FLORIDA DEPARTMENT OF HEALTH

Florida Onsite Sewage Nitrogen Reduction Strategies Study

Task A.2

Literature Review of Nitrogen Reduction Technologies for Onsite Sewage Treatment Systems

Final Report

August 2009

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Environmental Engineers & Scientists

In association with

SCHOOL OF MINES COLORADO 1874

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
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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

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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.

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).

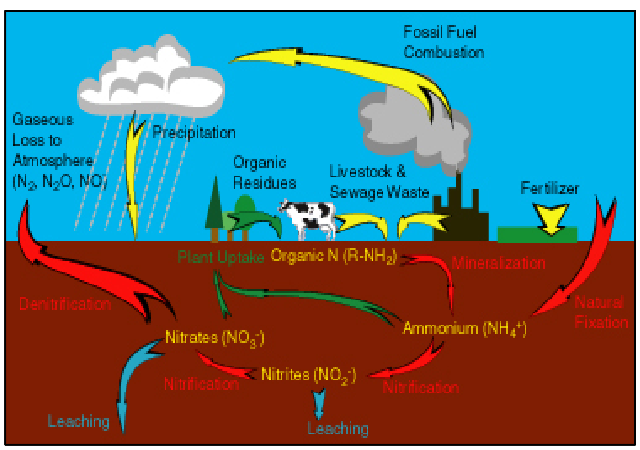

Figure 2-1: The Nitrogen Cycle (Harrison, 2003)

(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.

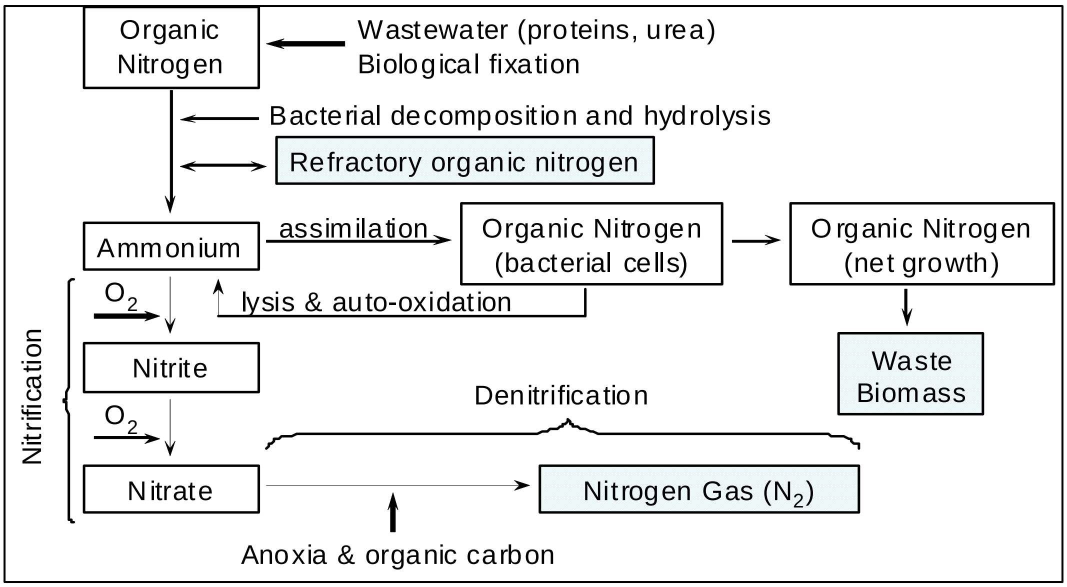

Figure 2-2: Nitrogen Transformation in Biological Processes
(Eckenfelder and Argaman, 1991)

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).

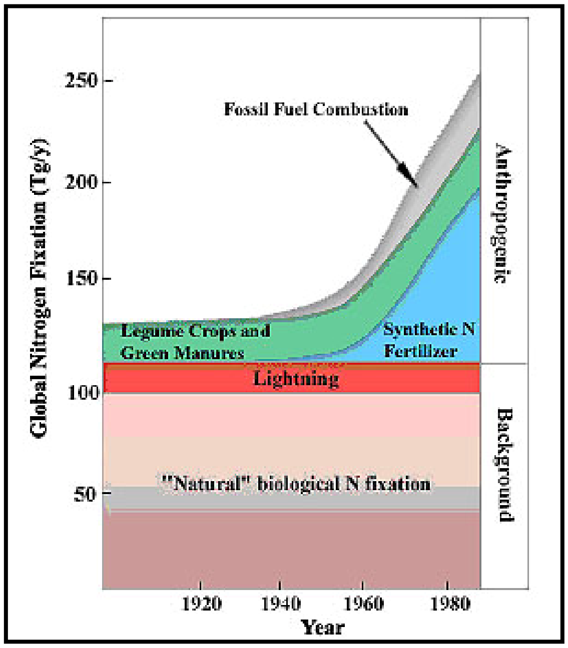

Figure 2-3: Recent Increases in Anthropogenic N Fixation in Relation to "Natural" N Fixation
(Harrison, 2003)

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 (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 dominate 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-

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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.

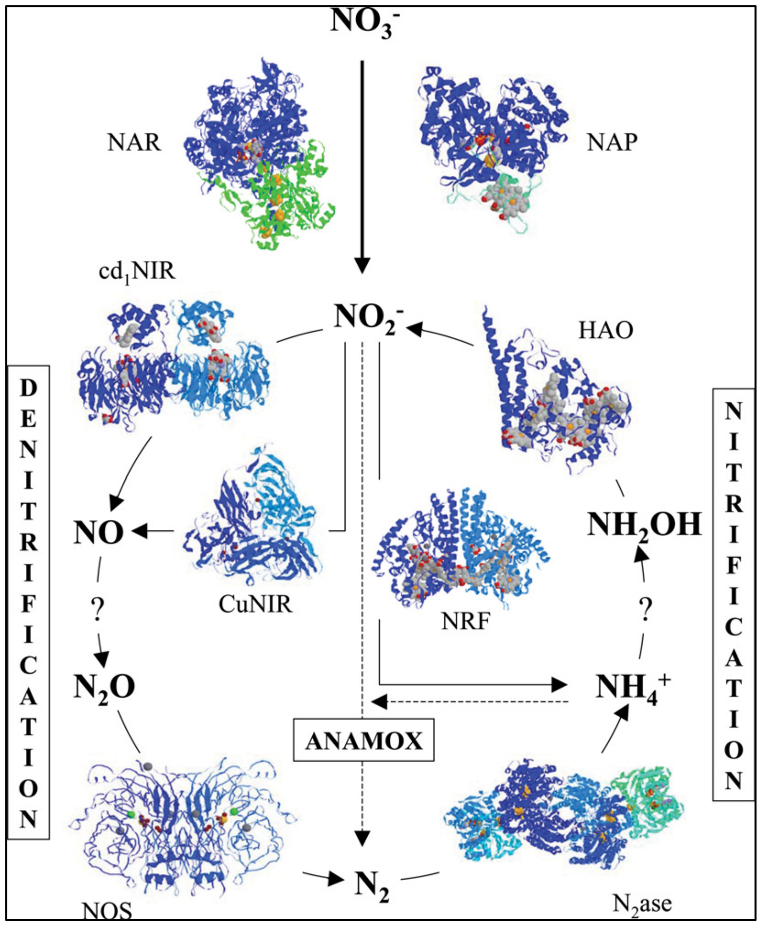

Figure 2-4: Illustration of the Classic Heterotrophic Denitrification and Anammox Denitrification (Structures of the enzymes are shown in each step; question marks represent unsolved structures) (Butler and Richardson, 2005)

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).

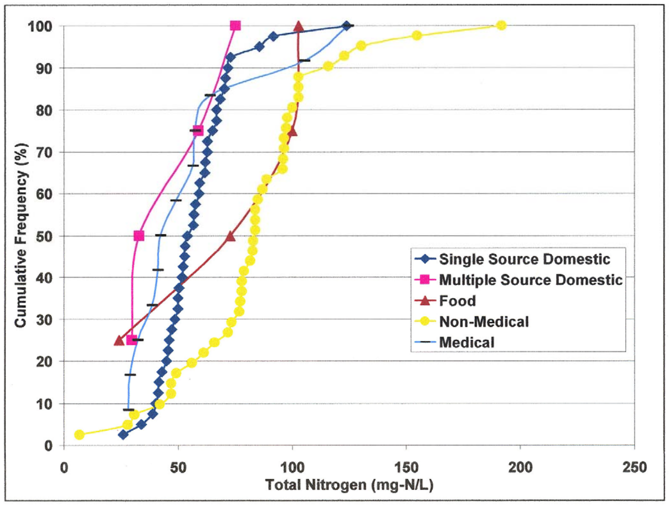

Figure 3-1: Cumulative Frequency of Total Nitrogen Concentrations in Septic Tank Effluent
(Lowe, Rothe et al., 2006)

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	-	13.6 8	-	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	-	-	-	-	-	-	-	-	-	-

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

	Median	Average	Standard Deviation	Range	Number of Reported Values
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.

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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.

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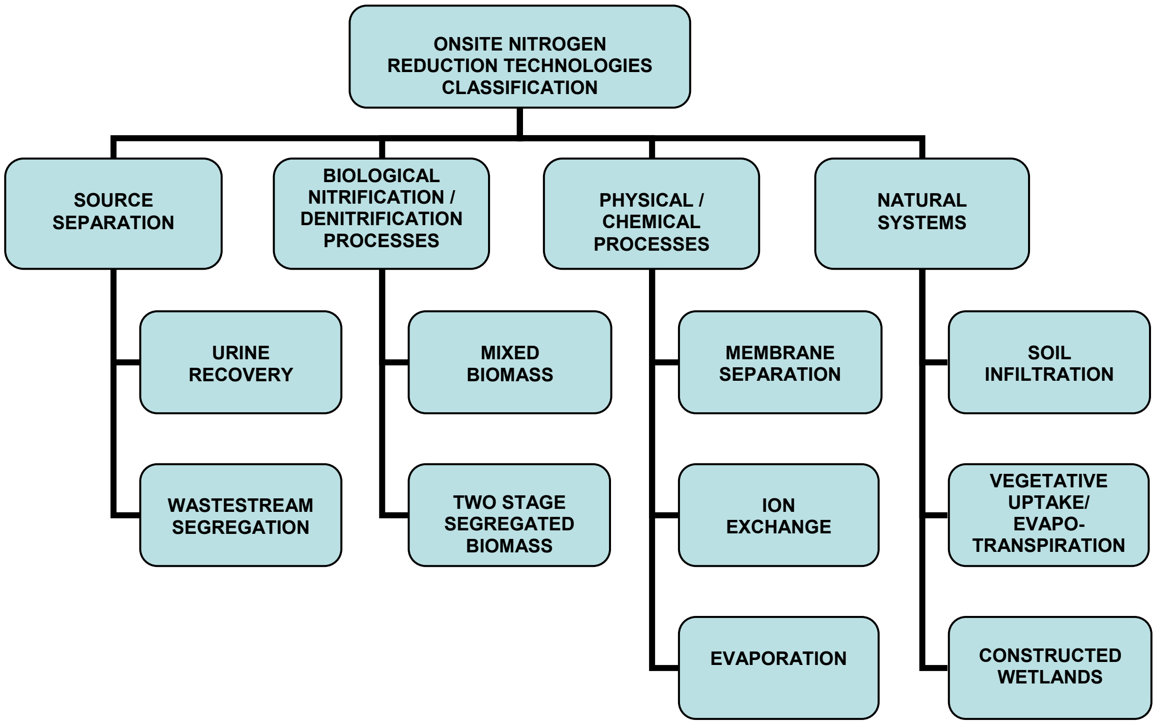

Figure 4-1: Treatment Options for Reducing Nitrogen in Household Sewage

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.

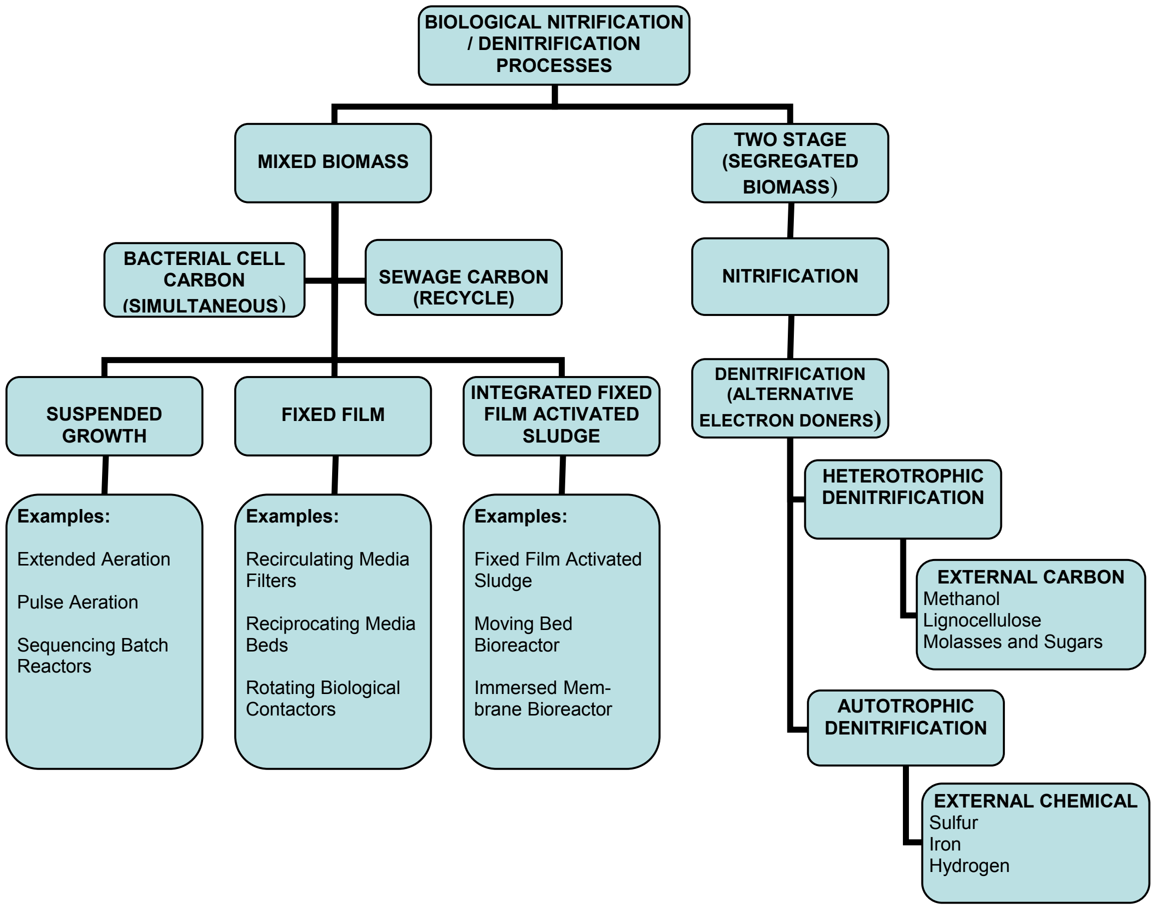

Figure 4-2: Onsite Treatment Technology Categories for Biological Nitrification/Denitrification Processes

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 oxic/anoxic environments (simultaneous denitrification), 2) mixed biomass with recycle 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 $CaCO_3$) 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 typically less than 50 percent nitrogen removal with these systems.

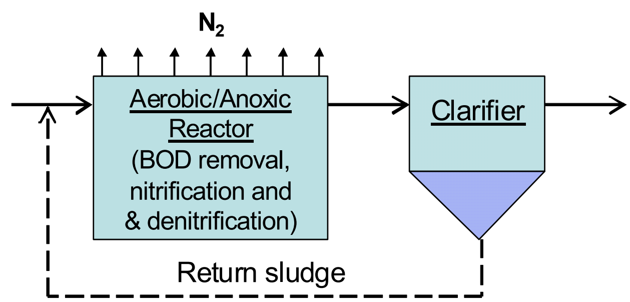

Figure 4-3: Alternating Oxic / Anoxic Reactor Denitrification

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 usually a smaller 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.

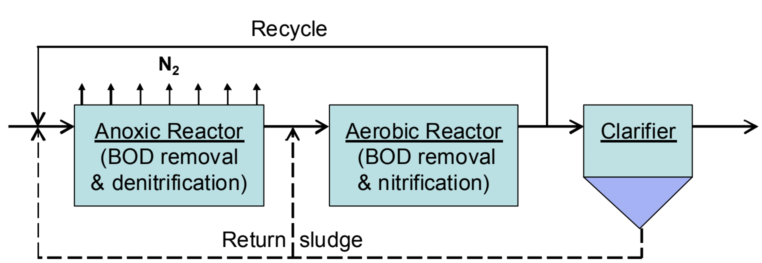

Figure 4-4: Mixed Biomass Recycling Denitrification Process

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. This approach requires 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.

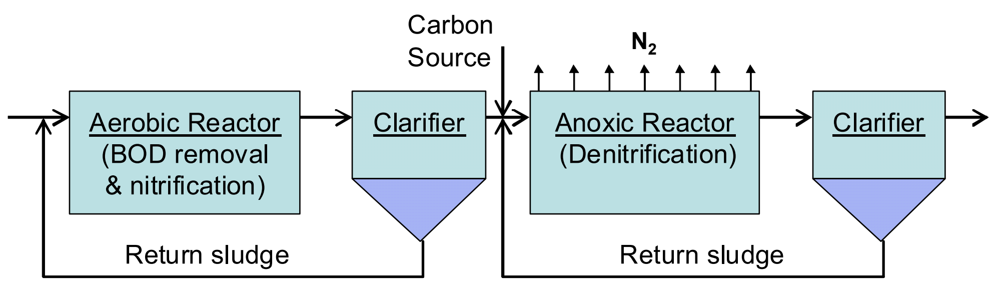

Figure 4-5: External Electron Donor Denitrification Process

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, Nitisoravut 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.

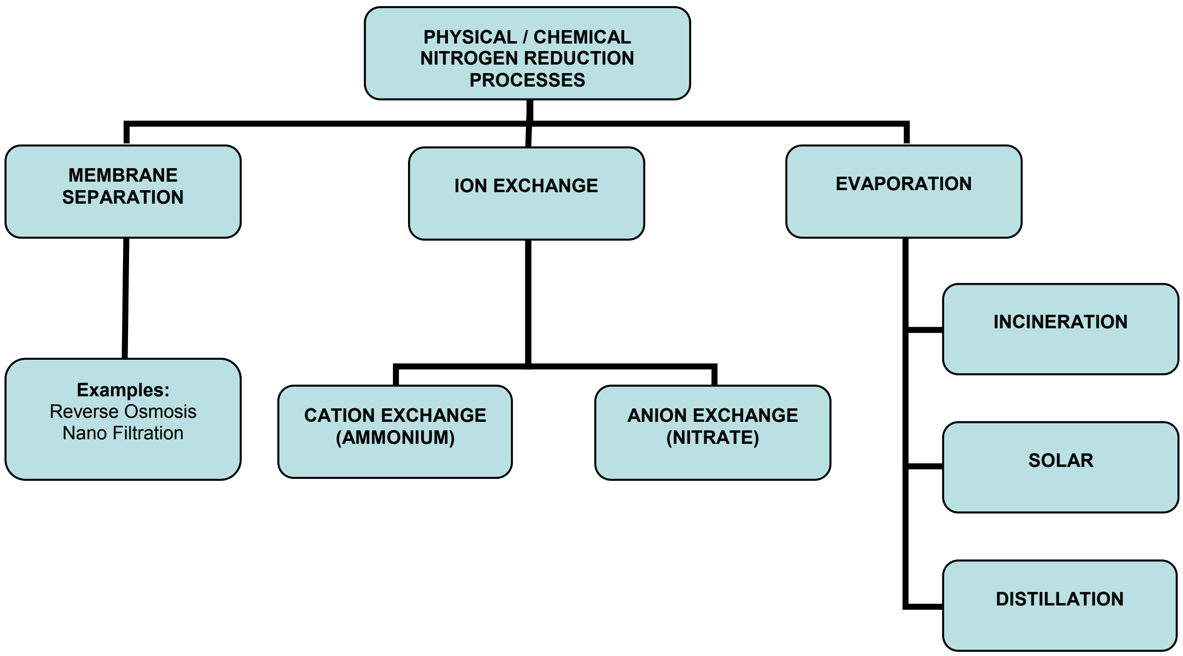

Figure 4-6: Onsite Treatment Technology Categories for Physical/Chemical Processes

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. It is becoming more popular in wastewater combined with biological treatment as membrane bioreactors. Ion exchange also requires pre-treatment and commercial regeneration of the exchange resins. Evaporation technologies can be effective in warm climates, but require periodic removal and appropriate disposal of the evaporates and are typically very energy intensive. Solar evaporation and distillation are emerging options for households but are early in their 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. Source separation is an option gaining more attention with the availability of urine separating toilets. For common separation options, see Figure 4-7.

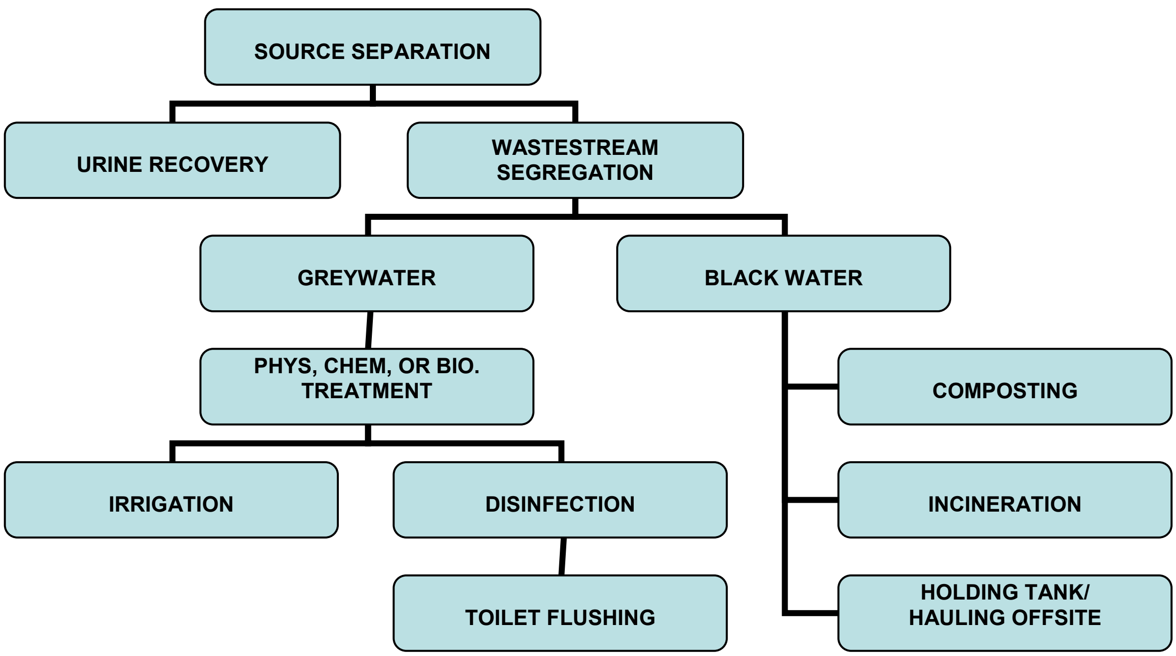

Figure 4-7: Nitrogen Source Separation Categories

4.4 Natural Systems

Natural systems are included as a separate classification because they are the last step in the process sequence of an OSTDS. They utilize a combination of physical, chemical and biological processes that occur naturally in the soil. 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.

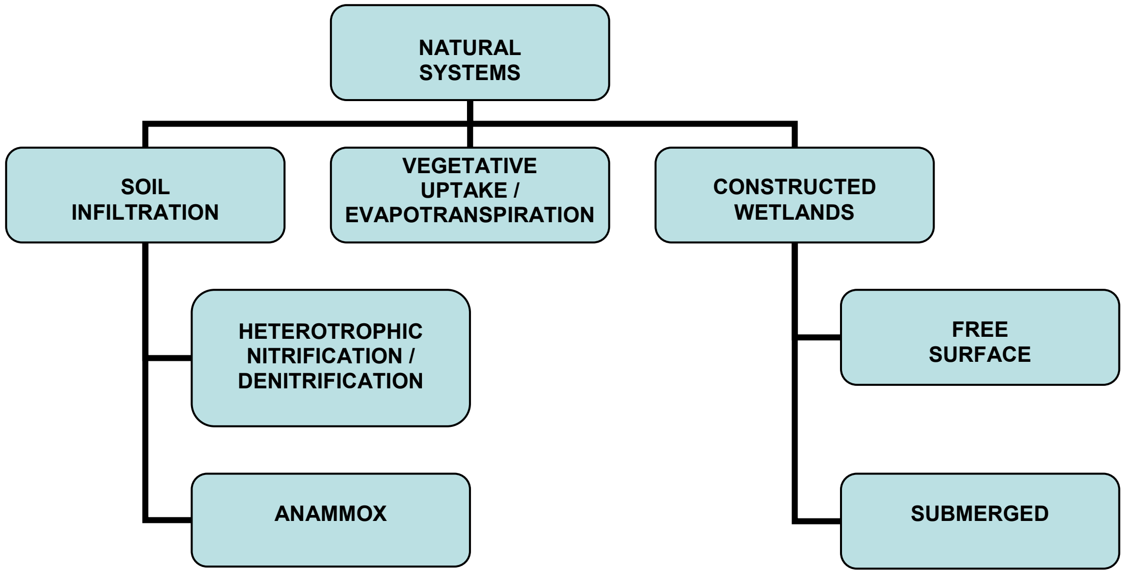

Figure 4-8: Categories of Natural Systems for Nitrogen Reduction

4.5 Passive Nitrogen Removal

Treatment systems can be either "passive" or "active". The Florida Department of Health (FDOH) has studied passive treatment systems for nitrogen removal under PNRS I. It defined "passive" for the purpose of this study 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.

Passive in this definition needs to be distinguished from adjectives such as consistent, reliable, low-energy or low-maintenance. Systems with such characteristics may be active or passive. Low-maintenance systems are generally preferred for onsite wastewater treatment because if well designed, they run largely on their own with less frequent need for inspection or servicing, as compared to active systems. 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. Onsite systems tend to be designed conservatively large because there are few operational remedial measures that can be taken if undersized.

This definition precludes most nitrogen reduction options primarily because of the requirement for reactive media. Only biological two-stage systems would qualify as passive biological treatment under this definition (Figure 4-4). Cation exchange (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 (Smith, 2008). 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. While the FDOH definition of “passive” is followed in describing and comparing the different nitrogen reduction processes and technologies in this review, it is recommended not to focus exclusively on this criterion in evaluating nitrogen reduction strategies.

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. Variations of this configuration are possible. In the septic tank, proteins are hydrolyzed releasing the organic nitrogen, which is reduced 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 four 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 though the remainder of the system so that flow through the system occurs by gravity, which eliminates the need for additional pumps. Fourth, its use can be proportional to flow, reducing the energy need compared to continuously aerating systems. 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 and additional treatment can occur.

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 $CaCO_3$ 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 $CaCO_3$, would be necessary to nitrify all of the TN. The wastewater itself can add 60-120 mg/L alkalinity as $CaCO_3$ (Crites and Tchobanoglous, 1998) but there may be many areas where sufficient alkalinity is unavailable for nitrification.

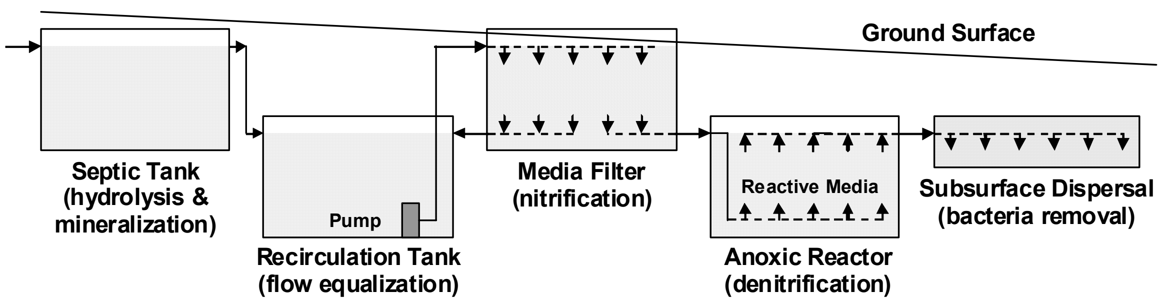

Figure 4-9: Passive Two-Stage Denitrification System

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.

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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.

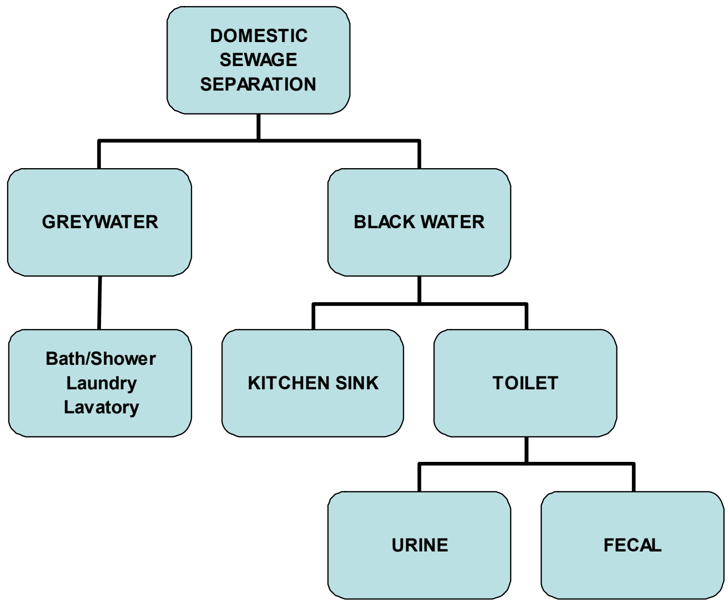

Figure 5-1: Domestic Wastestream Components

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$_5$	TSS	Total N (as N)	Total P (as P)
A	Non-kitchen sinks, clothes washer, shower, bath-tubs	32	11.4	5.2	0.8	0.2
B	Kitchen sinks, dishwasher, garbage grinder	10.3	35.1	38.5	1.7	0.3
C	Toilet: non-urine	17.5	12.5	80	1.1	0.4
D	Toilet: urine	0.6	4.2	0.1	10.9	1.2
Sum		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, wastestream 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_5	TSS	Total N (as N)	Total P (as P)	C-BOD_5	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.

Although not typically referred to as a “wastestream”, 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 Czemiel 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 a 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 electrodialysis/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 [$(NH_4)MgPO_4 \cdot 6H_2O$], a mineral called struvite, which has direct use as plant fertilizer (Ronteltap, Maurer et al. 2007a; Yetilmezsoy and Sapci-Zengin 2009). Hydroxyapatite [$Ca_{10}(PO_4)_6(OH)_2$] 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, Schuurbiers 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 potential 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). 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) 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 recovering 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 recoveries from the wastestream were 74 to 99 percent while ammonia recoveries 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, i.e. little nutrient reduction, and effluent with low TSS and high soluble COD.

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 settable and floatable suspended solids in the raw wastewater. These solids collect as sludge and scum within the tank where the organic nitrogen 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)
Electron Donor	Organic carbon from bacterial cells	Organic carbon from influent wastewater	External electron donor (Organic carbon; Ligno-cellulose; Sulfur; Iron, Other)
Typical N Reductions	40 to 65%	45 to 75%	70 – 96%
Typical Technologies	• Extended aeration[1] • Pulse aeration[2] • Recirculating media filters[3] • Sequencing batch reactors[4] • Reciprocating media beds[5] • Membrane bioreactor[6]	• Extended aeration with recycle back to septic tank • Recirculating media beds with recycle back to septic tank[7] • Moving bed bioreactor	• Heterotrophic suspended growth[8] • Heterotrophic packed bed fixed film • Autotrophic packed bed fixed film[9]

[1] *Leverenz, et al., (2002); USEPA (2002)*
[2] *California State Water Resources Control Board (2002)*
[3] *USEPA (2002)*
[4] *Ayres Associates (1998)*
[5] *Behrends, et al. (2007)*
[6] *Abbeggen, et al., (2008); Sarioglu, et al. (2009)*
[7] *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); Louden et al. (2005)*
[8] *USEPA, (1993)*
[9] *Rich (2007); Heufelder et al. (2008)*

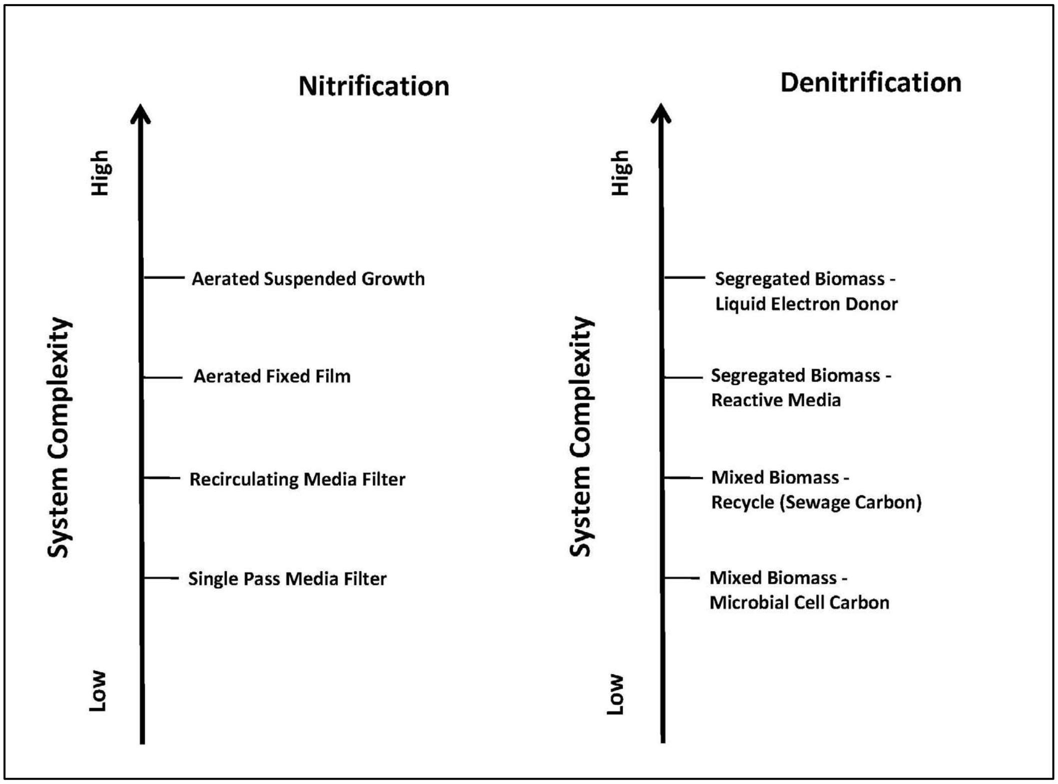

Figure 5-3: Relative Complexities of Nitrification / Denitrification Unit Operations

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 better treatment than septic tanks alone in order to reduce clogging of the infiltrative surface in the drainfield by removing BOD_5. 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.

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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 internatant 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 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. When looking at cross sections of a media filter, carbonaceous BOD is depleted first in the percolating wastewater, then nitrifying microorganisms thrive deeper in the filter. Under some conditions deep in the filter, oxygen may be depleted and denitrification could possibly occur.

"Single pass" or "intermittent" filters alone are not typically used for nitrogen removal. This is because the wastewater passes through the filter media only once before being discharged for further treatment or dispersal. This generally results in good nitrification, but does not provide overall nitrogen reduction sufficient to meet nitrogen reduction standards. Recirculating filters recycle the filtrate through the filter several times. The recirculation provides the needed wastewater residence times in the media to achieve greater nitrification. Recirculation provides more control of treatment process by adjustments that can be made to recirculation 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 (the "recirculation" part) results in significant nitrogen removal. Also, 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 recirculating 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_3-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 NH_3-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).

In column studies with a variety of different media, 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^2-day and 0.0045 to 0.0119 lb BOD_5/ft^2-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^2-day (Rodgers, Lambe et al. 2006).

Table 5.4
Summary of Media Filter Performance

Media Type	Features	Typical Performance Range
Single Pass Media Filters		
Expanded Clay Filters (single pass)[1]	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^2-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
Zeolite Filters (single pass)[2]	20 - 30 in. media depth 2.9 to 6.1 gal/ft^2-day	TN: Removal: 36.1% Influent: 72.2 mg/L Effluent: 43.6 mg/L NH3-N: Removal: 98.6 to 99.9% Influent: 63.4 to 70 mg/L Effluent: 0.036 to 1 mg/L NO3-N: Effluent: 38.8 to 57 mg/L
Peat Filters (single pass and/or recirculation)[3]	24 - 36 in. depth 3 to 6 gal/ft^2-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
Recirculating Media Filters		
Recirculating Sand Filters[4]	1.5 - 3 mm media 18 - 36 in. depth 3 - 5 gal/ft^2-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 Filters (with recirculation)[5]	2 - 3 in. cubes 36 - 72 in. depth 8 - 17 gal/ft^2-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 Filters (single pass and/or recirculation)[3]	24 - 36 in. depth 3 to 6 gal/ft^2-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

Table 5.4
Summary of Media Filter Performance

Media Type	Features	Typical Performance Range
Open Cell Foam Filters (with recirculation)[6]	3 - 4 in. cube media 48 in. depth 11 gal/ft^2-day	TN: Removal: 62% Effluent: 14 mg/L NH3-N: Effluent: 2.4 mg/L NO3-N: Effluent: 10 mg/L
Coir Filters (with recirculation)[7]	Coconut coir media 18 gal/ft^2-day 5.88 gal/ft^3-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 Filters (with recirculation)[8]	2 in. cube media 18 gal/ft^2-day 5.88 gal/ft^3-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 Filters (with recirculation)[9]	24 in media depth Polystyrene sphere media 2.5 - 4.5 mm 6.6 gal/ft^2-day	NH3-N: Removal: 97.7% Influent: 92.5 mg/L Effluent: 2.1 mg/L

1 *Smith et al. 2008; Smith, 2008*

2 *Philip and Vasel 2006; Smith et al. 2008; Smith, 2008*

3 *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. Some data is single pass, some data is recirculating. Unclear from Literature Review which is which.*

4 *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; Osesek, 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*

5 *(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*

6 *NSF-International 2003e*

7 *(NSF-International 2006; Sherman 2006; Talbot, Pettigrew et al. 2006; Sherman 2007)137,180,181,196*

8 *(NSF-International 2005)136*

9 *E-Z Treat Company, 2009*

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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 Vasel 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 6.1gal/ft^2-day (Philip and Vasel 2006). In an eight month bench scale study, a clinoptilolite media biofilter treating septic tank effluent and operated at 2.8 gal/ft^2-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_5 surface loading rate of 0.0037 to 0.0041 lb/ft^2-day without surface ponding or observable material accumulations of the media surface, which contrasts to reported COD loadings of 0.0039 lb/ft^2-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^2-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.

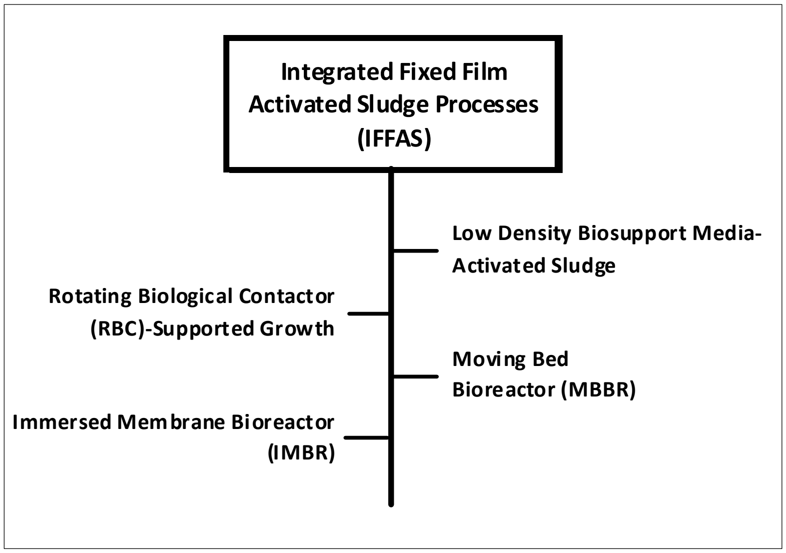

Figure 5-4: Common Integrated Fixed Film Activated Sludge (IFAS) Processes

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 nitrogen "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 re-cycled 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)[1]	1 liter bench column synthetic wastewater upflow single pass	Sulphur/oyster shell media (75/25% by volume) Sulphur: 4.7 mm	anoxic only NO3-N Removal: 80% Influent: 50 mg/L Effluent: 10 mg/L
Sulfur/oyster shell filter (bench scale)[2]	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^2-day	anoxic only NO3-N Removal: 99.9% Influent: 38.8 mg/L Effluent: 0.030 mg/L
Sulfur/oyster shell filter (bench scale)[3]	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^2-day	anoxic only NO3-N Removal: 99.9% Influent: 58.8 mg/L Effluent: 0.031 mg/L
Sulfur/oyster shell filter (bench scale)[4]	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^2-day	anoxic only NO3-N Removal: 89.9% Influent: 47.7 mg/L Effluent: 4.3 mg/L
Sulfur/limestone column[5]	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^2-day	anoxic only NO3-N Removal: >95% Influent: 60 mg/L Effluent: < 1 mg/L NO2-N Effluent: < 1 mg/L
Sulfur/oyster shell filter[6]	185 gal. column aerobic effluent upflow single pass 18 hr. HRT	Sulfur/oyster shell media (75/25% by volume) 47 gal/ft^2-day	anoxic only NO3-N Removal: 88% Influent: 20 mg/L Effluent: 2.4 mg/L
Sulfur/limestone column[7]	237 gal. column groundwater upflow single pass Residence time: 13 hr.	Sulfur/limestone media (67/33% by volume) 63 gal/ft^2-day Sulfur: 2.5 to 3.0 mm Limestone: 2.38 to 4.76 mm	anoxic only NO3-N Removal: 96% Influent: 64 mg/L Effluent: 2.4 mg/L NO2-N Effluent: 0.2 mg/L

Table 5.5
Summary of Saturated Anoxic Media Reactors

System Type	Description	Features	Treatment Performance
Nitrex™ [8]	aerobic effluent gravity flow upflow single pass	Nitrex wood-based media 24 to 30 inch media depth (est.) 4.6 gal/ft^2-day (est.)	aerobic+anoxic TN Removal: 79 to 96% Effluent: 3 to 18 mg/L NO3-N Effluent: 0.3 to 8 mg/L
Black& Gold™ [9]	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^2-day	aerobic+anoxic TN Removal: 98% Influent: 414 mg/L Effluent: 7.1 mg/L NH3-N Effluent: 4.4 mg/L NO3-N Effluent: 0.05 mg/L

[1] *Sengupta and Ergas 2006*
[2] *Smith et al. 2008, Smith, 2008*
[3] *Smith et al. 2008, Smith, 2008*
[4] *Smith et al. 2008 Smith, 2008*
[5] *(Moon, Shin et al. 2008)*
[6] *Brighton 2007*
[7] *Darbi, Viraraghavan et al. 2003a*
[8] *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*
[9] *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), saw-dust (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(e-caprolactone) (Horiba, Khan et al. 2005), and bacterial polyesters (Mergaert, Boley et al.

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 NO3-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^2-day; the effluent NO3-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^2-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 NO3-N; nitrate removal averaged 96 percent and average effluent NO3-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^2-day surface loading rate and removed greater than 95 percent of nitrate that was at 60 mg/L NO_x-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 NOx 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 NOx-N were 0.03, 0.031 and 4.3 mg/L. These denitrification filters operated at hydraulic loading rates of 4.9 gal/ft^2-day and at average NOx-N loadings of 0.003 to 0.005 lb/ft^2-day, which are similar to loading rates applied to acetic acid amended sand denitrification filters that achieved 94 to 99 percent NOx 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 NH_4^+ or 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 effective 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 Jenssen 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^2-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-

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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 under estimate 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 by Soil Infiltration Below OSTDS

Pretreatment	TN Removal
Traditional	0-35%[1]
Recirculating Sand Filter	71-97%[2]
Low Pressure Dosing Shallow	46%[3]
Low Pressure Dosing At-Grade	98%[4]
Mound	36[5]-86%[6]

1 *Ritter and Eastburn, 1988*
2 *Wert and Paeth, 1985*
3 *Brown and Thomas, 1978*
4 *Stewart and Reneau, 1988*
5 *Converse, et al., 1994*
6 *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

Drainage Class	**Description**	**Expected Impact on Heterotrophic Denitrification**
Excessively drained	Water is removed from the soil very rapidly. The soils are very porous. These soils tend to be droughty.	• Well aerated soil capable of achieving complete nitrification of applied TKN • Provides little organic carbon and will likely degrade any added organic matter within the aerobic zone • Short retention time
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.	• Well aerated soil capable of achieving complete nitrification of applied TKN • Provides little organic carbon and will likely degrade any added organic matter within the aerobic zone • Short retention time
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.	• Sufficiently aerated soil capable of achieving complete nitrification • May allow some organic matter to reach a saturated zone where it would be available for denitrification if a shallow water table is present
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.	• Sufficiently aerated soil capable of achieving complete nitrification • Denitrification would be enhanced with a fluctuating water table for a "two sludge" process or with slow drainage for a "single sludge" process

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

Drainage Class	Description	Expected Impact on Heterotrophic Denitrification
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.	• Ample organic matter for a carbon source and to create anoxic conditions in saturated zones for significant nitrogen reduction • Insufficiently aerated soil to nitrify TKN requiring nitrification of the wastewater prior to application to the soil
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.	• Ample organic matter for a carbon source and to create anoxic conditions in saturated zones for significant nitrogen reduction • Insufficiently aerated soil to nitrify TKN requiring nitrification of the wastewater prior to application to the soil
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.	• Ample organic matter for a carbon source and to create anoxic conditions in saturated zones for significant nitrogen reduction • Insufficiently aerated soil to nitrify TKN requiring nitrification of the wastewater prior to application to the soil

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 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, typical practice includes 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^2-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_5, NH^3-N, and NO^3-N. Median value reductions of 99 percent for fecal coliform, 99 percent for fecal strep, 86 percent for BOD_5, and 85 percent for NH^3-N + NO^3-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^2-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). Anommox 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 cellulous 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.

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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).

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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 and vegetative uptake / evapotranspiration. 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 if 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.

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 section (technologies with processes that are know 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
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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).