FLORIDA DEPARTMENT OF HEALTH

Florida Onsite Sewage Nitrogen Reduction Strategies Study

Literature Review of Onsite Nitrogen Reducing Technologies

DRAFT REPORT

May 2009

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Environmental Engineers & Scientists

In association with

COLORADO SCHOOL OF MINES 1874

AET Applied Environmental Technology

OTIS ENVIRONMENTAL CONSULTANTS, LLC

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Literature Review of Onsite Nitrogen Reducing Technologies

Prepared for:

Florida Department of Health
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Bureau of Onsite Sewage Programs
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Table of Contents

Section 1: Study Background 1-1

Section 2: Nitrogen in the Environment.. 2-1

Nitrogen Fixation 2-2
Nitrogen Uptake 2-3
Nitrogen Mineralization (Ammonification) 2-3
Nitrification 2-3
Denitrification 2-4

Section 3: Nitrogen in Wastewater.. 3-1

Section 4: Wastewater Nitrogen Reduction Technologies.. 4-1

Biological Nitrification / Denitrification Processes....................... 4-2

Mixed Biomass with Alternating Aerobic / Anoxic Environment 4-3
Mixed Biomass Recycling Systems............................ 4-4
Two-stage External Electron Donor Denitrification 4-5
Anaerobic Ammonium Oxidation................................ 4-5

Physical / Chemical Nitrogen Removal Processes 4-5
Source Separation.. 4-7
Natural Systems. 4-7
Passive Nitrogen Removal. 4-8

Section 5: Review of Onsite NItrogen Reduction Technologies 5-1

Source Separation.. 5-1

Urine Separation and Recovery 5-5

Urine Separation 5-5
Urine Treatment 5-7

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

Direct Nitrification 5-7
Precipitation 5-7

Greywater Collection and Reuse................................ 5-9

Greywater Storage 5-11
Greywater Treatment 5-11
Greywater Systems.................................... 5-14

Black Water Separation and Treatment 5-14

Primary Treatment (Septic Tank)... 5-15
Biological Nitrification / Denitrification Processes..................... 5-15

Mixed Biomass Nitrification / Denitrification............... 5-16

Suspended Growth (Activated Sludge) Reactors.. 5-16
Recirculating Media Filters 5-19
Integrated Fixed-Film Activated Sludge (IFAS) 5-30

Segregated Biomass (Two Stage) Denitrification 5-31

Anoxic Packed Bed Reactors...................... 5-31
Heterotrophic Denitrification 5-32
Autotrophic Denitrification 5-37

Physical / Chemical Nitrogen Reduction Processes 5-39
Natural Systems 5-39

Denitrification in Soil 5-39

Modifications to Conventional Onsite Treatment Systems 5-46

Drainfield Modifications.. 5-46

Section 6: Conclusions and Recommendations... 6-1

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

Appendix A: Glossary A-1

References: R-1

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

List of Tables

Table 1: Nitrogen Species Concentrations in Raw Wastewater and Septic Tank Effluent by Source (Lowe, Rothe et al., 2006) 3-2

Table 2: Daily Septic Tank Effluent Flows by Source in Gallons/Day (Lowe, Rothe et al., 2006) 3-2

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

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

Table 5: Biological Denitrification Processes and Typical Treatment Limits 5-17

Table 6: Summary of Media Filter Performance 5-23

Table 7: Factors Influencing Performance of Unsaturated Aerobic Filters 5-28

Table 8: Factors Influencing Performance of Saturated Anoxic Filters 5-33

Table 9: Summary of Saturated Anoxic Media Reactors 5-34

Table 10: Total Nitrogen Removals below Soil Infiltration Zones (Siegrist and Jenssen, 1989) 5-41

Table 11: Estimates of TN Removal Based on Soil Texture 5-42

Table 12: Total Nitrogen Removal Found in Various Studies of OWTS 5-42

Table 13: NRCS Drainage Classes and Descriptions 5-44

Table 14: Drainage Class and Expected Impacts on Denitrification 5-45

List of Figures

Page

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

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

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

Figure 4: Cumulative Frequency of Total Nitrogen Concentrations in Septic Tank Effluent (Lowe, Rothe et al., 2006) 3-1

Figure 5: Treatment Options for Reducing Nitrogen in Household Sewage 4-2

Figure 6: Onsite Treatment Technology Categories for Biological Nitrification / Dentrification Processes 4-3

Figure 7: Alternating Oxic / Anoxic Reactor Denitrification 4-5

Figure 8: Mixed Biomass Recycling Denitrification Process 4-6

Figure 9: External Electron Donor Denitrification Process 4-6

Figure 10: Onsite Treatment Technology Categories for Physical / Chemical Processes 4-7

Figure 11: Nitrogen Source Separation Categories 4-9

Figure 12: Categories of Natural Systems for Nitrogen Reduction 4-10

Figure 13: Passive Two-Stage Denitrification System 4-12

Figure 14: Domestic Wastestream Components 5-2

Figure 15: Two Swedish Urine Separating Toilets (EcoSan and Novaquatis) 5-6

Figure 16: Relative Complexities of Nitrification / Denitrification Unit Operations 5-18

Figure 17: Common Suspended Growth Treatment Processes (USEPA, 1993).........5-18

Figure 18: Common Integrated Fixed Film Activated Sludge (IFFAS) Processes 5-30

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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 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% 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 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 1).

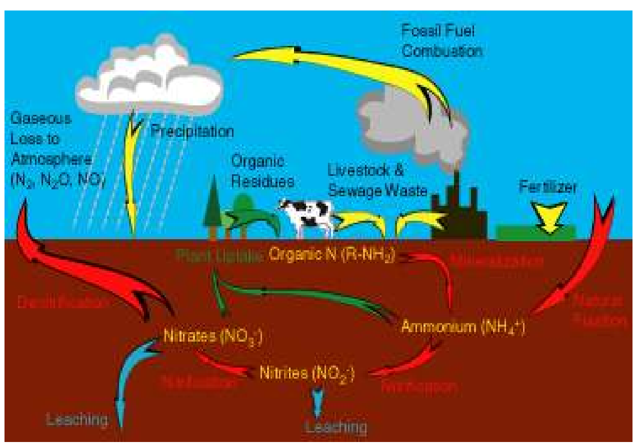

Figure 1: The Nitrogen Cycle (Harrison, 2003)

(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). Microorganisms, particularly bacteria, play major roles in all of the principal nitrogen transformations. As microbial mediated processes, the rates of these nitrogen transformations are affected by environmental factors that influence microbial activity, such as temperature, moisture, and resource availability.

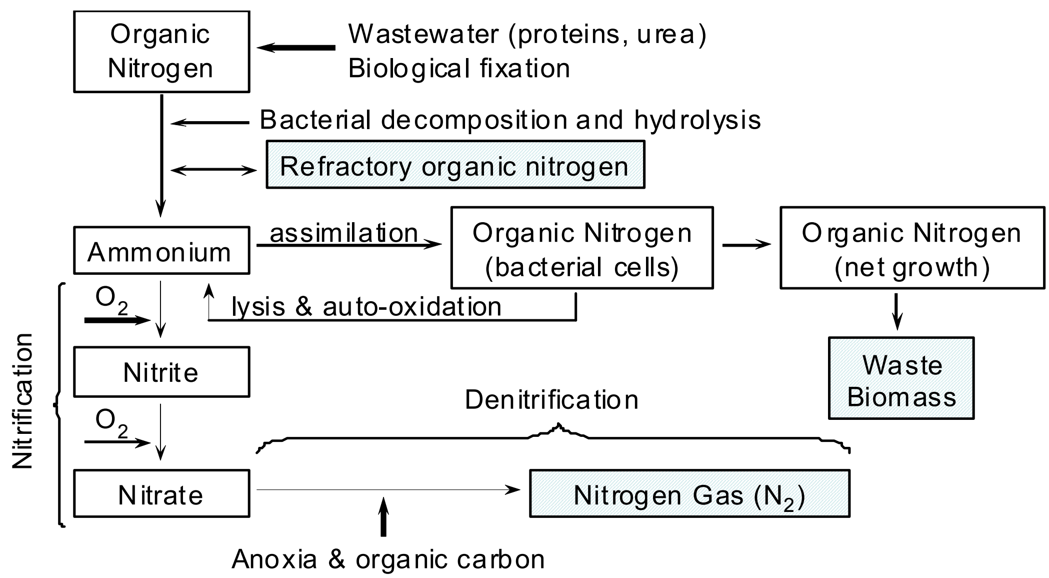

Figure 2: Nitrogen Transformation in Biological Processes
(Eckenfelder and Argaman, 1991)

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 form 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 that 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 3).

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 organisms for incorporation into organic biomass or conversion to nitrate.

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.

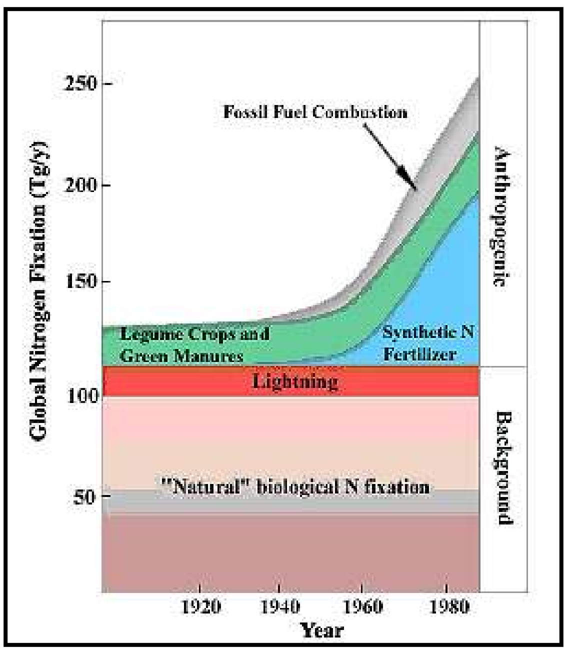

Figure 3: Recent Increases in Anthropogenic N Fixation in Relation to "Natural" N Fixation (Harrison, 2003)

NITRIFICATION

Nitrification is biological process that converts ammonium into nitrate. This process is used by chemoautotrophic bacteria that use the energy released by the conversion 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 low-

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ered to a point where the nitrifying bacteria can no longer thrive. Therefore, sufficient alkalinity is needed to buffer the pH so that the acidic conditions do not 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 and in soils, is not adsorbed but travels with the soil water until captured or taken up by plant roots.

DENITRIFICATION

Denitrification also is a biological process used by facultative heterotrophic bacteria to obtain their energy for growth. It is the only nitrogen transformation that removes nitrogen from ecosystems. Under anoxic (no free oxygen) conditions, heterotrophs, which use organic carbon for energy use the oxygen from the nitrate molecule and resulting breakdown compounds as an electron accepter in the degradation of the organic carbon. This process will ultimately break nitrate down to nitrogen gas following the sequence $NO_3^- \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2$. For this to occur, organic carbon and an anoxic environment is necessary. If the process is interrupted before the sequence is complete, nitric oxide (NO) and nitrous oxide (N_2O) can be released, which contributes to smog or is a greenhouse gas respectively. However, once converted to N_2, the nitrogen is not likely to be reconverted a biologically available form except through nitrogen fixation.

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. Each of us discharges approximately 11.2 gms of nitrogen into wastewater each day (EPA, 2002). Nearly 80% of this is discharged as toilet wastes (Lowe, Rothe et al., 2006; U.S. EPA, 2002). Another 15% 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 4 and Table 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 decreases but the nitrogen loading does not. This results in higher TN concentrations in homes with more residents. 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.

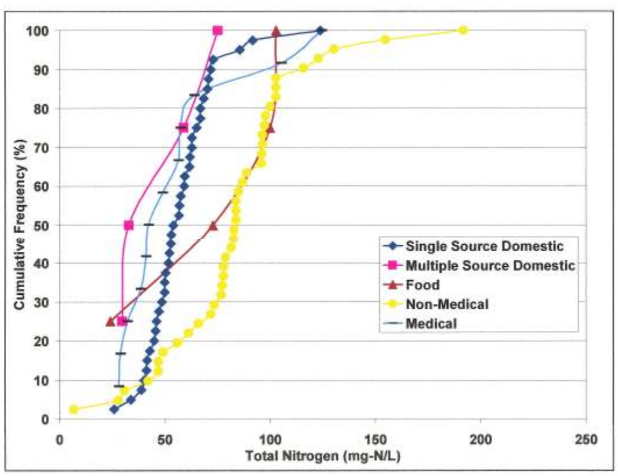

Figure 4: Cumulative Frequency of Total Nitrogen Concentrations in Septic Tank Effluent (Lowe, Rothe et al., 2006)

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Table 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	-	-	-	-	-	-	-	-	-	-

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

(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 concentrations of 59 to 42 mg-N/L respectively. In commercial establishments, the daily wastewater flow will vary by use (Table 2).

Table 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.

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 categories; source separation, physical/chemical processes, biological nitrification/denitrification, and natural systems (Figure 5). 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 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. These technologies are briefly described here.

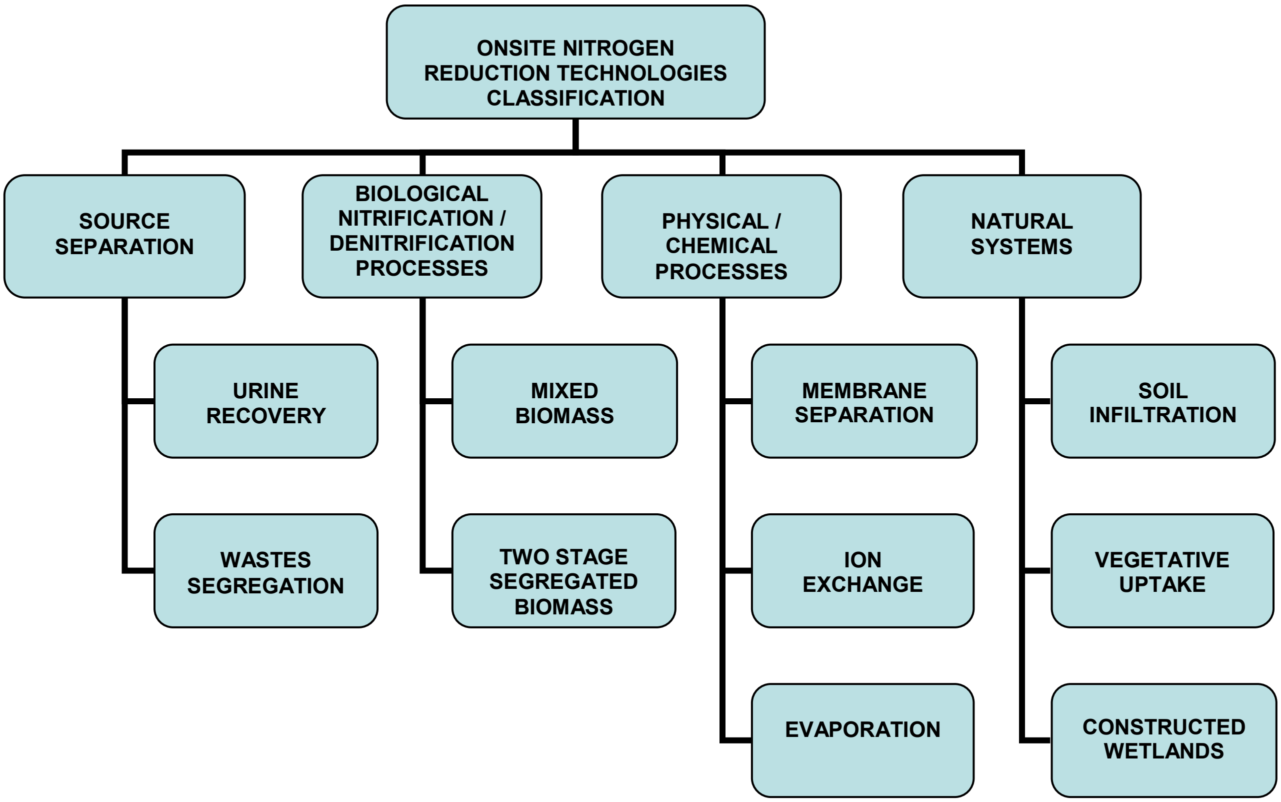

Figure 5: Treatment Options for Reducing Nitrogen in Household Sewage

BIOLOGICAL NITRIFICATION / DENITRIFICATION PROCESSES

There are many different nitrification/denitrification technologies available. Figure 6 lists commonly used groups of systems for each of the biological nitrification/denitrification processes described here.

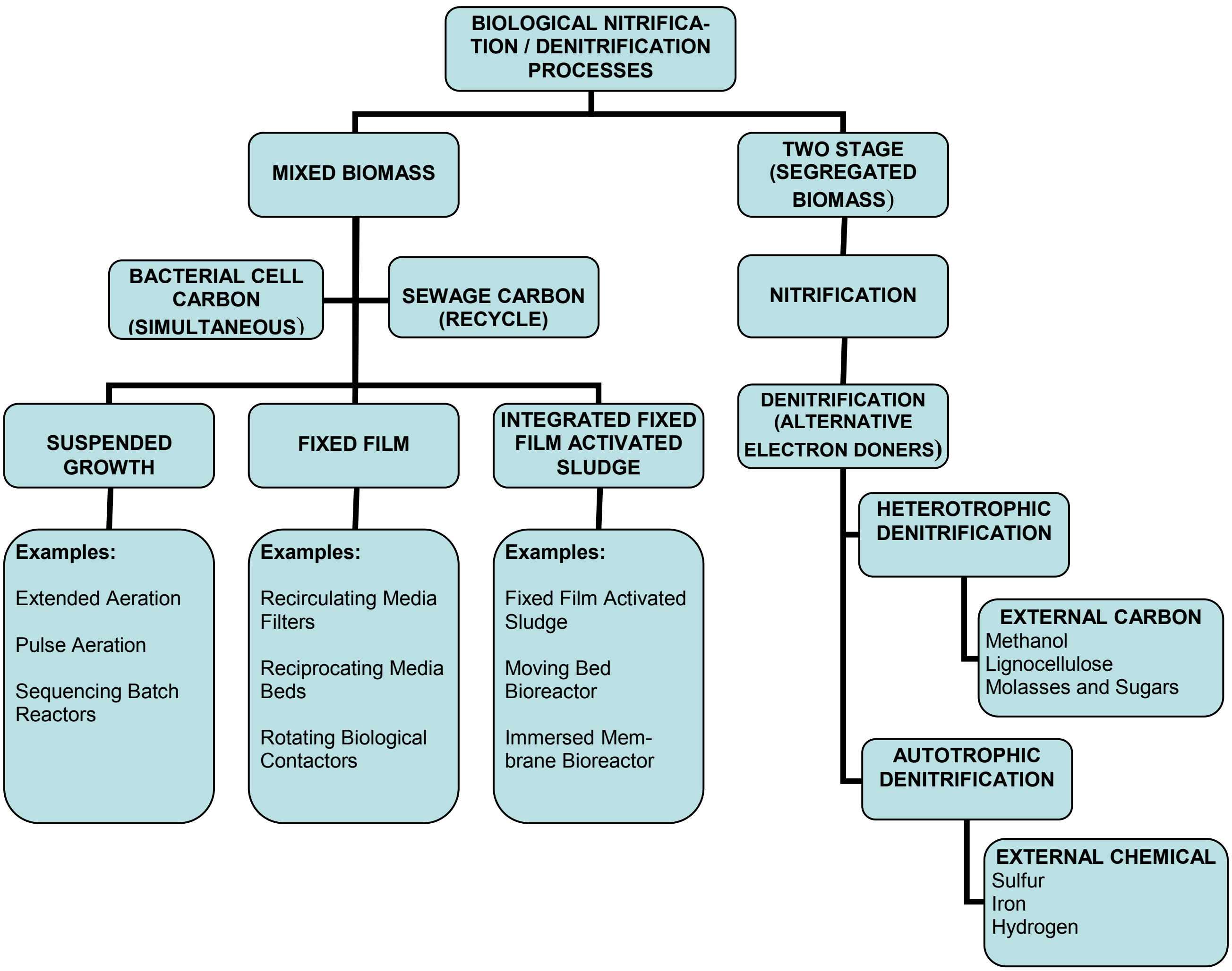

Figure 6: Onsite Treatment Technology Categories for Biological Nitrification/Denitrification Processes

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 to achieve biological nitrification and denitrification but they all generally fit into three process types: 1) mixed biomass with alternating oxic/anoxic environments (called simultaneous denitrification), 2) mixed biomass with recycle back to the treatment headworks, and 3) two-stage (separated biomass) using external electron donors (Figures 7-9).

"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 oxic and anoxic environments, the process is heavily dependent on microbial cell carbon for the electron donor during nitrification. A single stage process with recycle relies heavily on the organic carbon from the fresh incoming wastewater as the electron donor for denitrification. In a two stage process, external electron donors are necessary because the organic carbon is removed during nitrification in the first stage.

Reactor pH has a significant affect on nitrification. 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.

Mixed Biomass with Alternating Aerobic/Anoxic Environments

This nitrification /denitrification process combines the aerobic and anoxic reactors of the mixed biomass recycling system into one reactor (Figure 7). Periods of aeration when cBOD oxidation and nitrification occur alternate with periods of no aeration during which the active biomass is allowed to deplete the oxygen to create anoxic conditions for denitrification. The treatment performance is similar to the mixed biomass recycling process.

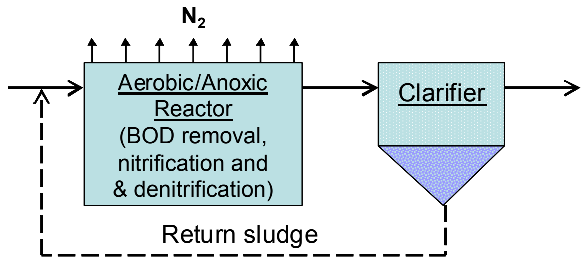

Figure 7: Alternating Oxic / Anoxic Reactor Denitrification

Mixed Biomass Recycling Systems

In 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 8). From this reactor, the wastewater flows to the aerobic reactor where the ammonium and organic nitrogen are nitrified. As the nitrified effluent exits the aerobic reactor, it is split with a small fraction directed to the final discharge while the majority is directed back to the anoxic tank where the nitrate can be reduced to nitrogen gas using the incoming wastewater cBOD as the electron donor. Also, the alkalinity consumed by nitrification is recovered during denitrification thereby reducing the alkalinity requirements. However, total nitrogen removal cannot be achieved with this process because “new” nitrogen is continuously introduced into the flow from fresh raw influent of which a portion is not recycled but discharged from the system. The amount of nitrate that can be removed through the system ranges from approximately 60% to 85% with ratios of recycled flow to forward flow of 5:1 and 2:1 respectively (USEPA 1993).

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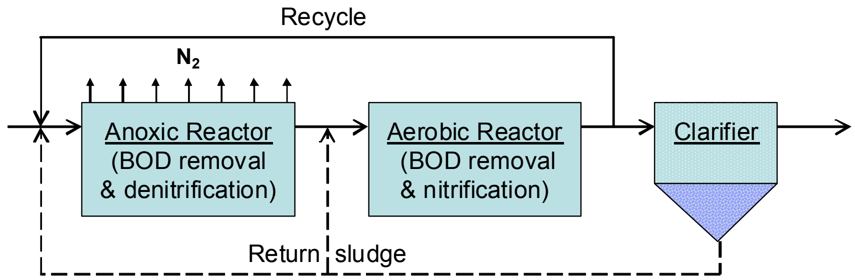

Figure 8: Mixed Biomass Recycling Denitrification Process

Two-stage External Electron Donor Denitrification

The two-stage process cultivates two separate bacteria populations; one for nitrification and the other for denitrification (Figure 9). This configuration allows nearly complete nitrogen removal because nitrate cannot by-pass denitrification as it can in the mixed biomass options. However, during the nitrification step nearly all the organic carbon in the raw wastewater is oxidized. As a result, is not available as an electron donor in denitrification thereby requiring a donor from an external source to be added directly into the denitrification reactor. A number of organic carbon sources have been used successfully. For larger treatment systems, liquid sources are typically used. The more popular are methanol, ethanol, and acetate. For smaller systems where less operation attention is desired, solid reactive media are used such as cellulous and elemental sulfur.

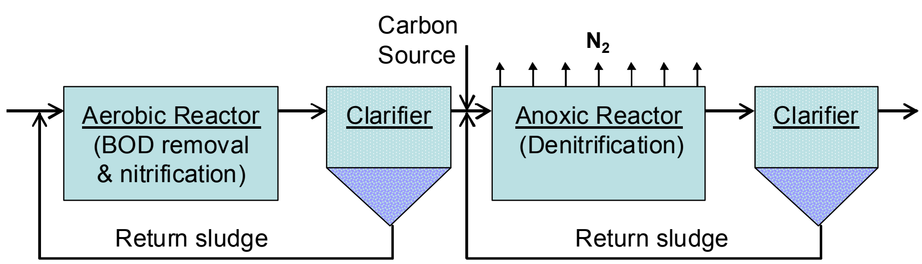

Figure 9: External Electron Donor Denitrification Process

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.

PHYSICAL / CHEMICAL NITROGEN REMOVAL PROCESSES

Physical/chemical (P/C) processes use non-biochemical approaches to wastewater nitrogen reduction. A fundamental difference from biological denitrification is that while biological denitrification converts the biodegradable organic nitrogen to ammonium, P/C processes do not, 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).

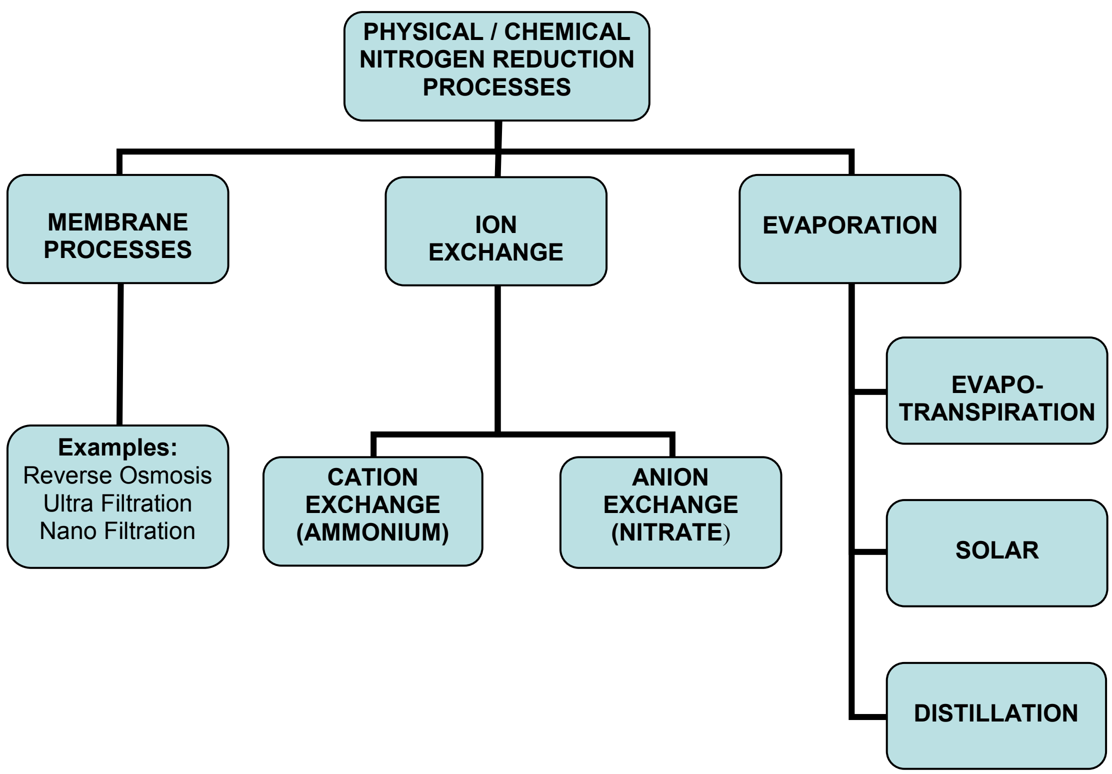

Figure 10: Onsite Treatment Technology Categories for Physical/Chemical Processes

P/C process options that might be appropriate for onsite sewage treatment are shown in Figure 10.

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. Of these, source separation is an option gaining more attention with the availability of urine separating toilets. Membrane separation requires substantial and costly pretreatment and therefore is most commonly used for drinking water treatment at the household level. Ion exchange also requires pre-treatment and commercial regeneration of the exchange resins. Evapotranspiration can be effective in warm climates with year round growing seasons, but require periodic removal and appropriate disposal of the evaporates. Distillation is an emerging option for households but it is early in its development.

SOURCE SEPARATION

The source of the majority of nitrogen in household wastewater is the toilet, which accounts for 70-80 percent of the total daily discharge of nitrogen (Wisconsin, 1978; U.S. EPA, 2002; Lowe, Rothe et al., 2006). Nitrogen from food wastes that are discharged through the kitchen sink or dishwasher accounts for an additional 15 percent. These sources can be segregated from the total household waste flows for separate treatment and handling. For common separation options, see Figure 11.

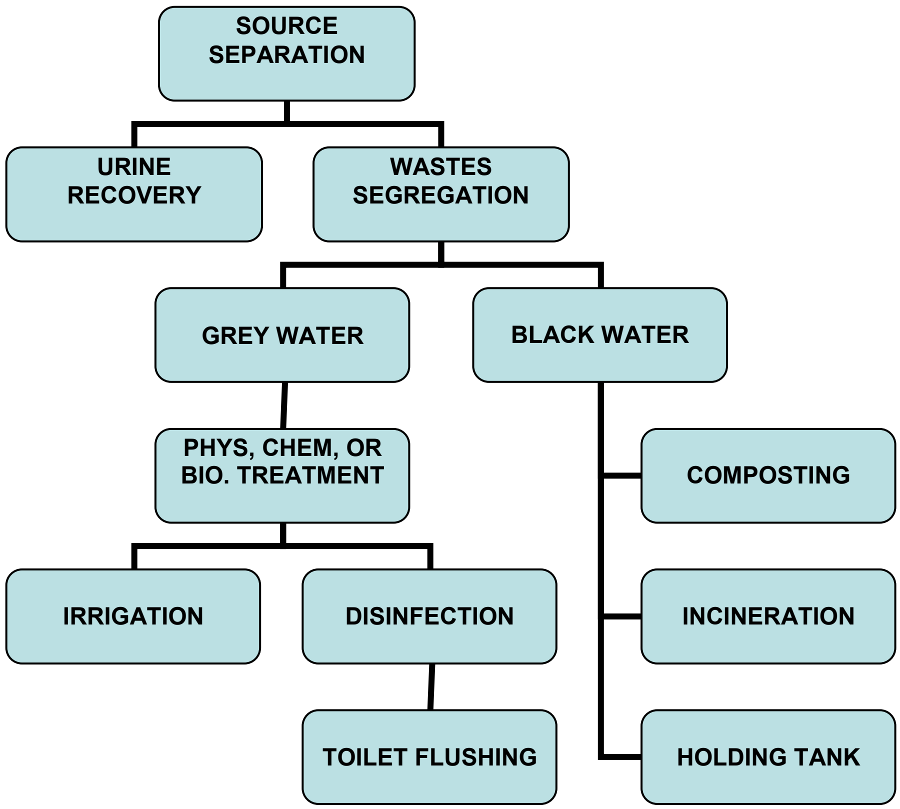

Figure 11: Nitrogen Source Separation Categories

NATURAL SYSTEMS

Natural systems are included as a separate classification because they are capable of significant nitrogen reduction. They utilize a combination of physical, chemical and biological processes that occur naturally in the environment. Natural biological processes can mimic both single and two-stage processes depending on the soil conditions (Briggs, Roeder et al., 2007; Otis, 2007). Categories of technologies that are practical for onsite sewage treatment are presented in Figure 12.

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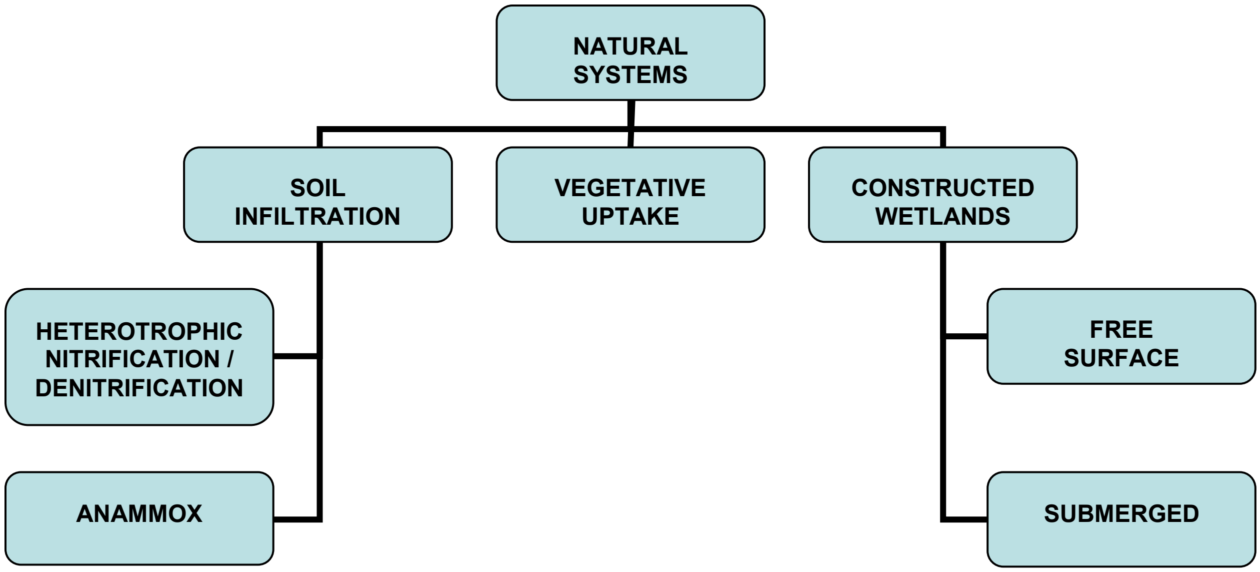

Figure 12: Categories of Natural Systems for Nitrogen Reduction

PASSIVE NITROGEN REMOVAL

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

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

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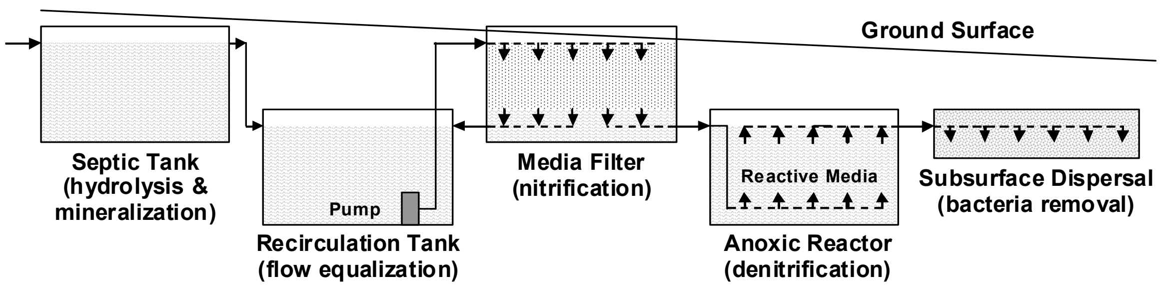
uses a reactive media to assist in nitrogen removal." Reactive media is defined as media that reacts with wastewater to reduce nitrogen concentrations.

This definition is restrictive in that it precludes most nitrogen reduction options primarily because of the requirement for reactive media. Only biological two-stage systems would qualify as passive under this definition (Figure 8). 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 eliminating it as a passive system. In any event, the added cost of the pretreatment would likely make ion exchange impractical for household applications. Most mixed biomass systems would be "passive" except for the requirement for reactive media, but these systems have less ability to meet very low total nitrogen concentrations. Where the total nitrogen requirements are above 10 mg N/L, these systems would be acceptable options. Mixed biomass systems also have the advantage that they recycle the alkalinity, which may be important in areas with low alkalinity in drinking water. The FDOH definition of "passive" is followed in describing and comparing the different nitrogen reduction processes and technologies in this review.

A two-stage denitrification system for household use that meets the FDOH "passive" definition probably would consist of a septic tank, recirculating media filter, anoxic denitrification reactor followed by soil infiltration as shown in Figure 13. In the <u>septic tank</u>, proteins are hydrolyzed releasing the organic nitrogen, which is oxidized to ammonium. Any nitrate or nitrite present in the influent is denitrified. The <u>media filter</u> is an unsaturated aerobic media, which removes most of the BOD, nitrifies the ammonium and removes up to 50% of the total nitrogen. Where low total nitrogen concentrations are necessary the filtrate must be returned to the <u>recirculation tank</u> to be recycled onto the media filter since nitrification may not be complete after a single pass through the filter. This requires a <u>pump</u> and a passive filtrate flow splitter that can divert the flow for recycling or discharge to the next treatment stage. The advantage of using the pump here is three fold. First, it can dose the media filter based on time (rather than demand) and under pressure, which achieves uniform distribution over the filter surface both spatially and temporally significantly enhancing treatment performance. Second, it provides <u>flow control</u> (equalization) through the remainder of the system, which also enhances system performance. Third, it can be used to raise the <u>hydraulic grade line</u> though the remainder of the system so that flow through the system occurs by gravity, which eliminates the need for additional pumps. The nitrified filtrate flows to the anoxic reactor, which is filled with saturated <u>reactive media</u> that provides the electron donors for denitrification to occur. After this reactor, the treated wastewater is discharged for <u>subsurface dispersal</u>

where bacteria in the water are removed by processes in the soil as the water percolates to the groundwater.

Availability of alkalinity is an important consideration in any nitrification/denitrification treatment process. It is an important buffering agent that is necessary to maintain pH concentrations in an acceptable range for nitrifying organisms to thrive. During nitrification, hydrogen ions are created and if not controlled by a buffering agent, will increase the acidity of the water to the point that nitrification ceases. Nitrification consumes approximately 7.14 grams of alkalinity as $CaCO_3$ 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 13: 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% 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 requiring more frequent replacement and the effluent quality will be reduced by excess dissolution product. If the rate of dissolution is too slow, denitrification will decline. 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 to allow 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 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).

FLORIDA DEPARTMENT OF HEALTH

Section 5.0
REVIEW OF ONSITE NITROGEN REDUCTION TECHNOLOGIES

The following is a review of what are considered technically and economically feasible nitrogen reduction technologies suitable for household and small commercial onsite sewage treatment and disposal systems. A nitrification/denitrification treatment system consists of a series of unit operations and processes that maybe packaged together in a single or several treatment system components or technologies. In this review, the technologies are presented in an order that they would appear in an onsite wastewater treatment train designed for denitrification.

SOURCE SEPARATION

Onsite domestic sewage treatment traditionally 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. For some time, wastewater segregation to isolate gray water for reuse has been practiced predominately in water short areas. More recently, recovery of urine for its nutrient content using urine separating toilets is gaining attention as a sustainable solution to the 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 at least 50%

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Domestic sewage can be subdivided into four separate wastestreams based on options for segregation that are likely to provide most appropriate treatment and reuse combinations. The four domestic wastestreams are illustrated in Figure 14. The quantity and

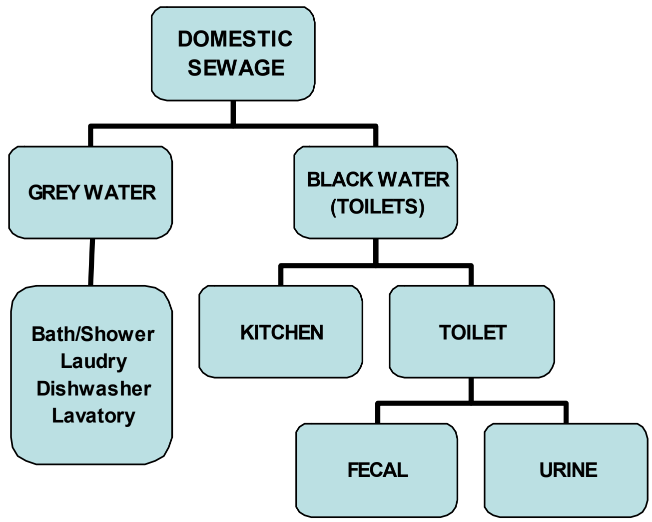

Fig. 14: Domestic Wastestream Components

constituent mass of these wastestreams were summarized from published data for typical U.S. households (Mayer, DeOreo et al. 1999; USEPA 2002; Tchobanoglous, Burton et al. 2003). Table 3 shows four waste source groupings based on quality characteristics of the wastestreams representing typical U.S. conditions, feasibility of waste source segregation or separation, and nitrogen reduction options (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). For example, Source A is referred to as greywater and has different levels of contaminants and would require less treatment for reuse than toilet wastestreams (i.e. source C + D).

Table 3:
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 small volumes and high nitrogen concentrations. Separation of wastestream components with relatively low pollutant concentrations enables their onsite reuse with limited treatment, which reduces the mass and volume of the remaining, more concentrated wastestreams that require smaller sized treatment units. Thus, waste segregation can reduce nitrogen loading to the environment through recovery and beneficial use of nutrients in the wastestreams and by decreased nitrogen loadings to onsite soil treatment and dispersal units.

Components of domestic wastestreams are shown in Table 4 for a typical 4 person household in the U.S. based on the Table 3 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 4 designations can be applied to analysis and selection of nitrogen reduction technologies that are advantageous for different source separation options.

Table 4:
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
Blackwater	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
Blackwater 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, separation of the domestic wastestream is into greywater (A) and black water (B+C+D) (Table 4). Here, the kitchen wastestream is not included in the greywater designation because of its associated with production and consumption of food and the BOD, TSS and pathogens that may be found in kitchen waste. Greywater comprises over half of the water volume while contributing relatively small fractions of total pollutant mass. With lower constituent concentrations, greywater requires less intensive treatment than black water to meet a given level of water quality. Greywater may be rendered suitable for onsite reuse (irrigation or indoor toilet flushing) with relatively simple aerobic biological treatment.

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

Black water (B+C+D) contains a majority of the constituent mass but less than half of the volume of the whole domestic wastestream (A+B+C+D), resulting in higher constituent concentrations (Table 4). Treatment of black water would require generally similar treatment as combined domestic wastestreams, although the necessary capacity of treatment processes 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 4). The treatment plant required for removal of BOD and TSS would not be greatly affected, but nitrogen reduction plant requirement 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% 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 4). Beneficial use of urine could also provide a future funding mechanism for onsite treatment infrastructure.

Greywater separation removes over half of the water volume and a small fraction of nitrogen. Segregated greywater is less polluted, which reduces its treatment requirements and provides options for reuse. After greywater separation, the remaining sanitation water stream has a much reduced daily volume and a higher nitrogen concentration.

Urine Separation and Recovery

<u>Urine Separation</u>

Urine separation systems include urine separating toilets and waterless urinals (Table 3). Urine separation technologies include toilets with separate collection bowls (Figure 15) and effluent lines for urine and feces, and waterfree urinals with a single effluent lines. 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 primary sewer line or into a composting bin. Urine separation systems have lower energy requirements than alternative systems (Tidåker, Sjöberg et al. 2007).

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, 476 liters of urine were collected per person with a coefficient of variation of 11%. When combined with feces collection, 60% of the nitrogen was

Figure 15: Two Swedish Urine Separating Toilets (EcoSan and Novaquatis)

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 138 ml/flush in households and 225 ml/use with waterfree urinals. Mean urine collection rates in households were 6.37 l/day on weekdays and 9.22 l/day on weekends. Urine recovery in households was maximally 70 to 75% 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 a pipeline (Justyna Czemiel Berndtsson 2006). Offsite transport can be followed by use of urine as a liquid fertilizer or collection and treatment 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).

Urine Treatment

A number or 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).

Direct Nitrification

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

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% recovery efficiency for ammonia nitrogen and 85 to 98% 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% 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 reserve 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% 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% 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.

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.

Greywater Collection and Reuse

Domestic sewage can be subdivided into greywater and black water. Greywater includes waste streams from lavatories, laundry and bathing. Black water includes waste streams from toilets, kitchen sinks, dishwashers, and garbage grinders. Greywater contains over half of the domestic effluent volume while black water contains the majority of pollutant mass (Tables 3 & 4). Generally, , greywater contains far less nitrogen, fewer pathogens, and biodegrades more rapidly than black water. The 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. Separate collection of effluent from all kitchen and toilet sources is typical. California guidance on a standard greywater irrigation system design includes a surge tank, filter, pump, and irrigation system (CSWRCB 1995).

A universally accepted definition of greywater does not exist. Excluding kitchen waste from greywater is consistent with Florida requirements. 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 the greywater stream should be determined.

Modeling predicted that a 40% 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 reduced potable water use by 29 to 35% 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 and 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% 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.

Greywater Storage

Storage of greywater is an important element of all greywater recycling systems. Greywater quality was found to be affected by four major processes during 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 improved suspended solids, but dissolved oxygen depletion after 48 hrs could result in odor problems. These results suggest that practical greywater systems could benefit from low intensity aerobic treatment, such as mild or intermittent aeration. This would serve to oxidize BOD in the influent greywater, and oxidize organics and odors that are released from underlying settled organic matter.

Since greywater contains only a small portion of the nitrogen in household sanitation water, the total impact of greywater separation on nitrogen reduction is limited and could reduce the amount of organic carbon needed for electron donors during denitrification of the black water. When used for irrigation, plant and soil processes can eliminate a part of the greywater nitrogen that is applied. Indoor greywater use for toilet flushing would transfer greywater nitrogen back to the black water stream. Use of greywater for any process that cycles back to a greywater stream, such as first-cycle clothes washing, could lead to an internal loop of nitrate buildup. Although greywater can be treated to any desired level of quality, the use of multiple process greywater treatment trains may be more appropriate for multiple family residences than individual homes. Separating and treating greywater with active treatment processes would result in duplication of effort if black water had to be treated with another treatment system. From a nitrogen reduction perspective, the most likely beneficial use of greywater may be a rapid application to soil or plant systems.

Greywater Treatment

Greywater treatment has been examined with a variety of treatment technologies applied in many different schemes for overall water recycling. Some of these are listed here:

- A bathroom greywater stream had flowrates of zero to 34 l/min and COD concentrations that ranged from 26 to 650 mg/L; a treatment process was reported that consisted of a primary settling tank, RBC, secondary settling tank, sand filter, and UV disinfection (Eriksson, Andersen et al. 2008).

- Treatment was examined for reuse of low strength greywater (public shower effluent) using direct filtration by ultrafiltration membranes (30, 200, and 400 kDa

MWCO) and nanofiltration (200 Da MWCO). Mean influent COD was 170 mg/L, and COD reduction was 45 to 70% by UF and over 90% by NF (Ramona, Green et al. 2004)

- Treatment of greywater was one component of ecological sanitation (EcoSan) concepts, where most of the COD and nitrogen removal is by microbial activity in soil filters and phosphorus retention is by sorption to soil particles (Benetto, Nguyen et al. 2009).

- Greywater from lavatories and showers was treated with hypochlorite and sand filtration, mixed with membrane reject and used for toilet flushing in a tourist hotel. The greywater was found to be safe and socially acceptable. Nitrogen reduction was not reported (Gual, Moià et al. 2008).

- A greywater treatment system was examined that employed a series of shallow ponds alternating with riparian zones. Nitrogen reduction efficiency was over 99% but the system required over 400 ft^2 per person (Günther 2000).

- A laboratory scale microfiltration/oxidation process was evaluated for greywater treatment to provide reuse for fire fighting, water for plants, water for toilets and car washing. Greywater was defined as all domestic sewage excluding toilet waste. Removals of color, COD, suspended solids, *E. coli*, *total coliform*, *Salmonella* and *Staphylococcus* were 99 to 100% (Kim, Song et al. 2009).

- An untreated laundry wastestream was used to irrigate soils in residential garden beds, resulting in substantial retention of salts in the sand/silt/clay Ferrosol soil and a decrease in hydraulic conductivity that was attributed to sodium salts (Misra and Sivongxay 2009). The authors caution that the widespread and continued use of untreated laundry greywater for irrigation could have undesirable effects on soil quality and suggests that with greywater treatment or reduction of the sodium content of laundry detergents should be considered.

- A system was employed to treat greywater from sink, bathtub/shower, and clothes for toilet flushing for 70 persons, employing sedimentation, RBC biotreatment, and UV disinfection. The system operated effectively for over 10 years with no public health risk or aesthetic issues (Nolde 1999). The author noted that most problems with greywater reuse were with simple systems for single family homes that employed mainly aeration and required high maintenance.

- Chemical treatment of greywater with coagulation and magnetic ion exchange resin was examined (Pidou, Avery et al. 2008). The authors concluded that these processes alone would be unsuitable for treatment of a greywater with a medium to high organic content but could have applications in specific circumstances where greywater was low in organic matter.

- Addition of nitrogen and phosphorus to greywater was found to stimulate oxygen uptake rate and COD removal, suggesting that the greywater was nutrient limited (Jefferson, Burgess et al. 2001). The source of greywater was baths, showers and hand basins.

- The advantages of natural zeolites for treatment of greywater were reviewed particularly focusing on their ion adsorption capability and microorganism retention (Widiastuti, Wu et al. 2008).

- Three greywater treatment systems were evaluated for indicator bacteria reductions (total coliforms, *Escherichia coli*, Enterococci, Clostridia, and heterotrophs over a 2 year period. The treatment systems were a constructed wetland, membrane bioreactor (MBR), and a membrane chemical reactor (MCR). The MBR provided the highest level of treatment, and the aerobic unsaturated wetland provided the best wetland technology for pathogen removal (Winward, Avery et al. 2008a).

- An upflow anaerobic sludge blanket (UASB) reactor treating greywater removed 85% COD and 15 to 21% of the total nitrogen. Greywater appeared to have included kitchen waste and had a TKN of 27 mg/L (Elmitwalli and Otterpohl 2007).

- Treatment of greywater by a rotating biological contactor, sand filtration and disinfection. The effluent was examined and found suitable for toilet flushing (Friedler, Kovalio et al. 2005).

- The efficacy of chlorine disinfection of greywater as measured by total coliform inactivation was most closely related to particle size. Particle associated coliforms (PACs) comprised 91% of the total (Winward, Avery et al. 2008b). Disinfection efficiency decreased with increasing particle size.

- Bisphenol A retention was 30 to 45% in short term laboratory experiments with submerged direct ultrafiltration of a synthetic greywater solution (Schäfer, Nghiem et al. 2006).

Greywater Systems

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. 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 operations and maintenance. One source recommends mulch type planting beds (Oasis Designs; http://oasisdesign.net/greywater)

The preferred practice for separate disposal of residential grey water 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 (page 13, pages 11-14 *Builder's Grey Water Guide (book)).* 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 be most effective.

In Australia, greywater collection systems are required to use disinfection (UV or chlorine) if greywater is held for longer than 24 hrs. Some systems are equipped filtration. A delineation can be drawn between systems intended only for outdoor irrigation versus indoor applications.

Black Water Separation and Treatment

Different techniques were examined for separation of fecal material from flush water. Included were filtration and the Aquatron system (which uses surface tension, gravitation and a whirlpool effect) produced a solids stream that contained 70 to 80% 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 rejections were 74 to 99% while ammonia rejections were 21 to 94%. Onsite anaerobic treatment of black water (Luostarinen and Rintala 2005) is similar to treatment of whole domestic sewage, albeit with higher constituent concentrations. Traditional onsite primary treatment systems (i.e. septic tanks) perform passive anaerobic digestion with little collection of methane gas. Primary effluent (i.e. septic tank effluent) from black water would be expected to have higher nitrogen levels than primary effluent from whole domestic sewage treatment, and possibly higher biochemical oxygen demand. The options for nitrogen reduction from primary effluent treating black water are similar to those for whole effluent. Three combinations of biological treatment and membrane filtration were compared for separate black water treatment: a UASB followed by membrane filtration, anaerobic MBR, and aerobic MBR (van Voorthuizen, Zwijnenburg et al. 2008). All three systems exhibited high nutrient conservation and effluent with low TSS and high soluble COD in the effluent.

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 carbon is degradation via hydrolysis, acidogenesis, acetogenesis and methanogenesis. During hydrolysis, the protein molecules are broken apart to release the organic carbon, much of which is converted to ammonium through acidogenesis. Nitrate in the influent is quickly denitrified by the heterotrophic denitrifiers. Consequently, the form of nitrogen in domestic septic tank effluent is approximately 70% ammonium and 30% organic nitrogen (University of Wisconsin 1978; Lowe, Rothe et al. 2006). Nitrate is typically negligible. About 15% 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% 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.

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 OSDTS where small and intermittent sewage discharges into the treatment system can easily result in extended aeration periods during low or no flow periods with the result that the organic carbon is oxidized before the denitrification step. Consequently, OSTDS that use mixed biomass processes are less likely to achieve low total nitrogen effluent concentrations, particularly those processes that rely on microbial cell carbon as the electron donor in denitrification. Table 5 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 16).

Mixed Biomass Nitrification / Denitrification

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). Figure 17 provides a listing of the types of suspended growth processes that are commonly used in municipal operations.

Many manufacturers offer suspended growth treatment units for onsite use. Most were developed to provide additional treatment after septic tanks to remove BOD_5 to reduce clogging

Table 5:
Biological Denitrification Processes and Typical Treatment Limits

Process	Simultaneous (Mixed Biomass)	Recycle Mixed Biomass	External Donor Two Stage
Electron Donor	Organic carbon from bacterial cells	Organic carbon from influent wastewater	Cellulose, Sulfur, Iron, Other
Typical Removal	40 - 60%	60 - 80%	70 – 96%
Technologies	• Recirculating media filters w/o recycle to septic tank[1] • Reciprocating media beds[2] • Extended aeration[3] • Pulse aeration[4] • Sequencing batch reactors[5] • Membrane bioreactor[6]	• Recirculating media filters with recycle to septic tank[7] • Extended aeration with recycle • Moving bed bioreactor	• Heterotrophic suspended growth • Heterotrophic packed bed reactive media8 • Autotrophic packed bed reactive media

[1] *(USEPA 2002)*

[2] *(Behrends, Houke et al. 2007)*

[3] *(Leverenz, Tchobanoglous et al. 2002; USEPA 2002)*

[4] *(CSWRCB 2002)*

[5] *(AyresAssociates 1998)*

[6] *(Abegglen, Ospelt et al. 2008; Sarioglu, Insel et al. 2009)*

[7] *(Ronayne, Paeth et al. 1982; Gold, Lamb et al. 1992; Piluk and Peters 1994; Roy and Dube 1994; CRWQCB 1997; AyresAssociates 1998; Loudon, Bounds et al. 2005)*

[8] *(Rich 2007; Heufelder, Rask et al. 2008)*

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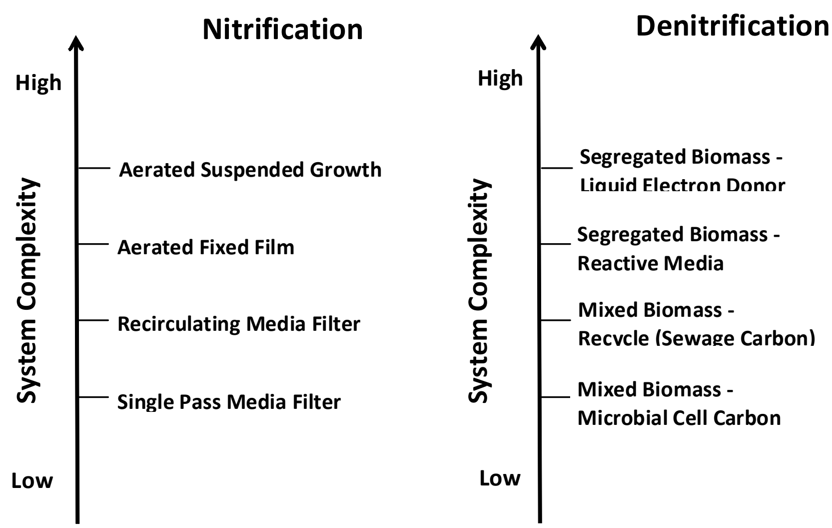

Figure 16: Relative Complexities of Nitrification / Denitrification Unit Operations

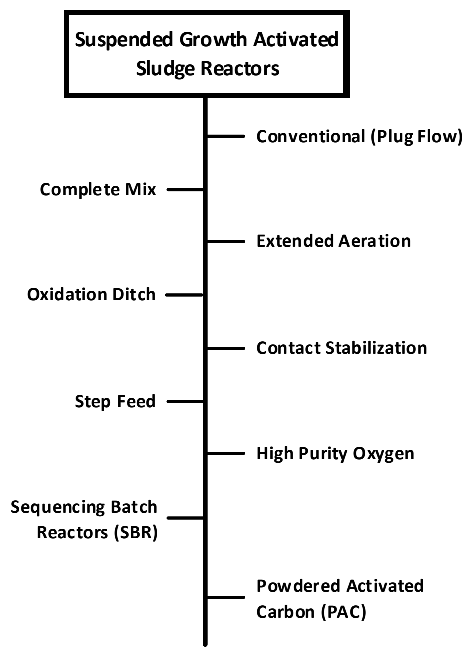

Figure 17: Common Suspended Growth Treatment Processes (USEPA 1993)

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of the infiltrative surface in the drainfield. Most of the manufactured units use the extended aeration process because of its simplicity and less sludge production. Extended aeration is similar to conventional activated sludge and complete mix processes except the hydraulic residence times are one to more than two days as compared to less than 10 hours for the conventional and complete mix systems. The extended reaction times are used to maximize endogenous respiration, which reduces the amount of sludge accumulation.

More recently sequencing batch reactors (SBR) have been manufactured for onsite use, which are more complex in operation but can be easily automated. This process uses two or more reactor tanks in which aeration, sedimentation and decanting occur in each reactor. This allows the treatment to occur in batches. A decanted reactor (active biomass is retained in the reactor after decanting) is filled after which 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 organic carbon during nitrification (AyresAssociates 1998; Habermeyer and Sánchez 2005).

Recirculating Media Filters

Media filters are unsaturated, aerobic fixed film bioreactors, which accept settled raw wastewater or septic tank effluent for treatment. They consist of a lined excavation or container filled with a bed of porous media that is placed over an underdrain system surrounded by coarse rock. 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 the most common materials, but peat, textile and open cell phone 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 for coconut husks). Most filters using media other than sand or gravel are proprietary systems.

Aerobic biochemical transformations that occur within the filter are the primary treatment mechanism but physical filtration and chemical sorption are also significant mechanisms. Oxygen is supplied by diffusion and mass flow of air behind wetting fronts through pore spaces in the media. Bio-slimes from the growth of microorganisms develop as films on the porous media. The microorganisms in the slimes absorb soluble and colloidal waste materials in the wastewater as it percolates over the surfaces of the media. The absorbed materials are incorporated into new cell mass or degraded under aerobic conditions to carbon dioxide and water. The BOD is nearly completely removed if the wastewater retention times in the media are sufficiently long for the microorganisms to absorb the waste constituents. With depleting carbonaceous BOD in the percolating wastewater, nitrifying microorganisms thrive deeper in the surface layer where nitrification readily occurs.

“Single pass” and “recirculating” filters are used. With single pass or “intermittent” filters, the wastewater passes through the filter media only once before being discharged for further treatment or dispersal. Recirculating filters recycle the filtrate through the filter several times. The recirculation provides the needed wastewater residence times in the media to achieve nitrification requirements. Recycling provides more control of treatment process through adjustments in recycle ratios and frequency. BOD and TSS removals are somewhat greater than those achieved by single pass filters and nitrification is nearly complete. In addition, the mixing of the return filtrate with fresh influent in the recirculation tank results in significant nitrogen removal. Also, the filtrate can be recycled back to the treatment head works to mix with undiluted raw wastewater or to an anoxic reactor between the septic tank and recirculation tank to increase nitrogen removal significantly. Therefore, because of these advantages of recirculation without a loss of passivity, only recirculating filters are considered here. 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 filters using various media types is presented in Table 3. Typical filter effluent concentrations treating domestic wastewater treatment are <10/10 mg/L for BOD and TSS respectively and approximately 50% total nitrogen removal. With recycle back to the septic tank total nitrogen removal can increase to 70% (USEPA 2002).

Recirculating sand filters (RSF) are capable of achieving ammonia removals of 98% and Total N removals of 40 to over 70% (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% or greater from septic tank effluent, with effluent NH_3-N in some cases of 1 mg/L or less ((Lacasse, Bélanger et al. 2001; Lindbo and MacConnel 2001; Loomis, Dow et al. 2004; Patterson 2004; Rich 2007) and can also bind phosphorus ((Kõiv, Vohla et al. 2009). TN reductions of 29 to 41% have been reported in modular recirculating peat filters (Monson Geerts, McCarthy et al. 2001a); 44% in peat filters using pressurized dosing (Patterson 2004); and 15 and 21% in two single pass modular peat filters.

Recirculating textile filters were shown to achieve 44 to 47% 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).

A variety of different media including slag, polonite (a calcium silicate based mineral material), limestone, opoka, and sand; greater than 98% 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/ft2-day and 22 to 58 gram BOD5/m2-day; over 90% 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% when operated at 3.8 gal/ft2-day (Rodgers, Lambe et al. 2006).

Synthetic media generally have smaller footprints and higher area hydraulic loading rates than traditional sand filters. However, smaller footprints usually come at the cost of energy input to power pumps or blowers. Issues involved include the use and need for recirculation, effluent levels of organic and ammonia N achievable, and reliability of performance. With some media, recirculation may be more important to maintaining a given level of ammonia removal, which may be related to reaeration at attachment sites of active nitrifying microorganisms. A rational basis for comparison of aerobic media could potentially be developed using the effective media surface area and residence time within the filter bed, as perhaps modified by factors effecting media reaeration and by recirculation. The overriding requirement for the aerobic treatment performance is to produce low effluent levels of organic N and ammonia N prior to treatment in anoxic reactive media filters.

Table 6:
Summary of Media Filter Performance

Media Type	Features	Typical Performance Range	Citations
Sand	1.5 - 3 mm media 18 - 36 in. depth 3 - 5 gal/ft2-day 40 - 120 dose/day	TN: Removal: 40 to 75% Effluent: 15 to 30 mg/L NH3-N: Effluent: 1 to 5 mg/L	(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)
Textile	2 - 3 in. cubes 36 - 72 in. depth 8 - 17 gal/ft2-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	(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)
Peat (single pass or recirculation)	246 36 in. depth 3 to 6 gal/ft2-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	(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)
Open Cell Foam (single pass or recirculation)	3 - 4 in. cube media 48 in. depth 11 gal/ft2-day	TN: Removal: 62% Effluent: 14 mg/L NH3-N: Effluent: 2.4 mg/L NO3-N: Effluent: 10 mg/L	(NSF-International 2003e)

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Table 6:
Summary of Media Filter Performance

Media Type	Features	Typical Performance Range	Citations
Zeolite	20 - 30 in. depth 6.1 gal/ft2-day	NH3-N: Removal: 98.6% Influent: 70 mg/L Effluent: 1 mg/L NO3-N: Effluent: 57 mg/L	(Philip and Vasel 2006)
Zeolite	24 in. media depth Stratified media size 8 in. 2.3-4.8 mm 8 in. 1.2-2.4 mm 6 in. 0.5-1.2 mm 2.9 gal/ft2-day	TN: Removal: 36.1% Influent: 72.2 mg/L Effluent: 43.6 mg/L NH3-N: Removal: 99.9% Influent: 63.4 mg/L Effluent: 0.036 mg/L NO3-N: Effluent: 38.8 mg/L	Smith et al. 2008 Smith, 2008
Expanded Clay	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/ft2-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	Smith et al. 2008 Smith, 2008
Coir	Coconut coir media 18 gal/ft2-day 5.88 gal/ft3-day	TN: Removal: 55% Influent: 38 mg/L Effluent: 17 mg/L TKN: Removal: 83% Influent: 38 mg/L Effluent: 6.5 mg/L	(NSF-International 2006; Sherman 2006; Talbot, Pettigrew et al. 2006; Sherman 2007)137,180,181,196

Table 6:
Summary of Media Filter Performance

Media Type	Features	Typical Performance Range	Citations
Aerocell biofilter	2 in. cube media 18 gal/ft2-day 5.88 gal/ft3-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	(NSF-International 2005)136
Polystyrene	24 in media depth Polystyrene sphere media 2.5 - 4.5 mm 6.6 gal/ft2-day	NH3-N: Removal: 97.7% Influent: 92.5 mg/L Effluent: 2.1 mg/L	E-Z Treat Company, 2009

Factors affecting performance of recirculating media filters are listed in Table 6. The hydraulic, organic and nitrogen loading rates critical operating parameters, 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 colloidal constituents in the wastewater, the need for sufficient pore space for assimilation of solids materials and their biodegradation between doses, the water retention capacity of the media, and the pore space that is available for aeration. The characteristics of media that influence performance of unsaturated filters are listed in Table 5. The performance of any unsaturated media filter is determined by the interactions of media characteristics (Table 5) with system parameters (Table 6). A significant interaction that occurs between the media and the system is the water retention capacity of media versus the hydraulic application rate. High water retention capacity is desirable to retain wastewater within the filter and achieve low effluent levels. The water retention capacity of media must exceed the hydraulic application rate per dose to prevent rapid movement of applied wastewater through the filter. More frequent doses (lower volume per dose), higher recycle to forward flow rations and high water retention media, offers the most favorable combination.

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 hydraulic capacity treatment efficacy (USEPA 2002; Kang, Mancl et al. 2007). A recent model was used to predict the clogging of intermittent sand filters as a function of the total suspended solids loading rate (Leverenz, Tchobanoglous et al. 2009). Conclusions drawn form this study would apply to recirculating media filters as well.

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% could have an aeration porosity of only 2.5% of the total media volume, depending on sand size and the hydraulic application rate. Such conditions could decrease nitrification effectiveness and perhaps also increase denitrification within microzones with limited contact with the gas phase. Denitrification within an unsaturated filter would improve total nitrogen removal but could result in less efficient nitrification and higher effluent ammonia concentrations. By contrast, media with high total porosity would be more likely to have a sufficiently high aeration porosity to allow effective utilization of all media surface area and better ammonia removal performance. If the goal is to achieve total nitrogen removal in an overall system containing an unsaturated filter followed by an anoxic, reactive media de-nitrification filter, then the goal of low effluent ammonia should take precedence over de-nitrification in the unsaturated first stage filter. An example media with high total porosity and high water retention capability is sphagnum peat moss. The total porosity of sphagnum peat is greater than 85%, and percolating water might occupy two thirds of this available pores. Under these conditions, pore space available for aeration would be over 25% of the total volume of the filter bed. The very low effluent ammonia levels that peat filters appear capable of producing may be related to these factors.

Table 7:
Factors Influencing Performance of Unsaturated Aerobic Filters

Feature	Affect
Hydraulic loading rate	Water retention time is inversely proportional and pollutant loading directly proportional to the hydraulic loading rate
Organic loading rate	Oxygen requirements is directly proportional to the organic loading and media clogging potential
Nitrogen loading rate	Oxygen and alkalinity requirements are directly proportional to the nitrogen loading
Media depth	Media depths greater than 3 ft provide marginally greater treatment
Specific surface area	Active media surfaces are directly proportional to the specific surface area of the media
Average water residence time	Longer residence time gives more time for biochemical reactions and better treatment
Uniformity of Dosing	Promotes full utilization of all elements of the filter media
Wastewater	
Suspended solids	Accumulated within pores, may lead to clogging if not biodegraded
BOD	High values require more room for attached growth and metabolism between doses, particularly in upper filter layers
Organic and ammonia nitrogen	Significant component of total oxygen supply requirement
Alkalinity	Consumed by nitrification and restored by heterotrophic denitrification; adequate supply needed to prevent pH decline by nitrification
Particle size distribution	Larger particles less subject to clogging Smaller particles have greater surface area per volume for treatment
Uniformity coefficient	Effects flow uniformity
Specific surface area	Higher values give greater attachment surfaces for microorganisms
Air filled porosity	Oxygen supply throughout media depth for BOD oxidation and nitrification in unsaturated filters
Water retention capacity	Higher water retention in unsaturated media filters provides longer time of contact of water with microorganisms and better treatment; affected by intrinsic porosity that favors capillary water retention

Media with significant ion exchange capacity may offer a method to superior removal of ammonia nitrogen in flowing systems, and zeolite media are excellent surface for biofilm attachment, 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% 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 pilot 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% (Smith 2008; Smith, Otis et al. 2008). In these studies, the filters were able to sustain a BOD_5 surface loading rate of 18 to 20 gram/m^2-day without surface ponding or observable material accumulations of the media surface, which contrasts to reported COD loadings of 19 gram/m^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 such 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% 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). Synthetic fiber materials could have many of the same advantages as a media as coir.

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.

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 18. All have been adapted for use in onsite treatment.

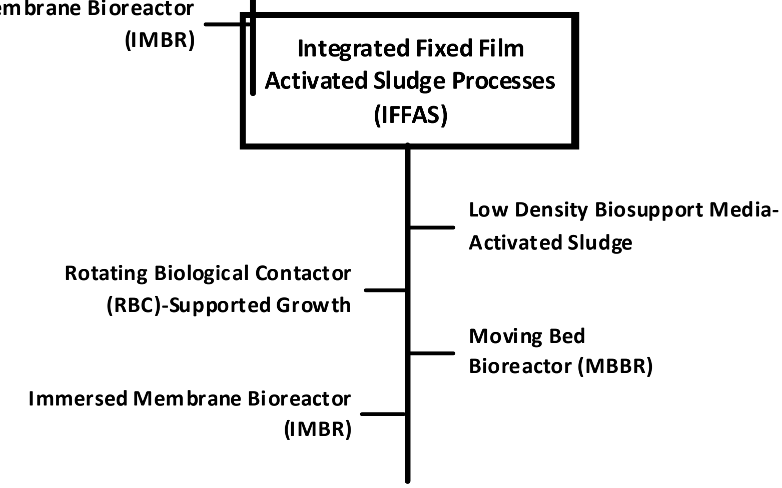

Figure 18: Common Integrated Fixed Film Activated Sludge (IFAS) Processes

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.

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 is eliminates the problem of nitrate "leakage" in the discharge, which can occur in mixed biomass systems. Consequently, a high degree of treatment is achieved more easily. However, organic carbon that is used in single stage (mixed biomass) processes does not reach the second anoxic stage requiring that an external donor be supplied to the second stage. Also alkalinity, which is recovered during denitrification, cannot be recycled. If it is needed to buffer the nitrification stage, an external source of alkalinity would need to be added.

Two groups of processes are used for denitrification; heterotrophic denitrification that 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.

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. Factors influencing the performance of anoxic denitrification filters are listed in Table 9. 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 (BOD for wood-based filters; sulfate for sulfur based filters). An aerobic process effluent low in BOD and suspended solids would be less likely to lead to channeling within the anoxic filter. 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 10.

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% 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 gallons/ft2-day; the effluent NO3-N averaged 0.6 mg/L. Other heterotrophic denitrification systems have been successfully tested at laboratory scale.

Table 8:
Factors Influencing Performance of Saturated Anoxic Filters

Feature	Effect
Hydraulic loading rate	Higher rates lower water retention time and treatment
Organic loading rate	Higher loading rates increase rate at which heterotrophic biomass could accumulate
Solids loading rate	Higher loading rates increase rate at which solids could accumulate
Nitrogen loading rate	Higher loading rates require higher denitrification rates and higher rates of electron donor dissolution
Media depth	Deeper beds can give better treatment; uppers layers often more reactive
Specific surface area	Higher values give greater surface area for attachment of microorganisms and dissolution of media
Superficial velocity	Effects mass transfer between wastewater and biofilms
Average linear velocity	Effects mass transfer between wastewater and biofilms
Average water residence time	Longer residence time gives more time for biochemical reactions and better treatment
Wastewater	
Suspended solids	Accumulated within pores, may lead to preferential flow if not biodegraded
BOD	Will create more heterotrophic biomass and may increase potential for preferential flow
Nitrate nitrogen	High loadings require greater surface areas and higher levels of denitrifying activity
Alkalinity	Consumed by autotrophic denitrification; must be balanced by sum of influent alkalinity and alkalinity provided by solid source

Table 9:
Summary of Saturated Anoxic Media Reactors

System Type	Description	Features	Treatment Performance	Citations (Refer to Appendix B)
Sulfur/oyster shell filter (bench scale)	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	(Sengupta and Ergas 2006)
Sulfur/oyster shell filter (bench scale)	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/ft2-day	anoxic only NO3-N Removal: 99.9% Influent: 38.8 mg/L Effluent: 0.030 mg/L	Smith et al. 2008 Smith, 2008
Sulfur/oyster shell filter (bench scale)	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/ft2-day	anoxic only NO3-N Removal: 99.9% Influent: 58.8 mg/L Effluent: 0.031 mg/L	Smith et al. 2008 Smith, 2008
Sulfur/oyster shell filter (bench scale)	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/ft2-day	anoxic only NO3-N Removal: 89.9% Influent: 47.7 mg/L Effluent: 4.3 mg/L	Smith et al. 2008 Smith, 2008
Sulfur/limestone column	22.4 gal. column Simulated groundwater	Sulfur/limestone media (75/25% by volume)	anoxic only NO3-N Removal: >95%	(Moon, Shin et al. 2008)

Table 9:
Summary of Saturated Anoxic Media Reactors

System Type	Description	Features	Treatment Performance	Citations (Refer to Appendix B)
	upflow single pass Residence time: 24 to 48 hr.	Sulfur: 5 to 10 mm 5 to 10 gal/ft2-day	Influent: 60 mg/L Effluent: < 1 mg/L NO2-N Effluent: < 1 mg/L	
Sulfur/oyster shell filter	185 gal. column aerobic effluent upflow single pass 18 hr. HRT	Sulfur/oyster shell media (75/25% by volume) 47 gal/ft2-day	anoxic only NO3-N Removal: 88% Influent: 20 mg/L Effluent: 2.4 mg/L	(Brighton 2007)
Sulfur/limestone column	237 gal. column groundwater upflow single pass Residence time: 13 hr.	Sulfur/limestone media (67/33% by volume) 63 gal/ft2-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	(Darbi, Viraraghavan et al. 2003a)
NitrexTM	aerobic effluent gravity flow upflow single pass	Nitrex wood-based media 24 to 30 inch media depth (est.) 4.6 gal/ft2-day (est.)	aerobic+anoxic TN Removal: 79 to 96% Effluent: 3 to 18 mg/L NO3-N Effluent: 0.3 to 8 mg/L	(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

Table 9:
Summary of Saturated Anoxic Media Reactors

System Type	Description	Features	Treatment Performance	Citations (Refer to Appendix B)
				Foreman 2007)
Black& GoldTM	wood-based media single pass downflow gravity	Influent: STE 280 gal. column Sand/tire crumb/woodchip (85/11/5% by volume) 8.3 gal/ft2-day	aerobic+anoxic TN Removal: 98% Influent: 414 mg/L Effluent: 7.1 mg/L NH_3-N Effluent: 4.4 mg/L NO_3-N Effluent: 0.05 mg/L	(Shah 2007)

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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 shall 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 a Clean Solution aerobic treatment system that was treating septic tank effluent. The sulfur/oyster shell filter removed 82% of influent TN, while the aerobic/sulfur treatment train removed 89.5% 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% and average effluent NO3-N was 2.4 mg/L (Darbi, Viraraghavan et al. 2003a). An 85 liter upflow column packed with sulfur/limestone at a 3:1 vol./vol. ratio treated a simulated groundwater at 0.9 to 1.8 gal/ft2-day surface loading rate and removed greater than 95% of nitrate that was at 60 mg/l in the influent (Moon, Shin et al. 2008). A laboratory sulfur/oyster shell column was operated at and Empty Bed Contact Time of 0.33 to 0.67 days and removed 80% 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 (Sengupta and Ergas 2006; Smith 2008; Smith, Otis et al. 2008). Average NOx reductions were 99.9, 99.9 and 88.9% 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 0.20 m/day and at average NOx-N loadings of 17 to 26 gram/m2-day, which are similar to loading rates applied to acetic acid amended sand denitrification filters that achieved 94 to 99% 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). Versus wood based organics electron donors, elemental sulfur could possibly remain physically intact for longer time periods. 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 wood 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.

PHYSICAL / CHEMICAL NITROGEN REDUCTION PROCESSES

Because of the complexity and cost associated with physical/chemical treatment systems very few available OSTDS systems were identified. Manufacturers of these processes focus on larger wastewater treatment plants; therefore, very few small flow designs exist. Three manufacturers of physical/chemical systems were identified in the search, all three were foreign manufacturers. One manufacturer, Wallax of Sweden, manufactures a complete physical/chemical treatment system. The other two firms, Columbio and Biovac A/S, are from Norway and both manufacture a biological/chemical system. These systems and their relative performance are discussed by Paulsrud and Haraldsen (1993), however, they are no specifically designed for N removal. Therefore, they provide little benefit over a conventional system as nitrogen reducing technology.

NATURAL SYSTEMS

Denitrification in Soil

Biological denitrification is a complex process that requires mineralization and nitrification the nitrogen before denitrification can occur. With the decay of organic matter, nitrogen is released into the environment as organic nitrogen (principally proteins and urea). Bacteria and fungi in the soil quickly “mineralize” the organic nitrogen by converting it to ammonium. The ammonium is nitrified by autotrophic bacteria, which use carbon dioxide for their carbon source instead of organic carbon. These bacteria are obligate aerobes that require an aerobic environment because oxygen is used as the final electron acceptor. Since hydrogen ions are created by this reaction, which can lower the pH to levels that inhibit the biological process, it is essential that sufficient alkalinity be available to buffer the soil solution so that nitrification can be complete. After nitrification, heterotrophic bacteria are able to convert the nitrate to gaseous nitrogen and NOX as they oxidize available organic matter. However, for this conversion, an anoxic or anaerobic environment is required since the oxygen associated with the nitrate is used as the final electron acceptor in oxidizing the organic matter. If either anoxic conditions or organic carbon are not available, denitrification does not proceed via this pathway. Other pathways exist, but they are far less prevalent.

The heterotrophic bacterial process models were used to define the mechanisms and the necessary conditions for biological denitrification to occur. By understanding these, the literature could be reviewed for the occurrence of the requisite conditions in soils from which the potential for nitrogen removal could be estimated. The most critical conditions for which data are available were selected to investigate. These included the soil’s internal drainage, depth to saturated conditions, and the availability of organic materials. 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 soil to allow the autotrophs to nitrify the ammonium nitrogen. The shallower the depth to the water table, the more likelihood organic matter will be leached to where the soil moisture is high enough to restrict soil reaeration to the point that aerobic organic matter decomposition is inhibited preserving the carbon for heterotrophic denitrification. The availability of organic carbon determines the occurrence and extent of denitrification that will occur.

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 nitrogen 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%. While Anammox quite likely could contribute substantially to the reduction of nitrogen below OWTS, little is known about the conditions under which it is likely to occur. Until the process requirements are better understood, detection of denitrification via the Anammox process would requires actual monitoring data where the nitrogen reduction by the heterotrophic processes can be separated out. Such data were not available so the estimates of nitrogen removal below OWTS reported in this study may under estimate the actual removals.

The extent to which denitrification occurs in soils varies depending on the specific environmental conditions at the particular site, and the design and operation of the OWTS. Numerous investigations into the fate of nitrogen below soil infiltration zones have been undertaken. However, the results are quite variable even for sites that appear similar. Gold and Sims (Gold and Sims 2000) point out that the dynamic and open nature of soil water infiltration designs results in uncertainties with in-situ studies of the fate of nitrogen in soil. The affects of dispersion, dilution, spatial 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 5 to 215 cm/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 10).

Table 10:
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 OWTS to develop a methodology for predicting OWTS 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% 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 10.

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 OWTS. They found denitrification removals varied substantially depending on the type of pretreatment and the design of the soil water infiltration system (Table 11).

Table 11:
Estimates of TN Removal Based on Soil Texture (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.

Table 12:
Total Nitrogen Removal Found in Various Studies of OWTS

System Type	TN Removal	Source
Traditional	0-35%	(Ritter and Eastburn 1988)
Sand filter	71-97%	(Wert and Path 1985)
Low Pressure Dosing Shallow	46%	(Brown and Thomas 1978)
Low Pressure Dosing At-Grade	98%	(Stewart and Reneau 1988)
Mound	44-86%	(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. OWTS 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' 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 12 provides a brief description of each of the classes.

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 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 typically flows deeper within the subsoil and does not intersect the reduced and organic enriched surface horizons. The groups and their expected impacts on denitrification are given in Table 13.

Heterotrophic bacterial denitrification is often limited by organic matter (Burford and Bremner 1975; Gambrell, Gilliam et al. 1975; Christensen, Simkins et al. 1990; Bradley, Fernandez et al. 1992) The organic carbon is necessary as an energy source for bacterial metabolism. 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. Studies indicate that denitrification is inhibited where the nitrate to dissolved organic carbon ratio is below 0.73 to 1.3 (Burford and Bremner 1975).

The amount of organic matter in the soil is greatest in the root zone and above (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 water 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% or the moisture content is greater than 60 to 75% of the soil's water holding capacity, reaeration is sufficiently restricted

**Table 13:
NRCS Drainage Classes and Descriptions**

Drainage Class	Description
Excessively drained	Water is removed from the soil very rapidly. The soils are very porous. These soils tend to be droughty.
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 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.
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.
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.
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.
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.

Table 14:
Drainage Class and Expected Impacts on Denitrification

Drainage Class Group	Expected Impact on Heterotrophic Denitrification
Excessively/ Somewhat excessively	• 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	• 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	• 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
Somewhat poorly/ Poorly/ Very poorly	• 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

that anoxic conditions can result (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 one meter, 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. 1973; Reneau 1977; Cogger 1988)Cogger, 1988; Reneau, 1977, 1979; Walker et al.,1973a).

The type of infiltration system used can affect the soil's potential for nitrogen removal. Traditional in-ground trench systems are installed with their infiltrative surfaces typically below the A horizon and thus below where organic matter can be expected to be the highest. At-grade and mound systems are typically installed above the O and A horizon thereby gaining the advantage of having a high organic layer available to create anoxic conditions with organic carbon available (Harkin, Duffy et al. 1979; Converse 1999). However, in Florida, the OWTS rules for mound construction require the removal of the O and A horizons, which removes most of the available organic carbon. Also, "digouts", which are systems on sites where a restrictive horizon in the soil profile is removed, can result in reducing a particular soil's nitrogen removal potential because quite often the restrictive horizon removed is a spodic layer, which can have a sufficiently high organic content and be restrictive enough to create a saturated zone where anoxic conditions may be created for denitrification.

MODIFICATIONS TO CONVENTIONAL ONSITE TREATMENT SYTEMS

Drainfield Modifications

Modifications to drainfields entail the in-situ addition of a permeable 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 modified drainfield design using a sulfur/limestone layer beneath a sand layer provided greater than 95% 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 saturated condition.

A wood based system using a mixture of sand, wood chips, and tire crumb (85/11/4% 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% 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. The consequences of uncertainty in the life of an in-situ denitrification zone located within a drainfield could be relatively more significant than for an in-tank denitrification filter, where media replacement would not require disruption of other treatment system components. Another issue of possible concern is the ability to definitively monitor in-situ nitrogen removal in subsurface locations.

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Section 6.0
CONCLUSIONS AND RECOMMENDATIONS

A review was conducted of passive technologies that enhance removal of nitrogen from on-site wastewater treatment systems. The review included searches of peer reviewed literature and conference proceedings, procuring technical reports, searches on the world wide web, discussions with vendors and national experts, and a site visit to the Massachusetts Alternative System Test Center. These efforts provided the basis for a critical assessment of the present state of technology. The following summarize the significant conclusions of this effort.

- To achieve high nitrogen removals from septic tank effluent using "passive" systems as defined by the study goals, a promising approach is a two stage filter system consisting of an unsaturated first stage media filter followed by a directly connected second stage anoxic filter with reactive media for denitrification; pressure and timed dosing to the first stage; with possible recirculation around the first stage.

- The two stage filter system could be configured in various manners, including an above ground system in separate tanks, as an unsaturated filter stacked above a saturated filter, or with the saturated second stage in the subsurface.

- Filter media that appear promising for passive nitrogen removal include zeolites, expanded clays and shales, peat, coir, synthetic fabrics, and tire crumb (first aerobic stage), and elemental sulfur and cellulosic based materials (sawdust and woodchips) in the second stage.

- As defined by FDOH, a passive system includes only one liquid pump and no aerator pumps. These constraints may limit performance or reduce reliability. Studies of actual field installations are required to ascertain their ability to perform satisfactorily over extended time periods.

- Passive systems to remove nitrogen from septic tank effluent (STE) must consider the entire nitrogen transformation process, including ammonification and nitrification (aerobic stage) and denitrification (anoxic stage), and the integration of these processes into a system that achieves total nitrogen reduction.

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- Aerobic, unsaturated filtration technologies have been well studied and in some cases can achieve effluent ammonia nitrogen levels of five milligrams per liter or less. Most prominent current technologies include sand, peat, textile and foam media, and often employ recirculation. Alternative media offer exciting possibilities for improved performance.

- Passive denitrification filters employ solid phase electron donors to produce saturated anoxic environmental. Passive technologies are currently under development or in early stages of deployment. Promising filter systems include cellulosic based media (wood, sawdust), other organic media, and elemental sulfur based systems.

- Passive denitrification technologies have not been deployed for sufficiently long periods of time to fully evaluate longer term performance, operation and maintenance requirements, media longevity, and media replacement The ability of passive denitrification media to maintain a long term supply of carbon or electrons for denitrification is a significant factor affecting their longevity. Theoretical stochiometric calculations provide an initial estimate of longevity, but longer term studies are needed to verify these results in practice.

- The longevity of passive denitrification filter systems may be affected by the long term accumulation of organic and inorganic solids within the filter media. This could be more important than the duration of the carbon or electron donor supply. Solids accumulation can result in the development of preferential flow paths, reduced contact of wastewater with solid media, and deterioration of performance. Longer term studies are needed to verify continued performance of denitrification filters in practice, and to determine filter maintenance needs and media replacement requirements.

- Constituents released by passive denitrification media include biodegradable organic matter (BOD) from carbon-based systems, and sulfate and possibly sulfide from sulfur-based systems. The environmental acceptability of constituent release must be ascertained.

- The practicality and life cycle costs of media replacement must be evaluated for all systems, including frequency of replacement, site access issues, replacement volumes, and management of used media.

- Modifications to soil treatment units have been evaluated in limited laboratory systems and some field studies are underway, using denitrification media similar to those used in in-tank treatment processes.

- In-soil denitrification is highly dependent on the specific environmental conditions at a particular site and operation of the onsite wastewater treatment and disposal system.

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).

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This document summarizes the scientific background information that led to these conclusions, reviews the reasons the proposed bioassay is considered inappropriate, outlines the factors that need to be considered in developing appropriate bioassays, and identifies gaps in our knowledge currently impeding the development of appropriate bioassays.

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