of nitrogen species in both the vadose and saturated zones is limited. However, some models have been developed either on the watershed or field scale that does in fact couple the two zones for simulations of contaminant transport. Among these are several models that could effectively simulate nitrogen transport.

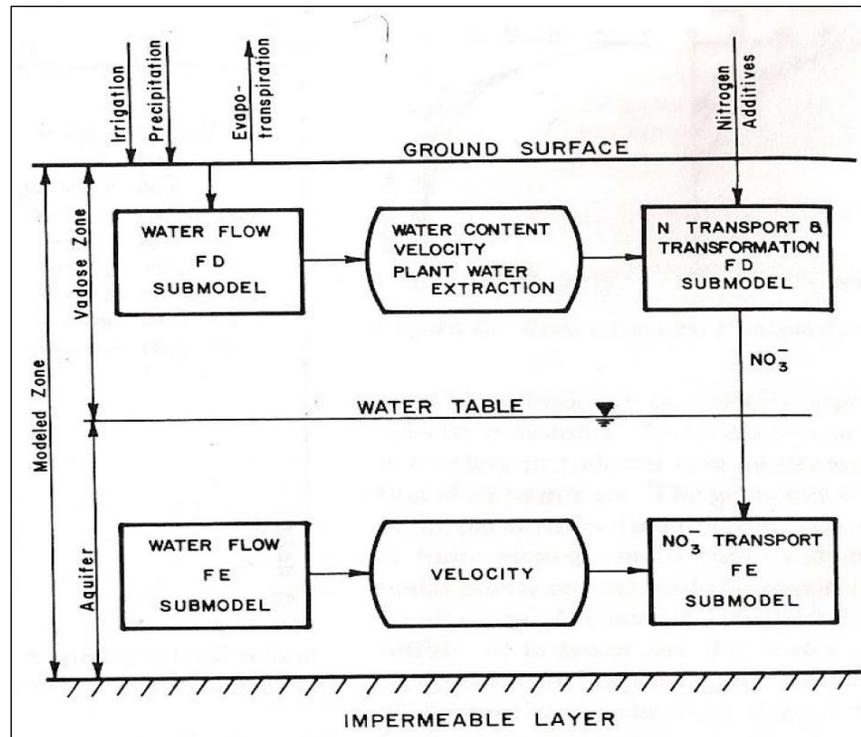
Mehran et al. (1983-1984) developed a two-dimensional numerical solution for the fate and transport of soluble nitrogen species in both the vadose and saturated zones. This model resulted in two separate codes that simulate the vadose zone and saturated zone simultaneously. The vadose zone is represented by the finite-difference code UCD-RANN and the saturated zone by the finite-element code FLOWS.

The model considers numerous parameters for both flow zones including all relevant flow parameters indicated in Darcy's Law (i.e., hydraulic conductivity, gradient, porosity), and transport and transformative parameters in both the vadose zone and saturated zone. Among others, these parameters include saturation index, pressure head, and root zone uptake of nitrates as well as first-order rate constants for nitrification and denitrification, and the retardation factor for ammonium transport in the vadose zone. The retardation factor is defined by:

$$R = 1 + \left( \frac{\rho_b}{n} \right) (K_d) \quad \text{(equation 10)}$$

Where R = retardation factor,  $\rho_b$  = bulk density of the soil, n = effective porosity, and  $K_d$  = soil distribution coefficient.

The model, shown conceptually in Figure 2-4, also provides equations for boundary conditions that are necessary to define the model dimensions.



**Figure 2-4: Conceptual Schematic of the Model (Mehran et al. 1983-1984)**

The outputs of the model include nitrate concentrations at various depths through time for the vadose zone and time-variable depth and distance nitrate concentrations in the aquifer. The researchers provide a model demonstration on a hypothetical aquifer system, shown in Figure 2-5. In this case, a constant nitrate concentration is imposed on the water table, however, it is probable that a varying rate of concentration input can be applied using this code. An example output for the vadose zone simulation is shown in Figure 2-6.

O:\44237-001R005\Wpdocs\Report\Draft

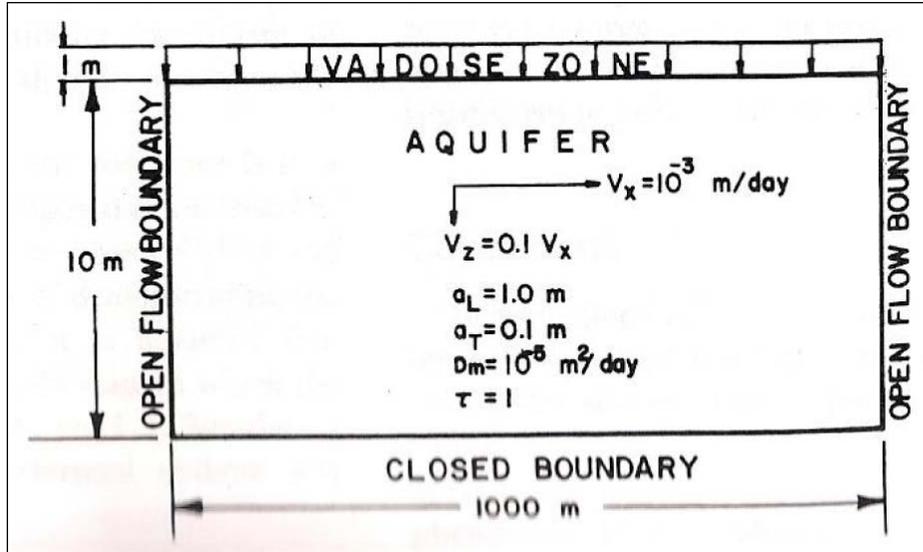


Figure 2-5: Hypothetical Aquifer for Model Example Simulation (Mehran et al. 1983-1984)

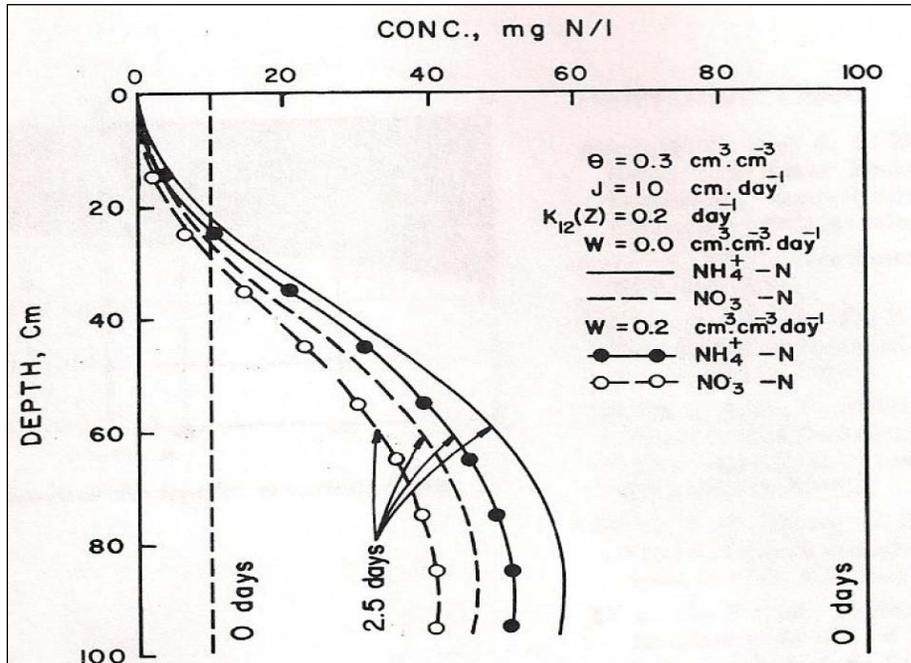
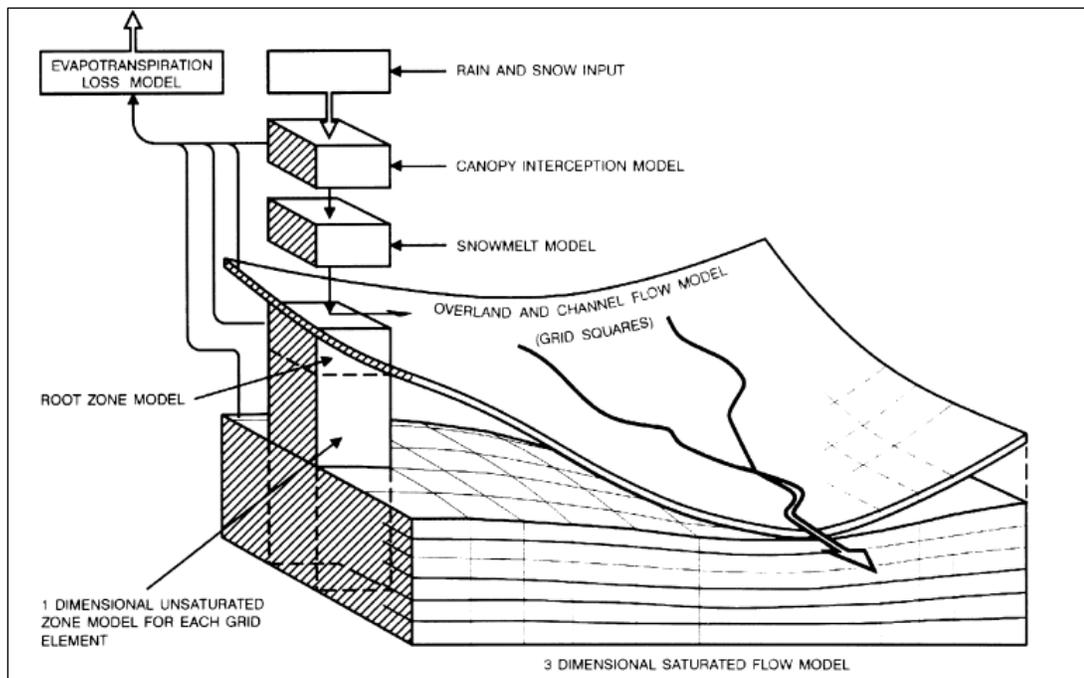


Figure 2-6: Example Output for Unsaturated Zone (Mehran et al. 1983-1984)

O:\44237-001R005\Wpdocs\Report\Draft

## 2.5 Nitrate Modeling at the Watershed Scale

Numerical modeling codes also have been applied to the watershed scale for the simulation of groundwater flow and nitrate leaching in the unsaturated zone and the resulting impacts to surface water bodies. These models represent perhaps the most complex of the models reviewed, requiring large input data sets and complex numerical codes for the simulations. Sonnenburg et al. (2003) and Refsgaard et al. (1999) present modeling of watersheds in Denmark using large scale models. The researchers utilize the code MIKE SHE, which uses numerical solutions for overland 2-D and channel 1-D flow, 1-D unsaturated flow, and 3-D saturated flow. The conceptual model is illustrated in Figure 2-7.



**Figure 2-7: Conceptual Schematic of MIKE SHE Model (Refsgaard, 1999)**

Refsgaard et. al. conceptualized and constructed groundwater flow and transport models to consider nitrate impacts on two watersheds in Denmark. The intention of the study was to show that such a model could be shown to be a reliable tool for specific watersheds, and that reasonable model performance at such a scale was possible. This model was a coupled model using the MIKE SHE code and the Daisy code, which simulates the percolation of water and nitrate at the bottom of the vadose zone. Input, verification, and calibration data for the model were found in GIS-linked databases which provided data for agricultural practices, topography, groundwater data, stream-flow and climatic

O:\44237-001R005\Wpdocs\Report\Draft

variables. Representative model grids were constructed using topography and catchment delineation data from the databases. The researchers use an up-scaling method to transfer some field-scale data to the catchment scale. Model parameters were assessed using various transfer functions, and the results of the simulation were validated by comparing model simulated results to observed results for annual water balance, river run-off, and groundwater nitrate concentrations.

Simulations were done for the Karup and Odense watersheds and validation of the model based on watershed-specific water balances over a five-year period and groundwater nitrate concentrations over the same period was also performed. Validation results for the Karup watershed were extremely good, and the simulations of the Odense watershed were acceptable although not at the same level as the simulations for the Karup watershed. The results indicate that similar models can be useful tools for assessing nitrate contamination at such a scale for other watersheds, provided access to adequate databases is available.

Conan et al. (2003) used MODFLOW and MT3DMS coupled with the watershed modeling code SWAT (Soil and Water Assessment Tool) to simulate nitrogen fate in a watershed in Brittany, France that has been impacted by livestock practices. The model considers the full range of transformative processes for nitrogen species, such as ammonification, nitrification, and denitrification. Hydraulically, the combined models simulate groundwater flow, nitrogen transport, and surface water flows and concentrations. Model structure is constructed using digital elevation models and used field data sets for stream flow, groundwater levels and nitrogen concentration over a three-year period. Data collected was used for initial input parameters and calibration of the model. Results were generally good, with some exceptions. The stream flow simulations consistently overestimated flows in June and underestimated flows in December. Simulations done with the SWAT model alone consistently underestimated nitrogen levels in the surface waters, but the coupled model performed much better, perhaps due to being able to account for the stream baseflow. Groundwater concentrations were also well simulated by the coupled model.

Heng and Nikoladis (1998) developed a highly complex, multidimensional watershed scale model for the transport and transformation of nitrogen from non-point sources. The model (NTT-watershed) generates a grid system based on topography and subsurface properties can also be vertically discretized to represent vegetation, overland flow, and the groundwater zones. Flow and solute inputs into the model are temporally and spatially variable and can consider transport and transformation of organic nitrogen, ammonium, and nitrate. For this study, the model was applied to the Muddy Brook watershed in Connecticut. Input, calibration, and validation data was collected over a two-year pe-

O:\44237-001R005\Wpdocs\Report\Draft

riod and included nitrogen species concentrations, field parameters, stream-flow values, precipitation rates, and land use practices and applications (i.e. fertilizer use). Simulations were performed and the model matched groundwater, stream-flow, and concentration data reasonably well. Results indicate that future models for other watersheds can be developed in a similar manner to assist with management planning.

Weintraub et al. (2004) used the GIS-based watershed modeling tool WARMF for investigating watershed-scale impacts from OWTS in Summit County, Colorado. WARMF is a modeling tool that estimates total maximum daily loads (TMDL) based on a series of modules with various inputs. The model was constructed using a Digital Elevation Model (DEM) of the Blue River Watershed and included input data for regional meteorology, point sources, and land use. Additional data was collected in the field for surface water quality, soil properties, well data and spatial distribution of OWTS. The model was enhanced by including a biozone module that simulated the transformative processes in the soil column beneath the OWTS. After constructing the model, simulations were performed representing a period from fall 1998 through fall 2002. Model simulations were compared to observed data, and a calibration was done to adjust the input parameters to improve the fit to observed hydraulic and concentration data. The calibrated model was then used to assess various management scenarios including converting housing subdivision from OWTS to a centralized sewer system. The results for nitrogen indicate that although nonpoint loading is reduced, the loading to the river increases due to the increased nitrogen loading from the treatment plant. This suggests that conversion to a sewer system would require a higher level of nitrogen treatment at the plant.

Geza and McCray (2007) also applied the WARMF model to assess the influence of various point and non-point sources including OWTS in the Turkey Creek watershed in Colorado. As with the previous study, a DEM was used to build the model that included land cover and soil type data. Data was also input regarding population, wastewater loading per person, and effluent concentration. Stream flow and water quality data from a previous study was used and a simulation period of five years (1998-2003) was used as a calibration run. Calibration was done using UCODE, an automatic calibration tool and a sensitivity analysis was also done. The analysis showed that groundwater concentrations for nitrate were most sensitive to soil parameters, land cover parameters, and input concentrations of ammonium. However, the model was not sensitive to denitrification rates. Stream concentrations were most sensitive to sediment transport parameters.

Once a calibrated model was completed, four management scenarios related to OWTS were performed. Stream concentrations of nitrate were shown to be highest when the stream segment was located close to an area with a high density of OWTS as compared to locations downstream, an effect likely due to dilution. Soil water concentrations in-

creased with increasing population, but decreased when OWTS were converted to sewer systems.

## 2.6 General Fate and Transport Modeling

Numerical or analytical codes for modeling groundwater flow and solute transport have had numerous applications. The use of a particular method, solution, or transport parameter depends largely on the contaminant of interest, the objectives of the modeling, and in some cases site-specific characteristics. For the most part, models in this category are limited to the saturated zone. Within the summary below, several models have been identified that do not model nitrate fate and transport specifically. However, because the codes can consider the important fate and transport processes, they can be useful in developing a model to simulate nitrate in the subsurface. Among the codes identified, 13 analytical solutions to the ADE were found in the literature. These solutions consider different methods, spatial and temporal simulations, and different transport parameters such as retardation, decay, or dispersion. Analytical solutions are appealing in that they can be programmed into a spreadsheet relatively easily, and can therefore be part of a user-friendly modeling tool. Four such spreadsheet programmed solutions are discussed below.

The governing equation in most cases is a variation of the ADE with either chemical (production of solute or degradation) or physical reaction (sorption) or both. An example featuring one-dimensional advection-dispersion with retardation and first-order degradation is shown from Elmore (2007):

$$R\left(\frac{\partial}{\partial t} C\right) = -v_x\left(\frac{\partial}{\partial x} C\right) + D_x \frac{\partial^2}{\partial x^2} C - kRC \quad (\text{equation 11})$$

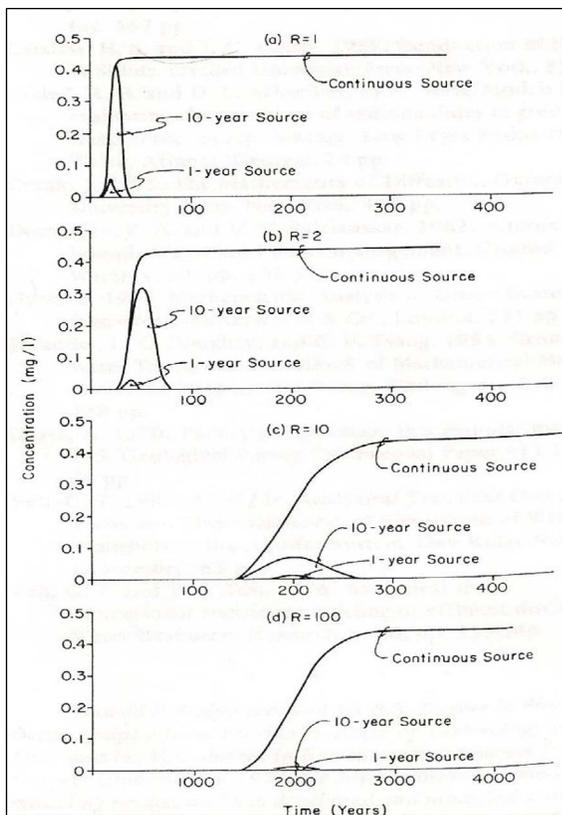
Where R= retardation factor; C = solute concentration; t = time;  $v_x$  = average pore water velocity; x = distance from source;  $D_x$  = dispersion coefficient; and k = first-order degradation constant.

Virtually all the analytical solutions considered either first-order reaction, equilibrium linear reversible sorption, or a combination of both. Leij et al. (1993) provide expressions for the partitioning of the solute in the mobile and immobile phases. For degradation, the majority of solutions use a first-order degradation rate constant for the solute. Sun et al. (1999) develop a three-dimensional reactive model for the saturated zone via an analytical solution that can simulate degradation as sequential first-order reactions. Example contaminants that can be simulated with this process include tetrachloroethylene (PCE) and its degradation daughter products and the denitrification process from nitrate to nitrite to ammonia to nitrogen gas. A unique three-dimensional code for the simulation of a reactive solute in a variably saturated porous media was developed by Srivasta and Yeh

(1992). Equations are based on the Richards' equation for flow in variably saturated media and the conventional ADE. This model also considers adsorption via a mobile-immobile partitioning condition and employs the use of a first-order decay function. The solutions are performed using a Galerkin finite element method and a Picard iterative process.

A number of solutions simulate transport considering different source orientations. This may be an important consideration for nitrate contamination from OWTS, as the impacts to groundwater may be from multiple sources that are not necessarily point sources. Carslaw and Jaeger (1959) derived solutions to the three-dimensional advection-dispersion equation utilizing Green's functions, which were extended to develop an aquifer transport model for a contaminant from a horizontal plane source at the top of the water table (HPS)(Galva 1987). This results in estimates of contaminant transport that are more accurate than simulations using a point source when considering a groundwater contaminant source that may be distributed over a relatively large planar area, such as a source associated with a landfill or a development utilizing OWTS.

Galva provides the mathematical background to the development of the HPS model, and provides example numerical simulations using a FORTRAN code to illustrate the applicability of the model. Simulation one uses various retardation rates, zero decay, and a continuous source rate. Simulation two varies the retardation rate and the period of source emissions. Finally, simulation three considers constant source input rates with variations in retardation and decay coefficient. Results for the second simulation are shown in Figure 2-8. Results are presented as predicted solute concentrations at a specified point (200 meters down-gradient) through time.



**Figure 2-8: Model Output for HPS Model, Simulation Two (Galya 1987)**

The model simulations indicate the sensitivity of the varied parameters such as decay factor, retardation factor and the temporal period of source emission. The authors also point out that while this model is useful for quick estimates, the simplifying assumptions required for the analytical solution make it impractical for hydrogeologically complex systems.

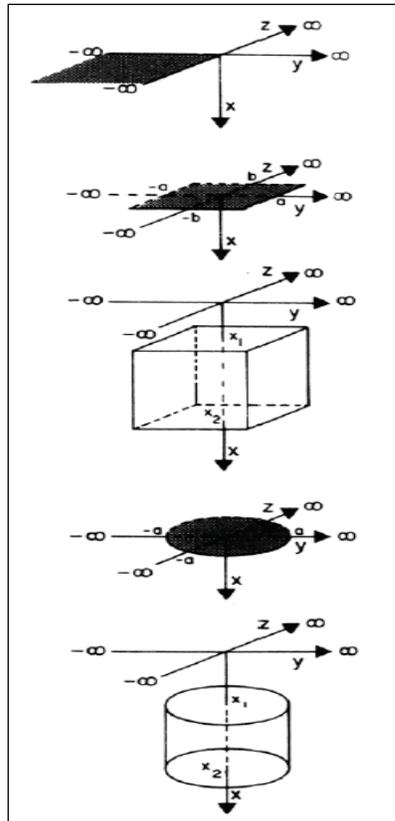
Heatwole and McCray (2006) applied the HPS to the problem of wastewaters derived from OWTS. The research demonstrates that the HPS model has the appropriate parameters and model structure to accurately simulate fate and transport of nutrients from OWTS, including nitrogen, phosphorus, and fecal coliform. Using an analytical solution of the HPS model in a FORTRAN code, simulations of an example nitrate plume resulting from an OWTS are run using baseline parameter inputs. Additionally, a sensitivity analysis of input parameters was included. Groundwater velocity is indicated as the most sensitive parameter, and is therefore indicated as an important factor when trying to es-

O:\44237-001R005\Wpdocs\Report\Draft

estimate potential groundwater impacts. The HPS model is identified as a potentially useful tool for simple simulations in support of regulatory compliance and OWTS planning.

Similarly, Domenico (1987) provided an three-dimensional analytical solution that allows the user to input source dimensions in the x, y, and z directions creating a vertical plane source in contrast to the horizontal plane source described above. A further development of this source orientation was presented in Ollila (1996) for estimating natural attenuation of groundwater contaminants. Superposition of the rectangular source orientations developed by Domenico can provide for simulation of concentration profiles and asymmetric concentration cross sections.

More complex source orientations are proposed by Leij et al. (1991). In the development of this analytical solution, solutions are provided for representing rectangular, circular, cylindrical, and parallelepipedal source regions (Figure 2-9).



**Figure 2-9: Source Orientations  
(Leij et al. 1991)**

O:\44237-001R005\Wpdocs\Report\Draft

Some researchers utilized novel approaches to the solution of the contaminant transport problem using analytical solutions. Hwang et al. (1985) incorporate a local analytical solution in a numerical model framework. By developing an analytical element at a node in the framework, a relationship with that node and its neighboring nodes is developed via a mathematical relationship. However, such an approach may require large amounts of computing power for more complex problems. Tang and Aral (1992) provide a solution for a layered aquifer that includes the main aquifer body and surrounding aquitards. Different values for degradation and retardation and other parameters are input for the aquifer and aquitards. This approach can therefore simulate different flow regimes in the same system and also can simulate the effects of diffusion into and out of aquitards.

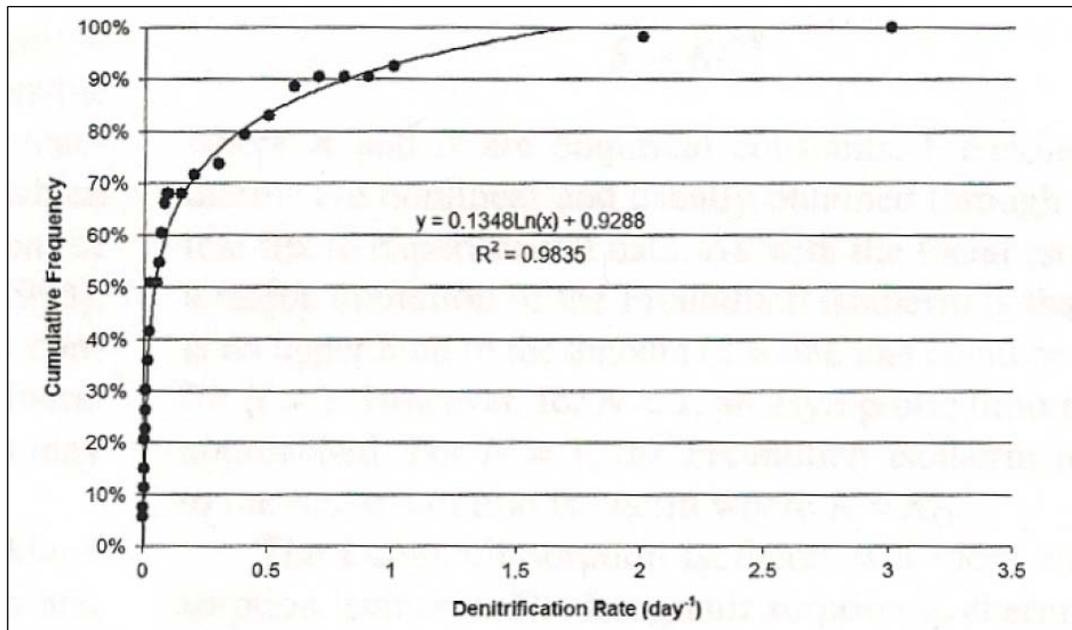
Among the analytical solution models, some researchers created spreadsheet solutions to the ADE that can potentially be useful, simple tools for simulating nitrogen transport in the subsurface. These studies include Ollila (1996), Elmore (2007), Karahan (2006), and Karahan and Ayyvaz (2005). While the equations themselves are typical solutions to the partial differential equation of the ADE, the use of a spreadsheet can be a very efficient method of calculating the solutions. Spreadsheets have the advantage of being relatively easy to program and use, wide availability, and iterative solution capabilities.

## 2.7 Parameter Estimation

Determining appropriate input parameters and the process of parameter estimation for fitting a groundwater model to observed data are an important yet difficult process. Parameter values can never be completely accurate, due to natural variations or incomplete data sets. Uncertainty related to input and calibration parameters leads to uncertainty in outputs. As such, some research has been performed specifically regarding parameter estimation. The objective of this research is to reduce the uncertainty as much as possible when estimating parameters, either as inputs to a model or through the calibration process.

A process of estimating parameters related to the transport of nitrogen and phosphorus from OWTS was developed by McCray et al. (2005). Cumulative frequency distributions (CFDs) were developed from data collected in the available literature to create statistical distributions for effluent concentrations, and nitrification and denitrification rates. In cases where inadequate data was available to produce a CFD, the mean, median and standard deviation was reported. These diagrams indicate the frequency of a reported value. The 50% value is the most frequently reported, whereas the 80% value means that 80% of the reported values are less than that value. Considering this for effluent concentrations as an example, the selection of the 90 percentile value would be considered as very conservative. This may be an appropriate choice if a simulation is intended to provide a protective concentration for a drinking water well.

The resulting CFD diagram (for an example see Figure 2-10) can be useful for selecting appropriate water quality input parameters when data is limited either for a site or in the literature. This is particularly true for nitrification and denitrification rates.



**Figure 2-10: Example Cumulative Frequency Distribution (McCray et al. 2005)**

Yanyong et al. (1992) describe a statistical process in the estimation of model parameters based on the observed data of the actual system. The method suggests using probability distributions of errors in the observed data to appropriately adjust parameters to reduce model residuals as much as possible. The researchers provide this as a direct method of estimating parameters, as opposed to performing a trial and error method to fit the model to the data. Therefore, the task of parameter fitting is less tedious, creates more optimal parameter sets, and has a statistical justification for the parameter values that were selected.

Regardless of the method chosen to estimate parameters, the analysis of uncertainty should also be included with any modeling effort. It is important to quantify the level of uncertainty as related to the simplifying assumptions of the model used and the quality of the data and how this influences the results.

O:\44237-001R005\Wpdocs\Report\Draft

## 2.8 Nitrification and Denitrification Modeling

The simulation of the nitrification process has not received much research beyond the use of Monod kinetics, zero-order or first-order mathematical expressions and is mostly expressed as part of a larger model if the transformation of ammonium to nitrate needs to be represented. However, the simulation of denitrification has resulted in numerous approaches to the problem. This is perhaps due to the complexity of the biogeochemical processes associated with denitrification. The methodology ranges from empirically-based expressions to complex numerical codes. The review of the literature identified 20 studies related to the modeling of denitrification. Heinen (2006) provides a comprehensive summary of over 50 denitrification models, mostly simple empirical expressions. Heinen and other researchers identify numerous influencing factors such as pH, water content, dissolved oxygen, and others that may be beyond the scope of such a modeling tool. The modeling of denitrification is not discussed in great detail in this review. The research is provided for possible reference if desired.



## Section 3.0

### Discussion and Analysis

---

The models addressed in the review represent potential approaches to the model tool being developed for this project. The models were grouped in such a manner since each zone of interest requires different parameters, inputs, and assumptions; the modeling tool being developed will likely consider simulations in the vadose and saturated zones. In addition to identifying models for the different zones, various modeling methods were also identified; mass-balance modeling, analytical modeling, numerical modeling, and transfer function modeling. Each of these has their own advantages and disadvantages.

Mass-balance models are probably the simplest of the models found in the research review. These models are largely based on estimates of mass flux or volumes, and as such do not necessarily require prior knowledge of the subsurface characteristics. These types of models can have value for predicting mass flux in a generalized sense. In fact, many researchers use mass- or flux- balance calculations to assist in estimating input rates and concentrations for transport modeling. However, these models require numerous simplifying assumptions and typically cannot be used to predict solute concentrations at specified points in time and space. Furthermore, they do not account for hydro-geochemical processes; thus, any change in mass flux that may be influenced by such characteristics is represented as a fractional loss that is assumed to be constant over time.

Analytical models are a deterministic approach that simulates systems based on relatively simple, but widely used equations of flow and transport. This approach does require prior knowledge of subsurface characteristics, but the mathematics behind the analytical solution is relatively simple and flexible. Input parameters are relatively few and can be readily adjusted. Therefore, an analytical solution can be applied to multiple sites or hypothetical sites. Furthermore, the solutions can be programmed into a spreadsheet program for ease of use. However, analytical solutions cannot simulate highly heterogeneous systems. Therefore, it is important to consider the level of spatial variability of a system and how accurately this variability must be represented.

Numerical modeling has the most applicability when considering heterogeneous systems that operate under non-steady conditions that are impractical with analytical solutions. This is usually the case when estimating flow and transport in the vadose zone. As with analytical solutions, these models are very flexible. Often, however, numerical models

C:\44237-001R005\Wpdocs\Report\Draft

are extremely complex, require relatively large amounts of input data (when compared to analytical solutions or mass-balance models), and require the use of computational methods. Furthermore, numerical models are often site-specific and not easily extended to other sites. Nonetheless, when considering a system that is highly heterogeneous or if boundary conditions cannot be simplified to constant, steady conditions, they are extremely useful.

The final category, transfer function models, operates on the assumption that deterministic models cannot appropriately account for the spatial variation in subsurface characteristics. This modeling approach, instead of relying on broadly applied parameter values, generates a probability distribution to account for water and solute movement. The major drawback to this model type is the lack of field studies to validate them and the likely resistance from regulatory agencies to their use since they do not rely on physical site characteristics. However, it is possible to couple stochastic methods with analytical models to better assess the uncertainty associated with the results.

When considering the modeling of denitrification, four approaches are possible. The first considers the details of the process, which may require inputs of temperature, pH, microbial population and growth dynamics, soil-moisture, and carbon availability in the modeling to capture the reduction functions that lead to denitrification. Secondly, models such as the analytical solution by Sun et al. (1999) could consider the reaction as a sequential first order process. The third approach simply identifies the process as a single first-order reduction, and therefore uses rate constants as an input parameter. Finally, recent work at Colorado School of Mines involves linking denitrification and nitrification rate constants to soil type and water content in analytical models. This provides an improvement over using only rate constants, without the need to consider all the biogeochemical processes at play. Considering the complexity of the denitrification process and the relative lack of understanding of how it works, the first approach is likely impractical. This may also be true when attempting to model denitrification as a sequential process, as the modeling would require rate constants for each step. Given the goals of the project and the level of influence of the process, the third approach may be sufficient to capture the denitrification process, while requiring the least amount of data collection. Field data and application of various models will be undertaken in this project to determine the appropriate level of complexity.

O:\44237-001R005\Wpdocs\Report\Draft



## Section 4.0 Conclusions

---

A review of the literature, the conceptual understanding of the transport of nitrogen as related to OWTS, and the goals of the project are all taken into consideration when beginning to describe the tool that will be developed. From this, several conclusions and some suggestions for the modeling tool can be developed.

The literature review was intended to identify the state-of-knowledge of nitrate fate and transport modeling, identify past models that may have provide good templates for the model developed by the FOSNRS Study, and assist in identifying key parameters and processes that need to be represented in a predictive tool.

As with any model development project, the appropriate approach can depend on numerous factors. When conceptualizing a model, several key questions need to be posed, such as:

- Will this model be constructed to represent a specific site of interest or be a predictive tool with broad applicability to a variety of sites?
- What is the desired output?
- What is the most appropriate method of calculating the output?
- Will this model require calibration to existing data sets?
- What, if any, regulatory requirements constrain the model choice?

The modeling tool that is being developed to simulate nitrate fate and transport will require certain features, some of which include:

- ease-of-use;
- ability to simulate time-variable OWTS inputs;
- simulation of transport and fate in both the vadose zone and saturated zones;

O:\44237-001R005\Wpdocs\Report\Draft

- representation of the numerous advective-dispersive and transformative processes that affect nitrate transport;
- simulation of temporal and spatial concentrations and mass loading downgradient of the source;
- include the impacts of seasonal rainfall variation on the source function; and
- incorporate critical OWTS operating characteristics that strongly influence nitrogen reduction.

Based on the above questions and objectives, many conclusions about the models and model types in the research summary can be made. No simple model (analytical or mass-balance) identified in the literature can currently achieve all of the above-described goals. Also, numerical models are generally not considered a useful tool for system design or regulatory compliance where broad applicability is desired. Thus, development of a new modeling tool is likely required and rigorous numerical modeling may be needed as a first step to determine the most important parameters to include.

A strictly mass-balance modeling approach will likely be inappropriate, as it either does not consider the known physical processes that influence nitrate transport or makes simplifying assumptions about these processes. Furthermore, the output will not satisfy the objectives of the model (time-variable estimations of concentrations at specific spatial points). Nonetheless, these approaches have value in the conceptualization of model inputs and should not be ignored. Transfer function models have not been widely applied and will likely encounter regulatory resistance, since they are based strictly on probabilities and do not directly consider measured site characteristics. Both analytical and numerical modeling methods are the most promising approaches when considering the FOSNRS Study model to be developed. These approaches will have wide applicability, regulatory acceptance, and are capable of estimating the important hydrogeochemical properties associated with nitrate fate and transport.

The modeling tool will need to consider transport and transformation (chemical and physical) in the vadose zone, because the Nitrogen transformations that occur in this zone have considerable influence on the mass-flux input into the underlying aquifer. This can be a numerical one-dimensional solution of the Richards' Equation as suggested by (Bakhsh et al. 2004) or (Heatwole and McCray 2007) coupled with the ADE applied to the unsaturated zone as found in (Selim and Mansell 1976). A one-dimensional formulation can likely be implemented in a spreadsheet. Additionally, the modeling will need to consider temporally and spatially variable inputs for multiple OWTS, as would be found

C:\44237-001R005\Wpdocs\Report\Draft

in a community development. This could be addressed through a series of one-dimensional vadose zone models that could provide input to a multi-dimensional groundwater flow and transport model such as those suggested by (Ollila 1996) or (Galya 1987). Both of these studies use the horizontal plane source model or some variation and are also capable of transient simulations. However, the models likely will not be capable of interacting with each other in the vadose zone (i.e., strictly vertical flow is assumed). Nonetheless, the value of including these model features is important when simulating the areal distribution of OWTS in a potential housing development and the temporal variation of source input due to changes in wastewater input rate and precipitation recharge. These combined models can likely be implemented in a spreadsheet or using Fortran or C++ programming while maintaining simple and straight-forward input requirements. Of course, no similar model is available to our knowledge, so considerable model research and development must be achieved by this project. Within the models identified by the research review, the model developed by (Mehran et al. 1983-1984) is an example of a coupled modeling code for the transport and transformation of nitrogen but it lacks certain features for simulating nitrogen fate and transport related to OWTS.

The literature review has suggested the most likely processes and parameters that will need to be considered when developing the modeling tool. The fate and transport of nitrogen products is a result of advective movement, retardation via adsorption, and the transformative processes of nitrification and denitrification. These processes are to be calculated in the model tool via the solutions of the appropriate equations using the necessary parameters, described below. Key parameters to consider for simulation should consist of:

- physical parameters of the media such as bulk density, water content, and soil characteristics;
- advective-dispersive parameters such as hydraulic conductivity, hydraulic gradient, porosity (or groundwater velocities), and dispersivity values;
- retardation factor values for ammonium sorption; and
- rate coefficients for transformative reactions, typically first-order rate constants

A majority of the parameter values needed for model input can be collected during site characterization. McCray et al. (2005) utilize CFD's for the estimation of initial parameter values if utilizing literature values but the approach results in an uncertain model output where the degree of uncertainty must be quantified.

Within the models identified by the research review, the model developed by (Mehran et al. 1983-1984) is a representative example of a coupled modeling code for the transport and transformation of nitrogen. Additionally, many analytical models were found in the literature review (nitrate-specific and general analytical solutions) that are appropriate for the modeling tool, since these can be programmed into a spreadsheet and can be user-friendly.



## Section 5.0 References

---

- Addiscott, T.M., and Wagenet, R.J. (1985). Concepts of Solute Leaching in Soils: A Review of Modelling Approaches. *Jour. of Soil Science*, 36, 411-424.
- Anderson, P.F., Mercer, J.W., and White, H.O.J. (1983). Numerical Modeling of Salt-Water Intrusion at Hallandale, FL. *Ground Water*, 26(5), 619-630.
- Bakhsh, A., Hatfield, J.L., Kanwar, R.S., Ma, L., and Ahuja, L.R. (2004). Simulating Nitrate Drainage Losses from a Walnut Creek Watershed Field. *J. Environ. Qual.*, 33, 114-123.
- Bear, J. (1979). *Hydraulics of Groundwater*, McGraw Hill, New York.
- Beggs, R.A., Tchobanoglous, G., Hills, D., and Crites, R.W. (2005). Modeling Subsurface Drip Application of Onsite Wastewater Treatment System Effluent. *ASABE*.
- Bibby, R. (1981). Mass Transport of Solutes in Dual-Porosity Media. *Water Resources Research*, 17(4), 1075-1081.
- Briggs, G.R., Roeder, E., and Ursin, E. (2007). Nitrogen Impact of Onsite Sewage Treatment and Disposal Systems in the Wekiva Study Area." In: *FDOH*.
- Canter, L.W. (1996). Nitrates in Groundwater.
- Carey, M.A., and Lloyd, J.W. (1985). Modeling Non-Point Sources of Nitrate Pollution of Groundwater in the Great Ouse Chalk. *Journal of Hydrology*, 78(1/2), 83-106.
- Carslaw, H.W., and Jaeger, J.C. (1959). *Conduction of Heat in Solids*.
- Celia, M.A., Kindred, J.S., and Herrera, I. (1989). Contaminant Transport and Biodegradation. 1. A numerical model for Reactive Transport in Porous Media. *Water Resources Research*, 25(6), 1141-1148.

C:\44237-001R005\wpdocs\Report\Draft

Conan, C., Bouraoui, F., Turpin, N., de Marsily, G., and Bidoglio, G. (2003). Modeling Flow and Nitrate Fate at Catchment Scale in Brittany (France). *Jour. Environ. Qual.*, 32, 2026-2032.

Crites, R. and G. Tchobanoglous (1998). *Small and Decentralized Wastewater Management Systems*. McGraw-Hill, San Francisco, CA.

DeSimone, L.A., and Howes, B.L. (1998). Nitrogen Transport and Transformations in a Shallow Aquifer Receiving Wastewater Discharge: A Mass Balance Approach. *Water Resources Research*, 34(2), 271-285.

Domenico, P.A. (1987). An Analytical Model of Multidimensional Transport of Decaying Contaminant Species. *Journal of Hydrology*, 91(1/2), 47-58.

Dyson, J.S., and White, R.E. (1987). A Comparison of the Convection-Dispersion Equation and Transfer Function Model for Predicting Chloride Leaching Through an Undisturbed, Structured Clay Soil. *Jour. of Soil Science*, 38(2), 157-172.

Elmore, A.C. (2007). "Applying a One-Dimensional Mass Transport Model Using Groundwater Concentration Data." *Jour. Environ. Engineering*, April, 372-379.

Frimpter, M.H., Donahue, J.J., and Rapacz, M.V. (1990). A Mass Balance Model for Predicting Nitrate in Ground Water. *Journal of N.E. Water Works Assoc.*, Winter, 1990, 219-232.

Galva, G.P. (1987). A Horizontal Plane Source Model for Groundwater Transport. *Ground Water*, 25(6), 733-739.

Geza, M., and McCray, J.E. (2007). Modeling the Effect of Population Growth on Stream Nutrient Concentration in Turkey Creek Watershed using the WARMF Model. Proc. Eleventh National Symposium on Individual and Small Community Sewage Systems, ASABE.

Hansen, S., Jensen, H.E., Nielsen, N.E., and Svendsen, H. (1991). Simulation of Nitrogen Dynamics and Biomass Production in Winter Wheat using the Danish Simulation Model DAISY. *Nutrient Cycling in Agroecosystems*, 245-259.

Hantzche, N.N., and Finnemore, E.J. (1992). Predicting Ground-Water Nitrate-Nitrogen Impacts. *Ground Water*, 30(4), 490-499.

O:\4237-001R005\Wpdocs\Report\Draft

Heatwole, K.K., and McCray, J.E. (2006). A Simple Model for Predicting Nitrate Plumes. Proc. NOWRA 16<sup>th</sup> Annual Technical Education & Exposition Conference.

Heatwole, K.K., and McCray, J.E. (2007). Modeling Potential Vadose-zone Transport of Nitrogen from Onsite Wastewater Systems at the Development Scale. *Jour. of Contam. Hydrology*, 91, 184-201.

Heinen, M. (2006). Simplified Denitrification Models: Overview and Properties. *Geoderma*, 133, 444-463.

Hendry, M.J., Gillham, R.W., and Cherry, J.A. (1983). An Integrated Approach to Hydrogeologic Investigations - A Case History. *Journal of Hydrology*, 63(3/4), 211-232.

Heng, H.H., and Nikoladis, N.P. (1998). Modeling of Nonpoint Source Pollution of Nitrogen at the Watershed Scale. *Journal of the Amer. Water Resources Assoc.*, 34(2), 359-374.

Huntzinger, D.N., and McCray, J.E. (2003). Numerical Modeling of Unsaturated Flow in Wastewater Soil Absorption Systems. *Ground Water Monitoring and Remediation*, 23(2).

Huyakorn, P.S. (1983). Computational Methods in Subsurface Flow.

Hwang, J.C., Chen, C.J., Sheikholeslami, M., and Panigrahi, B.K. (1985). Finite Analytic Numerical Solution for Two-Dimensional Groundwater Solute Transport. *Water Resources Research*, 21(9), 1354-1360.

Jabro, J.D., Stout, W.L., Fales, S.L., and Fox, R.H. (2001). SOIL-SOILN Simulations of Water Drainage and Nitrate Nitrogen Transport from Soil Core Lysimeters. *Jour. Environ. Qual.*, 30, 584-589.

Johnson, A.D., Cabrera, M.L., McCracken, D.V., and Radcliffe, D.E. (1999). LEACHN Simulations of Nitrogen Dynamics and Water Drainage in an Ultisol. *Agronomy Jour.*, 91, 597-606.

Johnsson, H., Bergstrom, L., and Jansson, P.E. (1987). Simulated Nitrogen Dynamics and Losses in a Layered Agricultural Soil. *Agriculture, Ecosystems, and Environ.*, 18(4), 333-356.

Jury, W.A. (1982). Simulation of Solute Transport using a Transfer Function Model. *Water Resources Research*, 18(2), 363-368.

Jury, W.A., and Sposito, G. (1985). Field Calibration and Validation of Solute Transport Models for the Unsaturated Zone. *Soil Sci. Soc. Am. Journal*, 49, 1331-1341.

Karahan, H. (2006). Implicit Finite Difference Techniques for the Advection-Diffusion Equation Using Spreadsheets. *Advances in Engin. Software*, 37, 601-608.

Karahan, H., and Ayvaz, M.T. (2005). Transient Groundwater Modeling Using Spreadsheets. *Advances in Engin. Software*, 36, 374-384.

Leij, F.J., Skaggs, T.H., and van Gnuchten, M.T. (1991). Analytical Solutions for Solute Transport in Three-Dimensional Semi-infinite Porous Media. *Water Resources Research*, 27(10), 2719-2733.

Leij, F.J., Toride, N., and van Gnuchten, M.T. (1993). Analytical Solutions for Non-Equilibrium Solute Transport in Three-Dimensional Porous Media. *Jour. Of Hydrology*, 151, 193-228.

Leismann, H.M., and Frind, E.O. (1989). A Symmetric-Matrix Time Integration Scheme for the Efficient Solution of Advection-Dispersion Problems. *Water Resources Research*, 25(6), 1133-1139.

Lerner, D.N., and Papatolios, K.T. (1993). A Simple Analytical Approach for Predicting Nitrate Concentrations in Pumped Ground water. *Ground Water*, 31(3), 370-375.

Lowe, K., Rothe, N., Tomaras, J., DeJong, K., Tucholke, M., Drewes, J., McCray, J., Munakata-Marr, J. (2007). *Influent Constituent Characteristics of the Modern Waste Stream from Single Sources: Literature Review*. WERF, 04-DEC-1. 89pg. PDF available at: [www.ndwrcdp.org/publications](http://www.ndwrcdp.org/publications)

McCray, J.E., Kirkland, S.L., Siegrist, R.L., and Thyne, G.D. (2005). Model Parameters for Simulating Fate and Transport of On-Site Wastewater Nutrients. *Ground Water*, 43(4), 628-639.

Mehran, M., Noorishad, J., and Tanjii, K.K. (1983-1984). Numerical Technique for Simulation of the Effect of Soil Nitrogen Transport and Transformations on Groundwater Contamination. *Environmental Geology*, 5(4), 213-218.

Metcalf and Eddy, Inc. (1991). *Wastewater Engineering: Treatment, Disposal and Reuse*. 3<sup>rd</sup> Ed. McGraw-Hill, New York.

- Mironenko, E.V., and Pachepsky, Y.A. (1984). Analytical Solution for Chemical Transport with Non-equilibrium Mass Transfer, Adsorption and Biological Transformation. *Journal of Hydrology*, 70, 167-175.
- Molénat, J., and Gascuel-Odeux, C. (2002). Modelling Flow and Nitrate Transport in Groundwater for the Prediction of Water Travel Times and of Consequences of Land-use Evolution on Water Quality. *Hydro. Processes*, 16, 479-492.
- Moreels, E., De Neve, S., Hofman, G., and Van Meirvenne, M. (2003). Simulating Nitrate Leaching in Bare Fallow Soils: A Model Comparison. *Nutrient Cycling in Agroecosystems*, 67(2), 137-144.
- Ollila, P.W. (1996). Evaluating Natural Attenuation with Spreadsheet Analytical Fate and Transport Models. *Ground Water Monit. and Remed.*, Fall, 69-75.
- Pickens, J.F., and Lennox, W.C. (1976). Numerical Simulation of Waste Movement in Steady Groundwater Flow Systems. *Water Resources Research*, 12(2), 171-180.
- Pinder, G.F., and Frind, E.O. (1972). Applications of Galerkin's Procedure to Aquifer Analysis. *Water Resources Research*, 8(1), 108-120.
- Puckett, L.J., and Lowder, T.K. (2002). Transport and Fate of Nitrate in a Glacial Outwash Aquifer in Relation to Groundwater Age, Land Use Practices and Redox Processes. *J. Environ. Qual.*, 31(3), 782-796.
- Reneau, R.B., Hagedorn, C., and Jantrania, A.R. (2001). Performance evaluation of two pre-engineered onsite treatment and effluent dispersal technologies. Proceedings for the Ninth National Symposium on Individual and Small Community Sewage Systems. ASAE, St. Joseph, MI.
- Refsgaard, J.C., Thorsen, M., Jensen, J.B., Kleeschulte, S., and Hansen, S. (1999). Large-Scale Modeling of Groundwater Contamination from Soil Leaching. *Journal of Hydrology*, 221, 117-140.
- Reynolds, C.M., and Iskandar, I.K. (1995). A Modeling-based Evaluation of the Effect of Wastewater Application Practices on Groundwater Quality. In: *CRREL Report #95-2*.
- Selim, H.M., and Mansell, R.S. (1976). Analytical Solution of the Equation for Transport of Reactive Solutes Through Soils. *Water Resources Research*, 12(3), 528-532.

- Siegrist, R.L. (2001). Perspectives on the science and engineering of onsite wastewater systems. *Small Flows*, 2(4): 8-13.
- Sonnenburg, T.O., Christensen, B.S.B., Nyegaard, P., Henriksen, H.J., and Refsgaard, J.C. (2003). Transient Modeling of Regional Groundwater Flow using Parameter Estimates from Steady-State Automatic Calibration. *Journal of Hydrology*, 273(3), 188-204.
- Srivasta, R., and Yeh, T.C.J. (1992). A Three-Dimensional Numerical Model for Water Flow and Transport of Chemically Reactive Solute Through Porous Media under Variably Saturated Conditions. *Advances in Water Res.*, 15(5), 275-287.
- Sun, Y., Petersen, J.N., and Clement, T.P. (1999). Analytical Solutions for Multiple Species Reactive Transport in Multiple Dimensions. *Jour. of Contam. Hydrology*, 35, 429-440.
- Tang, Y., and Aral, M.M. (1992). Contaminant Transport in Layered Porous Media. 1. General Solution. *Water Resources Research*, 28(5), 1389-1397.
- Taylor, J.R. (2003). Evaluating Groundwater Nitrates from On-Lot Septic Systems, A Guidance Model for Land Planning in Pennsylvania.
- Tinker, J.R.J. (1991). An Analysis of Nitrate-Nitrogen in Ground Water Beneath Unsewered Subdivisions. *Ground Water Monitoring Review*, Winter, 141-150.
- U.S. EPA (1997). Response to Congress on Use of Decentralized Wastewater Treatment Systems. U.S. Environmental Protection Agency, Office of Water, Washington, D.C.
- U.S. EPA (2002). *Onsite Wastewater Treatment Systems Manual*. Report No. 625/R-00/008. U.S. Environmental Protection Agency, Cincinnati, OH.
- Wehrmann, H.A. (1984). Managing Ground Water Quality by Mass Balance Modeling in the Rockton-Roscoe Area, Illinois. *Proceedings of the NWWA Eastern Regional Conference on Ground Water Quality*, 558-587.
- Weintraub, L.H.Z., Chen, C.W., Goldstein, R.A., and Siegrist, R.L. (2004). WARMF: A Watershed Modeling Tool for Onsite Wastewater Systems. Proceedings of the 10th National Symposium on Individual and Small Community Sewage Systems, ASAE, pp. 636-646.

White, R.E. (1987). Transfer Function Model for the Prediction of Nitrate Leaching under Field Conditions. *Journal of Hydrology*, 92(3/4), 207-222.

White, R.E., Heng, L.K., and Magesan, G.N. (1998). Nitrate Leaching from a Drained, Sheep-grazed Pasture. II. Modelling Nitrate Leaching Losses. *Aust. Jour. Soil Res.*, 36, 963-977.

Yanyong, X., Thomson, N.R., and Sykes, J.F. (1992). Fitting a Groundwater Transport Model by L1 and L2 Parameter Estimators. *Advances in Water Res.*, 15(5), 303-310.

Young, C.P., Oakes, D.B., and Wilkinson, W.B. (1976). Prediction of Future Nitrate Concentrations in Ground Water. *Ground Water*, 14(6), 426-438.