Hazen

Technology Assessment for New York State Center for Clean Water Technology Final Report

Stony Brook University

Revised Final Report
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1. Introduction

New York State has established the New York State Center for Clean Water Technology (CCWT) at Stony Brook University in order to develop and promote wastewater technologies to reduce nitrogen loads and other contaminants to groundwater and surface waters. Nitrogen from onsite wastewater treatment systems (OWTS) has been a growing concern due to the sensitivity of Long Island's water resources to nutrient-induced challenges. Consequently, CCWT has commissioned a report to review and assess existing and emerging nitrogen reducing OWTS, with systems spanning from conceptual to implementable. Hazen and Sawyer was retained to provide this review and technology assessment, the results of which are summarized in this report. The report is divided into the following sections:

- Section 1: Introduction
- Section 2: Background
- Section 3: Literature Review of Existing Onsite Wastewater Technologies and Practices
- Section 4: Onsite Wastewater Treatment Patent Search
- Section 5: Technology Ranking Criteria
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This report provides a detailed synthesis of the literature, which incorporates, updates and expands the scope of previous reviews that were prepared as part of the Suffolk County IA System Evaluation (H2M 2013), Florida Department of Health Onsite Sewage Nitrogen Reduction Strategies study (Hazen and Sawyer 2009), Chesapeake Bay Watershed study (USEPA 2013), Suffolk County Comprehensive Water Resources Management Plan (2015) and other assessments. The review catalogued well over 1,300 papers, proceedings, reports, and manufacturers' technical materials regarding existing and emerging technologies, which can be accessed on the database accompanying this report. As this review indicates, a wide variety of nitrogen reduction technologies exist and are available for use in OWTS. These existing and emerging approaches span a range of nitrogen reduction processes which are summarized into four main categories: biological processes; soil, plant and wetland processes; source separation; and physical/chemical processes.

An overview of pathogen, pharmaceutical, and personal care product occurrence in the environment and in municipal wastewater is presented and removal technologies in natural and engineered systems are discussed. These insights, gained from bench-scale research to full-scale monitoring programs, enabled the identification of potential collaborators in the private sector for future development and evaluation of pathogen, pharmaceutical, and personal care product removal efficiencies by OWTS. Additionally, the results of an in-depth patent search are presented, based on the use of the United States Patent and Trademark Office (USPTO) and the European Patent Office (EPO) database, Espacenet. The intent of the patent search is to discover novel and unique wastewater treatment ideas which have been published but not necessarily marketed or produced. The patents highlighted in Section 4 illustrate and describe aspects of different onsite approaches and emerging technologies which may be considered in the development of further OWTS research and pilot testing.

Knowledge gained from the literature review was used to conduct a technology assessment, in which evaluation matrices were used to compare OWTS across multiple criteria. The matrices were developed by categorizing the major processes of each OWTS, and then quantifying treatment effectiveness, operability, complexity, energy use, and other considerations based on reported data and experience. The matrices were used to evaluate and rank nitrogen reduction technologies for further evaluation. Matrix criteria and corresponding rankings are presented herein. In addition, knowledge gaps pertaining to OWTS are discussed, thus highlighting opportunities for CCWT to engage in pilot- and full-scale testing of systems that are at the implementation stage, as well as fundamental research needs for newer OWTS concepts.

2. Background

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

The concentration of total nitrogen (TN) in household wastewater will depend on the number of residents in the home, number and model of water-using appliances, and water use characteristics. As the number of residents increases, water use per capita typically decreases but the nitrogen loading does not. Consequently, homes with more residents often have higher TN concentrations in their wastewater. Therefore, using TN concentration without good flow estimates based on expected occupancy of the home can result in under or over sizing of the OWTS. Measured average per capita daily indoor residential water use (a surrogate for wastewater flows) show that it typically ranges from 40 to 70 gpd per person (Brown and Caldwell 1984; Anderson and Siegrist 1989; Anderson, Mulville-Friel et al. 1993; Mayer, DeOreo et al. 1999; Foundation 2014), with lower values in more recent years. These per capita flow values result in estimated raw wastewater nitrogen concentration of 75 to 42 mg-N/L respectively. In commercial establishments, the daily wastewater flow and nitrogen concentrations vary considerably and are based on specific activities in the establishment.

A variety of nitrogen reduction technologies exist and are available for use with OWTS. The technologies can be grouped into four general process categories; 1) engineered biological processes 2) physical/chemical processes, 3) natural systems consisting of soil, plant and wetlands processes, and 4) source separation, (Figure 2-1). Biological nitrification/denitrification processes have been the most common approach for nitrogen reduction technology OWTS applications. Soil treatment systems, which primarily rely on the biological treatment processes and the assimilative capacity of the receiving environment, have been the most prevalent of the OWTS used to protect public health and our water resources in the past. 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. Physical/chemical (P/C) reduction methods have been generally less favored because of the greater need for operator attention, greater chemical and energy costs and larger volumes of residuals that may be generated. Source separation is an emerging option as the technologies improve and the nutrients recovered are increasingly valued.

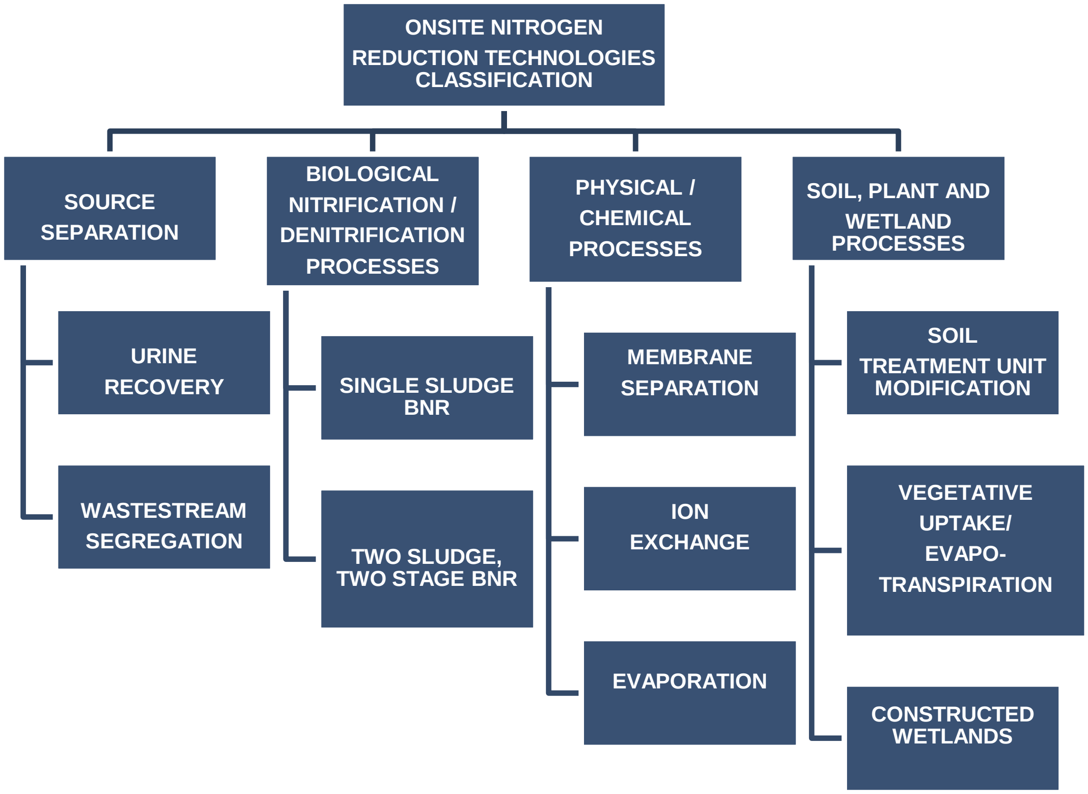


Figure 2-1: Onsite Wastewater Treatment Technology Classifications for Reducing Nitrogen in Household Wastewater

2.1 Biological Processes

2.1.1 Biological Wastewater Treatment

The overall objectives of the biological treatment of domestic wastewater are to 1) transform (i.e. oxidize) dissolved and particulate biodegradable constituents into acceptable end products, 2) capture and incorporate suspended and nonsettleable colloidal solids into a biological floc or biofilm, 3) transform or remove nutrients, such as nitrogen and phosphorus, and 4) in some cases, remove specific trace organic constituents and compounds. The removal of dissolved and particulate carbonaceous BOD and the stabilization of organic matter found in wastewater is accomplished biologically using a variety of microorganisms, principally bacteria. Microorganisms are used to oxidize (i.e. convert) the dissolved and particulate carbonaceous organic matter into simple end products and additional biomass. The principal biological nitrification and denitrification processes typically used for general wastewater treatment can be divided into three main categories: suspended growth, fixed film (or attached growth) processes, and aerobic granular sludge (AGS). Integrated fixed film activated sludge (IFAS) is a group of technologies that combine both fixed film and suspended growth microbial communities (Figure 2-2).

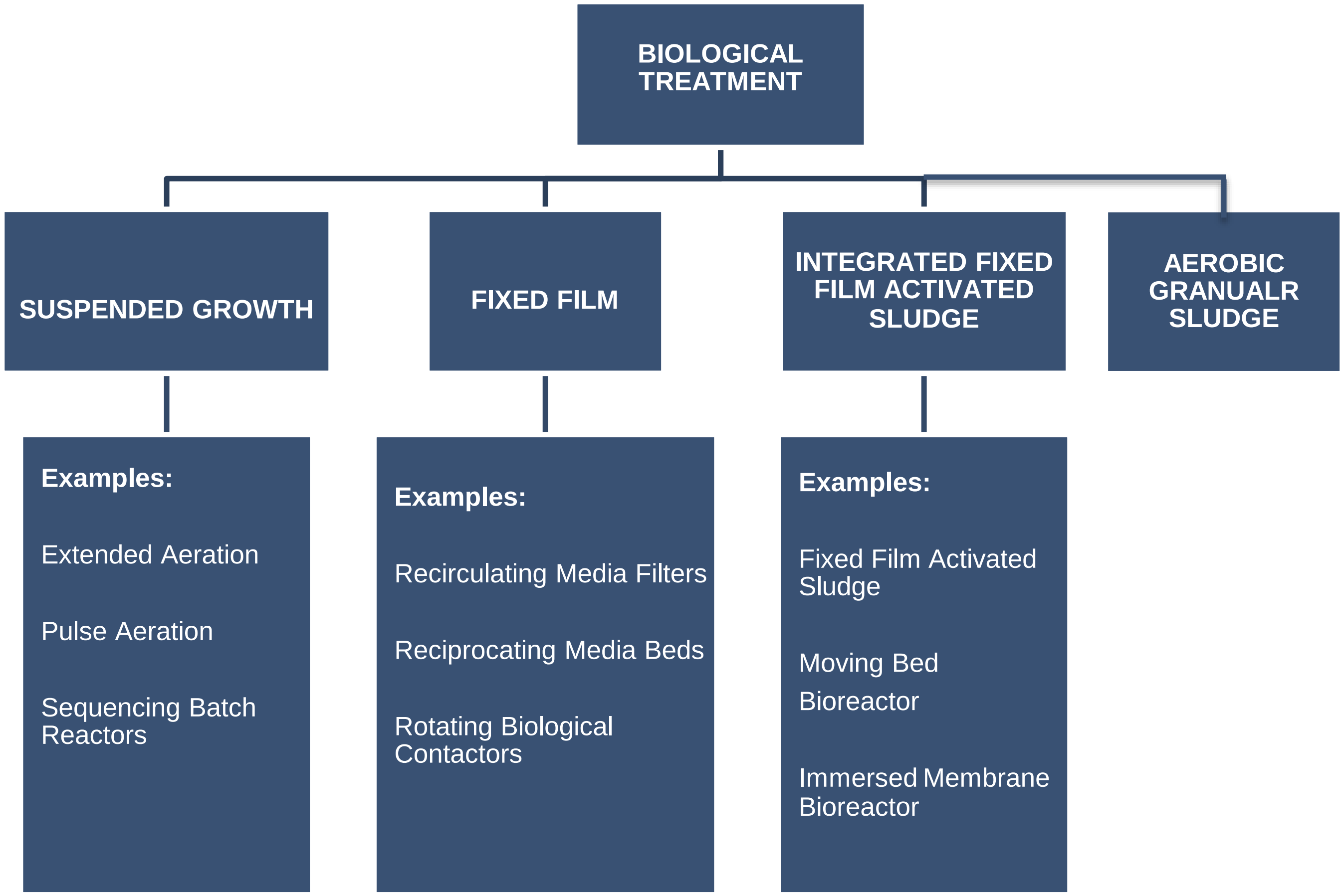


Figure 2-2: Biological Wastewater Treatment Processes

2.1.1.1 Suspended Growth Processes

In suspended growth processes, the microorganisms responsible for treatment are maintained in liquid suspension by appropriate mixing methods. Many suspended growth processes used in wastewater treatment for biodegradation of organic substances are operated with dissolved oxygen (aerobic) or nitrate/nitrite (anoxic) utilization. The most common suspended growth process used for municipal wastewater treatment is the activated sludge process. The activated sludge process was so named because it involved the production of an activated mass of microorganisms capable of stabilizing a waste under aerobic conditions. In the aeration tank, contact time is provided for mixing and aerating influent wastewater with the microbial suspension, generally referred to as the mixed liquor suspended solids (MLSS). Mechanical equipment is used to provide the mixing and transfer of oxygen into the process. The MLSS then flows to a clarifier where a fraction of the microbial suspension is settled and thickened. The settled biomass, described as activated sludge because of the presence of active microorganisms, is returned to the aeration tank to continue biodegradation of the influent organic material.

2.1.1.2 Fixed Film Processes

In fixed film (or attached growth) processes, the microorganisms responsible for the conversion of organic material or nutrients are attached to an inert packing material. The organic material and nutrients are removed from the wastewater flowing past the attached growth, also known as biofilm. Packing materials used in attached growth processes include rock, gravel, slag, sand, redwood, and a wide range of plastic and other synthetic materials. Attached growth processes can also be operated as aerobic or anaerobic processes. The packing can be submerged completely in liquid or partially submerged, with air or gas space above the biofilm liquid layer. Air circulation in the void space, by either natural draft or blowers, provides oxygen for the microorganisms growing as an attached biofilm. Influent wastewater is distributed over the packing and flows as a nonuniform liquid film over the attached biofilm.

2.1.1.3 Integrated Fixed Film Activated Sludge (IFAS) Processes

Integrated fixed film activated sludge (IFAS) processes 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 single sludge processes. The most common process design immerses low density bio support 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. Alternatively, the media may be fixed and contribute to aeration and mixing similar to the packaged STM Aerotor™ system, which has already been approved for onsite use in Suffolk County.

2.1.1.4 Aerobic Granular Sludge

Aerobic granular sludge (AGS) has emerged as a cost effective alternative to activated sludge processes for meeting strict nutrient limits. In AGS systems, the airlift configuration helps to stratify biomass into granules which allows for carbon, nitrogen and phosphorus removal to occur within a compact footprint that is reported to be 75% smaller versus activated sludge systems. Additionally, it has been reported that AGS systems reduce energy requirements by 25 to 35% versus activated sludge.

While this technology is not yet available for license in the USA, Hazen and Sawyer anticipates that AGS systems will become more widespread. With this in mind, our technical specialists have evaluated pilot data generated from AGS systems with a view to understanding how these systems can be integrated into municipal treatment facilities for nutrient removal. These considerations include providing flexibility in influent equalization and control systems that manipulate feed, reaction and settling times so as to stimulate and sustain granule formation. Academic research of these concepts may be warranted for OWTS.

2.1.2 Biological Nitrification and Denitrification

Specific bacteria are capable of oxidizing ammonia (nitrification) to nitrite and nitrate, while other bacteria can reduce the oxidized nitrogen to gaseous nitrogen (denitrification). The conventional nitrification/denitrification approach of two-step nitrification (nitritation and nitratation) and two-step denitrification (denitratation and denitritation) are displayed in Figure 2-3. In this strategy, ammonia is oxidized to nitrite and then nitrate by aerobic ammonia oxidizing bacteria (AOB) and aerobic nitrite oxidizing bacteria (NOB) respectively. This nitrate is then denitrified to nitrogen gas using heterotrophic or autotrophic bacteria.

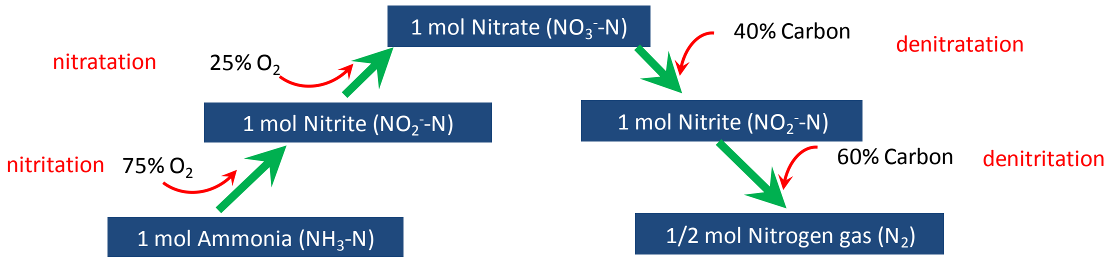


Figure 2-3: Conventional Biological Nitrification and Heterotrophic Denitrification Processes

2.1.2.1 Nitrification

The nitrification process is composed of two steps. In the first step, *Nitrosomonas*, *Nitrosococcus*, and *Nitrosospira* bacteria convert NH_4 to NO_2, which in the second step is converted to NO_3 by *Nitrospira* and *Nitrobacter* bacteria (USEPA 1993). The bacteria that perform nitrification are chemolithoautotrophs, meaning they use carbon dioxide (CO_2) as the C source and derive energy from chemical reactions in which inorganic compounds are used as the electron donor. For the nitrification process, NH_4 is used as the electron donor and oxygen is the electron acceptor. The energy-yielding two-step oxidation of NH_4 to NO_3 is represented by Equation 2-1 and Equation 2-2 and combined in Equation 2-3 (Metcalf & Eddy 2014) as follows:

Step 1:

$$2NH_4^+ + 3O_2 \rightarrow 2NO_2^- + 4H^+ + 2H_2O \qquad (2\text{-}1)$$

Step 2:

$$2NO_2^- + O_2 \rightarrow 2NO_3^- \qquad (2\text{-}2)$$

Total oxidation reaction:

$$NH_4^+ + 2O_2 \rightarrow NO_3^- + 2H^+ + H_2O \qquad (2\text{-}3)$$

Based on this equation, for each gram of NH_4-N oxidized to NO_3-N approximately 4.57 g O_2 and 7.14 g alkalinity as $CaCO_3$ are consumed. Since the conversion produces hydrogen ions, the pH can be lowered to a point where the nitrifying bacteria can no longer thrive. Therefore, sufficient alkalinity is needed to buffer the pH so that acidic conditions do not occur to inactivate the nitrifiers and prevent complete nitrification. The nitrifying bacteria are also sensitive to cold temperatures, which can slow the reactions. In addition, nitrification is susceptible to organic and inorganic inhibitors present in the wastewater (Li et al. 2016). The bioavailability of inhibitors is a function of many factors, including exposure time, sludge age, hydraulic conditions, and specific inhibitor properties including hydrophobicity, sorption, internalization, solid/solution partitioning, and chemical speciation. Though nitrate can be utilized by organisms for growth, the nitrate produced is negatively charged, which in soils is not adsorbed but travels with the soil water until captured, taken up by plant roots or denitrified.

2.1.2.2 Denitrification

Biological denitrification involves the biological oxidation of soluble organic substrates in wastewater treatment using nitrate and/or nitrite as the electron acceptor instead of oxygen. This process reduces the nitrate to nitrogen gas following the sequence of $NO_3^- \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2$. If the process is interrupted before the sequence is complete, nitric oxide (NO) and nitrous oxide (N_2O) can be released, which contribute to smog and greenhouse gases, respectively. Biological denitrification is the only nitrogen transformation that removes nitrate from ecosystems in the form of nitrogen gas, which is released to the atmosphere. Deammonification is another nitrogen transformation that removes nitrogen from ecosystems in the form of nitrogen gas, which is discussed in Section 1.1.4. Once converted to N_2, the nitrogen is not likely to be reconverted to a biologically available form except through nitrogen fixation.

The most common biological denitrification process is performed by facultative heterotrophic or autotrophic bacteria under anoxic conditions (no free oxygen). The electron donor is typically one of three sources: (1) the biodegradable soluble chemical oxygen demand (bsCOD) in the influent wastewater, (2) the bsCOD produced during endogenous decay, or (3) an exogenous source (Figure 2-4).

The heterotrophs use organic carbon as an electron donor and the oxygen from the nitrate molecule and its resulting breakdown compounds as the electron acceptors to obtain energy necessary for their growth. Under reducing conditions NO_3 can be converted to N_2O and then N_2 by the microbially mediated process of heterotrophic denitrification which use an organic carbon source (in this case glucose) for energy and cell synthesis which can be generally represented by the following equation (Schmidt and Clark 2012):

$$5C_6H_{12}O_6 + 24NO_3^- + 24H^+ \rightarrow 12N_2 + 42H_2O + 30CO_2 \qquad (2\text{-}4)$$

Autotrophs use inorganic compounds such as sulfur, iron and hydrogen as electron donors in place of organic carbon to obtain their energy for growth. The combined oxygen on the nitrate molecule and its breakdown compounds are still used as the electron acceptors. The advantage of using autotrophs over heterotrophs is primarily in the management of the electron donors. Inorganic compounds are easier to manage and maintain than organic carbon in onsite wastewater treatment applications. A number of common soil bacteria, such as *Thiobacillus denitrificans* and *Thiomicrospira denitrificans*, are able to use reduced S compounds as electron donors and respire on NO_3 in the absence of oxygen. Equation 2-5 is a stoichiometric equation for autotrophic denitrification using sulfur as an electron donor (Batchelor and Lawrence 1978):

$$NO_3^- + 1.10\ S^0 + 0.40\ CO_2 + 0.76\ H_2O + 0.08\ NH_4^+ \rightarrow 0.08\ C_5H_7O_2N + 0.50\ N_2 + 1.10\ SO_4^{2-} + 1.28\ H^+ \qquad (2\text{-}5)$$

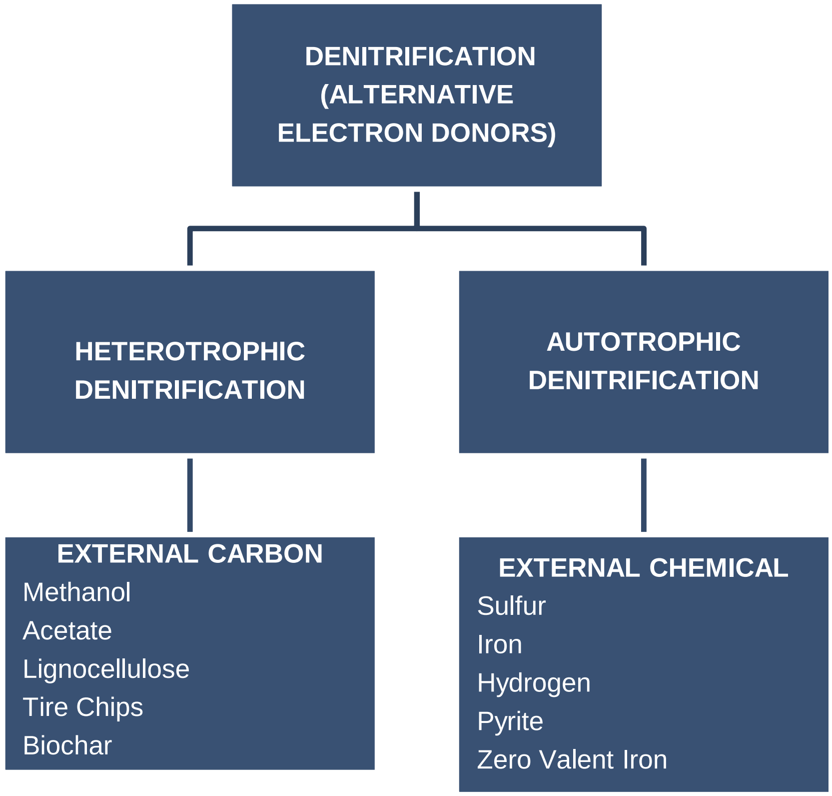


Figure 2-4: Denitrification Exogenous Electron Donors

External carbon sources such as methanol, acetate, lignocellulose, tire chips and biochar can be used to help achieve heterotrophic denitrification. Methanol (CH_3OH) and acetate (CH_3CO_2) are available in liquid form and often used as supplemental carbon sources for wastewater treatment (Metcalf & Eddy 2014). Lignocellulose is composed of biomass from woody plants and is one of the most abundant materials on earth while tire chips are an easily accessible waste product. Biochar is a solid material obtained from thermochemical conversion of biomass in an oxygen limited environment. These five external carbon sources allow carbon supplementation which can increase the rate of denitrification. The rate of denitrification depends primarily on the nature and concentration of the carbonaceous matter undergoing degradation. Rates are highest with a readily biodegradable source, such as methanol. Rates also increase with increasing temperature and with decreasing oxygen concentration.

External chemicals and minerals used for autotrophic denitrification include sulfur, iron, hydrogen, pyrite and zero valent iron. A significant amount of the world's supply of elemental sulfur for human uses formerly came from S-bearing limestone deposits found in the Gulf Coast region of North America. Currently, however, elemental S is produced primarily through its recovery from the hydrogen sulfide (H_2S) in sour natural gas or from coal fired power plant emission controls and by refining of petroleum using the Claus process (Nabikandi and Fatemi 2015). Pyrite is one of the most abundant minerals in the earth’s crust.

2.1.3 Biological Nitrification/Denitrification Processes

To effect biological denitrification in wastewater, OWTS 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. The methods of incorporation of these processes into treatment of onsite wastewater may be grouped into the following two basic categories (Figure 2-5) based on method of denitrification: 1) combined carbon oxidation, nitrification, and denitrification, referred to as the "single-sludge" process, and 2) nitrification and denitrification in separate unit processes, referred to as the "two sludge, two-stage" process. In the single sludge processes, the active microorganisms are a mixture of autotrophs (nitrifiers) and facultative heterotrophs (organic degraders and denitrifiers) while in the two sludge, two-stage processes, the two groups of microorganisms (nitrifiers and denitrifiers) are segregated in separate reactors. Note that simultaneous nitrification/denitrification has the advantage of reducing electron donor requirements with both reactions occurring within the same reactor, at the same time.

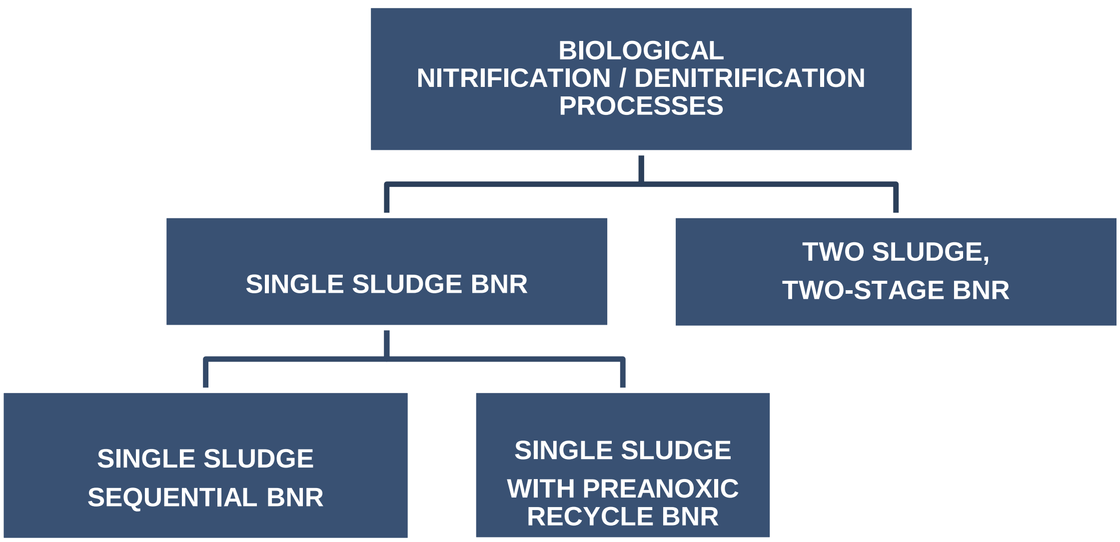


Figure 2-5: OWTS Technology Categories for Biological Nitrification/Denitrification Processes

A single sludge process with preanoxic nitrified effluent recycle relies on the organic carbon from the fresh incoming wastewater as the electron donor for denitrification. In a two sludge, two-stage process, external electron donors are necessary in the second stage (denitrification) because the organic carbon is removed during the first stage (nitrification); however, nitrification is more complete, which results in more complete denitrification than is possible in single sludge systems, especially in OWTS applications where there is little operational control of dissolved oxygen and ammonia conversion.

Reactor pH has a significant effect on nitrification. If the reactor is too acidic, nitrification may cease. Therefore, it is important that the pH be controlled during treatment. The optimum pH range is 6.5 to 8.0 (USEPA 1993). The pH is often controlled naturally by alkalinity in the wastewater itself. However, the nitrification reactions consume approximately 7 mg of alkalinity (as $CaCO_3$) for every mg of ammonium oxidized because of the hydrogen ions released by the oxidation reaction. Thus, there is a risk in low alkalinity waters that the pH could become too acidic and inhibit biochemical nitrification. Typical household wastewater nitrogen (organic and ammonium as N) concentrations range from 40 to as much as 70 mg/L or higher, which would require approximately 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 TN concentrations in the treated water are required.

Numerous commercially available proprietary systems with nitrogen removal processes have been developed and installed nationwide. Several documents have been compiled summarizing long-term full-scale performance monitoring data for installations of several such systems (Rich 2007; Harden, Chanton et al. 2010; Ursin and Roeder 2013; State of New Jersey Pinelands Commission 2015). Barnstable County, Massachusetts has a septic system database for tracking and compliance which includes effluent nitrogen data on innovative/alternative (I/A) septic systems installed within the County and summarizes performance in relation to the 19 mg/L effluent TN standard (Barnstable County Department of Health and Environment 2016). Anderson and Otis (2000) have previously summarized the various types of onsite wastewater treatment systems and provided an overview of the expected performance of many of the systems available.

2.1.3.1 Single Sludge Sequential BNR

A single sludge system carries out nitrification and denitrification in a single sludge reactor by alternating between aerobic and anaerobic environments with a single tank. These systems can be designed as a sequencing batch reactor (SBRs) or flow through reactor with sludge recycle as shown in Figure 2-6. 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 for OWTS applications is typically less than 50 percent nitrogen removal, depending on the configuration and cBOD availability, with these systems (Harden, Chanton et al. 2010; Ursin and Roeder 2013; State of New Jersey Pinelands Commission 2015; Barnstable County Department of Health and Environment 2016).

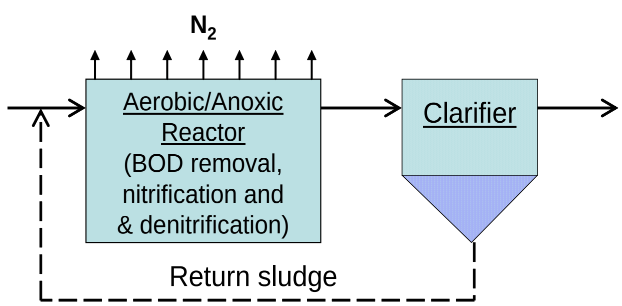


Figure 2-6: Single Sludge Sequential BNR

2.1.3.2 Single Sludge with Preanoxic Recycle BNR

Single sludge systems combine nitrification and denitrification 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 2-7). From this reactor, the wastewater flows to the aerobic reactor where the carbonaceous organics are further reduced and ammonium and most organic nitrogen are nitrified. As the nitrified effluent exits the aerobic reactor, it is split with usually a smaller fraction directed to the final discharge while the majority is directed back to the anoxic tank where the nitrate can be reduced to nitrogen gas using the incoming wastewater cBOD as the electron donor. Also, some of the alkalinity consumed by nitrification is recovered during denitrification thereby reducing total alkalinity requirements. However, TN 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 nitrogen removed by onsite systems utilizing this process typically ranges from approximately 40 to 75 percent (Harden, Chanton et al. 2010; Ursin and Roeder 2013; State of New Jersey Pinelands Commission 2015; Barnstable County Department of Health and Environment 2016).

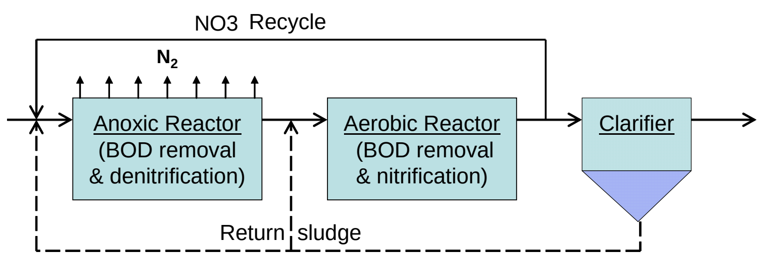


Figure 2-7: Single Sludge with Preanoxic Nitrified Effluent Recycle BNR

2.1.3.3 Two Sludge, Two-Stage BNR

The two sludge, two-stage process cultivates two separate bacteria populations; one for nitrification and the other for denitrification (Figure 2-8). This configuration allows nearly complete nitrogen removal because nitrate cannot by-pass denitrification as it can in the single sludge processes. Since most organic carbon is removed in the first stage aerobic reactor, this approach requires an electron 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 include: methanol, ethanol, acetate, and glycerol. For smaller systems where less operation attention is possible or desired, solid reactive media have been used such as lignocellulose and elemental sulfur.

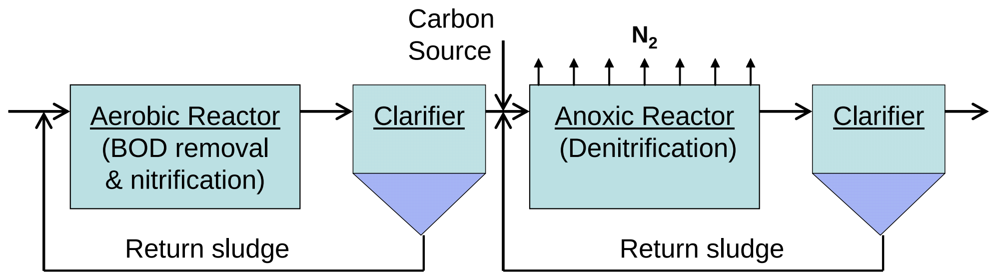


Figure 2-8: Two Sludge, Two-Stage BNR

2.1.4 Deammonification Process (Anammox)

An alternative approach for biological nitrogen removal is the use of the deammonification process (Figure 2-9) which is currently mostly used for high strength (ammonia) wastewaters such as the liquid stream of the centrifugation dewatering process "centrate" at municipal wastewater treatment plants as a sidestream treatment. The deammonification process requires conversion of approximately 50% of the influent ammonia into nitrite by AOB using nitritation, followed by the simultaneous removal of ammonia and nitrite by anammox bacteria. Anammox bacteria, short for anaerobic ammonia oxidizing bacteria, are the catalyst behind the deammonification process. Under anoxic conditions anammox bacteria have the ability to simultaneously reduce nitrite and ammonia to nitrogen gas (equation 2-6). Anammox is thought to be a significant factor in the conversion of nitrogen compounds to nitrogen gas in soils, wetlands, and marine, freshwater, and estuarine sediments (Kuenen 2008).

$NH_4^+ + 1.32\ NO_2^- + 0.066HCO_3^- + 0.13H^+ \rightarrow 1.02N_2 + 0.26NO_3- + 0.066CH_2O_{0.5}N_{0.15} + 2.03H_2O$ (2-6)

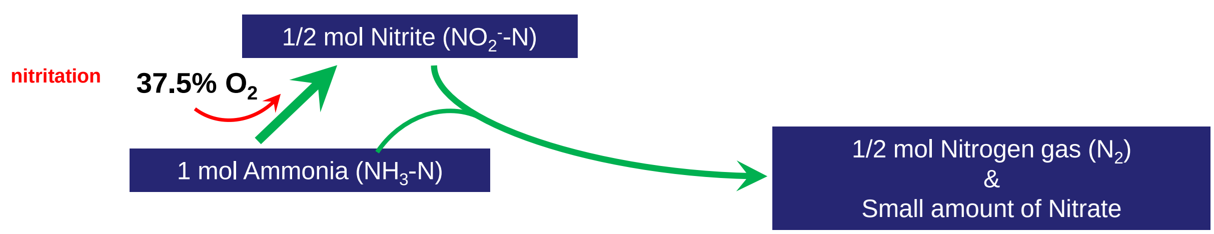
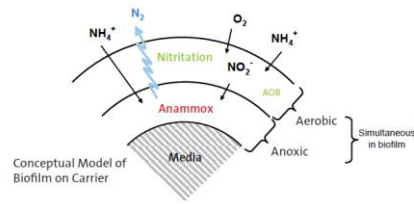


Figure 2-9: Deammonification Process

Using this process, a theoretical savings of 66.5% in aeration, 50% in alkalinity, and 100% in supplemental carbon are possible, resulting in a substantial savings in energy and chemicals. For deammonification processes to be successful, stable production of nitrite is needed. To accomplish this, deammonification processes repress the growth of NOB which are responsible for converting nitrite to nitrate in the nitrification process. Further, since anammox bacteria have extremely low growth rates, deammonification processes must provide sufficiently long solids retention times (SRT) for anammox bacteria growth. Due to the very slow growth rate of anammox bacteria, media is used to retain the microorganisms and increase mixed liquor concentrations in IFAS and MMBR systems. The



deammonification process has not yet been considered for development of a treatment unit for OWTS; however, the resultant energy and chemical savings associated with incorporating this treatment technology for OWTS and research directed toward the role and mechanisms of the specialized media could lead to lower cost, scalable installations. For decentralized or onsite use constraints to this process include operational oversight requirements, low temperature sensitivity, pH sensitivity, and slow growth rate of anammox bacteria. Typically, most deammonification processes are dependent on adjusting aeration through control of at least one parameter such as pH, conductivity or ammonia. This approach is illustrated within patent application 2016/0023932 whose inventor, Dr. Charles Bott, is considered a leading expert on deammonification and anammox process.

2.2 Physical/Chemical Processes

Physical/chemical (P/C) processes use non-biochemical approaches to wastewater nitrogen reduction. A fundamental difference from biological processes is that biological nitrification/denitrification converts the biodegradable organic nitrogen to ammonium prior to nitrification; P/C processes typically do not make this conversion, which can make reduction of TN to very low concentrations more difficult. Though P/C processes were initially equally acceptable compared to biological processes, they have found limited application for municipal applications because they have been found to be more expensive and more problematic when treating dilute wastestreams (USEPA 1993). P/C processes are not typically used for OWTS. Recently, electrochemical processes for wastewater treatment have been explored and will be discussed in a later Sections. P/C process options that might be appropriate for onsite wastewater treatment are shown in Figure 2-10.

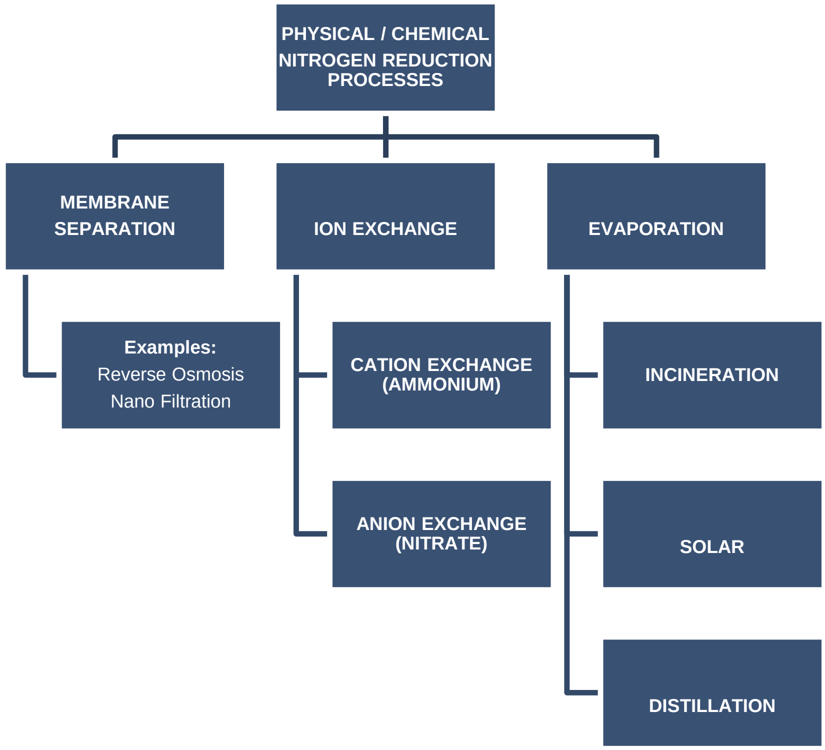


Figure 2-10: Onsite Treatment Technology Categories for Physical/Chemical Processes

There are several P/C options that are capable of reducing TN in wastewater. However, many are not practical for household applications including ammonia stripping and breakpoint chlorination. The more suitable P/C options for household/on-site use are 1) membrane separation, 2) ion exchange, and 3) evaporation. Membrane separation requires substantial and costly pretreatment, and therefore is most commonly used for drinking water treatment at the household level. It is becoming more popular in wastewater combined with biological treatment as membrane bioreactors. However, membrane bioreactors require activated sludge to remove/separate nitrogen. They do allow for a small footprint but require operation at very high mixed liquor concentrations.

Ion exchange also requires pre-treatment and commercial regeneration of the exchange resins, but can be potentially successful at high concentrations typically found in source separated streams (discussed in Section 2.4). Evaporation technologies can be effective in warm, dry climates, but require periodic removal and appropriate disposal of the residuals and typically require a large footprint. Solar evaporation and distillation are emerging options for households but are early in their development. The one OWTS area where P/C options are being considered and show potential is for urine treatment where urine source separation processes are used. From a research perspective, P/C methods could be investigated further in an academic setting and are further discussed in the urine source separation section of this document.

2.2.1.1 Membrane Processes

While membrane treatment is used for water and wastewater treatment, physical membrane separation processes have not been applied effectively for nitrogen removal in onsite wastewater. Membranes are a separation technology based on filtration through synthetic membranes. However, most are not capable of removing ammonia molecules from water. Reverse osmosis (RO) is one membrane process that is capable of ionic species removal including NO3 and is used in wastewater treatment, but has not been applied to onsite treatment. However, RO membranes are used for treatment of household drinking water.

Membrane bioreactors (MBR), also referred to as submerged membrane bioreactors, have gained widespread application in municipal treatment facilities and recently have been introduced to the onsite treatment market. MBRs are very common in decentralized WWTPs, mostly for housing developments that beneficially reuse the effluent for irrigation of landscaping etc. Ultrafiltration membranes are used in activated sludge processes as a substitute separation process in lieu of the final clarifier. The membranes retain the volatile suspended solids in the biological treatment vessel through filtration rather than sedimentation, which allows the process to maintain significantly higher biomass concentrations that facilitate both nitrification and denitrification, and also require a smaller footprint. Because the membranes themselves do not remove the nitrogen but rather support more effective biological denitrification, this type of process is reviewed under “Biological Nitrification/Denitrification Processes” (Section 2.1.3).

2.2.1.2 Ion Exchange

Ion exchange for removal of either NH_4 or NO_3 nitrogen from wastewater has been studied by several investigators. The natural zeolite clinoptilolite has been shown to have a high selectivity for ammonium with a total exchange capacity of approximately 2 meq/g. It can be regenerated with sodium chloride or an alkaline reagent such as sodium or calcium hydroxide. However, without prior treatment, the zeolite is easily fouled (University of Wisconsin 1978; Eckenfelder and Argaman 1991). Wu et al. (2008) found that the addition of powdered zeolite added to a contact stabilization activated sludge plant was effective in removing ammonium and during the anoxic stage was biologically regenerated. However, the powdered zeolite was continuously lost from the system. Removal of low concentrations ammonium typically found in municipal wastewater were not effective (Zhang, Wu et al. 2007). A more recent lab-scale study examining an anaerobic/ion exchange process found nearly complete (99.4%) ammonium removals and TN reductions exceeding 95%, but the ammonium capacity of the tested clinoptilolite, 11.3 mg N/g (< 1 meq/g), was lower than previous observations (Smith and Smith 2015). In any event, the added cost of the pretreatment would likely make ion exchange impractical for OWTS applications. However, some have approached the use of ion-exchange for capturing nitrogen in a variety of different configurations. For example, Patent application 2015/0239761 described in Section 4 illustrates an approach to use multi-chamber ion exchange zeolite bioreactors for an onsite system and which claims reuse of the spent media for land application as agricultural slow release fertilizer.

2.2.1.3 Electrolysis

Electrolysis (Figure 2-11) is an electro-chemical process that can be used to remove ionic compounds from solution. At least two inventors have developed electrolysis processes to remove ammonium (and/or nitrate) from wastewaters (Jeon, Kim et al. 2012; Spielman and Summers 2012) and Section 4 highlights other grants 8460520 B2 and patent application 2012/0160706 which claim electrolysis for wastewater treatment which could be applied to different aspects of onsite wastewater treatment Although electrolysis is a well-established process, application to onsite wastewater treatment appears to be largely theoretical, and no electrolysis systems have been marketed to date.

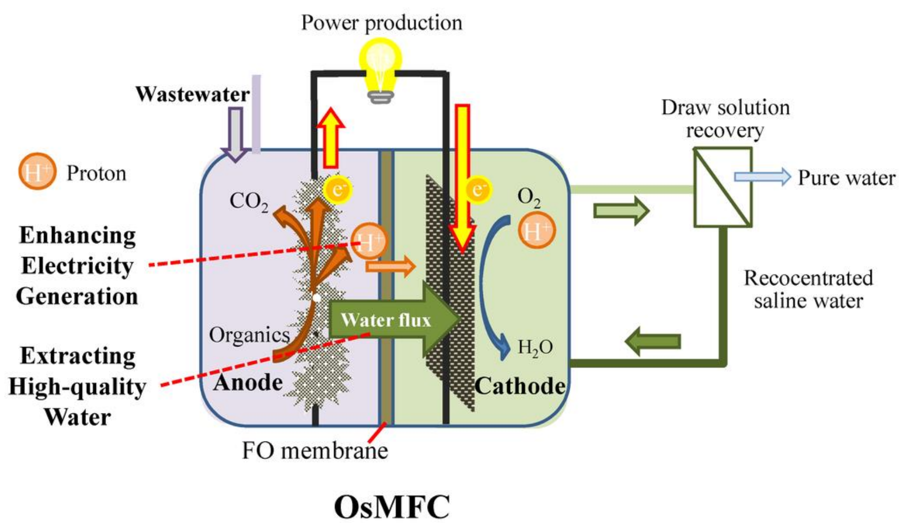


Figure 2-11: Conceptual Process Flow Diagram for Electrodialysis Extraction Process (From Lu et al. 2015)

2.2.1.4 Distillation

Distillation is another P/C process that has been considered for onsite wastewater treatment. Attempts have been made to develop an effective proprietary mechanical distillation unit, but no distillation technologies have been marketed to date. In addition, the disposal of the distillation residuals containing high levels of nitrogen and other contaminants have not been addressed.

2.3 Soil, Plant and Wetland Processes

Natural systems such as soil, plant and wetland systems are included as a separate classification because they utilize a combination of physical, chemical and biological processes that occur naturally in the soil and/or plant. Soil treatment systems are typically the last step in the process sequence of an OWTS for final treatment and dispersal of effluent. Natural biological processes can mimic both single sludge and two-sludge, two-stage processes depending on the soil conditions (Briggs, Roeder et al. 2007; Otis 2007).

Constructed wetlands are wastewater treatment systems consisting of shallow ponds or channels that are usually less than a meter deep; have been planted with aquatic plants; and rely upon natural microbial, biological, physical, and chemical processes to treat wastewater. They typically have impervious clay or synthetic liners, as well as engineered structures to control the flow direction, liquid detention time, and water level (Wu 2015b). Patent 8252182 in Section 4 illustrates a novel OWTS constructed wetland coupled with a mixture of layered media and plant species that was developed at the University of Central Florida.

Algae has also been proven to remove both nitrogen and phosphorous from both water and wastewater while providing the benefit of producing oxygen. Originally patented (4,333,263) in 1982 by Dr. Walter Adey, the Algal Turf Scrubber® (ATS) was developed based on natural algal mats over coral reefs. The algae mat from the ATS process can be continuously recovered, processed and utilized for biofuel production, digestion, feedstock, soil amendments, Omega 3s and other products. NYCDEP recently conducted a 3.2 acre ATS pilot at the NYC Rockaway WWTP pilot which demonstrated that 100 kg of wet algae can yield 1 L biobutanol fuel, 1 kg yields 0.3 L methane while providing 4% effluent Nitrogen removal. Algae is considered an emerging technology which could be considered more as a wastewater polishing process with sustainable, energy recovery features (May et al. 2015). Categories of technologies that are practical for onsite wastewater treatment are presented in Figure 2-12.

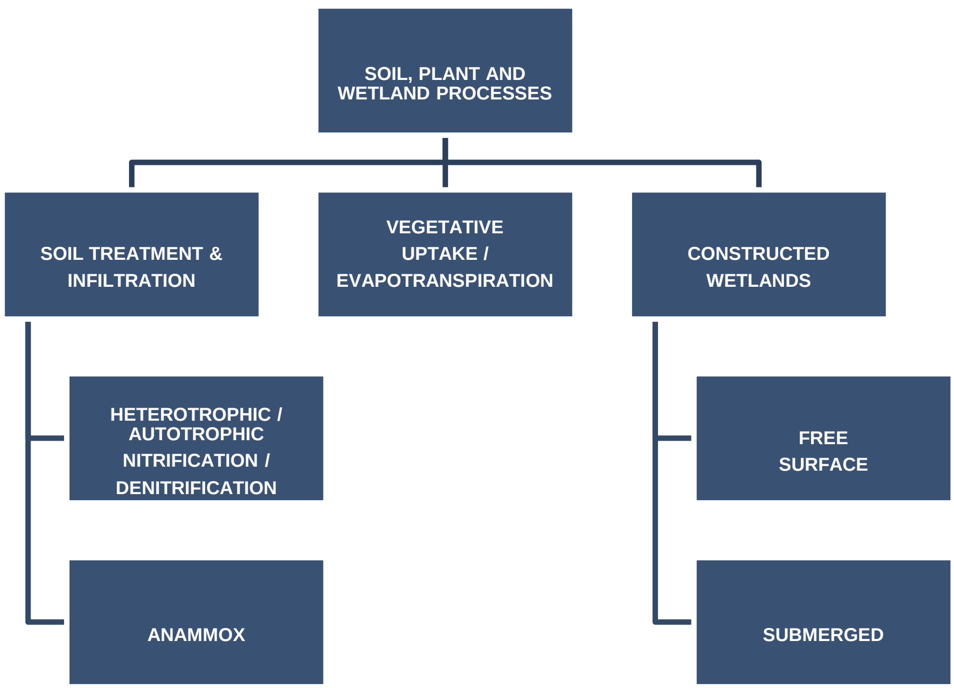


Figure 2-12: Categories of Natural Systems for Nitrogen Reduction

2.4 Source Separation

The source of the majority of nitrogen in household wastewater is the toilet, which accounts for 70 to 80 percent of the total daily discharge of nitrogen (University of Wisconsin 1978; Lowe, Rothe et al. 2006). Nitrogen from food wastes that are discharged through the kitchen sink or dishwasher account for an additional 15 percent. These sources can be segregated from the total household waste flows for separate treatment and handling. Source separation is an option gaining more attention with the availability of urine separating toilets. For common waste separation options, see Figure 2-13.

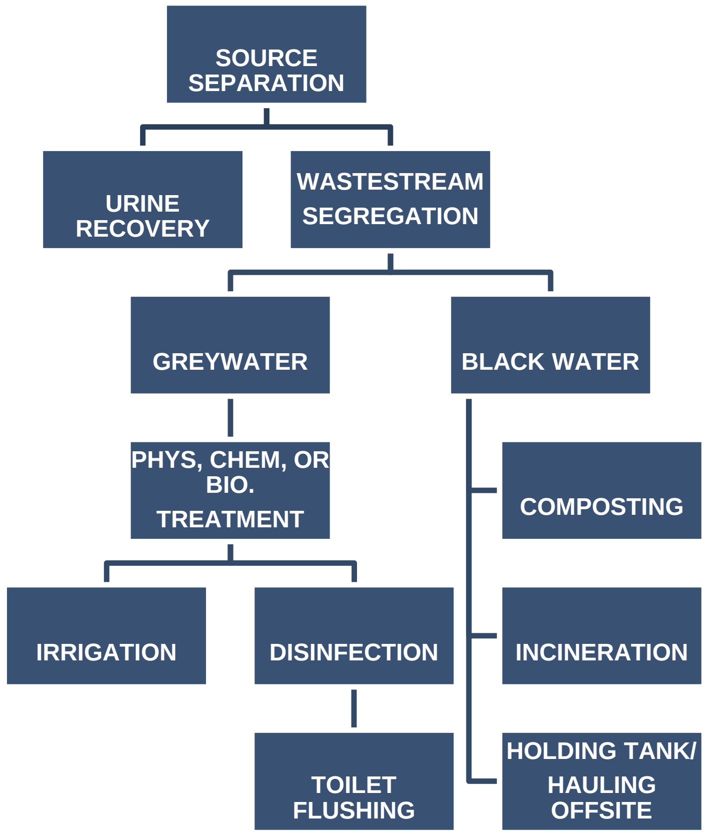


Figure 2-13: Nitrogen Source Separation Categories

2.5 Passive Nitrogen Reducing OWTS

Treatment systems can be either “passive” or “active”. There is no strict definition of passive versus active, however the Florida Department of Health has adopted a definition of passive as “a type of onsite wastewater treatment system that utilizes no mechanical components other than one effluent pump and uses a reactive media for denitrification”. Reactive media was defined as media that provides an electron donor for the denitrification process such as a carbon or other energy source.

Passive in this definition is also meant to imply consistent, reliable, low-energy, and low-maintenance. However, in actuality systems with such characteristics may be active or passive. Nevertheless, passive systems are generally preferred for onsite wastewater treatment because if well designed, they run largely on their own with less frequent need for inspection or servicing, as compared to active systems that include numerous mechanical parts and controls. By design, they should have a minimum of moving parts to avoid breakdowns typically using hydraulics of the influent water as the driving force through the system and natural air draft through media for oxygen supply. Onsite systems tend to be designed conservatively large because there are few operational remedial measures that can be taken if undersized.

This passive definition precludes many existing nitrogen reduction options primarily because of the requirement for no aeration equipment and use of reactive media. Of the systems currently in use, only biological two sludge, two-stage systems would qualify as passive treatment under this definition. US Patent 5318699 *Denitrification of Septic Tank Effluent* published in June 1994 and licensed to Nitrex™ is an early example of passive nitrogen reducing treatment systems. Cation exchange (NH_4), a physical/chemical process is another reactive medium process but to be effective, pre-filtration and treatment is necessary to prevent resin fouling, which may require additional mechanical components beyond one pump and would eliminate it as a passive system. In any event, the added cost of the pretreatment would likely make ion exchange impractical for household applications (Smith 2008). Most single sludge systems would be “passive” except for the requirement for reactive media and/or aeration equipment, but these systems have less ability to meet very low TN concentrations. Where the TN requirements are above 10 mg N/L, these systems could be acceptable options. Single sludge with preanoxic recycle systems also have the advantage that they recycle the alkalinity, which may be important in areas with low alkalinity in drinking water. While the definition of “passive” is followed in describing and comparing the different nitrogen reduction processes and technologies in this review, it is recommended not to focus exclusively on this criterion in evaluating nitrogen reduction strategies.

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 exceedances 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. The Water Environment Research Foundation has studied this issue because of challenges to its inclusion by municipal treatment plants (WERF 2008).

3. Literature Review of Existing Onsite Wastewater Technologies and Practices

3.1 Nitrogen Reducing OWTS

The following is a review of what are considered technically feasible nitrogen reduction technologies and practices suitable for single households, small multi-dwelling developments, and small commercial establishments.

3.1.1 Primary Treatment (Septic Tank)

A septic tank (Figure 3-1) is commonly used as the first treatment step in an OWTS. Its principal function is to remove, store, and digest settable and floatable suspended solids in the raw wastewater. These solids collect as sludge and scum within the tank where the organic nitrogen is degraded via hydrolysis, acidogenesis, acetogenesis and methanogenesis. During hydrolysis, the protein molecules are broken apart to release the organic nitrogen, much of which is converted to ammonium. Any nitrate in the influent is quickly denitrified by the heterotrophic denitrifiers. Consequently, the form of nitrogen in domestic septic tank effluent varies, but is approximately 70 percent ammonium and 30 percent organic nitrogen (University of Wisconsin 1978; Lowe, Rothe et al. 2006). Nitrate is typically negligible. As much as 10 to 15 percent of the influent nitrogen is retained in the tank within the sludge and scum (Otis 2007).

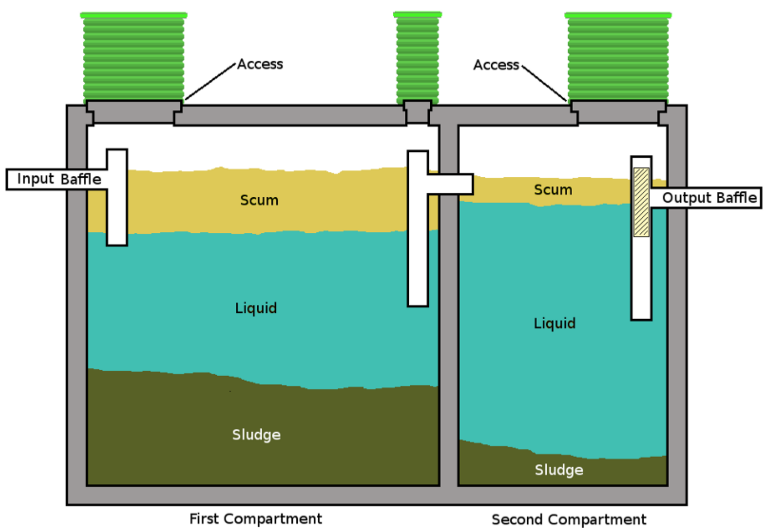


Figure 3-1: Primary Tank (Septic Tank) (Express Septic Service 2016)

In denitrification systems, the septic tank is often used as a carbon source for heterotrophic denitrification of nitrified wastewater returned from downstream nitrification processes. The nitrified wastewater is returned to the septic tank inlet to mix with the influent and septage in the tank. Up to 70 percent reduction of the TN in the wastewater can be achieved with recycle if relatively complete nitrification is achieved prior to 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.

3.1.2 Biological Processes

As discussed in Section 1.1.3, the biological nitrification/denitrification processes that are most practical and commonly used for onsite wastewater treatment are single sludge BNR and two sludge, two-stage BNR. The principal difference between the two is the source of the electron donor used by the denitrifying microorganisms. The single sludge systems use organic carbon that is available in the wastewater being treated; either microbial cell carbon and/or wastewater carbon. Two sludge, two-stage systems require external sources of organic carbon or chemical electron donors.

Management of wastewater carbon is critical to successful denitrification in OWTS. This is difficult in single sludge 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 OWTS where small and intermittent wastewater discharges into the treatment system can easily result in extended periods of aeration during low or no flow periods with the result that the organic carbon is oxidized before the denitrification step. Consequently, without careful carbon management, OWTS that use single sludge processes are less likely to achieve low TN effluent concentrations, particularly those using processes that rely on microbial cell carbon as the electron donor in denitrification. Table 3.1 summarizes TN removal results from OWTS using single sludge sequential BNR, single sludge with preanoxic nitrified effluent recycle BNR, and two sludge two-stage BNR, which shows the differences in treatment capability due to the source of the electron donor.

Table 3.1: Biological Processes and Typical Nitrogen Reduction Limits of OWTS[1,2,3,4,5]

Process	Single Sludge Sequential BNR	Single Sludge with Preanoxic Nitrified Effluent Recycle BNR	Two Sludge, Two-Stage BNR
Electron Donor	Organic carbon from bacterial cells	Organic carbon from influent wastewater	External electron donor (Organic carbon; Lignocellulose; Sulfur; Iron, Other)
Typical N Reductions	40 to 65%	45 to 75%	70 – 96%
Typical Technologies	• Extended aeration • Pulse aeration • Porous media biofilters • Sequencing batch reactors • Membrane bioreactor	• Extended aeration with recycle back to septic tank • Recirculating media biofilters with recycle back to septic tank • Moving bed bioreactor	• Nitrification followed by: • Heterotrophic suspended growth denite • Heterotrophic porous media fixed film denite • Autotrophic porous media fixed film denite

1 USEPA (2002)
2 Behrends, et al. (2007)
3 Abbeggen, et al. (2008); Sarioglu, et al. (2009)
4 Piluk and Peters (1994);
5 Rich (2007); Heufelder et al. (2008)

3.1.2.1 Single Sludge BNR

As discussed in Section 1.1.3, a single sludge BNR system carries out nitrification and denitrification in a single sludge reactor by alternating between aerobic and anoxic environments with a single tank and can incorporate recycle of nitrified effluent where the nitrate is reduced to nitrogen gas using the incoming wastewater cBOD as the electron donor. Many reactor configurations are possible incorporating suspended growth and fixed film biological treatment and a combination of both fixed film and suspended growth microbial communities (IFAS).

3.1.2.1.1 Suspended Growth (Activated Sludge) Reactors

Activated sludge processes are well developed and have proven capabilities to remove TN from wastewater to very low concentrations via various biological nitrification/denitrification configurations (USEPA 1993). Many manufacturers offer suspended growth treatment units for onsite use. Most were developed to provide better treatment than conventional OWTS alone, and in order to reduce clogging of the infiltrative surface in the soil treatment unit (aka drainfield) by removing BOD_5. Most of the manufactured units use the extended aeration process because of its simplicity and lower sludge production. Extended aeration is similar to conventional activated sludge and complete mix processes except the hydraulic and mean cell residence times are significantly longer than 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. SBRs are more complex in operation (Figure 3-2) but can be easily automated. This process uses two or more reactor tanks in which aeration, sedimentation and decanting occur in each reactor. This allows the treatment to occur in batches. A decanted reactor (active biomass is retained in the reactor after decanting) is filled. Once filled, it receives no more influent and is allowed to aerate and settle off and on over timed cycles. In the meantime, another reactor is filled. When the treatment period is complete, the supernatant is discharged. Seventy-four installations in New Jersey of one proprietary SBR system achieved a median effluent TN concentration of 11.9 mg/L, while another proprietary SBR system had a median concentration of 31.5 mg/L across sixty-two installations; following retrofits, the latter system's median effluent TN concentration was reduced to below 20 mg/L (State of New Jersey Pinelands Commission 2015).

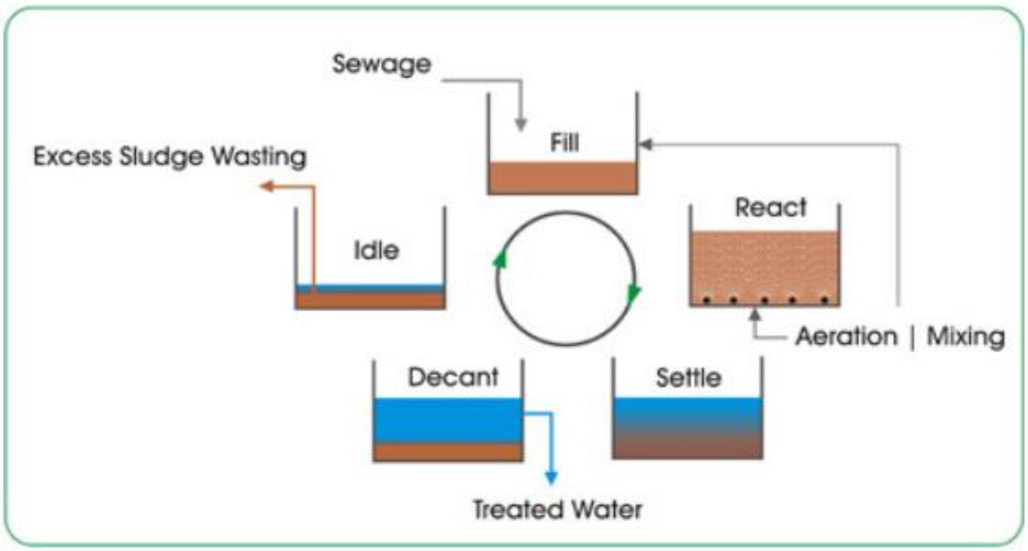


Figure 3-2: Sequencing Batch Reactor (SBR) Operating Principle (Rogers 2016)

Both SBRs and conventional activated sludge systems can achieve complete nitrification with extended aeration times. They are also used to denitrify, but denitrification by these processes requires careful management of the organic carbon during treatment. The high diurnal variation in wastewater quality, specifically the BOD_5/nitrate ratio, reduces TN removal efficiency in small community wastewater treatment plants (Raboni, Torretta et al. 2013), and this same obstacle applies to small-scale onsite wastewater treatment. Both extended aeration and SBR processes can incorporate recycling back to the septic tank to reduce TN, but TKN added during recycling will not be completely denitrified and will enter the discharge stream. If only microbial cell carbon is relied upon, addition of TKN is avoided, but without attention to carbon oxidation, sufficient carbon may not be available to support denitrification. Pulse or intermittent aeration can be an effective way to reduce the loss of organic carbon during nitrification (Ayres Associates 1998; Habermeyer and Sánchez 2005).

3.1.2.1.2 Fixed Film (Recirculating Porous Media Biofilters)

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

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

Aerobic biochemical transformations and physical filtration are the dominant treatment mechanisms within porous media biofilters, but chemical sorption also can be significant depending on the media selected. Oxygen is supplied by diffusion and mass flow of air behind wetting fronts through pore spaces in the media. Biofilms from the growth of microorganisms develop on the porous media and retain/accumulate the biology needed to carry out biological treatment. The microorganisms in the biofilm absorb soluble and colloidal waste materials in the wastewater as it percolates over the surfaces of the media. The absorbed materials are incorporated into new cell mass or degraded under aerobic conditions to carbon dioxide and water. The BOD is nearly completely removed if the wastewater retention times in the media are sufficiently long for the microorganisms to absorb the waste constituents. When looking at cross sections of a porous media biofilter, carbonaceous BOD is depleted first in the percolating wastewater, then nitrifying microorganisms thrive deeper in the biofilter. Under some conditions deep in the biofilter, oxygen may be depleted and denitrification could possibly occur.

3.1.2.1.2.1 Single Pass Operation

"Single pass" or "intermittent" filters alone are not typically used for nitrogen removal. This is because the wastewater passes through the filter media only once before being discharged for further treatment or dispersal. This generally results in good nitrification, but may not provide overall nitrogen reduction sufficient to meet nitrogen reduction standards. Low hydraulic loading rates and deeper media depths can increase nitrification in single pass biofilters.

3.1.2.1.2.2 Recirculation Operation

Recirculating biofilters recycle the nitrified filtrate back to a recirculation tank, which allows the wastewater to pass through the filter several times. The recirculation provides the needed wastewater residence times in the media to achieve greater nitrification. Recirculation provides more control of treatment process by adjustments that can be made to recirculation ratios and dosing frequencies. BOD and TSS removals are somewhat greater than those achieved by single pass filters and nitrification is nearly complete. The mixing of the return filtrate with fresh influent in the recirculation tank (the "recirculation" part) results in significant nitrogen removal via denitrification with wastewater carbon. Also, filtrate can be recycled back to the treatment head works to mix with undiluted raw wastewater or to an anoxic reactor between the septic tank and recirculation tank to increase nitrogen removal significantly. Summaries of media filter applications, design, operation and performance can be found elsewhere (Crites and Tchobanoglous 1998; Leverenz, Tchobanoglous et al. 2002; USEPA 2002; Jantrania and Gross 2006). Typical filter effluent concentrations treating domestic wastewater treatment are <10/10 mg/L for BOD and TSS, respectively, and approximately 50 percent TN removal. With recycle back to the septic tank, TN removal up to 75 percent has been shown (USEPA 2002).

Recirculating sand filters

Recirculating sand filters (RSF) are capable of achieving ammonia removals of 98 percent and TN removals of 40 to over 70 percent (Anderson, Siegrist et al. 1985; Piluk and Peters 1994; Kaintz and Snyder 2004; Loudon, Bounds et al. 2004; Richardson, Hanson et al. 2004). Effluent ammonia levels less than 3 mg/L are typical (USEPA 2002; Urynowicz, Boyle et al. 2007). Low temperatures typically inhibit nitrification but recirculating media biofilters appear to overcome the effects of low temperatures by increasing residence time in the biofilters through recirculation.

Textile biofilters

Recirculating textile biofilters were shown to achieve 44 to 47 percent TN reduction (Loomis, Dow et al. 2004) from septic tank effluent. In some cases, textile filters treating septic tank effluent have produced effluents with NH_3-N levels of less than 1 mg/L (Rich 2007). Textile filters also produce nitrified effluents (McCarthy, Monson Geerts et al. 2001; Wren, Siegrist et al. 2004; Rich 2007) and are often operated at higher hydraulic loading rates. More recent sampling of seventeen installations of one commercial recirculating textile filter system showed a median effluent TN concentration of 17.4 mg/L (Lancellotti, Loomis et al. 2015), and for single family home testing an overall reduction of TN over 60% (Maryland Department of the Environment 2016).

Zeolites

Media with significant ion exchange capacity may offer a method of superior removal of ammonia nitrogen in flowing systems. Natural zeolites provide excellent surfaces for biofilm attachments, and have relatively high porosities (Philip and Vasel 2006; Smith 2006; Zhang, Wu et al. 2007; Smith 2008; Smith, Otis et al. 2008; Hazen and Sawyer 2014; Hirst, Smith et al. 2014; Hirst 2015). 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 biofilter for onsite wastewater treatment removed 98.6 percent of ammonia and produced an effluent ammonia nitrogen concentration of 1 mg/L when operated at 6.1gal/ft^2-day (Philip and Vasel 2006). In an eight month bench scale study, a clinoptilolite a type of zeolite media biofilter treating septic tank effluent and operated at 2.8 gal/ft^2-day and 48 dose per day reduced ammonia by an average of 99.9 percent (Smith 2008; Smith, Otis et al. 2008). In an eighteen month pilot scale study, a clinoptilolite single pass media biofilter treating septic tank effluent and operated at 3 gal/ft^2-day and 24 dose per day reduced ammonia by an average of 99.9% (Hazen and Sawyer 2014; Hirst, Smith et al. 2014). A plenum-aerated biofilter with a hydraulic loading rate of 17 gal/ft^2-day was found to reduce ammonium by 99% over a study period of 200 days (Smith 2015). 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 Clay

Expanded mineral media may also have significant sorption potential for ammonium ions (Kietlinska and Renman 2005; Hinkle, Böhlke et al. 2008). Pilot scale single pass expanded clay biofilters reduced ammonia by 99.3 percent when operated on septic tank effluent at 3 gal/ft^2-day with dosing every hour (Hazen and Sawyer 2014; Hirst, Anderson et al. 2015). A recirculating column study resulted in total inorganic nitrogen removal of 11% (Polonite filter media) and 23% (Sorbulite filter media) corresponding to removal capacities of 1.32 mg/g and 6.68 mg/g, respectively (Nilsson, Lakshmanan et al. 2013). Nine filter beds were monitored in the Nordic countries which resulted in TN removal ranging from 32 to 66% (Jenssen, Krogstad et al. 2010).

In column studies with a variety of different media, including slag, polonite (a calcium silicate based mineral material), limestone, opoka, and sand, greater than 98 percent ammonia transformation to nitrate was achieved in all columns (Renman, Hylander et al. 2008).

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

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

Peat filters

Peat filters can achieve ammonia nitrogen removal efficiencies of 96 percent or greater from septic tank effluent, with effluent NH_3-N in some cases reduced to 1 mg/L or less (Lacasse, Bélanger et al. 2001; Lindbo and MacConnel 2001; Loomis, Dow et al. 2004; Patterson 2004; Rich 2007). Peat filters can also bind phosphorus (Kõiv, Vohla et al. 2009). TN reductions of 29 to 65 percent have been reported in modular single-pass and recirculating peat filters (Monson Geerts, McCarthy et al. 2001a; Barnstable County Department of Health and Environment 2016); 54 percent in peat filters using pressurized dosing (Patterson 2004).

3.1.2.1.3 Integrated Fixed-Film Activated Sludge (IFAS)

IFAS is a group of technologies that combine both fixed film and suspended growth microbial communities. The combination of these communities results in very stable treatment processes that achieve more reliable and consistent performance than other single sludge 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. Sampling from forty-one full-scale installations of one such system revealed a median effluent TN concentration of 11.2 mg/L (State of New Jersey Pinelands Commission 2015).

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. Limited sampling from ten installations of one commercial IMBR system has shown a median effluent TN concentration of 19.7 mg/L (State of New Jersey Pinelands Commission 2015). A novel lab-scale sulfur-oxidizing autotrophic denitrifying AnFB-MBR system which integrates membranes and an elemental sulfur-based autotrophic denitrifying anaerobic fluidized bed for treatment of nitrate-contaminated groundwater reported a high nitrate removal efficiency (100%) was maintained throughout the operation (Zhang, Zhang et al. 2015).

3.1.2.2 Two Sludge, Two-Stage BNR

Two sludge, two-stage BNR processes consist of two separate stages of treatment that segregate the nitrification (Stage 1) from denitrification (Stage 2) as depicted in Figure 3-3. This type of process eliminates the problem of nitrogen “leakage” in the discharge, which can occur in single sludge systems due to recycling. Consequently, a high degree of treatment is achieved more effectively. However, organic carbon that is used in single sludge processes does not reach the second anoxic stage requiring that an external donor be supplied to the second stage. Also alkalinity, which is recovered during denitrification, cannot be recycled to buffer the nitrification stage in a two sludge, two-stage system. If it is necessary to buffer the nitrification stage, an external source of alkalinity would be needed.

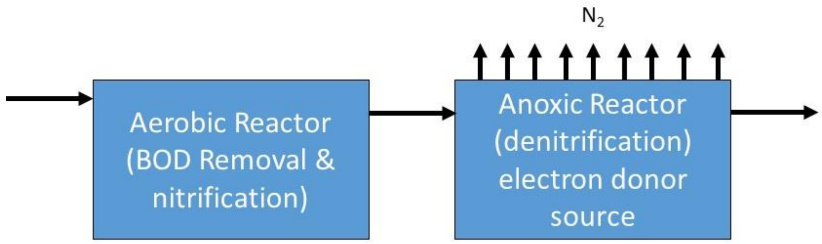


Figure 3-3: Two Sludge, Two-Stage BNR Example Process Flow Diagram

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

A two sludge, two-stage BNR system for household use that meets the "passive" definition consists of a septic tank, porous media biofilter, anoxic denitrification reactor followed by a soil treatment unit for final treatment and dispersal. An example of such a system is shown in Figure 3-4. Variations of this configuration are possible. In the septic tank, proteins are hydrolyzed releasing the organic nitrogen, which is reduced to ammonium. The Stage 1 porous media biofilter is an unsaturated aerobic media, which removes most of the BOD, nitrifies the ammonium and removes a portion of the TN. Where low TN concentrations are necessary, nitrified filtrate must be returned to the recirculation tank to be mixed with incoming septic tank effluent for denitrification using organic carbon from the wastewater. Recycling or recirculation of filtrate also increases nitrification since it may not be complete after a single pass through the filter. This requires a pump and a passive filtrate flow splitter that can divert the flow for recycling/recirculation or discharge to the next treatment Stage 2.

The advantage of using the pump here is four fold. First, it can dose the media filter based on time (rather than demand) and under pressure, which achieves uniform distribution over the filter surface both spatially and temporally significantly enhancing treatment performance. Second, it provides flow control (equalization) through the remainder of the system, which also enhances system performance. Third, it can be used to raise the hydraulic grade line though the remainder of the system so that flow through the system occurs by gravity, which eliminates the need for additional pumps. Fourth, its use can be proportional to flow, reducing the energy need compared to continuously aerating systems. The Stage 1 nitrified filtrate flows to the Stage 2 anoxic reactor, which is filled with saturated reactive media that provides the electron donors for denitrification to occur. After this reactor, the treated wastewater is discharged to a soil treatment unit (STU) for dispersal where additional treatment occurs and bacteria in the water are removed by processes in the soil as the water percolates to the groundwater.

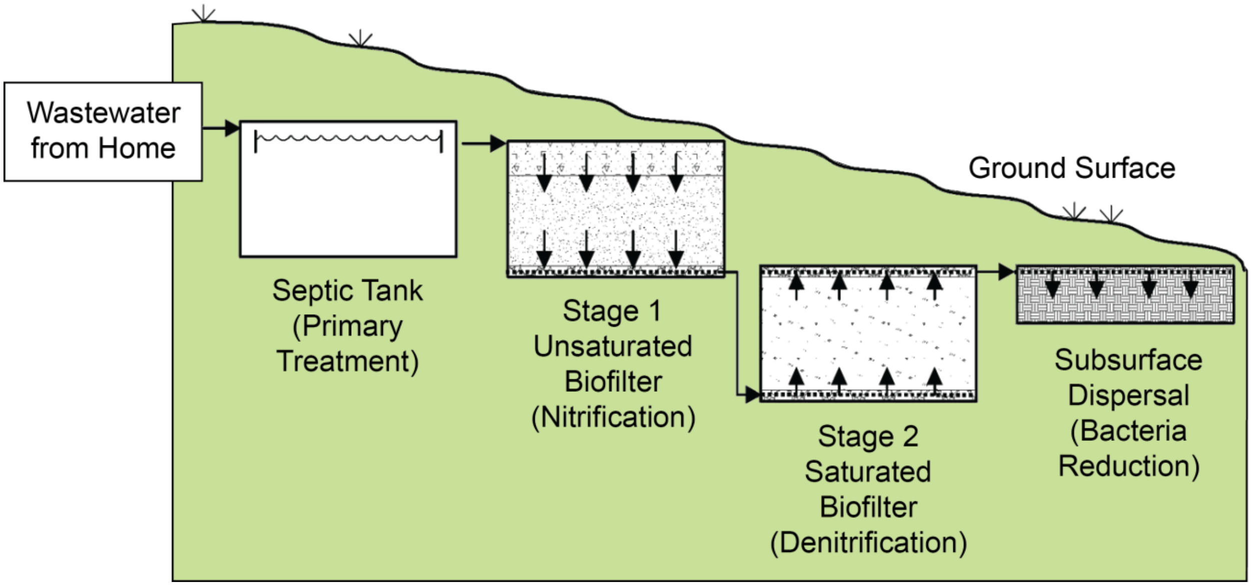


Figure 3-4: Two Sludge, Two-Stage BNR Example OWTS Process Flow Diagram

Pilot-scale results over a test period of 18 months indicated that a two sludge, two-stage biofiltration process was effective in greater than 95 percent TN removal from wastewater primary effluent (Hirst, Smith et al. 2014). Full scale demonstrations of three two sludge, two-stage passive nitrogen removal systems effluent average TN concentrations ranged from 1.8 to 7.4 mg/L with an average reduction in TN ranging from 89% to 98% over an 18 month period (Anderson and Hirst 2015; Hirst and Anderson 2015; Hirst, Anderson et al. 2015; Hazen and Sawyer 2015c).

Stage 1: Nitrification

Candidate media for the Stage 1 unsaturated media nitrification biofilters include sand, zeolite, expanded clay, expanded shales, tire crumb, and glass which should possess many of the desirable characteristics that have been discussed previously for porous media biofilters. 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. Pilot scale testing of clinoptilolite and expanded clay nitrification biofilters mean effluent ammonia nitrogen levels ranged from 0.01 to 0.5 mg/L, with many analyses at or below method detection limits (Hirst, Smith et al. 2014). Full scale demonstrations of three Stage 1 expanded clay biofilters effluent average ammonia concentrations ranged from 0.9 to 8.1 mg/L. The recirculating expanded clay biofilters effluent average ammonia concentrations were below 1 mg/L (Hirst, Anderson et al. 2015).

Stage 2: Denitrification

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

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

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; Robertson, Vogan et al. 2008; Cameron and Schipper 2010; Elgood, Robertson et al. 2010; Moorman, Parkin et al. 2010; Schipper, Cameron et al. 2010), sawdust (Kim, Seagren et al. 2003; Eljamal, Jinno et al. 2006; Greenan, Moorman et al. 2006; Jin, Li et al. 2006; van Driel, Robertson et al. 2006; Eljamal, Jinno et al. 2008; Cameron and Schipper 2010), cardboard (Healy, Ibrahim et al. 2012; Healy, Barrett et al. 2015), 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; Wang, Wang et al. 2015), poly(e-caprolactone) (Horiba, Khan et al. 2005), bacterial polyesters (Mergaert, Boley et al. 2001), and chitin (Robinson-Lora and Brennan 2009). The use of lignocellulose material has been generally recognized as a viable approach to engineered heterotrophic denitrification (Schipper, Robertson et al. 2010). Zhang et al. (2012) have suggested that biodegradable plastic (60% starch and 30% polypropylene) is a more suitable carbon source for denitrification due to its associated higher nitrate removal efficiency, longer service life, and lower nitrogen release relative to sawdust, straw wheat, and chitin.

The nitrate removal rate in denitrification biofilters incorporating lignocellulosic media are commonly reported as g N m^{-3} media day^{-1}. Cameron and Schipper (2012) tested nine different carbon substrates including softwood and hardwood which showed no statistical difference. Mean nitrate removal rates tested at two temperatures 14°C and 23.5°C were 3.0 and 4.9 g N m^{-3} day^{-1} for softwood and 3.3 and 4.4 g N m^{-3} day^{-1} for hardwood, respectively. Schmidt and Clark (2013) found similar results of 3.0 and 3.61 g N m^{-3} day^{-1} for softwood and hardwood, respectively. Both studies determined that temperature and carbon availability of the media are more important for controlling nitrate removal rate than hydraulic efficiency. Schipper, Cameron et al. (2010) summarized that nitrate removal rates supported by denitrification beds incorporating wood generally range from 2 to 10 g N m^{-3} day^{-1}. These values are within the range reported by other investigators (Robertson and Cherry 1995; Schipper and Vojvodic-

Vukovic 1998; Robertson, Blowes et al. 2000; Robertson, Vogan et al. 2008; Cameron and Schipper 2010; Moorman, Parkin et al. 2010; Robertson 2010; Schipper, Robertson et al. 2010; Long, Schipper et al. 2011; Schmidt and Clark 2012; Schmidt and Clark 2013).

A recent meta-analysis of denitrifying woodchip bioreactors determined (through categorical and linear assessments) significant nitrate removal effects with bed temperature and yielded a Q_{10} of 2.15 (i.e., the factor by which the removal rate increases for each 10˚C increase) which was similar to that reported in other studies (Addy 2016).

In-tank cellulosic-based systems have produced average TN removals of 88 to 96 percent from septic tank effluent, with average effluent NO3-N concentrations of 2 to 5.4 mg/L (WDOH 2005; Rich 2007). An upflow-anaerobic filter filled with coconut shells with influent being a combination of raw wastewater and nitrified effluent from an intermittent sand filter resulted in a nitrate reduction of 98% (da Silva, Tonetti et al. 2015).

In-ground pilot-scale studies in Washington State of vegetated denitrifying woodchip beds achieved nitrate removals of 39 to 98 percent with a correlation to wastewater temperature (Jones 2015). A full-scale demonstration of an in-ground system design in Florida consisting of a vertically stacked media arrangement, with the Stage 1 sand nitrification biofilter directly above the Stage 2 wood chip biofilter, underlain by an impermeable liner produced mean effluent TN of 6.5 mg N/L representing a 90% reduction from the STE concentration (Anderson and Hirst 2015; Hazen and Sawyer 2015c).

Autotrophic Denitrification

The autotrophic denitrification systems that have received the most attention are elemental sulfur-based media filters, which are under development with full-scale demonstrations. 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; Nisola, Redillas et al. 2011). 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 biofilter containing elemental sulfur and oyster shell at a 3:1 ratio was operated for 11 months at the Massachusetts Alternative Septic System Test Center (Brighton 2007). The filter received nitrified effluent from an aerobic fixed film treatment system that received septic tank effluent. The sulfur/oyster shell biofilter removed 82 percent of influent TN, while the overall aerobic/sulfur treatment train removed 89.5 percent TN from the septic tank effluent. A 22.5 gallon upflow column packed with sulfur/limestone at a 3:1 volume ratio treated a simulated groundwater at 0.9 to 1.8 gal/ft^2-day surface loading rate and removed greater than 95 percent of nitrate that was at 60 mg/L NO_x-N in the influent

(Moon, Shin et al. 2008). A laboratory sulfur/oyster shell column was operated at an empty bed contact time of 0.33 to 0.67 days and removed 80 percent of influent nitrate (Sengupta and Ergas 2006). Three saturated denitrification biofilters containing sulfur and oyster shell media were operated for eight months on septic tank effluent that was pretreated with unsaturated media filters that provided ammonification, nitrification, and carbonaceous biochemical oxygen demand reduction (Smith 2008; Smith, Otis et al. 2008). Average NOx reductions were 99.9, 99.9 and 88.9 percent respectively for treatment of effluent from unsaturated biofilters containing clinoptilolite, expanded clay, and granular tire chip media, respectively. Corresponding average effluent NOx-N were 0.03, 0.031 and 4.3 mg/L. These denitrification filters operated at hydraulic loading rates of 4.9 gal/ft^2-day and at average NOx-N loadings of 0.003 to 0.005 lb/ft^2-day, which are similar to loading rates applied to acetic acid amended sand denitrification filters that achieved 94 to 99 percent NOx reduction (Aslan and Cakici 2007).

In another study, four saturated denitrification biofilters containing sulfur mixed with either oyster shell or limerock media for alkalinity buffering were operated for eighteen months on septic tank effluent that was pretreated with unsaturated nitrification media biofilters (Hazen and Sawyer 2014; Hirst, Anderson et al. 2015). Average NOx reductions were greater than 99.5% in the four sulfur upflow packed bed denitrification biofilters. In another recent study, a tire-sulfur upflow packed bed bioreactor column study resulted in NO3 removal efficiencies under various operating conditions of steady state (90%), variable flow (89%) and variable concentration (94%). The scrap tires were determined to have an adsorption capacity of 0.66 g NO3-N per kg of scrap tires (Krayzelova, Lynn et al. 2014). A pilot-scale column study with zeolite media installed above soil, sawdust and iron scraps resulted in TN removal of 61.5% (Luo, Yang et al. 2014).

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

Alkaline Material

Typically active treatment systems incorporate a constant input of an alkaline material like lime, caustic soda, soda ash, etc. to neutralize the acidity associated with the nitrification process and autotrophic denitrification process. As previously discussed, passive biofilters have employed limestone or oyster shell as a solid phase alkalinity source to buffer the alkalinity consumption. The use of solid phase alkalinity sources obviates the need for careful dosing control that would pertain for liquid alkalinity sources. Crustacean shells (such as crab and shrimp) are composed of a complex solid matrix of chitin, protein, and CaCO3 and potentially provide a slow-release source of C and alkalinity in one substrate (Robinson and Brennan, 2009; Robinson and Brennan, 2010). However, previous testing using chitin for denitrification indicated that effluent nitrogen was still approximately 40% of the influent nitrogen, mainly due to the protein content of chitin and associated ammonia release resulting from protein degradation.

3.1.3 Soil, Plant and Wetland Processes

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

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

3.1.3.1 Soil Treatment Unit Infiltration

Biological nitrification in soils below STUs readily occurs where the requisite conditions exist, which include unsaturated, aerobic soils with adequate permeability. Complete nitrification generally occurs in the first 30 cm depth if these conditions prevail. The capacity of the soil to denitrify varies depending on the specific environmental conditions at the particular site and the design and operation of the STU. Numerous investigations into the fate of nitrogen below STUs have been undertaken. However, the results are quite variable even for sites that appear similar. Gold and Sims (2000) point out the dynamic and open nature of STU designs that create uncertainties with in-situ studies of the fate of nitrogen in soil. The effects of dispersion, dilution, special variability in soil properties, wastewater infiltration rates, inability to identify a plume, uncertainty of whether the upstream and downstream monitoring locations are in the same flow path, and temperature impacts are a few of the problems that challenge the in-situ studies. As a result, even when small differences in concentrations are observed, the spatial and temporal variability can result in large changes in estimates of the mass loss of nitrogen.

3.1.3.1.1 STU Hydraulic Loading

In a study investigating the effects of effluent type, effluent loading rate, dosing interval, and temperature on denitrification under STU, Degen et al. (1991) and Stolt and 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 treatment unit infiltration (Table 3.2).

Table 3.2: TN Removals by Soil Infiltration below STU

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

[1] Ritter and Eastburn (1988)
[2] Wert and Paeth (1985)
[3] Brown and Thomas (1978)
[4] Stewart and Reneau (1988)
[5] Converse, et al. (1994)
[6] Harkin, Duffy et al. (1979)

The more significant environmental factors that determine whether nitrogen removal occurs and to what extent include the soil's texture, structure, and mineralogy, soil drainage and wetness, depth to a saturated zone and the degree to which it fluctuates, and amount of available organic carbon present. OWTS design and operation factors include the species of nitrogen discharged to the STU 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.

Simple-to-use tools developed by the Colorado School of Mines (CSM) offer an easy user interface, but incorporate complex and robust evaluation of treatment scenarios and operating conditions. The outputs from HYDRUS-2D simulations (Hazen and Sawyer 2015a) are an example of the simplest-to-use tools providing the user a visual representation of subsurface behavior in the unsaturated zone (also referred to as the vadose zone) for selected conditions. The treatment information provided by these simulation outputs (graphical and tabular) is based on data generated by numerical models that incorporate complex and robust treatment and operating conditions. Because the choices for representative OWTS conditions are limited, the user must decide how/if their OWTS system fits within the limited treatment estimations displayed by the graphics. The approach used was based on a modified factorial design to highlight treatment performance as effected by soil texture, depth to the water table, distribution configuration, and effluent quality which demonstrates conditions that achieve up to a predicted 100% TN removal (Hazen and Sawyer 2015a). Figure 3-5 displays the output generated for a trench system with equal distribution in a less permeable sand with water table at 2 ft below the infiltrative surface, which estimated a 27% TN reduction.

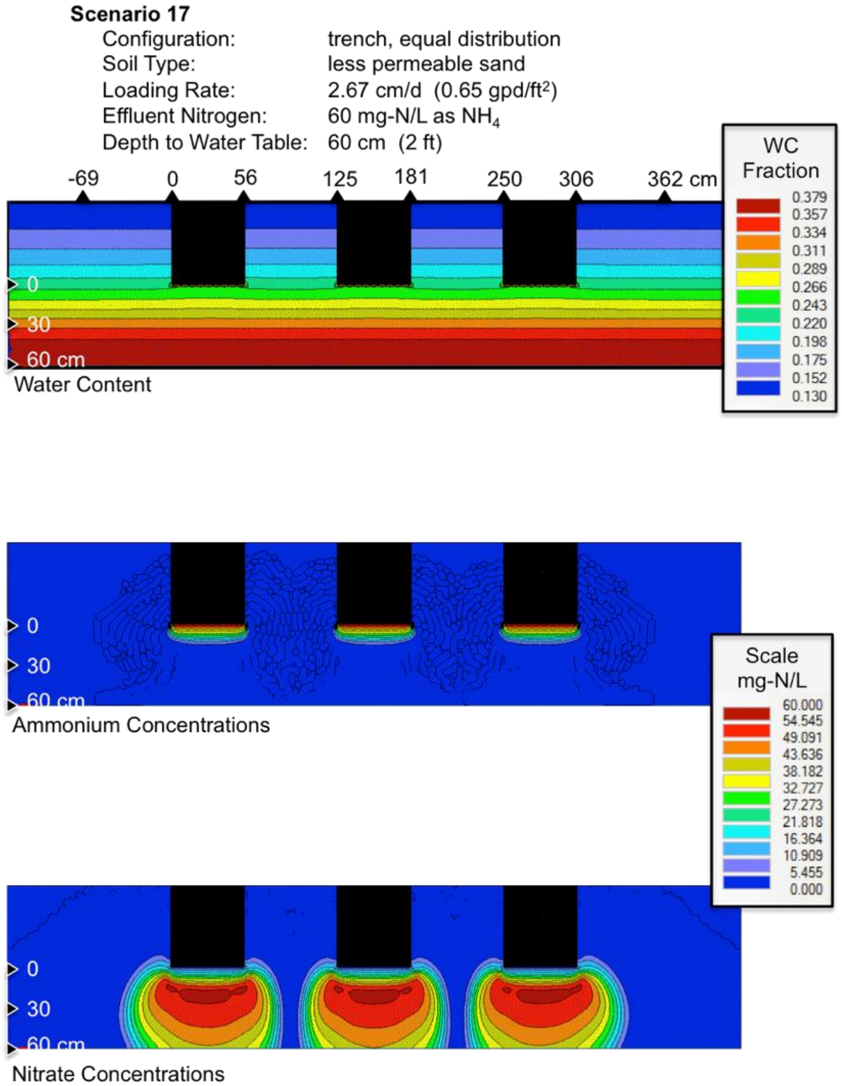


Figure 3-5: Simple Soil Tools, Example HYDRUS-2D Output (Hazen and Sawyer 2015a)

Alternatively, the impacts of these factors can be simulated with a simple to use complex model such as STUMOD-FL-HPS applied to a specific site which calculates the nitrogen species concentrations and the fraction of TN reaching the aquifer or a specified soil depth (Geza, Lowe et al. 2014; Geza, Lowe et al. 2014; Hazen and Sawyer 2015a). Default values are populated into the STUMOD-FL-HPS graphic user interface (Figure 3-6). However, user specified inputs can be added instead of default parameters allowing model calibration/corroboration to site specific data.

The output is simulated steady state performance (i.e., constituent concentration) at the center under the point of effluent application with down gradient transport through the saturated zone. Model outputs provide insight into the behavior of soil treatment, groundwater fate and transport, and quantitative estimations of nitrogen removal as affected by a range of conditions (Geza, Lowe et al. 2014; Geza, Lowe et al. 2014; Hazen and Sawyer 2015b).

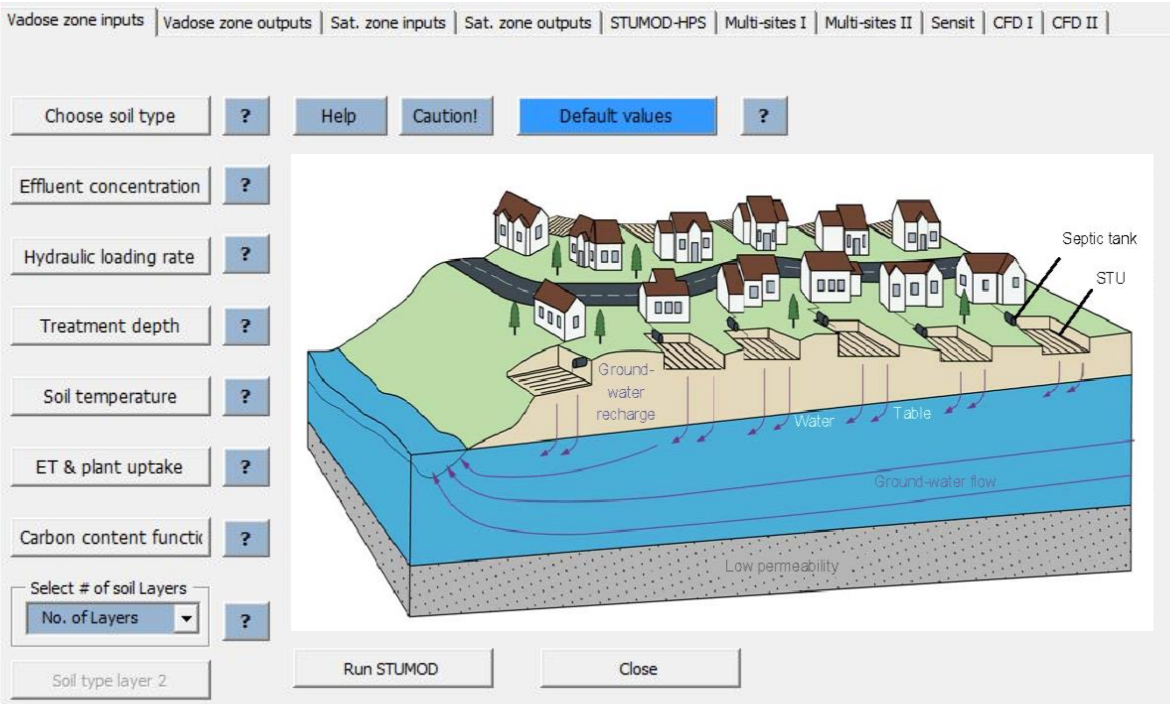


Figure 3-6: STUMOD-FL-HPS Graphical User Interface Showing Module Tabs (Hazen and Sawyer 2015b)

Radcliffe and Bradshaw (2014) modeled the effects of hydraulic loading rates for various soil textures in regards to nitrogen treatment. Nitrogen treatment varied widely among soils with denitrification losses ranging from 1% in the Group-I sand to 75% in the Group-IV sandy clay, due to water content limits on denitrification. Leaching losses were inversely related to denitrification losses, ranging from 97% in the sand to 27% in the sandy clay. Plant uptake and soil storage accounted for 5% or less of the N losses (Radcliffe and Bradshaw 2014).

3.1.3.1.2 STU Denitrification

Heterotrophic bacterial denitrification is often limited by the availability of sufficient quantities of organic matter (Burford and Bremner 1975; Gambrell, Gilliam et al. 1975; Christensen, Simkins et al. 1990; Bradley, Fernandez et al. 1992). Sources of organic matter in soil are either natural, which is continuously replenished in the soil from the decay of vegetative materials, or supplied by the wastewater itself. Research was conducted at the CSM to evaluate denitrification in soil treatment units and to what extent the N species in the effluent affects the potential and expressed denitrification rates. Four sand columns were dosed twice daily to yield a hydraulic loading rate of 2 cm/d with two columns receiving septic tank effluent and two columns receiving nitrified intermittent sand filter effluent. The highest recorded levels of representative denitrification rates, potential denitrification rates and denitrification genes *nirS*, *nirK*, and *nozZ* were all documented at a depth of 0-1 cm below the infiltrative surface of a column receiving septic tank effluent (Farrell, Siegrist et al. 2014).

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

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

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

The type of infiltration system used for the STU 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 typical practice includes the removal of the O and A horizons, which removes most of the available organic carbon. Also, "digouts", which are systems on sites where a restrictive horizon in the soil profile is removed, can result in reducing a particular soil's nitrogen removal potential because quite often the restrictive horizon removed is the illuvial accumulation of organic matter in the 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.

3.1.3.1.3 STU Effluent Dosing

Modifying the method by which wastewater is applied to the STU has been shown to enhance nitrogen removal in STU infiltration systems. By dosing septic tank effluent on timed cycles into the STU, alternating aerobic and anoxic conditions are created in the biomat and upper layer of the STU's soil infiltrative surface. With each dose the infiltrative surface becomes saturated during which time the soil can become anoxic due to the depletion of oxygen created by facultative heterotrophic bacteria degrading the organic matter. With the creation of anoxic conditions, nitrification of the ammonium ceases and the ammonium ion, which is positively charged, is adsorbed onto the negatively charged soil particles. As the soil drains and reaerates, the ammonium is nitrified but is not able to percolate downward because the soil has drained and is no longer saturated. However, the next dose adds fresh organic matter, which causes anoxic conditions to return creating the necessary conditions to enable the heterotrophic bacteria to denitrify the nitrate using the fresh septic tank effluent carbon as an electron donor. This intermittent dosing of septic tank effluent has been shown by several studies to reduce the TN applied. A study of various soil types using subsurface drip irrigation system for effluent application measured nitrogen removal rates in the range of 63 to 95 percent (Beggs, Hills et al. 2011). Another pilot scale study in Florida sand resulted in 60 percent TN removal in a mounded drip irrigation system (De and Toor 2015).

3.1.3.1.4 Intermittent Aeration

A patented method of rejuvenating ponded conventional septic tank STUs using forced air also was found to enhance TN removal (Potts 2004; Amador 2007; Amador 2008; Amador, Potts et al. 2010). In this method air is blown into the STU every 2 hours for 30 minutes. At traditional hydraulic loadings of septic tank effluent, 10 to 50 percent of the TN was found to be lost in the soil below the STU. When the hydraulic loading was increased, the TN reduction was increased up to 70 percent. The reason postulated for the increase was the increased organic carbon loading that prolonged the anoxic conditions favorable to biological denitrification. This method of operation was suggested to be similar to a sequencing batch reactor, which according to the investigators, would need regular attention if it were to be optimized for nitrogen removal.

3.1.3.2 Soil Treatment Unit Modification for Nitrification/Denitrification

Modifications to conventional STUs can entail the addition of a reactive media that supports denitrification through the release of carbon or electron donor. Wastewater (septic tank effluent) would initially pass through an unsaturated layer or zone (of sand or other porous media for example), where nitrification occurs. Following passage through the unsaturated zone, the wastewater would pass through a denitrification layer or zone which is either mixed with a media with high water retention capacity (creating a permeable layer) or underlain with an impermeable liner to promote saturated conditions (impermeable liner). Denitrification media carbon substrates (usually wood chips or sawdust) or inorganic electron donors (elemental sulfur) could be placed as an underlayment beneath the unsaturated soil, or as a subdivided treatment zone within a soil treatment unit (e.g. drainfield) through which nitrified effluent from the aerobic zone must pass (Robertson and Cherry 1995; Robertson, Blowes et al. 2000).

Typically these denitrification layers have a high water retention capacity to keep the media near saturation so that anoxic conditions are created as the septic tank effluent percolates through the permeable reactive layer. Nitrogen reductions of 60 to 100 percent were achieved in four field trials. The Massachusetts Alternative Septic System Test Center is currently performing full scale trials incorporating a sawdust-sand mixture integrated into a STU system without an impermeable liner which resulted in preliminary results of TN below 10 mg/L (G. Heufelder, personal communication, December 2015). A modified STU design using a sulfur/limestone layer beneath a sand layer provided greater than 95 percent TN removal in laboratory scale columns receiving primary effluent from a municipal wastewater treatment plant (Shan and Zhang 1998). Nitrification occurred in the upper sand layer, and the lower denitrification layer was not maintained in a saturated condition. A synthetic domestic wastewater laboratory-scale experiment with a sulfur/limestone mixture layer underlying a soil layer resulted in an overall nitrate and ammonia removal efficiency of 85% and 95%, respectively (Kong, Feng et al. 2014).

Chang, Wanielista et al. (2009) reported initial results for septic tank effluent treatment using a lined STU that contained a layer of lignocellulosic-based electron donor media underneath a layer of sand. The systems were operated at a surface loading rate of ca 0.5 gal/ft^2-day, with an influent TN of 46.3 mg/L. Ammonia removals were 85 to 90 percent in the two monitoring samples, while the corresponding TN removals were 60 and 85 percent. A prototype and full scale in-ground vertically stacked media arrangement, with a sand layer overlying a wood/sand mixture underlain by an impermeable liner and underdrain connected to a sulfur/oyster shell denitrification upflow biofilter, were each monitored over an 18 month period, receiving STE with an average TN concentration of 65.4 mg N/L for the prototype system and 50.5 mg N/L for the full scale system. The average TN concentration of the treated effluent prior to subsurface dispersal was 3.5 mg N/L for the prototype system and 1.9 mg N/L for the full scale system, representing a 95% and 96% reduction in nitrogen concentration, respectively (Anderson and Hirst 2015; Hazen and Sawyer 2015c).

A wood based system using a mixture of sand, wood chips, and tire crumb (85/11/4 percent by mass), was examined in bench scale columns to simulate treatment that would occur in a separate reactive media treatment zone established within a STU (Shah 2007). In this system, septic tank effluent would first pass through an unsaturated sand layer, and then through the treatment zone containing the reactive media. Laboratory column experiments with septic tank effluent supplied at a hydraulic residence time of 24 hours resulted in 98 percent TN removal. Average effluent ammonia and nitrate nitrogen concentrations were 4.4 and 0.05 mg/L, respectively (Shah 2007). A wood based pilot-scale system using a mixture of approximately 68% fine sand, 25% tire crumbs, 7% sawdust by volume resulted in an overall TN removal greater than 70 percent (Chang, Wanielista et al. 2010). Biochar addition to a woodchip bioreactor resulted in average NO3 removal of 86% and 97% compared with 13% and 75% in the woodchip control (Bock, Smith et al. 2015).

A study which incorporated peat below a subsurface wastewater infiltration system resulted in a TN removal efficiency as high as 92.67% (Chen, Cui et al. 2014). A study which incorporated coconut husk and basalt sediment to carbonate sand columns failed to show an effect on N removal relative to the carbonate sand control; however additional removal (~7%) occurred with the addition of a biochar filter (Tait, Shepherd et al. 2015).

Issues of concern for modified STUs with reactive denitrification media 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 STU, so the life of the reactive media in the denitrification zone would need to be at least as long as the other STU components. However, Robertson and Vogan (2008) report that after 15 years of use, a barrier consisting of a mixture of sawdust and sand was still achieving denitrification of septic tank effluent. Results from a study that evaluated woodchip media of varying age supported field experience indicating that woodchips loose about 50% of their reactivity in their first year of operation but for many years later rates remain relatively stable (Robertson 2010).

3.1.3.3 Constructed Wetlands

Subsurface flow constructed wetlands (Figure 3-7) are a system that has been used for single family and commercial applications. This system consists of a submerged rock bed that may be planted with wetland vegetation. Initially claimed to remove nitrogen from septic tank effluent, studies have shown that wetland plant roots do not supply enough excess oxygen to nitrify ammonium in septic tank effluent (McIntyre and Riha 1991; Burgan and Sievers 1994; Huang, Reneau et al. 1994; Johns, Lesikar et al. 1998; USEPA 2002; Behrends, Bailey et al. 2007; Kavanagh and Keller 2007; Austin and Nivala 2009). Nitrification seldom exceeds 50 percent, which limits denitrification. However, denitrification can reduce nearly all the nitrate that is available if adequate electron donors are present (Haunschild 2009; Leverenz, Haunschild et al. 2010).

Free water surface wetlands (reed beds) closely resemble natural wetlands in appearance with aquatic plants that are rooted in a soil layer on the bottom of the wetland. Water flows through the leaves and stems of the plants. Surface wetlands are typically used as a tertiary process in large wastewater treatment installations, and are mainly used for polishing secondary effluent. Surface water wetlands have the potential for vector attraction and public health concerns because of visible standing water. Therefore, subsurface flow wetlands are the recommended approach for decentralized wastewater nitrogen treatment.

Providing recirculating gravel filters, vertical wetlands, and/or or forced bed aeration to pre-nitrify the effluent has been successful in increasing TN reductions in subsurface vegetated beds up to nearly 90 percent (Askew, Hines et al. 1994; White 1995; Kantawanichkul, Neamkam et al. 2001; Van Oirschot, Wallace et al. 2014; Wu, Fan et al. 2015). A two-year study of pilot hybrid systems combining vertical flow, recirculation, and/or carbonaceous bioreactors achieved TN removals of 58 – 95% (Tanner, Sukias et al. 2012). Nitrogen removal in constructed wetlands is impacted by water temperature and dissolved oxygen (DO) concentration and can vary seasonally (Chang, Wu et al. 2013). Although constructed wetlands are potentially capable of high levels of nitrogen removal, a chief drawback of such systems is their substantial footprint. Austin and Navala (2009) examined energy and areal requirements for several types of constructed wetlands, finding there was generally a tradeoff between the two quantities. Anammox may be an alternative pathway for removing nitrogen in wetlands without the need for denitrification. Several alternative biochemical pathways may be involved, but development work is needed to optimize wetland design to successfully apply this process (Wallace and Austin 2008). Design guidelines may be found in USEPA's manual, *Constructed Wetlands Treatment of Municipal Wastewaters"* (2002).

For some applications, especially where inexpensive land is available, constructed wetlands are feasible because they are relatively inexpensive to construct and maintain, offer stable performance, provide a natural appearance, and potentially have some ecological benefits.

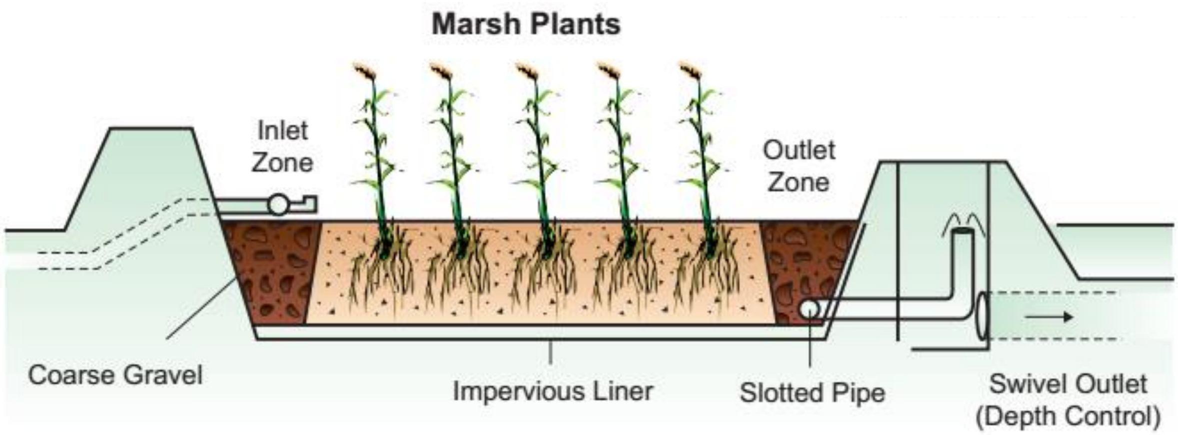


Figure 3-7: Typical Configuration of a sub-surface wetland system (Kadlec and Knight 1996)

3.1.3.4 Evapotranspiration and Vegetative Uptake

Lined evapotranspiration beds and vegetative uptake are two other methods that have been promoted for nitrogen removal. Both rely on plants to either transpire the water and uptake nitrogen for incorporation into the plant materials. However, the loss of water through evapotranspiration leaves a nutrient and salts rich liquid that must be removed periodically to prevent toxic conditions for the plants. Also the plants must be continually harvested to remove the nutrients taken up from the system. Studies have found that nitrogen removal is achieved by these systems but that other systems perform as well or better in removing nitrogen from the wastewater (Atkins and Christensen 2001; Barton, Schipper et al. 2005; Taylor 2006). While promoted heavily in the 1970's and early 1980's as an option for areas with slowly permeable soils or shallow water tables, evapotranspiration beds are infrequently used and seem to have been replaced by constructed wetlands. However, in southwestern states of the US they are primarily employed to reduce the hydraulic load on the STU (Rainwater, Jackson et al. 2005). Four pilot land treatment systems planted with different plant species indicated that nitrates accumulated in the soil profile and was dependent on plant species and may imply stimulated denitrification rates induced by rhizospheres of reeds (Tzanakakis, Paranychianakis et al. 2011). However vegetative uptake/evapotranspiration is not a feasible approach for the Suffolk County area because of high precipitation and low ET (Sanford and Selnick 2013).

3.1.4 Source Separation

Source separation involves the separate collection and treatment of wastewater streams to better target specific contaminants and/or resources for removal and/or recovery. Motivation stems from the fact that individual waste streams can have largely disproportionate impacts on the treatment requirements of combined wastewater, as well as a higher potential for effective resource recovery. For example, typically less than 1% of municipal wastewater volumetric flow is attributable to urine, yet urine contributes greater than 75% of the nitrogen load to combined wastewater; the remaining 25% of the nitrogen load is distributed between greywater (~5%) and blackwater (~20%) (Wilsenach and van Loosdrecht 2006). The disproportionate contributions of urine to the nitrogen content of combined wastewater indicate that efforts to remove nutrients are being applied to larger than necessary volumes of liquid when the entire flow is treated, thus resulting in increased energy and resource consumption, as well as reduced effectiveness. From a different perspective, source separation of greywater presents the opportunity for water reclamation due to the significant volumetric contribution of greywater to combined wastewater and the low organic pollutant and pathogenic content of this waste stream (Eriksson, Auffarth et al. 2002). Source separation of blackwater typically allows the organic carbon content of wastewater to be more pointedly addressed, either for general removal of oxygen demand or for energy recovery.

Source separation is expected to warrant increased attention as the need to view wastewater treatment as an opportunity to recover valuable, depletable resources becomes more apparent and wastewater discharge requirements become more stringent, e.g., nutrient waste load allocation and numeric nutrient criteria. The collection and treatment of combined household wastewater makes resource recovery a challenge because resources are diluted and contaminated by the mixing of waste streams. Source separation presents the opportunity to recover nitrogen and phosphorus from a low flow, nutrient-rich solution (urine), energy from a carbon rich stream (blackwater), and water from a minimally contaminated source (greywater). Source separated waste streams will be defined as follows hereafter:

- A: Non-kitchen sinks, clothes washer, shower, bathtubs (excludes toilets)
- B: Kitchen sinks, dishwasher, garbage grinder
- C: Toilet: non-urine
- D: Toilet: urine

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

Typically, domestic wastewater is separated into greywater (A) and black water (B+C+D) (Table 3.3). Here, the kitchen wastestream should not be included in the greywater designation because of its association with production and consumption of food and the BOD, TSS and pathogens that may be found in kitchen waste. Greywater comprises over half of the water volume while contributing relatively small fractions of total pollutant mass. With lower constituent concentrations, greywater requires less intensive treatment than black water to meet a given level of water quality. Greywater may be rendered suitable for onsite reuse (irrigation or indoor toilet flushing) with relatively simple aerobic biological treatment.

Although not typically referred to as a “wastestream”, urine (D) accounts for very small volumes but high fractions of nitrogen and phosphorus. Separation and recovery of urine as a concentrated nutrient source provides benefits for both onsite nitrogen reduction and beneficial nutrient recovery. Urine separation can be accomplished with or without the separation of greywater and black water, resulting in typical domestic wastestreams minus urine (A+B+C) or a black water wastestream minus urine (B+C).

Black water (B+C+D) contains a majority of the constituent mass but less than half of the volume of the whole domestic wastestream (A+B+C+D), resulting in higher constituent concentrations (Table 3.3). Treatment of black water would require generally similar treatment as combined domestic wastestreams, although the necessary treatment system capacity required to achieve a similar level of effluent quality could be smaller. Removal of urine from domestic wastestreams (A+B+C) or from black water (B+C) has relatively minor effect on total daily volume and BOD and TSS concentrations. The treatment plant required for removal of BOD and TSS would not be greatly affected, but the required nitrogen reduction treatment capacity would be reduced.

The primary options for household source separation are recovery of urine and segregation of greywater for reuse. Urine separation removes a majority of the nitrogen and a small fraction of the volume of total household wastestream (Larsen, Peters et al. 2001). The remaining household wastestream has a similar daily volume but only ~20% of the TN. 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, Schwegler 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. Beneficial use of urine could also provide a future funding mechanism for onsite treatment infrastructure, however, this requires that a market be identified for the urine based fertilizer product.

Table 3.3: Volume and Constituent Concentrations of Domestic Sewage Wastestreams for a Four Person Household in the US (Hazen and Sawyer 2009)

Description	Label	Production (gal/ 4 pers/day)	Constituent Concentration (mg/L)				Percent of Total Constituent Load (%)			
			$C\text{-}BOD_5$	TSS	Total N as N	Total P as P	$C\text{-}BOD_5$	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
Non-urine domestic sewage	A+B+ C	239	261	547	16	3.5	93	100	25	40
Non-urine blackwater	B+C	111	453	1,128	27	6.2	75	96	19	33
Urine	D	2.4	1,838	35	4,808	528	7	0.07	75	60

3.1.4.1 Urine Source Separation

3.1.4.1.1 Overview of Urine Treatment Objectives and Typical Approaches

The main objectives for a urine source separation system include collection, conveyance, disinfection, production of a nutrient-rich product, and product distribution (or product disposal if nutrient removal is prioritized over nutrient recovery).

Collection and conveyance: Effective collection and conveyance depends on the type of urine separating fixture in use, system operation and upkeep, and user behavior. Available fixtures include urine separating toilets, which have a divided bowl with dedicated effluent piping for urine and feces (Figure 3-8), and waterless urinals with a single effluent line. Vinnerås and Jönsson (2002a) describe the performance of a urine collection system for a urine separating toilet. Annually, 125 gallons of urine were collected per person with a coefficient of variation of 11 percent. When combined with feces collection, 60 percent of the nitrogen was recovered from the wastewater. In Switzerland, urine separating toilets and waterless urinals were tested in four households (Rossi, Lienert et al. 2009). Water recovery was 0.036 gal/flush in households and 0.059 gal/use with waterfree urinals. Mean urine collection rates in households were 1.68 gal/day on weekdays and 2.44 gal/day on weekends. Urine recovery in households was maximally 70 to 75 percent of the physiologically expected quantity.

Urination output and frequency can also be estimated based on work by Latini, Mueller et al. (2004) and FitzGerald, Stablein et al. (2002), the quantification of which is important for the design of urine source separation systems. Twenty-four-hour diaries of 300 and 284 racially diverse women and men, respectively, were collected to determine daily urination frequencies and volumetric outputs. Women reported a median value of 1.62 liters of urine per day and eight urination events per day; men reported a median value of 1.65 liters of urine per day and seven urination events per day. Total urine production for a household or cluster of buildings would depend on the number of people in the structures and the fraction of their day spent there.

Communication with the public regarding the installation of urine separating fixtures is critical for encouraging use and ensuring proper protocols to maximize urine collection and minimize contamination. Urine source separation has long been recognized as a significant change to conventional processes and, as such, has demanded early inclusion of sociological expertise in order to facilitate public acceptance. Overall, respondents in previously conducted survey studies appear highly accepting of urine source separation, especially when these systems are being considered for use outside of one's home. In a comparative analysis of 38 studies across seven Northern and Central European countries, Lienert and Larsen (2010) stated that "urine source separation and nutrient recycling is appealing to lay people and their willingness to support the NoMix bathroom innovation is large." For example, urine-separating toilets were installed at two Swiss organizations around the year 2000. Lienert and Larsen (2006) found that the majority (72%) of the users at these organizations, one school and one research institute, liked the idea of urine source separation. Furthermore, 86% of respondents would move into apartments with NoMix toilets and most users believed NoMix toilets to be equivalent to conventional toilets with respect to design, hygiene, and smell. Expert stakeholders in China have also expressed acceptance of urine source separation, expecting that the implementation of such technologies will increase over the next 20 years (Medilanski, Chuan et al. 2007). To the best of our knowledge, the first survey concerning acceptance of urine source separation in the United States was conducted by Lamichhane and Babcock (2013), in which it was also concluded that emotional support for urine source separation was high at the University of Hawaii. Although the majority of respondents in this study had no previous knowledge of urine source separation, 80% were willing to install a urine-diverting toilet in their home if there was no associated cost.

Figure 3-8: Two Swedish urine separating toilets (EcoSan and Novaquatis)

Once urine is separated from other household waste streams through the use of urine separating toilets and urinals, the next step is ensuring that the urine can be conveyed to its intended destination. This is typically done via a separate set of urine-only piping. A pipe material that is resistant to the corrosivity of urine is required (e.g., PVC), as is an understanding of precipitation potential. Spontaneous precipitation is expected to occur in urine conveyance pipes due to the rate at which urine undergoes hydrolysis and the resulting precipitation favoring conditions that develop, e.g., an increase in pH and conversion of urea to ammonia/ammonium. Precipitates are expected to be dominated by struvite and hydroxyapatite, which are controlled by the calcium and magnesium content in urine and urine flush water. If these precipitates adhere to piping, this represents a reduction in nutrient recovery potential, as well as a potential cause of system malfunction due to blockages. Prevention of pipe blockages depends on the design of the urine collection system, e.g., pipe sloping and urine flush water volume, and maintenance plans, e.g., procedures recommended by waterless urinal manufacturers. Frequent rinses with a mild acid, e.g., vinegar, and/or hot water can be used as a preventative measure, while stronger solutions are needed to remove blockages after they have already formed, e.g., >24% acetic acid (Nakatsuji, Salehi et al. 2015; Ren, Ni et al. 2015). Alternative approaches to ensuring the slowing of urine hydrolysis and/or the minimization of precipitation potential at the fixture are also available. For example, Boyer, Taylor et al. (2014) explored the use of *in-situ* urine softening through the use of a cation exchange cartridge in waterless urinals to reduce of availability of struvite and hydroxyapatite forming magnesium and calcium. Additionally, improvements have been incorporated into the design of urine diverting fixtures to prevent significant blockages, such as valve replacements (Nakatsuji, Salehi et al. 2015).

Disinfection: Although urine is assumed to have a far lesser pathogen content than non-urine blackwater, disinfection is an important aspect of urine source separation, if the end goal is to produce a urine-based fertilizer. If there intent of urine source separation is only to store urine onsite for subsequent transport to a centralized wastewater treatment facility, onsite disinfection is not necessary because disinfection will be achieved at the centralized facility.

There are a few viruses that can be excreted through urine, but pathogenic concerns related to source separated urine mostly pertain to pathogens that enter the waste stream through fecal contamination. Elongated storage of source separated urine is the predominant mode of disinfection that is commonly proposed, as it benefits from the naturally occurring urea hydrolysis process that takes place in stored urine and the biocide properties of the resulting ammonia. However, the storage time required for adequate disinfection is difficult to determine, as it depends on the pathogen content (type and concentration), ammonia content, pH, and temperature of the stored urine mixture, which are often highly uncertain and variable over time (Hoglund, Ashbolt et al. 2002; Maurer, Pronk et al. 2006; Vinneras, Nordin et al. 2008). As a result of varying degrees of pathogen sensitivity to ammonia and a distribution of expected storage conditions, a conservative storage time must be selected. For example, urine solution composition and storage conditions expected in Ishii and Boyer (2015) would require a storage time of only 1–2 days for Salmonella enterica inactivation, but 50–250 days for Ascaris egg inactivation, based on results seen by Vinneras, Nordin et al. (2008) and Sepehri, Heidarpour et al. (2014). Higher storage temperatures and minimal dilution with flush water facilitate faster disinfection, however, fluctuations in temperature can also benefit the disinfection process (Nordin, Niwagaba et al. 2013). Paruch (2015) found that mild conditions (10 degrees C or 50 degrees F) were the least effective in terms of E. coli inactivated when compared with cold (4 degrees C or 39 degrees F) and warm (22 degrees C or 72 degrees F) conditions. The conservative storage times required to target multiple pathogens over a range of storage conditions necessitate large urine storage tanks, which occupy more land area and also present a higher potential for human and environmental concerns should tanks malfunction, e.g., liquid and odor leaks. The World Health Organization recommends that source separated urine be stored for a minimum of six months at 20 degrees C for unrestricted use as crop fertilizer, the caveat being that urine should be applied at least one month prior to harvest for food crops that are consumed raw (WHO 2006). Shorter storage times and lower storage temperatures increase the potential for pathogenic contamination of source separated urine, and thus also increase constraints on recommended use as fertilizer. For example, source separated urine that has been stored for one month at 4 degrees C is only recommended for use as fertilizer on food and fodder crops that are to be processed (WHO 2006). A six month storage time translates to an onsite storage capacity of approximately 450 gallons for a four person household, assuming urine production rates shown in Table 3.3 (2.4 gallons of urine per four persons per day).

Alternative or additional treatment steps can be taken to encourage complete disinfection of source separated urine and/or to forego the resources required for elongated storage, e.g., space, long-term monitoring, phasing of urine collection/storage/use. Disinfection steps may pertain to the entire liquid and come before subsequent nutrient recovery steps (like the storage approach), may pertain to the entire liquid and happen concurrent/after nutrient recovery steps, or may pertain to treating just the nutrient end product itself. For example, nutrient-rich struvite (discussed below) can be produced from urine as an alternative fertilizer, the safety of which depends on its pathogen content. Decrey, Udert et al. (2011) found that the concentration of a human virus surrogate tended to be comparable in urine and the struvite precipitate; however, Ascaris eggs accumulated within the solid struvite matrix. Thus, disinfection can be solely applied to the struvite, e.g., via drying, as opposed to disinfection the entire urine solution. Inactivation of Ascaris egg survival is positively correlated with moisture content. Udert, Buckley et al. (2015) deemed the disinfecting capabilities of struvite precipitation as “medium” depending on the resulting moisture content, whereas nitrification/distillation was ranked as “high” due to the distillation process (discussed below with regard to nutrient recovery). Electrolysis (a method discussed below for nutrient removal, not recovery) also earned a ranking of medium strength with regard to disinfection capabilities because of the anodic oxidation of chloride to chlorine, which is damaging to viruses and bacteria.

Regardless of whether disinfection is achieved via elongated storage or an alternative method, some level of urine storage will likely be required as a central collection location. In order to protect public health and acceptance of urine source separation, odor production during urine storage must be addressed. The composition of urine and the hydrolysis process that takes place during storage results in the production of gaseous emissions (i.e., pressurized headspace) including odorous compounds (e.g., ammonia, volatile organic compounds), thus providing an opportunity for odor problems. Odor control may be exercised through the use of well-sealed and/or buried tanks or odor control devices (Zhang et al., 2013). Additionally, indoor toilets should be equipped with a seal (e.g., water, latex/silicon membrane) to prevent odorous air from flowing from the urine storage tank and collection system to the indoors (Jonsson et al., 2007). Acid traps can also be used at pipe outlets and in tanks to sequester/recover volatilized ammonia (Siegrist et al., 2013). Although odor control is typically discussed with regard to the urine storage phase, the need for odor control must be considered at all treatment steps, especially those involving ventilation (e.g., aeration/stripping). Alternatively, if a urine stabilization step is employed to prevent urea hydrolysis, such as urease inhibition, pH adjustment, or nitrification, the conversion of urea to ammonia is stymied, thus also minimizing odor production (Fewless, 2015).

Nutrient removal and recovery: The end goal of urine source separation may be simply nutrient removal from wastewater or it may be the removal and subsequent recovery of nutrients from wastewater. Nutrient removal (without recovery) is motivated by the need to divert nutrient loads away from the environment, e.g., via onsite treatment or centralized treatment prior to environmental discharge. Ideally, urine source separation allows for a large fraction of wastewater nutrients to be targeted for removal while only dealing with a small fraction of the total volumetric flow, thus efficiently minimizing the eutrophication potential of treated effluent. Nutrient recovery, i.e., nutrient removal followed by beneficial reuse, requires that efforts not only be dedicated toward removing nutrients from urine, but also that those nutrients end up in a contaminant-free, usable form. Nutrient recovery is driven by the fact that although nutrients represent contamination when released into the environment, they serve as a resource when applied to crops (Guest, Skerlos et al. 2009). Mihelcic, Fry et al. (2011) estimate that approximately 11% of the global demand for phosphate rock could be offset by the recovery and reuse of phosphorus in urine. The use of urine-based fertilizers instead of synthetic fertilizers has gained interest due to the non-renewable, globally unevenly distributed nature of phosphate rock (Ashley, Cordell et al. 2011; Cordell, Rosemarin et al. 2011). The following sections discuss various methods for achieving nutrient removal and recovery from source separated urine.

3.1.4.1.2 Direct Application and Precipitation

The nutrients in urine can be diverted away from the environment via application of disinfected urine or urine-based product, e.g., struvite, to croplands or conversion of urine-based nutrients into biologically unavailable forms, e.g., atmospheric nitrogen. Previous studies have typically focused on the direct application of liquid urine to croplands or the production of struvite ($MgNH_4PO_4 \cdot 6H_2O$). The nutrients in liquid urine can be taken up by plants and there have been multiple demonstrations of the use of liquid urine as a fertilizer (Pradhan, Nerg et al. 2007; Mnkeni, Kutu et al. 2008; Pradhan, Holopainen et al. 2009; Heinonen-Tanski, Pradhan et al. 2010; Richert, Gensch et al. 2010; Zheng, Ji et al. 2010). If adequate cropland (or hydroponic area) is established and urine is applied at a rate that corresponds with plant uptake rates and soil retention (thus minimizing nutrient-rich runoff), upwards of 80% of nitrogen can be removed from wastewater effluent simply due to the majority of wastewater nitrogen originating from urine.

However, the social barriers and costs of transporting liquid urine to agricultural sites typically highlight the fact that urine volume reduction is needed in order to have a sustainable nutrient recovery and redistribution program. Struvite precipitation is the most thoroughly researched method of nutrient recovery/volume reduction with studies spanning the overall effectiveness of the method (Udert, Larsen et al. 2003a; Udert, Larsen et al. 2003b; Wilsenach, Schuurbiers et al. 2007; Etter, Tilley et al. 2011; Grau, Rhoton et al. 2015; Xu, Luo et al. 2015) and the quality of the end-product as a fertilizer (Ronteltap, Maurer et al. 2007a; Decrey, Udert et al. 2011). Urine undergoes hydrolysis during conveyance and storage due to the presence of naturally occurring urease enzyme, which converts urea to ammonia/ammonium and bicarbonate, thus resulting in a pH increases, ammonia for pathogen inactivation, and ammonium for struvite formation. The duration of storage is determined by the kinetics of pathogen inactivation, as the time required for favorable precipitation conditions is only on the order of days (Udert, Larsen et al. 2003a; Udert, Larsen et al. 2003b).

Struvite is a slow-release mineral fertilizer with the chemical formula $MgNH_4PO_4 \cdot 6H_2O$ (Mg:P:N molar ratio of 1:1:1). Magnesium is typically added to stored urine until the Mg:P molar ratio is 1:1, thus enabling maximum precipitation of phosphorus and partial precipitation of available nitrogen. Magnesium can be added in the form of high purity magnesium oxide or magnesium chloride, as well as part of low-cost materials to minimize costs and/or take advantage of available resources, e.g., wood ash, bittern, seawater (Sakthivel, Tilley et al. 2012; Mackey, Zheng et al. 2014; Krahenbuhl, Etter et al. 2016). Phosphorus recovery as struvite in stored urine can be upwards of 100%, but nitrogen recovery as struvite with only the addition of magnesium can only be ~5% due the molar concentration of nitrogen in urine being much greater than that of phosphorus. The 1:1 molar ratio of N:P in struvite can be viewed favorably, i.e., one product with two essential nutrients, but it is also a challenge because of the far greater N:P molar ratio in urine and the fact that plants need more nitrogen than phosphorus. As a result, large nitrogen concentrations remain in urine solution post-struvite precipitation and a supplemental nitrogen source is needed in agriculture if struvite is used as the fertilizer. Nitrogen in urine can be further incorporated into struvite if both magnesium and phosphorus are added to stored urine (thus making the final Mg:P:N equal to 1:1:1 as needed ideally for struvite) (Liu, Zhao et al. 2008c), but the environmental and economic costs of the required chemical input to bring the molar concentrations of magnesium and phosphorus up to nitrogen are likely prohibitive (Ishii and Boyer 2015).

3.1.4.1.3 Sorption and Ion Exchange

The removal and recovery of nutrients from source separated urine are also possible via sorption and/or ion exchange. These processes benefit from the high concentrations of nutrients in urine relative to other constituents, i.e., a positive concentration gradient. Notably, these materials that facilitate sorption and/or ion exchange of nitrogen can be used as a standalone process or post-struvite precipitation to address the leftover nitrogen. Cation exchange for ammonium in source separated urine can be accomplished with clinoptilolite, a naturally occurring zeolite (Lind, Ban et al. 2000; Lind, Ban et al. 2001; Jorgensen and Weatherley 2003; Smith 2008; Smith, Otis et al. 2008); the mineral wollastonite (Lind, Ban et al. 2001), and polymeric ion exchange resins (Jorgensen and Weatherley 2003). Ion exchange can be applied as post treatment following struvite precipitation or as an integrated precipitation/ion exchange process. A combined process consisting of magnesium enhanced struvite crystallization and ion exchange adsorption was evaluated in laboratory experiments. Up to 80 percent of the nitrogen content of a synthetic human urine was removed (Lind, Ban et al. 2001).

Sorption and ion exchange strategies for the removal/recovery of nutrients tend to fall into one of two categories: sorption/ion exchange using natural media, e.g., zeolite, or engineered media, e.g., ion exchange resin. Engineered materials tend to have a higher loading capacity but come at a higher cost. Additionally, due to the higher cost and possible incompatibility of engineered materials with soils, one would typically opt to regenerate the media instead of directly applying it to agricultural operations – this further increases the cost and complexity of the process. (Tangsubkul, Moore et al. 2005) found that the zeolite clinoptilolite could remove 97% of ammonium from urine and that 88% of that ammonium could be recovered when applied to agriculture. The ammonium exhausted zeolite was comparable to synthetic fertilizer and benefited from reduced salinity when compared to liquid urine. The nitrogen loading of clinoptilolite can be further increased through the use of variable loading as opposed to constant loading (Enfield 1977). It is important to note that additional non-targeted constituents may also be attracted to sorptive or ion exchange materials. For example, Sendrowski and Boyer (2013) reported that hybrid anion exchange could achieve high levels phosphate removal (> 97%) in both fresh and hydrolyzed urine, however, co-removal of diclofenac (a commonly used pharmaceutical) was also occurring. These results highlight the need to exercise removal/destruction of pharmaceuticals prior to nutrient removal or to use nutrient recovery strategies that specifically target nutrients even in the presence of pharmaceuticals.

A remaining challenge is the identification of a media with an adequate nitrogen sorption or ion exchange capacity. The notably high concentration of nitrogen in urine and the constant production of urine require a media with a large capacity for nitrogen. As the allowable loading rate of nitrogen onto a material increases, the required frequency of media exchange or regeneration decreases and the nutrient value of the end product increases. As an example of how nitrogen capacity relates to maintenance requirements, Xu, Luo et al. (2015) found that zeolite could remove >85% of ammonia-nitrogen from stored urine at a zeolite dose of 375 g/L. Although this shows that zeolite can have a high nitrogen removal rate when dosed appropriately, these results also show that a urine production of 1.5 liters/person/day would require ~205,000 g zeolite/person/year, which either necessitates significant space allocations or frequent media change outs.

3.1.4.1.4 Aeration/Stripping

The hydrolysis of urine results in the conversion of urea nitrogen to ammonia/ammonium nitrogen. The ratio of ammonium to ammonia is important because ammonia serves as a biocide and is a volatile compound, whereas ammonium is available for precipitation. One can take advantage of ammonia's volatility and remove it from solution by aeration. The volatilized ammonia can subsequently be sequestered into a solution, e.g., sulfuric acid, for reuse (Larsen, Maurer et al. 2010). Ammonia stripping with air in a batch system is a function of air flow rate and pH (more ammonia at higher pH values). Jonsson and Vinneras (2007) saw 92% recovery of ammonia as ammonium sulfate in an absorption unit with hydrolyzed urine at pH 12 and an air flow rate of 0.21 m^3/hour. Ammonium sulfate is a marketable, plant-available fertilizer (Morales, Boehler et al. 2013). Similarly, Larsen, Lienert et al. (2004) found that 90% of nitrogen was removed from urine by stripping (3 hours of urine circulation through stripping column, 80 liter per hour air flow rate) and that 100% of the stripped nitrogen was recovered as liquid ammonium sulfate. However, the air flow rate required for stripping can be energy and cost intensive, thus making a nearby demand for the ammonium sulfate fertilizer essential (Zheng, Ji et al. 2010). As previously discussed, the potential for odor problems resulting from fugitive ammonia emissions should be addressed.

3.1.4.1.5 Nitrification with Distillation

Nitrification and distillation of source separated urine presents the opportunity to stabilize nitrogen, i.e., convert volatile ammonia to oxidized nitrogen, and achieve complete nutrient recovery. Additionally, distillation facilitates more complete disinfection than storage and drying with minimal production of nuisance and/or hazardous byproducts (Udert, Buckley et al. 2015). Udert and Wachter (2012) demonstrated the use of a membrane aerated biofilm reactor, the process of stability of which was controlled with pH, for the nitrification of urine. Nitrified urine was subsequently reduced in volume via distillation in a lab-scale reactor. Although the process was successful and a nutrient-rich product was produced, the energy requirement was four to five times higher than removing nitrogen and phosphorus in a conventional wastewater treatment plant (mostly due to energy-intense distillation). It was hypothesized that energy requirements could be made comparable to conventional treatment with the introduction of reverse osmosis prior to distillation. Ammonia oxidation in urine by biological means is typically limited to 50% due to the availability of alkalinity (Sun, Dong et al. 2012). The stabilization of nutrients in urine via biological processes with subsequent volume reduction is a promising pathway due to the accomplishment of complete nutrient recovery, e.g., phosphorus, nitrogen, potassium, calcium, magnesium. Nitrification can also be carried out in a packed column. Feng, Wu et al. (2008) saw nitrification of urine reach 95% in a packed column with pH adjustment (pH = 8), however, only 50% nitrification was achieved without artificial maintenance of pH.

3.1.4.1.6 Membrane Filtration

Membrane filtration provides the opportunity to separate nutrients from other constituents of concern, as well as bulk liquid, in urine. The Rich Earth Institute has ongoing research pertaining to the use of reverse osmosis for the concentration of nutrients. Results show that volume reduction can indeed by achieved, however, ~20% of nitrogen is lost as permeate due to the moderate rejection rate of reverse osmosis membranes against ammonia (Rich Earth Institute 2015). Forward osmosis has also be explored, using seawater and desalination brine as the draw water. In forward osmosis, bulk liquid is instead drawn through the membrane and nutrient-rich solution remains. In one application, forward osmosis resulted in high water fluxes across the membrane and high rejection of phosphate and potassium in both fresh and hydrolyzed urine. Nitrogen rejection was highest in hydrolyzed urine (50 to 80%) due to improved rejection of ammonia/ammonium versus urea (Nolde 2000). Research has also shown the separation of nitrogen (permeate) from micropollutants (concentrate) using nanofiltration membranes, however, phosphorus and sulfate are retained with the micropollutants (Pronk, Palmquist et al. 2006). Membrane filtration can also be combined with other processes, such microbial technologies, to further improve nutrient recovery. For example, substrate oxidation in a microbial electrolysis cell can provide energy for the separation of nutrients from solution through an ion exchange membrane (Haddadi, Nabi-Bidhendi et al. 2014).

3.1.4.1.7 Electrolysis and Microbial Fuel Cells

Recent research in the area of urine source separation has focused on the use of microbial electrochemical technologies, such as microbially mediated electrolysis and microbial fuel cells, for the concurrent treatment of urine and production of energy (Ledezma, Kuntke et al. 2015). These processes have to capacity to address multiple constituents in urine, such as nutrients, pharmaceuticals, and chemical oxygen demand. The revenue from removal/recovery of nitrogen in comparison with existing nitrogen technologies coupled with the production of electricity have made bioelectrochemical systems a promising pathway (CSWRCB 1995). Microbial fuel cells involve the conversion of organics in source separated urine to energy through the metabolism of microbial communities. Field trials were recently completed for “Pee Power Urinals”, which harness microbial fuel cell technologies for the provision of internal lighting. Results showed that chemical oxygen demand removals were highly variable (30 to 95%) depending on temperature and the frequency of use. Ammonia inhibition can be a challenge in microbial fuel cells, however, the capacity of bacteria to resist ammonia inhibition has been shown to be greater with high substrate concentrations and frequent feed (Hellström and Kärrman 1997). As previously mentioned, microbial fuel cells can also be coupled with other treatment processes to provide energy where needed (Haddadi, Nabi-Bidhendi et al. 2014). Balmer (2004) combined struvite precipitation with a microbial fuel cell to achieve 95% removal of phosphorus and 29% removal of nitrogen from urine with the addition of magnesium, and 43% removal of phosphorus and 40% removal of nitrogen with the addition of both magnesium and phosphorus (for additional struvite precipitation). For electrolysis of urine, the process can be configured such that nitrogen is converted to atmospheric nitrogen (and lost) or recovered, as well as configured for either indirect oxidation through an oxidation mediator produced on the anode or through direct electron transfer on the anode surface (Anglada, Urtiaga et al. 2009). The potential for graphite to be used as an inexpensive electrode for direct oxidation has been demonstrated by Zollig, Fritzsche et al. (2015), with results showing 30 to 40% oxidation of ammonia, ~80% of which went to nitrogen gas and the remainder to nitrate/nitrite. Indirect oxidation benefits from a lack of potentially expensive electrodes, as well as oxidation of chloride to chlorine which serves as a disinfectant, however, chlorine also reacts with organics to form chlorinated organic substances, chlorate, and perchlorate (all of which have adverse health effects) (Udert, Buckley et al. 2015).

The potential co-benefits of combining urine source separation with energy production have also been demonstrated for algal-based biofuels. In a life cycle comparison of algae and other bioenergy feedstocks, Clarens et al. (2010) found that although algae had a lower associated land requirement and eutrophication potential for use as a feedstock, other more conventional crops (e.g., switchgrass, canola, corn) had lower environmental costs related to greenhouse gas emissions and water use. The large environmental burden of algal-based biofuels mainly stemmed from algae's demand for carbon dioxide and fertilizer. The use of nutrient-rich, source separated urine as an alternative to synthetic fertilizer was found to make algae more environmentally beneficial than the other feedstocks from a life cycle assessment. The coupling of urine source separation with algae harvesting enables the beneficial reuse of nutrients for biofuel production, while also reducing nutrient discharges to the environment (Pittman et al., 2011).

3.1.4.2 Greywater Source Separation

3.1.4.2.1 Overview of Greywater Treatment Typical Approaches

As shown in Table 3.3, greywater contains only a small portion of the nitrogen in household wastewater, therefore the total impact of greywater separation on nitrogen reduction is limited. However, it does reduce the amount of organic carbon available to potential electron donors during denitrification of black water.

A universally accepted definition of greywater does not exist. Excluding kitchen waste from greywater is consistent with Florida requirements. Separate collection of effluent from all kitchen and toilet sources is typical. Some greywater definitions include kitchen waste, which would increase pollutant concentrations and lead to greater nuisance potential and greater requirement for treatment. Kitchen wastes have been further subdivided, where all wastes except garbage grinder wastes are included in greywater. Including kitchen wastes in greywater would necessitate more intensive treatment processes which would duplicate black water treatment processes and reduce the advantage of separating greywater. In reviewing any reports on system performance and feasibility, the composition of the greywater stream should be determined.

Rational for separate greywater collection is to reuse or dispose of the less polluted greywater onsite, through irrigation, application on land or indoor non-potable reuse. Modeling predicted that a 40 percent savings in potable water demand could result with greywater recycling in an urbanized area, although no attention was given to nitrogen reduction (Mah, Bong et al. 2009). Greywater recycling in a multi-story residential building for toilet flushing reduced potable water use by 29 to 35 percent and had a payback period of less than 8 years. Nitrogen reduction was not reported (Ghisi and Ferreira 2007).

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

3.1.4.2.2 Greywater Treatment

Greywater treatment has been examined by several investigators with a variety of treatment technologies applied in many different schemes for overall water recycling (Nolde 1999; Günther 2000; Jefferson, Burgess et al. 2001; Ramona, Green et al. 2004; Friedler, Kovalio et al. 2005; Schäfer, Nghiem et al. 2006; Elmitwalli and Otterpohl 2007; Eriksson, Andersen et al. 2008; Gual, Moià et al. 2008; Pidou, Avery et al. 2008; Widiastuti, Wu et al. 2008; Winward, Avery et al. 2008a; Winward, Avery et al. 2008b; Benetto, Nguyen et al. 2009; Kim, Song et al. 2009; Misra and Sivongxay 2009).

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

Storage of greywater is an important element of all greywater recycling systems. Greywater quality has been found to be affected by storage; sedimentation, aerobic microbial oxidation, anaerobic microbial processes in settled solids, and reaeration (Dixon, Butler et al. 2000). Storing greywater for a 24 hour period led to improved quality due to the reduction of suspended solids, but dissolved oxygen was depleted after 48 hours which 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. In Australia, greywater collection systems are required to use disinfection (UV or chlorine) if greywater is held for longer than 24 hrs. This would serve to oxidize BOD in the influent greywater, and oxidize organics and odors that are released from underlying settled solids.

In a review of technological approaches for the treatment and reuse of greywater, Roma, Philp et al. (2013) stated that physical treatment processes alone were not sufficient for greywater processing. The recommendation was that an aerobic biological process with physical filtration and disinfection be used to adequately reduce organics, nutrients, and surfactants. With regard to the biological treatment of greywater, it is important to note that bathroom and laundry greywater tends to be deficient in nitrogen and phosphorus, while kitchen greywater has a balanced chemical oxygen demand to nitrogen to phosphorus ratio for biological growth. In a different study, Li, Gulyas et al. (2009) found that ultrafiltration membrane filtration was capable of reducing greywater TN to 16.7 mg/L and Dalahmeh, Pell et al. (2012) determined that up-flow anaerobic sludge blanket treatment with a membrane bioreactor aerobic step produced greywater suitable for unrestricted use in Egypt. Electro-coagulation in series with a submerged bioreactor has been shown to bring greywater to better overall quality than a submerged bioreactor alone, however, nitrogen removal was actually better when only the submerged bioreactor was in use. It was hypothesized that the electrolysis condition needed to be optimized in order to avoid impediment of biological nitrogen removal (Sun, Dong et al. 2012). More passive, low maintenance approaches to greywater treatment are also possible, such as the use of a grease trap, followed by a sedimentation tank and constructed wetlands (Paulo, Azevedo et al. 2013). The main component that required ongoing attention was the wetland, as clogging would occur if not properly maintained. In a study of bark, activated charcoal, and sand filtration for greywater treatment and reuse, chemical oxygen demand and nutrient removal was a function of hydraulic and organic loading rates. Bark and charcoal filters performed better than sand filters when subjected to variable loading rates, but the charcoal filter alone was the best option if minimization of environmental eutrophication was a primary goal of the filtration step (Kalmykova, Harder et al. 2012; Lamichhane and Babcock 2012; Sakthivel, Tilley et al. 2012).

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

3.1.4.3 Black Water Source Separation

With the nutrient contributions from individual waste streams in mind, the diversion of blackwater without urine for subsequent treatment only provides the opportunity to target ~20% of the total wastewater nitrogen content. Blackwater (without urine) treatment tends to focus on the removal of chemical oxygen demand and/or energy production due to the high concentration of organics.

Different techniques were examined for separation of fecal material from flush water. The Aquatron system uses surface tension, gravitation and a whirlpool effect to produce a solids stream that contains 70 to 80 percent of the incoming dry matter thereby recovering the majority of nitrogen (Vinnerås and Jönsson 2002a). Black water treatment was investigated using anaerobic biotreatment followed by filtration using commercial nano-filtration and reverse osmosis membranes (van Voorthuizen, Zwijnenburg et al. 2005). Orthophosphate recoveries from the wastestream were 74 to 99 percent while ammonia recoveries were 21 to 94 percent. Onsite anaerobic treatment of black water (e.g., Luostarinen and Rintala 2005) is similar to treatment of whole domestic wastewater, albeit with higher organic matter and solids, as well as greatly reduced nitrogen if urine is separately collected. Three combinations of biological treatment and membrane filtration were compared for separate black water treatment: a UASB followed by membrane filtration, anaerobic MBR, and aerobic MBR (van Voorthuizen, Zwijnenburg et al. 2008). All three systems exhibited high nutrient conservation, i.e. little nutrient reduction, and effluent with low TSS and high soluble COD. The majority of the recent research defines blackwater as the entire toilet waste stream, i.e., urine, feces, and flush water.

3.1.5 Applicable Groundwater Remediation N Reduction Techniques

The in-situ addition of a permeable reactive media barrier (Figure 3-9) that supports denitrification through the release of carbon or electron donor has been used to intercept OWTS wastewater plumes both in the vadose zone and in shallow water tables. Permeable horizontal "barriers" consisting of cellulous materials such as sawdust or woodchips have been installed below the STU to intercept nitrified effluent and provide reactive media for electron donors for denitrification (Robertson and Cherry 1995; Robertson, Blowes et al. 2000).

These barriers have a high water retention capacity to keep the media near saturation so that anoxic conditions are created as the nitrified effluent percolates through. A side-by-side pilot-scale study evaluating sulfur and wood-based bioretention systems observed greater than 88% TN removal efficiencies in both units using synthetic storm water (Ergas, Sengupta et al. 2010). Vertical permeable reactive barriers have been installed in shallow groundwater downgradient of OWTS to intercept nitrate contaminated groundwater for denitrification. One such permeable reactive barrier has been removing almost all nitrate from an OWTS for over 20 years (Robertson, Vogan et al. 2008; Robertson 2010).

Lignocellulosic carbon sources are generally preferred for their immobility and long-term release of carbon. Even within the lignocellulosic category, there are multiple carbon-rich options. Grinnell (2013) stated that woodchips are the preferred selection because they supply a more consistent, sustained release of carbon over multiple years when compared to wheat straw, alfalfa, corn stalks, corn cobs, and rice husks. More soluble sources of carbon may enable high denitrification rates initially due to carbon being readily available, but the carbon supply is more quickly exhausted than in less soluble counterparts. In general, solid substrates are the longest-lasting carbon supply, but they are the most laborious in terms of replacement (e.g., trenching/excavation).

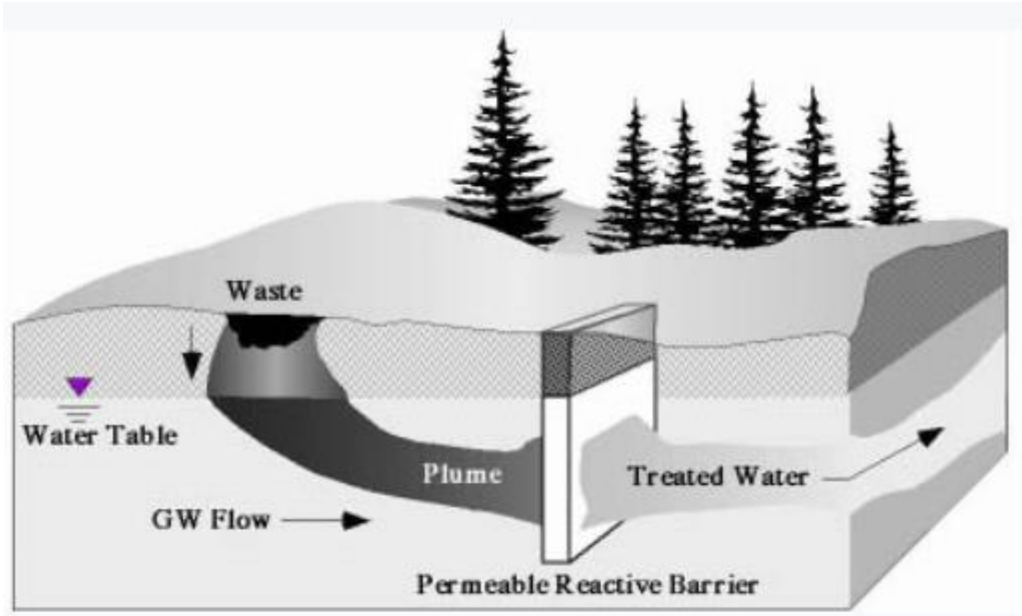


Figure 3-9: Schematic of a Permeable Reactive Barrier (Powell and Associates 2016)

Soluble substrates require more frequent replenishment, but replenishment is typically less intensive (e.g., injection). Table 3.4 provides a summary of potential substrates for denitrification, although the table was originally developed with regard to groundwater remediation (USEPA, 2013). It should be noted that the frequencies of injection given in Table 3.4 are approximate; true injection frequencies are site specific and depend on local biological activity, substrate retention by soil, and hydraulic loading.

Table 3.4: Substrates (electron donors) used for enhanced anaerobic bioremediation of groundwater

Classification	Substrate	Typical Delivery Techniques	Approximate Frequency of Injection
Soluble Substrates	Lactate and butyrate	Injection wells or circulation systems	Continuous to monthly
	Methanol and ethanol	Injection wells or circulation systems	Continuous to monthly
	Sodium benzoate	Injection wells or circulation systems	Continuous to monthly
	Molasses, high-fructose corn syrup	Injection wells	Continuous to monthly
	Whey (soluble)	Direct injection or injection wells	Monthly to annually
Slow-Release Substrates	HRC or HRC-X	Direct injection	Annually to biennially for HRC (typical), every 3–4 years for HRC-X, potential for one-time application
	Vegetable oils	Direct injection or injection wells	One-time application (typical)
	Vegetable oil emulsions	Direct injection or injection wells	Every 2 to 3 years (typical)
Solid Substrates	Mulch and compost	Trenching or excavation	One-time application (typical)
	Chitin (solid)	Trenching or injection of chitin slurry	Annually to biennially, potential for one-time application

Slow-release substrates, such as vegetable oils and vegetable oil emulsions, represent a combination of benefits related to solid and soluble substrates. Slow-release substrates are characterized by low solubility and high viscosity, thus encouraging immobility within the aquifer (like solid substrates). Additionally, slow-release substrates can be injected into the aquifer without requiring excavation/trenching, thus making it easier to replenish than solid substrates, but less frequently required than soluble substrates (USEPA, 2013). Additionally, eventual degradation of lignocellulosic carbon sources could potentially lead to land subsidence and clogging of the system, thus presenting treatment and safety concerns. On the contrary, liquid/emulsified slow-release substrates adhere to aquifer materials, but they do not offer any initial structural support that degrades with time. In accordance with this combination of benefits, slow-release substrates have received considerable attention in the realm of groundwater remediation and offer a potential opportunity for the enhancement of OWTS.

In a pilot-scale sand representation of an aquifer, Hunter (2011) found that soybean oil-coated sand could act as a permeable reactive barrier. Influent water containing 20 mg NO_3 -N/L was flushed through the model aquifer for 30 weeks at a flow rate of 1,112 L/week, ultimately achieving an overall nitrate removal rate of 39%. Nitrate removal was near complete during the first ten weeks of operation and decreased to insignificant removal during the last ten weeks. The pilot results are expected to translate to longer durations of nitrate removal in the field due to the fact that the tested flow rate (1,112 L/week) was substantially higher than what is typically observed in an aquifer. Similarly, Belloso (2005) conducted column tests using aquifer sand, natural groundwater with elevated nitrate levels, and five different electron donors: soy oil, emulsified soy oil, glycerine-lactic acid polymer, common sugar, and a control. The use of soy oil resulted in the greatest nitrate removal (96%), with emulsified soy oil demonstrating similar performance. Belloso (2005) attributed soy oil's high nitrate removal rates to its low solubility and tendency to adhere to aquifer materials.

Site-specific attributes must be taken into consideration when evaluating the use of any electron donor for denitrification, as well as a few attributes that are specific to vegetable oils and vegetable oil emulsions. For example, if a soil treatment area has higher concentrations of sulfate than nitrate, bacteria may preferentially reduce sulfate instead of nitrate, thus resulting in the production of hydrogen sulfide gas. Additionally, if phosphate is the limiting nutritional factor relative to nitrogen and carbon, nitrite accumulation may take place instead of complete denitrification (Belloso, 2005). More specific to slow-release substrates, such as vegetable oil and vegetable oil emulsions, effective operation requires that the substrate be distributed throughout the soil without substantially reducing soil permeability. Permeability is critical to soil treatment efficiency because if major losses in permeability occur, groundwater will preferentially flow around the oil instead of through, thus bypassing treatment. Coulibaly and Borden (2004) found that properly made soybean oil emulsions can be distributed through sands (with varying clay content) without excessive pressure building and permeability losses, while soybean oil as a non-aqueous phase liquid may require infeasible injection pressures and causes moderate permeability losses. Oil emulsions should be stable with oil droplets significantly smaller than the mean pore size of the sediment.

Issues of concern for permeable reactive barriers incorporating reactive denitrification media include media longevity, replacement intervals, and hydraulic issues related to preferential flow paths.

3.1.5.1 Zero Valent Iron

Over the past 20 years, zero-valent iron (ZVI) has been extensively applied for the remediation/treatment of groundwater and wastewater contaminated with various organic and inorganic pollutants (Figure 3-10). The major limitations of ZVI include low reactivity due to its intrinsic passive layer, narrow working pH, reactivity loss with time due to the precipitation of metal hydroxides and metal carbonates, low selectivity for the target contaminant especially under oxic conditions, limited efficacy for treatment of some refractory contaminants and passivity of ZVI arising from certain contaminants (Guan, Zhang et al. 2015). Fu et al. (2014) summarized the use of ZVI for the remediation of several groundwater contaminants, including nitrate. Two lab-scale studies conducted using nanoscale zero valent iron (NZVI) showed nitrate removals of 97% and 100% (Jiang, Lv et al. 2011; Zhang, Li et al. 2011). Hwang et al. (2011) demonstrated that the reduction of nitrate by NZVI yielded ammonium as the primary end-product under the tested conditions. Zhang (2011) demonstrated that NZVI supported on pillared clay (NZVI/PILC) composite is advantageous on removing nitrate over NZVI alone. Some studies have focused on reducing ammonium generation during nitrate reduction, but more research is needed before this technology is implemented for onsite treatment. OWTSs using ZVI to reduce nitrate will likely need to facilitate the reduction of nitrate to nitrogen gas or, alternatively, strip off the produced ammonium via another process, such as ion exchange.

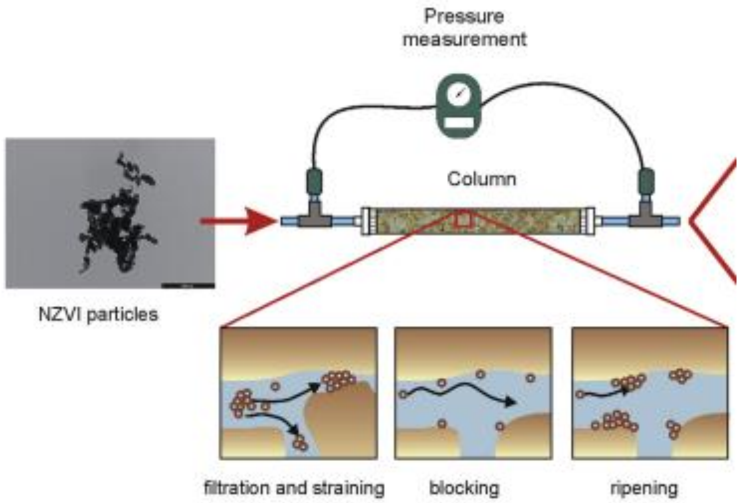


Figure 3-10: Schematic of a ZVI Colum Transport Tests (Tosco 2014)

Solid organic carbon and zero-valent iron (ZVI) have been used separately as reactive media in permeable reactive barriers (PRBs) to degrade nitrate in groundwater, but few studies have examined the combination of the two materials in one system for nitrate remediation. Batch tests were conducted to evaluate three common solid organic carbons and their combination with ZVI for nitrate removal from water; and the results show that the combined system achieves better denitrification efficiency than that measured with sawdust or cotton alone. When complete nitrate removal was achieved in the system that combined ZVI with sawdust or cotton, only 72 and 62.6 % of nitrate removal, respectively, were obtained in which the carbon (C) source was used alone (Wang, Wang et al. 2015).

Another study explored the efficacy of a biochemical remediation of a nitrate-contaminated aquifer by a combination of NZVI and bacteria supported by carbon substrates. Nitrate removal was first assessed in batch tests, and then in a laboratory bench-scale aquifer model. An array of non-pumping-reactive wells (NPRWs) filled with NZVI mixed with carbon substrates (beech sawdust and maize cobs) was installed in the bench-scale aquifer model to intercept the flow and remove nitrate (NO3- conc. = 105 mg/l). A nitrate degradation below the limit target concentration (10 mg/l) was obtained after 13 days (corresponding to 13 arrays of wells in the field). The results of this study demonstrated that using the NZVI-mixed-carbon substrates in the NPRW system has a great potential for in-situ nitrate reduction in contaminated groundwater (Hosseini and Tosco 2015).

3.1.6 Microbial-Earthworm Ecofilters

One of the alternatives for wastewater treatment in developing countries is microbial-earthworm ecofilters (MEEs). MEEs are a natural engineered system which is based on the symbiotic relationship between earthworms and microorganisms, which was first developed by Professor Jose Toha in 1992 at the University of Chile (Aguilera, 2003). Pilot-scale testing of MEEs in developing countries for various wastewaters has shown to provide improved wastewater treatment performance than conventional biofilter without earthworms and nitrogen removal rates up to 60.2% (Jiang, Liu et al. 2016).

3.2 PPCPs Removal OWTS

Pharmaceuticals and personal care products (PPCPs) represent a wide range of organic contaminants, often referred to as part of the larger “contaminants of emerging concern” category. The discharge of wastewater effluent containing PPCPs and the resulting occurrence of PPCPs in the aquatic environment and in downstream potable source waters has become a topic of increasing concern. Regulations, research, and treatment technologies geared toward PPCPs stem from evidence of associated environmental impacts, advances in analytical capabilities, and the fact that conventional (onsite and centralized) wastewater treatment systems are not designed to remove these compounds. OWTSs have been repeatedly identified as a source of PPCPs in the environment (Beardall 2015; Sui, Cao et al. 2015). The presence of PPCPs in groundwater and surface water is so frequently correlated with OWTSs that certain compounds have been recommended to serve as potential chemical tracers of septic contamination (Subedi, Codru et al. 2015); others have suggested current regulatory protection of domestic drinking water wells from pathogens in OWTS effluent is insufficient for protection against PPCPs (Schaider, Ackerman et al. 2016). Something to consider, however, is that PPCPs are present in higher concentrations from single family homes, decentralized apartment complexes, and housing developments because the wastewater does not have the contribution of dilution from widespread industry, wastewater collection systems, and associated infiltration and inflow sources. In response to concerns and reports of PPCPs in the environment, Suffolk County Department of Health Services initiated a monitoring and research based plan in 2001 to evaluate the impacts of emerging contaminants on local water resources. The 2015 Suffolk County Comprehensive Resource Management Plan (2015) demonstrates the County’s in-depth understanding of ongoing PPCP research, reported PPCP removal efficiencies for various OWTSs, and the variability of PPCP data resulting from inherent temporal and geographic variability (e.g., site specific OWTS characteristics, sporadic use of PPCPs). The County has used their understanding of PPCPs to create a suite of recommended OWTS design parameters for optimizing PPCP removal.

OWTSs and centralized wastewater treatment systems are also a potential source of pathogens to groundwater and nearby surface waters, hence the wealth of existing research and development related to pathogen die-off and persistence through various treatment processes. Pathogens, including bacteria, protozoans, and viruses, are primarily addressed by physical removal in the soil treatment stage of OWTSs due to the adsorption and filtration effects of soil; biodegradation of pathogens is also frequently observed (O'Keeffe, Akunna et al. 2015). As a result, pathogen removal in OWTSs is highly site specific. In an intensive water quality monitoring program including eleven stations in Florida coastal waters, Lipp et al. (2001) reported that the areas with the highest pathogenic risk were those associated with high densities of OWTSs. Enterovirus detection was the most prevalent across study sites, while *Cryptosporidium* and *Giardia* (which are larger in size than viruses) were detected in less than 10% of the samples. Historically, OWTS effluent has been identified as a dominant source of pathogenic contamination to groundwater, generally attributed to malfunctioning systems and a lack of multiple disinfection barriers (USEPA 1977).

3.2.1 Mechanisms of PPCP and Pathogen Removal

Potential PPCP removal processes are summarized in Table 3.5 based on PPCP designation as non-biodegradable or biodegradable. In OWTS applications, PPCPs are more commonly targeted by one of the following removal mechanisms, the extent of which depends on the chemistry of the specific compound in question: sorption and ion exchange, biotransformation, filtration. Similarly, OWTS removal of pathogens tends to occur in the soil treatment unit stage as a result of sorption, physical straining, and biotransformation/abiotic inactivation (Stevik, Aa et al. 2004). It should be noted that pathogen inactivation and/or removal is also commonly achieved by UV disinfection, chlorination, and ozonation, however, these practices tend to be reserved for centralized wastewater treatment applications due to the associated operation and maintenance requirements and energy footprint. Herein, PPCP removal refers to removal of parent compounds from the liquid phase. Thus, accumulation of PPCPs in the solid phase (e.g., sludge, soil) and conversion of parent compounds to daughter compounds are considered within the realm of PPCP removal. Pathogen removal includes physical removal from the liquid phase, as well as inactivation.

Table 3.5: Summary of potential PPCP treatment processes (Martz 2012)

PPCP Designation	Treatment Type	Treatment Processes
Non-biodegradable PPCP	Chemical/ physical processes	UV-activated H_2O_2 advanced oxidation
		Combustion
		Adsorption
		Ozonation
		Membrane filtration
Biodegradable PPCP	Biological processes	Activated sludge
		Biofilm processes on solid surfaces
		Biofilm processes on membrane surfaces

3.2.1.1 Sorption and Ion Exchange

Removal of PPCPs via sorption and ion exchange is a function of PPCP chemistry, as well as the properties of the media with which it comes into contact. PPCP removal tends to increase with PPCP hydrophobicity (e.g., as measured by the octanol-water coefficient), as well as with the clay and organic matter content of sorptive media (Loftus, Jin et al. 2015). PPCP acidity (e.g., as measured by the acid dissociation constant) can also control its removal from the aqueous phase depending on how it relates to the pH of the sorptive media and carrier fluid. For example, in onsite wastewater treatment systems, PPCPs with a net negative charge (low pKa values) have a higher tendency to stay in solution due to repulsion between the PPCP and negatively charged soil constituents (Schaider, Rodgers et al. 2013). In other scenarios, the low pKa and associated net negative charge of a PPCP may facilitate its removal, such as in an anion exchange system. Landry et al. (2015) demonstrated the removal of five pharmaceuticals from synthetic source separated urine using strong-base anion exchange resin, the extent of which is favored by the negative ionization and hydrophobicity of the pharmaceuticals. In OWTSs, sorption and ion exchange may take place in the soil treatment unit or in media-filled reactors specifically designed to facilitate these removal processes.

The factors that control pathogen removal via sorption in soil treatment units are similar to those controlling PPCP removal. Sorption is the dominant mode of pathogen removal in infiltration zones with pore sizes larger than pathogen cell size. Stevik et al. (2004) summarize the factors controlling sorption of pathogens as physical, chemical, and microbiological, all of which are listed in Table 3.6.

Table 3.6: Factors controlling pathogen removal via sorption

Category	Factor	Relationship with Pathogen Removal
Physical	Media size and surface area	↓ size, ↑ surface area = ↑ pathogen removal
	Soil organic matter	↑ soil organic matter = ↑ pathogen removal
	Dissolved organic matter	↑ dissolved organic matter = ↓ pathogen removal
	Presence of biofilm	Presence = ↑ pathogen removal
	Temperature	↑ temperature = ↑ pathogen removal
	Water flow velocity	↑ water flow velocity = ↓ pathogen removal
Chemical	Ionic strength	↑ ionic strength = ↑ pathogen removal
	pH	Varies based on bacterial species
Microbiological	Hydrophobicity	↑ hydrophobicity (of pathogen and/or media) = ↑ pathogen removal
	Pathogen concentration	↑ pathogen concentration = ↑ pathogen removal

The importance of sorptive and ion exchange media properties on the potential uptake of PPCPs and pathogens suggests that potential private sector partners could be those working with various natural and engineered materials. More specifically, private entities whose efforts are currently focused on developing materials for nutrient and overall BOD removal may be able to expand their horizons to PPCPs and pathogens in an OWTS application. Ixom Watercare serves municipal and industrial clients worldwide by supplying water and wastewater products and services, including MIEX® Resin and treatment systems (www.miexresin.com). These resins are typically used in centralized water and wastewater applications for the removal of dissolved organic carbon, color, nitrate, arsenic, sulfide, bromide, and chromium, with subsequent regeneration of the resin for reuse using a concentrated brine solution. There is a lack of research regarding the use of these resins in OWTS for PPCP removal, although the resin's kinetics, ease of fluidization, settlability, and selectivity for organic compounds may lend themselves to such an application. Similarly, Calgon Carbon (www.calgoncarbon.com/wastewater) is an industry leader in the area of activated carbon, and to a lesser extent ion exchange, for centralized water and wastewater applications. Powdered and activated carbon are advertised as effective removal strategies for soluble organic chemicals, endocrine disruptors and other contaminants of emerging concern. However, the behavior of these products is unknown in OWTS applications, which have configurations, loading rates, residence times, and other conditions that differ from centralized facilities. Lastly, biochar shows potential for the enhancement of soil treatment units due to its surface area and sorptive capacity, thus highlighting biochar producers as potential OWTS collaborators. For example, Mohanty et al. (2014) found that biochar filters retained up to 1,000 times more E. coli than sand filters and that removal was maximized by the use of biochar with low volatile matter and polarity. A list of biochar and biochar equipment manufacturers and retailers, primarily located in North America, has been provided by the US Biochar Initiative (http://biochar-us.org/manufacturers-retailers).

3.2.1.2 Biotransformation

Biotransformation of PPCPs has been widely demonstrated in both onsite and centralized wastewater treatment systems, especially under aerobic conditions. Biotransformation is expected to be especially important for the removal of endocrine disrupting compounds. For example, Dong et al. (2015) monitored PPCPs in water and soil samples from an effluent-dependent stream in Tucson, Arizona. Many PPCPs and overall estrogenic activity were quickly removed from the aqueous phase with distance of travel in the river, while those that were not quickly removed were characterized by low biotransformation probabilities and low hydrophobicity. The Barnstable County Department of Health and Environment (BCDHE) (2012) has also suggested that aerobic biotransformation is an important pathway for PPCP removal, as demonstrated by increased removal of two PPCPs in an aerated soil treatment unit as compared with a conventional soil treatment unit. The Suffolk County Comprehensive Water Resources Management Plan (2015) highlighted research suggesting that biotransformation of PPCPs may be particularly facilitated in a sequencing batch reactor (SBR) due to alternating "feast and famine" stages, during which complex chemical constituents are stored (feast period) for later use (famine period).

The reduced concentrations of PPCPs observed in drainfield effluent as compared with septic tank effluent can be primarily attributed to a combination of sorption, ion exchange, and biotransformation processes in the soil treatment unit. Thus, methods for extending soil treatment unit residence time and maximizing oxygen supply are often recommended for facilitation of PPCP removal (e.g., pressurized distribution of septic tank effluent into the soil treatment unit, soil treatment unit venting, maximization of unsaturated vertical travel in soil treatment unit, and minimization of hydraulic loading to the soil treatment unit) (BCDHE 2012). The variability of PPCP subjectivity to these processes, as well as the variability of soil treatment unit designs, results in a wide range of observed removal rates across PPCPs and across sites (Table 3.7).

Table 3.7: Summary of PPCP removal in drainfields across study sites (Schaider et al. 2013)[1]

PPCP	Percent Removal in Drainfield	Number of Systems Used to Generate Percent Removal
Acetaminophen	98 to > 99.9	9
Carbamazepine	10 to 60	2
Sulfamethoxazole	0 to > 95	3
Trimethoprim	33 to > 99.9	2
Caffeine	50 to > 99.9	16
DEET	0 to > 99	8
Nonylphenol	0 to > 99.9	6
TCEP	0 to 80	7
Triclosan	70 to > 95	4

[1] Note that PPCP removal was previously defined as removal of parent compounds from the liquid phase

The biotransformation of PPCPs, and bulk organic matter in general, does not only benefit from the availability of oxygen for aerobic degradation, but can also be further enhanced by oxidation pretreatment. However, this practice is not typically considered an onsite wastewater treatment option due to its active nature. Ozonation has been shown to significantly transform bulk organic matter, converting high-molecular-weight, hydrophobic organic fractions into simpler, low-molecular-weight, hydrophilic organic matter. These changes translate to an increase in the overall bioavailability of the organic constituents, thus benefitting PPCP and organic carbon removal in any downstream biotransformation processes (Snyder, Gunten et al. 2014). Thus, ozonation is commonly implemented prior to biofiltration in centralized treatment facilities. Some PPCPs are also susceptible to oxidation by UV advanced oxidation (e.g., diclofenac), but UV advanced oxidation tends to be less effective than ozone-based oxidation for PPCP removal overall. Barriers to the use of ozone in OWTSs pertain to capital costs and required operation and maintenance efforts. Effective ozonation requires a feed gas preparation unit, an injection pump, relatively high quality water, and dose control/off-gas destruction capabilities (Leverenz, Darby et al. 2006). Ozonia, recently renamed Suez Treatment Solutions (http://www.suez-environnement.com), manufactures and supplies ozone generation equipment which is capable of treating complex organic molecules such PPCPs and pathogens through a wide range of flows from 4 g/h to over 100 MGD for both water and wastewater treatment applications and hosts a local factory in Leonia, NJ. Ozonia has committed to major research efforts for a variety of different industries and could serve as a viable collaborator for onsite wastewater treatment research.

Pathogens in municipal wastewater are also subject to biotransformation in OWTSs, stemming from predation and exposure to inhibitory substances produced by other microbiological constituents. Pathogenic microorganisms may also be less equipped to compete for nutrients in a nutrient-limited environment (Stevik, Aa et al. 2004). For example, Thompson et al. (1990) found that two bacteria, *Flavobacterium* and *Arthrobacter*, survived better in heat-sterilized soil microcosms as opposed to non-sterile soil. The reduced survival rate in non-sterile soil was attributed to “competition from and predation by the indigenous community plus a lack of soluble nutrients” (Thompson, Cook et al. 1990). Biotransformation of pathogenic microorganisms is favored by aerobic conditions. For example, Pundsack, Axler et al. (2001) found that intermittently/vertically-fed, predominantly aerobic sand and peat filters were capable of 7 to 8 log removal of seeded *Salmonella*, while intermittently aerobic-anaerobic subsurface flow constructed wetlands achieved 2 to 5 log removal. Amador et al. (2014) hypothesized that climate change will negatively impact biological mediation of pathogens in soil treatment units due to higher temperatures lending themselves to lower oxygen solubility and higher microbial consumption of oxygen.

Research and findings related to biotransformation of PPCPs and pathogens suggest that this removal pathway is significant and potentially enhanced by SBR-type systems and aerobic conditions. Thus, three potential partners in the private sector include SoilAir, Norweco, and SBR Wastewater Technologies. SoilAir Systems (www.soilair.com) involve the intermittent aeration of the soil treatment unit, as opposed to tank aeration. The manufacturer claims that the result is “rapid rejuvenation of failed septic systems,” as well as enhanced BOD, pathogen, and nutrient removal. SoilAir has partnered with universities and third party test organizations in the past. Norweco (www.norweco.com) carries residential OWTSs with extended aeration and attached growth, which could be monitored and potentially optimized for maximum PPCP and pathogen removal. Systems by SBR Wastewater Technologies (www.sbrww.com) may also show potential for PPCP and pathogen monitoring and removal optimization due to evidence of SBR enhancement and the simplicity of designs by SBR Wastewater Technologies (e.g., one tank for aeration, settling, and decant cycle). In general, consistent removal efficiencies over time must be demonstrated because results typically show substantial temporal and geographic variability. Variation is often a function of factors outside user control, e.g., temperature, water table, formation of preferential pathways in natural media, thus highlighting the need for more controllable, predictable systems.

3.2.1.3 Filtration

Physical removal of PPCPs and pathogens by filtration typically refers to that achieved via soil treatment units, engineered granular media filtration, or membrane filtration. Membranes are being increasingly implemented as a PPCP control barrier in centralized systems. Microfiltration membranes are not intended to provide removal of organic chemical contaminants, though some ancillary removal can occur. Studies have reported that this ancillary removal is enhanced by the use of membrane bioreactors (Stanford, Pisarenko et al. 2013; Trussell, Salveson et al. 2013; Snyder, Gunten et al. 2014). For example, a pilot-scale MBR (flow rate of 1.2 m^3 of wastewater per day) was installed and operated for one year at a Swiss hospital to test the removal efficiencies of organic micropollutants. Influent and effluent flows were monitored for 56 pharmaceuticals, ten pharmaceutical metabolites, and two corrosion inhibitors. Iodinated x-ray contrast media was resistant to biotransformation, 11 out of 12 antibiotics were eliminated by less than 60%, beta-blockers/other cardiovascular system preparations were eliminated by less than 55%, and some nonsteroidal anti-inflammatory drugs and analgesics showed no evidence of being removed. However, trimethoprim (an antibiotic) was removed by 96%, other non-steroidal anti-inflammatory drugs and analgesics were removed by 92%. Overall, the MBR system reduced the PPCP load in the hospital wastewater by 22%, with the overall low removal rate being mostly attributed to the prevalence of iodinated x-ray contrast media and its resistance to biotransformation. Excluding influent/effluent loads from the x-ray contrast media, the overall PPCP mass elimination was 90%. MBR systems are even marketed for the treatment of wastewater from pharmaceutical manufacturers, not just municipal wastewater containing low levels of pharmaceuticals (e.g., ZeeWeed MBR system, GE Power & Water 2011). Ovivo (www.ovivowater.com) is also a leader in removing complex chemical compounds and pathogens in water and wastewater membrane treatment.

Multiple studies have demonstrated the ability of reverse osmosis membranes to reject organic and inorganic chemical contaminants, with the notable exception of NDMA, 1,4-dioxane, and several low molecular weight molecules including carbonaceous disinfection byproducts (Kimura, Amy et al. 2003; Steinle-Darling, Zedda et al. 2007; Plumlee, López-Mesas et al. 2008; Stanford, Pisarenko et al. 2013; Trussell, Salveson et al. 2013). Despite its well-known ability to remove PPCPs, high pressure membrane filtration tends to be practiced in centralized settings as opposed to onsite wastewater treatment systems due to its operational requirements, energy footprint, and production of a concentrate stream that requires disposal.

Although filtration of PPCPs typically pertains to membrane filtration, pathogen removal by filtration in OWTS more often refers to straining via infiltration in a soil treatment unit or filtration in an engineered granular media filter system. Pathogen removal by this mechanism depends on grain size of the porous media, pathogen cell size and shape, and the moisture content/hydraulics of the media (e.g., saturated vs. unsaturated, clogging, preferential pathways). Stevik et al. (2004) stated that “straining generally becomes an important removal mechanisms when the average cell size of the bacteria is greater than the size of 5% of the grains that compose the porous material”, the extent of which tends to be greater in unsaturated media. The combination of biological treatment and membrane filtration in MBRs has also proven to be effective for pathogens. Zhou et al. (2015) reported 0.2 to 0.4 log removal of pathogens by fine screens in preliminary treatment, and then 1.3 to 1.7 log removal of bacteria and viruses in biological treatment, followed by 0.7 to 4.7 additional log removal in the MBR.

The synergistic effects of biotransformation and membrane filtration on PPCP and pathogen removal suggest that additional research and comparative evaluations of existing and modified MBR systems are needed. MBR OWTSs are available on the market and a better understanding of how these systems compare in terms of removal of PPCPs, pathogens, and other wastewater constituents. Furthermore, large-scale (e.g., ZeeWeed) and small-scale system performance should be compared in order to identify any discrepancies between systems and further define factors requiring modification in scaled down systems. Potential private sector collaborators in this area are Smith and Loveless (www.smithandloveless.com; TITAN MBR), BioMicrobics (www.biomicrobics.com; BioBarrier MBR), General Electric Water (www.gewater.com; ZeeWeed) and Ovivo (www.ovivowater.com). Furthermore, Busse Green Technologies (http://www.busse-gt.com), manufactures small scale submerged membranes in the range of 250 to 2000 Gal/d therefore more directly oriented toward this market.

3.2.2 Literature Review Summary

As this review indicates, a large variety of nitrogen reduction and PPCPs removal technologies exist and are available for use with onsite wastewater treatment systems. These existing and emerging innovative and alternative wastewater treatment systems span a range of nitrogen reduction processes. Physical/chemical (P/C) nitrogen reduction processes are not typically used for OWTS, and they have found limited application for municipal applications because they have been found to be more expensive and more problematic when treating dilute wastestreams. From a research perspective, P/C methods could be investigated further in an academic setting and are included in the source separation technology classification. To simplify organization, three general onsite nitrogen reducing technology classifications were developed to include:

- **Biological Nitrification/Denitrification Processes**
  - Single Sludge Sequential BNR
  - Single Sludge with Preanoxic Recycle BNR
  - Two Sludge, Two-Stage BNR (denitrification with reactive media)
- **Soil, Plant and Wetland Processes**
  - Soil Treatment Unit Infiltration
  - Soil Treatment Unit Modification Nitrification/Denitrification
  - Constructed Wetlands
- **Source Separation**
  - Urine Source Separation and Recovery
  - Greywater Source Separation
  - Black Water Source Separation

Single sludge biological nitrogen removal systems managing biological nitrification/denitrification utilizing pumps, aerators and controls has been the preferred method for most nitrogen reduction applications for OWTS. Biological nitrogen removal through two sludge, two-stage BNR using reactive media for denitrification is gaining recognition as a robust treatment alternative for achieving low effluent nitrogen levels. Soil, Plant and Wetland processes showing promise are those utilizing STU modifications incorporating nitrification and denitrification layers with an external carbon source. Source separation is an emerging option as the technologies improve and the nutrients recovered are increasingly valued.

The assessment in Section 3 points out the challenges in developing a sustainable nitrogen reduction onsite wastewater treatment system which includes minimizing chemical, energy and labor (operation and maintenance) inputs; eliminating the need for frequent sludge handling; and creating an effluent quality suitable for reuse in non-potable applications. Therefore, based on the data gained from this review combined with the project team's experience, we identified several general system types recommended for further detailed analysis for the technology ranking assessment (Section 5). These include:

1. **Biological Nitrification/Denitrification Processes**
    a. Single Sludge BNR utilizing fixed film porous media recirculating biofilters
    b. Single Sludge nitrification or BNR units followed by a reactive media denitrification soil treatment system
    c. Two Sludge, Two-Stage porous media biofilter utilizing reactive media for denitrification
2. **Soil, Plant and Wetland Processes**
    a. Soil Treatment Unit Modification for Nitrification/Denitrification utilizing reactive media for denitrification
3. **Source Separation**
    a. Urine Source Separation and Recovery

4. Onsite Wastewater Treatment Patent Search

This section includes a summary of technologies related to onsite wastewater treatment which were discovered as a result of patent searches through the United States Patent and Trademark Office (USPTO) and the European Patent Office (EPO) database, Espacenet. The intent of the patent search is to discover novel and unique wastewater treatment ideas which have been published but not necessarily marketed or produced and are applicable to OWTS systems. An important point to remember regarding the technology presented in the patents is that the idea must be novel, unique and non-obvious and someone "skilled in the arts" must be able to reduce the idea to practice. However, the ideas do not necessarily have to have the proper scientific rigor or practical application to actually perform and are best reviewed for unique approaches toward developing products, licensing the technology for market or simply expanding knowledge for further research.

4.1 Overview

Patent searches originated through PUBS East, the USPTO examiner's search engine. This limited access search engine, available only through a Patent and Trademark Resource Center (PTRC), was used to search through over 1000 published patent applications and grants using onsite wastewater treatment search phrases noted in Table 4.1. Patent applications (distinguished by the year prefix) and grants summarized below do not include treatment technologies which are already on the market. The intent of searching for published patent grants and applications was to determine a variety of novel approaches, not already considered, but that could be applied toward OWTS. The selected technologies and approaches may be considered novel treatment systems in themselves; or simply unique approaches to enhance or resolve traditional process problems within wastewater treatment. Furthermore, the "assignee" of the patent, provides information on private enterprises that have developed the technology for further consideration for future collaboration with CCWT. Reference patents and cited literature within either the application or patent grant can also be used to pursue other related technology which might be considered useful for further research and development. The patents cited as examples are grouped into four different process categories, by most recent date of publication.

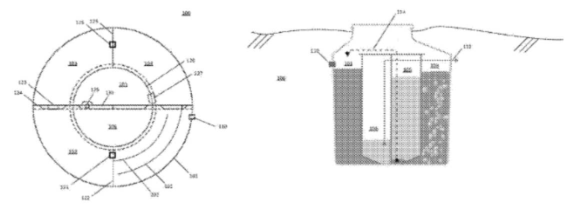
Table 4.1: Patent Search Terms Used

Search Phrases	
On-Site Wastewater Treatment	On-Site Sewage Treatment
On-Site Wastewater System	Residential Sewage Treatment
Small Scale Wastewater Treatment	Pharmaceutical Wastewater Treatment
Residential Wastewater Treatment	Decentralized Wastewater Treatment
Small Scale Sanitation	Septic Treatment System
Decentralized Sanitation	Decentralized Sewage Treatment

4.2 Patent Search Results

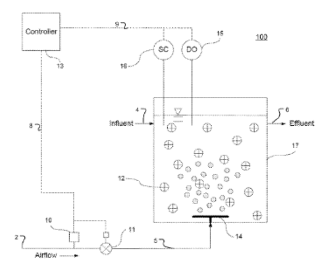
4.2.1 Biological Processes

SYSTEM AND METHOD OF TREATING WASTEWATER
2016/0039695A1
February 11, 2016



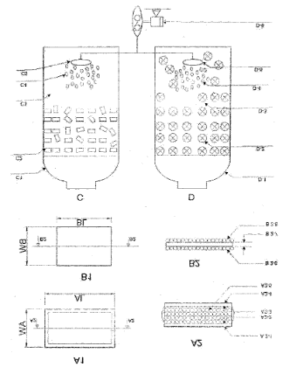
The patent application discloses an arrangement of concentric baffles and chambers within a structure to minimize dead space, reduce turbulent flow, and increase detention time to enhance the contact time of wastewater with air. This patent illustrates one novel approach in the design of wastewater structures to maximize treatment in smaller footprints.

METHOD FOR DEAMMONIFICATION PROCESS CONTROL USING PH, SPECIFIC CONDUCTIVITY, OR AMMONIA
2016/0023932 A1
January 28, 2016



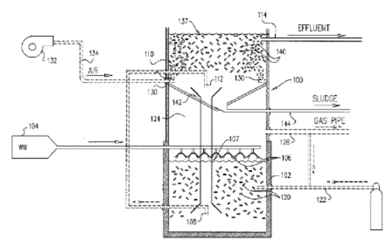
The patent application discloses methods and systems through a deammonification MBBR process where partial nitritation and anaerobic ammonium oxidation may occur simultaneously in a biofilm or in an integrated fixed film activated sludge process. Air is controlled using a control scheme that targets pH, alkalinity, specific conductivity, or ammonium. Further research and development in the area of a simplified control scheme and means to obtain these measurements could lead to lower energy and footprint for onsite systems such as an SBR which are well suited for single tank processing and deammonification.

IMPROVED FERMENTATION PROCESS AND PRODUCTS USEFUL FOR THE SAME
2016/0002079 A1
January 7, 2016



The patent application discloses an improvement in media used for Moving Bed Biofilm Bio Reactor (MBBR). Typically, MBBR processes use plastic “honeycomb” media which serve to increase the surface area and provide adhesion for the microorganisms. This patent improves upon previous methods by using a combination of activated carbon and other types of media. One particular claim describes method and materials using small sachets/pouches filled with activated carbon comparable to the cost of other commercially available MMBR media. Research and development through this approach has the potential to lead to more optimized processing and reduced footprint for a variety of different onsite system configurations.

DYNAMIC ANAEROBIC AEROBIC (DANA) REACTOR
8758613 B2
June 24, 2014



The patent discloses a combined reactor where wastewater is introduced in a lower anaerobic chamber followed by circulation of the wastewater to an upper aerobic MBBR chamber utilizing a combination of pressurized methane and carbon dioxide gas produced by the anaerobic treatment byproduct below and supplementary air to assist in the movement and mixing of the wastewater. The patent claims to be effective for many heavy industrial applications to include the treatment of emerging contaminants and pharmaceuticals. The patent illustrates a novel configuration and biological treatment within a single structure that could be similarly adapted to onsite systems with the potential to treat PPCPs if combined with, for example, ozonation.

WATER TREATMENT SYSTEM FOR SIMULTANEOUS NITRIFICATION AND DENITRIFICATION
2012/0145611 A1
June 14, 2012

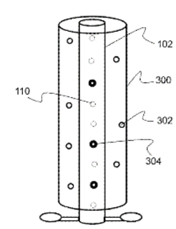
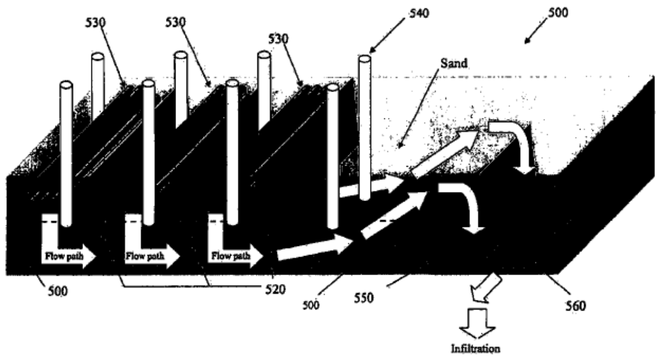


FIG. 3

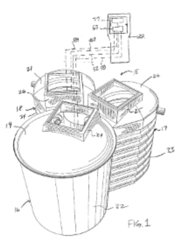
This patent application discloses the concept of a counter-current exchange through the mechanism of diffusion and convection by using one perforated column within another perforated column where one column is aerobic for nitrification and the other column is anoxic. The application claims that the present invention has been used in experiments with high ammonia concentration belt press filtrate and can be used for secondary wastewater treatment applications. The application describes the conversion of ammonia through a series of enzyme reactions in the presences of aerobic and anoxic conditions, but does not specify further the conditions or details of the mechanism. The application does use an external means of supplying aeration through a blower, pump or mixer device. The patent illustrates a novel geometry which could be adapted to onsite systems to minimize material and footprint, combined with other approaches that maximize nitrification and denitrification mechanisms.

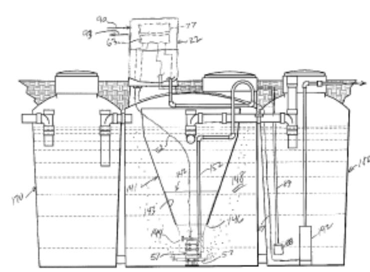
PASSIVE UNDERGROUND DRAINFIELD FOR SEPTIC TANK NUTRIENT REMOVAL USING FUNCTIONALIZED GREEN FILTRATION MEDIA
7927484 B2
April 19, 2011



The patent discloses methods, systems and compositions of green sorption media for use as bioretention soil amendments in drainfields for on-site waste water systems. The patent, based on research at the University of Central Florida, claims media compositions which include one or more recycled materials such as tire crumbs, sawdust, orange peels, coconut husks, leaf composts, oyster shells, soy bean hulls and one or more naturally occurring materials including peat, sands, zeolites, and clay. The media is contained within baffled cell compartments with air risers which provide alternating cycles of aerobic and anoxic environments. The patent illustrates alternative, readily available media in a novel configuration to facilitate passive wastewater treatment.

WASTEWATER TREATMENT SYSTEM
7998343 B2
August 16, 2011

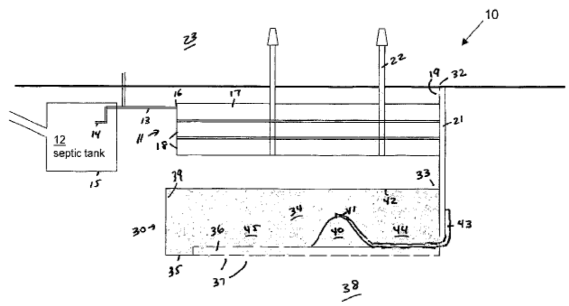




The patent discloses a novel configuration of three subsurface aeration tanks nested together where one aeration tank includes an inverted frusto-conical clarifier in the center. A pump within the center of the clarifier is used for aeration and mixing. Through hydraulic displacement, water rises upward through the clarifier and the inverted side slope drops solids to the bottom for further processing. Onsite system design could benefit from similar combinations of process treatment tanks within one structure.

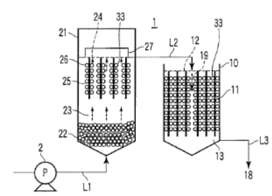
PASSIVE DRAIN FIELD SYSTEM FOR WASTEWATER TREATMENT AND ASSOCIATED METHODS
7632408 B1
December 15, 2009

The patent discloses a stacked passive nitrification and denitrification system where the discharge from the septic tank enters a multi-pipe bundle where air enters through vents. The denitrification chamber below is enclosed within a water impermeable layer which retains the wastewater for saturation of the media below to maintain anaerobic conditions. The patent illustrates alternative treatment configurations which reduce footprint and provide non-mechanical aeration.



AERATION-LESS WATER TREATMENT APPARATUS
2009/0032451 A1
February 5, 2009

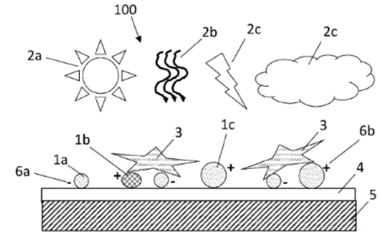
The patent application discloses a two tank configuration comprising of an anaerobic section from which wastewater is pumped up through different claimed mesh types and a suspended fixed carrier systems followed by discharge to an aerobic tank. Aeration is provided by pulling in atmospheric air from the top of the secondary tank. The invention claims that the suspended carrier also serves to capture the anaerobic bacteria from flowing from the anaerobic chamber to aerobic chamber and helps to retain the selected anaerobic or aerobic biomass in each respective chamber. Onsite design could benefit from physical mechanisms which supply necessary aeration from atmospheric air without the use of electrical or mechanical devices.



4.2.2 Physical/Chemical Processes

BIOCONTROL ACTIVITY SURFACE
2016/0050916 A1
February 25, 2016

The patent application promotes novel approach to applying different surface and electrical potentials on a substrate or media using nanoscopic features in a variety of configuration to effect biocontrol activities. The arrangement of nanoparticles would be influenced by exposure to different environmental conditions such as heat, light pH. Recent advances in 3D printing and surface deposition on substrates deserves further research and development to explore coatings to enhance and manipulate the surface of media to optimize biological wastewater treatment mechanisms so that onsite systems can perform a wider range of functions in smaller footprints.



PORTABLE UV DEVICES, SYSTEMS AND METHODS OF USE AND MANUFACTURING
2015/0359915 A1
December 17, 2015

The patent application discloses a novel approach to bringing a portable UV which can be enclosed within a variety of different types of spaces. Typically UV systems are flow through vessels or confined spaces. Onsite system design could benefit from this approach where disinfection and treatment of emerging contaminants are required and the lamps can be adapted to a variety of different shapes and sizes to effect different treatment levels.

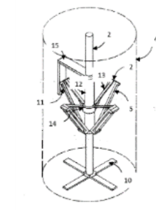


FIG. 7

RESIDENTIAL WASTEWATER PURIFICATION SYSTEM
8828240 B1
September 9, 2014

The referenced patent discloses a system directed to locations where land area is limited for drainage fields. This system claims to provide electrocoagulation and flocculation using electrodes and electrical energy to convert some of the dissolved material present in the wastestream to a suspended particulate form that can be subsequently filtered and separated out. The system also claims to uses anaerobic and aerobic digestion, filtration, exposure to ultraviolet radiation, reverse osmosis processing for a complete treatment system with a modest supply of externally supplied energy. The concept of using electrical energy to suspend wastewater particulate illustrated within this patent may have some application in onsite systems where solids must be removed throughout the treatment scheme provided the cost of materials and energy to support this method is less expensive compared to other traditional methods of solids removal.

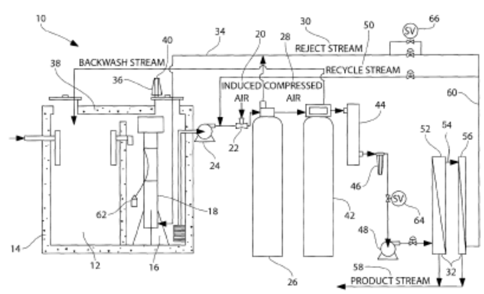


FIG. 1

ELECTROCHEMICAL SYSTEM AND METHOD FOR THE TREATMENT OF WATER AND WASTEWATER
8460520 B2
June 11, 2013

The patent discloses a comprehensive electrochemical treatment system which utilizes direct electrical current through electrochemical cells consisting of electrode rods which run parallel with the direction of wastewater flow. The mechanisms employed include electro-coagulation, electro-flocculation, electro-floatation, electrochemical oxidation, electro-charge reduction, electrolysis of water and production of free radicals electrical charge neutralization, and electroplating. The patent claims to treats a wide range of contaminants to include pharmaceuticals in urine and feces, destruction of pathogens, and long chain and complex organic compounds. This recent patent, which references a wide range of published patents and journals, illustrates the current state of art for treating emerging contaminants. While this noted patent is too sophisticated for low cost onsite treatment, further research and development directed toward

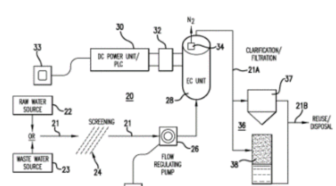


FIG.1

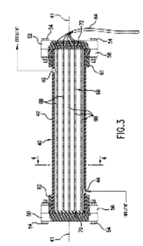


FIG.3

electrochemical methods for specific contaminants may reveal cost effective solutions applied to some aspects of onsite treatment.

POROUS COMPOSITE MEDIA FOR REMOVING PHOSPHORUS FROM WATER
2013/0098840 A1
April 25, 2013

The patent application discloses the use of nano-engineered porous ceramic composite filtration media for the removal of phosphorous from wastewater. The invention claims a variety of different compounds and porosity ranges that are modified and grown on a ceramic substrate. The application provides both experimental research conducted and methods of preparation. Novel approaches in the manufacturing of different substrates as illustrated in this disclosure would benefit the design of media specifically selected to alter physical parameters such as pH and ion exchange for different onsite wastewater treatment efficacies.

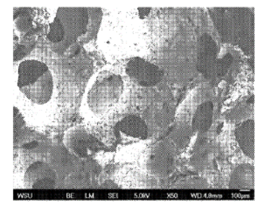


FIG. 1A

METHOD AND SYSTEM FOR REMOVING ORGANIC COMPOUNDS IN A CLOSED LOOP SYSTEM
8419858 B1
APRIL 16, 2013

The patent discloses the use of oscillating ultraviolet (UV) lamps combined with peracetic acid to accelerate the decomposition of organic compounds from water and wastewater. The patent claims a method utilizing UV lamps in the wavelength range of 300nm to 800 nm to excite electrons in the chemical compounds, are embedded within a light and water permeable porous cartridge. Further research and development of similar small scale UV systems and configurations will be necessary to resolve the issue of disinfection and treatment of emerging contaminants of concerns and pathogens.

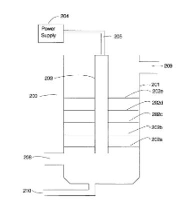


FIGURE 2

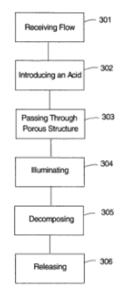


FIGURE 3

SOLAR ENCLOSURE FOR WATER REUSE
2012/0234771 A1
September 20, 2012

The patent application discloses a system and a method for treating wastewater using concentrated solar energy. The ideas within this patent are applicable for novel low energy onsite wastewater disinfection and reuse applications. The application preferably proposes attachment of the solar collector to a building facade. The disclosure claims disinfection would be limited to desalination, pasteurization, and viral/pathogenic removal from gray water building source water. The patent illustrates novel energy harvesting and reuse principles which are critical to reducing the cost of materials and energy applicable to onsite wastewater treatment technology.

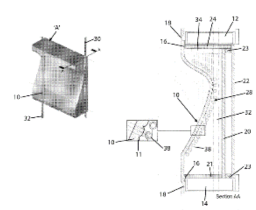
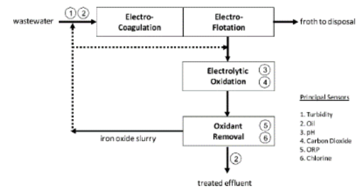


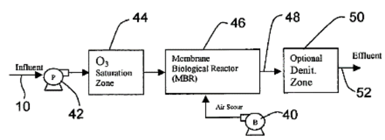
FIG. 8

APPARATUS AND METHOD FOR ELECTROCHEMICAL TREATMENT OF WASTEWATER
2012/0160706 A1
June 28, 2012



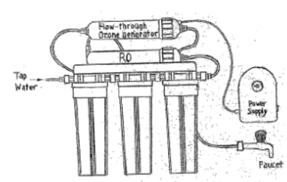
The patent application discloses the treatment of wastewater through electrocoagulation using anode and cathodes and an electro-oxidation unit for oxidizing contaminants. Referred to as a Wastewater Electrochemical Treatment Technology (WETT), the electrodes produce Hydrogen gas which combine with, precipitate or float a range of wastewater contaminants and particulates. The patent application reports among others, removal of TSS, Oil, BOD, COD, and Fecal Coliform for example. The application illustrates a novel approach to the use of electrodes and this type of treatment may be considered in some aspect of onsite wastewater treatment provided that further development reduces the cost of materials and energy associated with this mechanism.

WASTEWATER TREATMENT METHOD AND SYSTEM WITH OZONATION FOR MICROCONSTITUENT REMOVAL
8268174 B2
September 18, 2012



The patent discloses a system of treating personal care products and pharmaceutical, referred to as micro constituents, by combining ozone followed by a Submerged Membrane Bioreactors (MBR). The patent claims that ozone is introduced in an aerobic component of the plant at concentrations between 25 mg/l to 100 mg/l and serves to breakdown refractory micronutrients which are then more easily biodegradable by an MBR process**.** The patent also claims that ozone creates oxygen bubbles which serve to provide aeration to meet BOD and nitrogen removal and provide for required air scouring of the membranes. The Patent reports that MBR technology has a unique advantage over Conventional Activated Sludge (CAS) systems because sludge concentrations are more than three times higher allowing for longer SRT given the same volume which is important biodegradation of long complex organic compounds. Research directed toward understanding the cost effective method of delivering ozone in conjunction with current onsite membrane systems could be benefited through the approach this patent discloses.

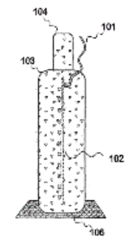
OZONE GENERATORS
2010/0135869 A1
June 3, 2010



The patent application discloses a low energy portable powered ozone generator that is claimed to provide wastewater disinfection by converting electrical energy to ozone. Research directed toward applications in onsite wastewater reuse, disinfection and emerging contaminants could benefit from this novel and simple approach.

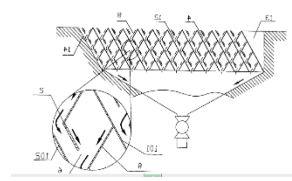
ELECTRODES FOR THE ELECTROLYSIS OF WATER
2012/0037512 A1
January 28, 2010

The patent application discloses a non-toxic concrete coated, metallic or carbon fibre core electrode used for facilitating secondary and tertiary wastewater treatment by encouraging the growth of algae and aerobic bacterial by freeing up available hydrogen and oxygen through electrolysis. Concrete is claimed to increase the life of the electrodes. The invention claims that the electrode may be powered by a 12 volt battery or solar power. Research directed toward durable electrolysis principles for onsite treatment would benefit from the ideas illustrated in this disclosure.



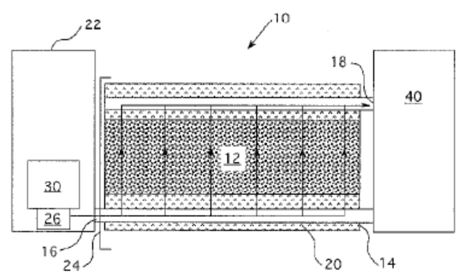
HORIZONTAL-TUBE SEDIMENTATION-SEPARATION APPARATUS
2010/0018916 A1
January 28, 2010

The patent discloses a unique geometric configuration which attempts to improve the sedimentation of wastewater treatment, in particular clarification by creating a static region within the flow patch thus capturing greater solids by gravity and limiting solids washout. Research directed toward alterations in changes to onsite structural geometry could benefit from the ideas illustrated in this patent in order to modify basic physical mechanisms which effect wastewater processes.



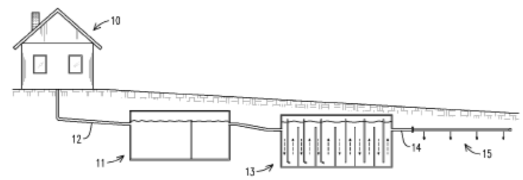
4.2.3 Soil, Plant and Wetland Processes

METHOD FOR REMOVING DRUGS FROM WASTEWATER UTILIZING NUETRALIZED BAUXITE RESIDUE
9187342 B2
November 17, 2015



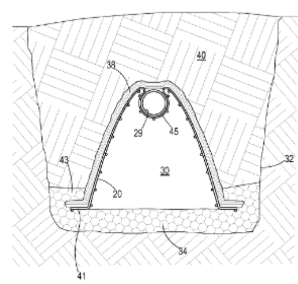
The referenced patent discloses a method and system using filtration media for removing pharmaceuticals, personal care products and microorganisms and pathogens. The application claims that the system can remove emerging contaminants using a mixed metal oxide bed (MMOB) when combined with at least 5 PPM dissolved oxygen. Metal oxides include iron oxide, alumina, silica, titanium dioxide zeolite, and bauxite ore. The patent references experimental data from a constructed natural wetland wastewater treatment system referred to as an Engineer Natural System with a design flow of 45,000 gpd. The patent illustrates a novel approach to media selection which could be further researched for onsite systems directed toward the treatment of PPCPs.

RECOVERY OF RESOURCE FROM WASTEWATER
2015/0239761 A1
August 27, 2015



The patent application discloses an onsite system using a multi-chamber ion exchange bioreactor followed by anaerobic section for the treatment of wastewater and recovery of nitrogen. The ion exchange media is granular zeolite or mixtures thereof. The invention claims that the spent media, trapped with nitrogen is then further used for direct soil application in agriculture to increases water retention capacity in the soil, cation exchange capacity and provides for a slow release of fertilizer. Further research directed toward full recovery of nutrients and water for reuse applications could benefit from the novel ideas in this approach.

ROLL-FORMED CONDUIT ARCH FOR LEACH FIELD
2014/0212219 A1
July 31, 2014



The patent application for a conduit arch skeletal framework which is utilized in excavated trenches to form a leach field allowing the dispersal of wastewater throughout the trench without a distribution pipe or crushed stone. Furthermore, the invention claims to allow for substantial evaporation and air infiltration into the trenches to encourage aerobic degradation of organics and ammonia. Materials used are readily available wire mesh fencing. The application provide descriptive design configurations and methods of construction for the arch tunnel formed and recommended depths of layered media above and below the arch to ensure proper loading on the surface. This novel approach could be applied toward passive layered soil treatment systems or other onsite configurations which provide non-mechanical supply of oxygen with simple off the shelf materials.

LEACH FIELD SYSTEM
2013/0126407 A1
May 23, 2013

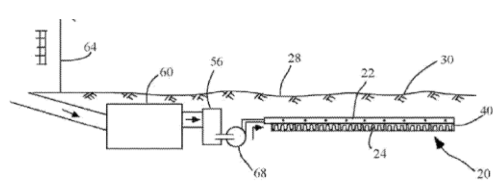
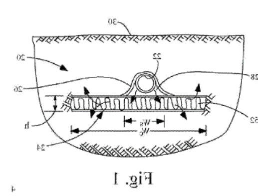
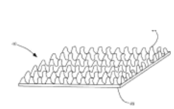
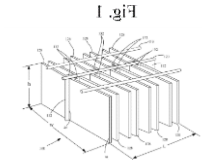


Fig. 9







The patent application discloses a leaching field form to facilitate the aerobic treatment of wastewater within soil. The invention also discloses a method of making a leach field using the form. The present invention improves the geometry and reduces the aspect ratio (height divided by width) so that the bottom of the conduit can be positioned closer to the surface of the soil and increases the void space and air-soil gas interchange so that oxygen is continuously supplied to the soil to enable good biodegradation treatment. Proposed advantages of this invention include utilizing shallow depths of native soil above high water tables or rock ledges. Shown below is perforated dosing pipe over a low aspect channel incorporating the geotextiles or referred to as geonets within this application provided by others. Onsite system design will benefit from the novel approach to altering structural geometry to increase non-mechanical means of aeration to natural systems.

SUBSURFACE UPFLOW WETLAND SYSTEM FOR NUTRIENT AND PATHOGEN REMOVAL IN WASTEWATER TREATMENT SYSTEMS
8252182 B1
August 28, 2012

The referenced patent claims novel methods and unique green sorption media filter blends to remove nutrients and pathogens using different mixtures of materials and plant species that provide for sorption, ion exchange, chemical precipitation, biological uptake and filtration in mixed aerobic, anoxic and anaerobic environments. Various mathematical models are presented based on experimental data from wastewater at the University of Central Florida which provide for a variety of different configurations of the layered media. Natural and passive onsite design can benefit from the data presented in this patent for the selection of media suited for different treatment efficacies.

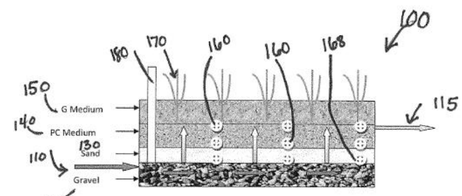
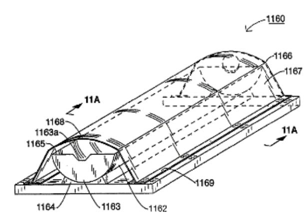


Fig. 1

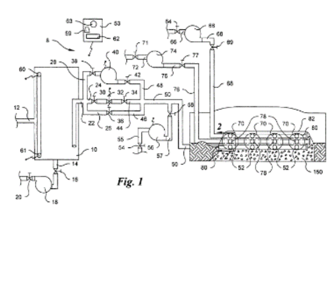
WASTEWATER TREATMENT AND DISPERSAL SYSTEM
7300577 B1
November 27, 2007

The patent discloses a structure, in-conjunction with a separate treatment system which replaces typical dispersal conduit and extends the detention time to maximize detention time for air/oxygen contact for further biological treatment while simultaneously routed to a dispersal field. The novel structure disclosed illustrates utilizing all structural aspects of the complete infrastructure to treat some aspect of the wastewater.



WASTEWATER BIOLOGICAL TREATMENT SYSTEM AND METHOD THEROF
7022235 B2
April 4, 2006

The patent discloses a method of using a variety of perforated drain field pipe configurations within one another which serve to extend and provide waste water treatment in addition to dispersal. Similar to patent, 730577, configuration of the onsite infrastructure illustrated in this patent can benefit design of onsite systems which extend beyond the tank.



4.2.4 Source Separation

SEPARATOR AND COMPOSTING SYSTEM AND METHOD
2015/0191386 A1
July 9, 2015

The patent application approaches source separation of urine and solids through a rotating drum in which liquid is separated through the slots in the walls of a core assembly referred to as a separator chamber. A parallel and connected draining chamber rotates as a different speed, which further removes liquid from the solids. The solids are discharged to a second rotating composting drum for further processing and disposal. The devices is designed to insert within sanitary plumbing. This approach illustrates alternative approaches to selectively removing urine from the source for further consideration in the removal of concentrated nitrogen in urine from the wastewater stream prior entering an onsite treatment scheme.

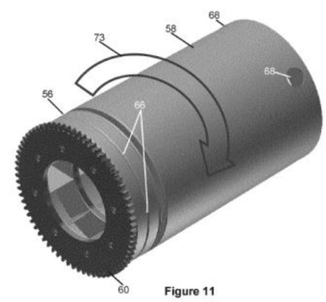


Figure 11

IN-SITU SELECTIVE CONTAMINANT ADSORPTION IN URINALS AND TOILETS
US 2014/0008303 A1
January 9, 2014

This patent application is a simple absorption device consisting of an ion exchange resin, either cation or anion which may be inserted into a standard toilet drain to selectively remove calcium and magnesium ions from urine in for waterless toilets to inhibit scale and blockage in the downstream piping. The patent application also claims use for selectively removing pharmaceuticals and has an accessible colormetric indicator which allows the user to determine when the urine has exceed the absorption of the exchange media device. This approach to removing select nutrients and PPCP removal from the waste streams prior to downstream onsite treatment systems; or the removal feature of the device may play a role in the actual configuration of the onsite system itself for removal.

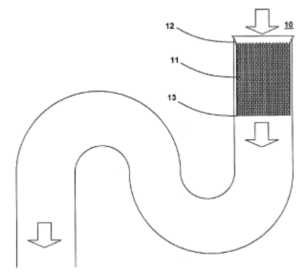
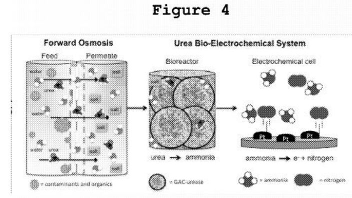


Figure 1

UREA-BASED SYSTEM FOR ENERGY AND
WASTE RECOVERY IN WATER RECYCLING
US 2014/0061127 A1
Mar. 6, 2014

This patent application is sponsored by NASA for the purpose of recovering ammonia from urine though a granular activated carbon (GAC)-urease bioreactor during space flight. The ammonia recovered from the urea is then used to feed an electrochemical cell for energy capture. This device would be used in conjunction with a forward osmosis subsystem for further reclamation of water. The application provides a thorough background on the development of different

Figure 4


technologies utilized for recovering water from urine during space flight with referenced experimental data. This represents another approach to harvesting components from a urine stream which may be utilized for energy or other applications for smaller scale onsite systems.

4.3 Patent Search Summary and Recommendations

The selected patents, referenced throughout the report and Section 4.2, illustrate novel approaches for resolving process problems which contribute to OWTS limitations. The following areas, categorized by main process, are recommended for further research.

- **Biological Nitrification/Denitrification Processes**
  - Approaches to novel geometric tank configurations which enhance hydraulics, nutrient and solids removal and aeration during wastewater treatment in order to minimize footprint and energy.
- **Physical/Chemical Processes**
  - Engineered material and surface properties which effect physical, chemical and biological behavior during wastewater treatment, such as biofilm extracellular polymeric networks (biofilm slime layers) and reactive media
  - Low cost approaches to adapting UV and Ozone combined with activated sludge OWTS to assist in PPCP treatment, pathogen disinfection and water reuse
  - Low cost approaches to electrical-chemical processes for treatment of wastewater nutrient and solids removal
  - Solar and wastewater treatment for both disinfection and low cost energy sources
- **Soil, Plant and Wetland Processes**
  - Hybrid plant, native and engineered media in natural systems which remove nutrients, PPCPs, and pathogens
  - Recovery of water, nutrients and energy from natural plant and wetland processes to achieve energy neutral wastewater treatment
- **Source Separation**
  - Novel structures and nutrient removal treatment methods adapted to household plumbing components which reduce the biological loading to OWTS.

5. Nitrogen Reduction Technology Ranking Assessment

The studies and data identified in the literature review were synthesized to develop matrices categorizing the major processes, stage of development, treatment effectiveness, operability, complexity, energy use, and other considerations for single family home nitrogen reducing OWTS These findings are synthesized into key insights, followed by identification of knowledge gaps and opportunities for the CCWT, including opportunities for pilot and full scale implementation as well as more fundamental research opportunities for existing conceptual ideas.

A simple numerical ranking system was developed to prioritize available nitrogen reduction systems based on thirteen selected criteria. Each criterion is scored against its particular attribute using a scale ranging from 1 to 5. To account for relative differences in significance of each of the criteria, the criteria are assigned weighting factors indicating relative importance, compared to the other criteria. The priority ranking for a technology is determined by its total score, which is the sum of the products of the individual criterion scores multiplied by the weighting factors for each criterion. The highest score represents the highest priority ranking.

5.1 Criteria Descriptions and Values

A description of each criterion is presented below together with the attributes for the criterion and the value scores that are the basis for scoring of individual technologies.

5.1.1 Effluent Total Nitrogen Concentration

The attribute of this criterion is the average concentration of TN in the final effluent prior to discharge to the STU or to effluent dispersal. It is based on performance that is achieved under suitable conditions with proper and adequate operation and maintenance. Percent removal could also be used to evaluate treatment performance, but literature references reporting effluent concentrations were more common. The criterion values for TN effluent concentration are listed in Table 5.1. TN values used to score a given technology were based on an average of values from various sources, ranging from peer reviewed publications with systems data to manufacturers' websites. The scores represent what the project team determined to be accurate reflections of the system potentials.

Table 5.1: Criterion Values for Total Nitrogen in Effluent

Mean Effluent TN (mg/L)	Score
< 5	5
5 – 10	4
11 – 15	3
16 – 30	2
> 30	1

5.1.2 Performance Consistency

The consistency of performance is defined here as the sensitivity of the treatment system to upset. The standard deviation of final effluent TN concentration provides a measure of the consistency of a technology. The sensitivity of a system is heavily influenced by the treatment process used. Therefore the attribute of the performance consistency criterion is either the standard deviation of final effluent TN (if available) or the type of treatment process used, based on a review of wastewater treatment design guidelines and onsite wastewater treatment performance. The categories for performance consistency are listed in Table 5.2.

Table 5.2: Criterion Values for Performance Consistency

System Process Type	Score
Physical/Chemical & Source Separation	5
Fixed Film Processes	4
MBR / IMB *	3
IFAS **	2
Activated Sludge Nite/Denite	1

* MBR/IMB: Membrane Bioreactor / Immersed Membrane Bioreactor
** IFAS: Integrated Fixed-Film Activated Sludge

5.1.3 Construction Cost

The attribute of this criterion is the total capital cost of system installation, including septic tank with a standardized add-on value used, where necessary. However, available data was not always complete, and therefore engineering judgment and cross-study comparisons were used to attempt to compare costs between technologies. The categories for construction costs are listed in Table 5.3.

Table 5.3: Criterion Values for Construction Cost

Construction Cost ($1000)	Score
< 5	5
5 - 10	4
11 – 15	3
16 – 20	2
> 20	1

5.1.4 CBOD/TSS Effluent Concentration

The attribute of this criterion are the final effluent concentrations of five day carbonaceous biochemical oxygen demand ($CBOD_5$) and total suspended solids (TSS) under suitable conditions with proper and adequate operation and maintenance. Categories for BOD and TSS effluent concentration are listed in Table 5.4.

Table 5.4: Criterion Values for CBOD/TSS Effluent Concentration

Effluent cBOD/TSS (mg/L)	Score
10 / 10	5
20 / 20	4
30 / 30	3
40 / 40	2
> 40	1

5.1.5 Mechanical Reliability

The attributes of the mechanical reliability criterion is expressed as the “mean time between unscheduled service calls”. The frequency of routine service and unscheduled call-outs provides a measure of the reliability of a technology. Factors that can increase the need for service include a high number of mechanical components (pumps, aerators, mechanical mixers), complexity of electrical systems, complexity of design, components prone to failure, and complex equipment that requires specialized parts and training of personnel. The categories for performance reliability are listed in Table 5.5.

Table 5.5: Criterion Values for Performance Reliability

Mean Time Between Unscheduled Service Calls	Score
Annually	5
Semi-annually	4
Quarterly	3
Monthly	1

5.1.6 Land Area Requirements

The attribute of this criterion is the plan area or the size of the additional footprint required for the treatment system, over and above the components for a conventional OWTS (septic tank and STU). Available data for this criterion was limited and significant judgment was required to compare relative land area requirements between technologies. Criterion values for land area required are the footprint area in square feet or the relative difference between land area requirements for system types, based on experience. These are listed in Table 5.6.

Table 5.6: Criterion Values for Land Area Requirements

Land Area Req. (ft^2)	Score
Low (single tank unit or STU modification within conventional STU area)	5
Medium (multiple tanks units)	3
High (STU modification requiring additional STU area or wetlands with pre-nitrification)	1

5.1.7 Restoration of Performance

Treatment technologies occasionally will fail to achieve their performance expectations. Such upsets may be due to electrical or mechanical problems or a process upset. In addition, systems may only be used on a seasonal basis. The time needed to restore treatment is an important criterion in preventing harm to the environment. The consequences of an operational failure are much less significant if treatment efficacy is restored rapidly. Data was generally unavailable for this criterion, so scoring was based on engineering judgment related to the treatment process utilized by a given technology, as noted in Table 5.7. The categories for performance restoration are listed in Table 5.7.

Table 5.7: Criterion Values for Restoration of Performance

System Type	Score
Physical/Chemical & Source Separation	5
Fixed Film	4
MBR/IMB *	3
IFAS **	2
Activated Sludge Nite/Denite	1

* MBR/IMB: Membrane Bioreactor / Immersed Membrane Bioreactor
** IFAS: Integrated Fixed-Film Activated Sludge

5.1.8 Operational Complexity

The attribute of this criterion is the degree of complexity required to operate the system in question. High scoring systems will allow operation by the homeowner with little or no effort or training, while low scoring systems will not. Criterion values for operation complexity are qualitative, and are listed in Table 5.8. Data for this criterion was generally unavailable in most literature reviewed, and engineering judgment was therefore used to score the various technologies based on the knowledge of the process utilized and perceived difficulty in maintaining treatment performance.

Table 5.8: Criterion Values for Operational Complexity

Description	Score
Simple operation with limited operator requirements annual scheduled visit	5
Some specialized operator training required; Scheduled visits by manufacturer's representative required twice per year	3
Complex operation with operator training required; Scheduled visits by manufacturer's representative required > quarterly	1

5.1.9 Energy Requirements

The attribute of this criterion is the annual energy usage of the entire treatment system, including pumps, aerators, and mixing devices. The annual energy requirement is the sum of all energy requiring components or the rate of energy usage in operating the component multiplied by the component operating time. Criterion values for energy requirements are listed in Table 5.9. Greater energy use is associated with more "active" technologies that employ greater numbers of liquid pumps, aeration pumps, and mechanical mixing, whereas unsaturated granular media filters that employ passive aeration would consume less energy.

Table 5.9: Criterion Values for Energy Requirements

Energy Req. (kW-hr/year)	Score
< 500	5
500 – 1,000	4
1,001 – 1,500	3
1,501 – 2,500	2
> 2,500	1

5.1.10 Construction Complexity

The attribute of this criterion is the degree of difficulty necessary to install the system in question. High scoring systems will be simple to install even by an untrained contractor or installer – put it in the ground, plug it in, and it works. Low scoring systems will require substantial training and require an extensive installation process. Criterion values for construction complexity are qualitative, and are listed in Table 5.10. Data for this criterion was generally unavailable in most literature reviewed, and engineering judgment was therefore used to score the various technologies based on knowledge of system components and the perceived difficulty of installation.

Table 5.10: Criterion Values for Construction Complexity

Description	Score
Simple to install by any contractor	5
Some specialized knowledge and training required	3
Complex installation, specialized training, sophisticated electrical and controls knowledge req., master septic tank contractor	1

5.1.11 Local Resources

The attribute of this criterion is the local availability of technology system components and/or materials of construction in Suffolk County. The categories for construction costs are listed in Table 5.11.

Table 5.11: Criterion Values for Local Resources

Local Resources	Score
Readily available	5
Available	3
Not available	1

5.1.12 Climate Resiliency

The attribute of this criterion is a general judgment of the treatment system to demonstrate coastal resiliency (ability to withstand storm damage, and/or long-term ability to mitigate impacts of rising sea level and groundwater tables). Categories for climate resiliency are listed in Table 5.12.

Table 5.12: Criterion Values for Climate Resiliency

Climate Resiliency	Score
No impact	5
Perceived impact	3
Impacted	1

5.1.13 Stage of Technology Development

The attribute of this criterion is the stage in development of the nitrogen reduction technology. Criterion values for stage of technology development are listed in Table 5.13. Systems used nationwide and are approved in other States, or thoroughly tested by NSF, ETV or MASSTC will be assigned the highest ranking, while the lower rankings allow room for consideration of meritorious ideas that have not yet been tested. This would include “experimental” systems, such as those tested in the FOSNRS pilot studies, or “conceptual” system ideas based on processes, components, or operational strategies that have yet to be tested.

Table 5.13: Criterion Values for Stage of Technology Development

Stage of Development	Score
NSF/State Approved	5
Demonstration (full scale)	4
Experimental (pilot/lab)	2
Conceptual	1

5.1.14 Summary of Criteria Descriptions and Values

For each of the thirteen criteria, scores were established based on cost and/or non-cost attributes for single family home nitrogen reducing OWTS. Table 5.14 presents a summary of score assignments for each criterion. The criterion assignments were the basis for scoring and ranking of the technology classifications.

Table 5.14: Summary of Criterion Scores

Criteria Number	Criteria	Score				
		1	2	3	4	5
1	Effluent Nitrogen Concentration (mg-N/L)	> 30	16 – 30	11 – 15	5 – 10	< 5
2	Performance Consistency[1]	Activated Sludge Nite/Denite	IFAS[2]	MBR/IMB[3]	Fixed Film	Physical/ Chemical & Source Separation
3	Construction Cost[4] ($1,000's)	>20	16-20	11-15	5-10	<5
4	BOD/TSS Effluent Concentration (mg/L)	>40	40/40	30/30	20/20	10/10
5	Mechanical Reliability	Monthly		Quarterly	Semi-Annually	Annually
6	Land Area Required[5]	High (STU modification requiring additional STU area or wetlands with pre-nitrification)		Medium (Multiple tanks)		Low (single tank unit or STU modification within conventional STU area)
7	Restoration of Performance[6]	Activated Sludge Nite/Denite	IFAS[2]	MBR/IMB[3]	Fixed Film	Physical/ Chemical & Source Separation
8	Operation Complexity	Complex operation with operator training required; Scheduled visits by manufacturer's representative required > quarterly		Some specialized operator training required; Scheduled visits by manufacturer's representative required twice per year		Simple operation with limited operator requirements; annual scheduled visit
9	Energy Requirement (kW-h/year)	>2500	1501-2500	1001-1500	500-1000	<500

Table 5.14: Summary of Criterion Scores (cont.)

Criteria Number	Criteria	Score				
		1	2	3	4	5
10	Construction Complexity	Complex installation, specialized training, sophisticated electrical and controls knowledge req., master septic tank contractor		Some specialized knowledge and training required		Simple to install by any Contractor
11	Local Resources	Not available		Available		Readily available
12	Climate Resiliency	Impacted		Perceived Impact		No impact
13	Stage of Tech. Development	Conceptual	Experimental (pilot/lab)		Demonstration (full scale)	NSF/State Use

[1] Since most of the natural systems include fixed film, the natural systems received a score of “4”.
[2] Integrated Fixed-Film Activated Sludge
[3] Membrane Bioreactor / Immersed Membrane Bioreactor
[4] Construction cost assumes a standard septic tank cost of $2000 installed, if needed.
[5] Land area is the area over and above that required for a conventional OWTS. Based on a treatment system not including effluent dispersal component.
[6] Since soil infiltration is fixed film, a score of “4” was used for the natural soil infiltration classifications. The constructed wetlands subsurface flow is not quite comparable; therefore it received a score of “3”.

5.2 Prioritization of Nitrogen Reduction Technologies

Prioritization of nitrogen reduction technologies was based on systematic application of the ranking criteria to the technologies identified in the literature review. Technologies were grouped according to the classification scheme developed.

5.2.1 Technology Evaluation Criteria

The technology evaluation criteria were individually discussed and edited, and a final consensus list of criteria was agreed to and adopted during the Technology Weighting Factor Workshop held with Stony Brook on March 2, 2016. Also agreed to and adopted at that meeting were the weighting factors for each individual criterion. The finalized criteria and weighting factors are listed in Table 5.15.

Table 5.15: Technology Criteria and Weighting Factor

Criteria	Weighting Factor
Effluent Nitrogen Concentration	12
Performance Consistency	11
Construction Cost	10
CBOD/TSS Effluent Concentration	8
Mechanical Reliability	7
Land Area Required	7
Restoration of Performance	7
Operation Complexity	6
Energy Requirement	4
Construction Complexity	3
Local Resources	2
Climate Resiliency	1
Stage of Technology Development	0[1]

[1] The weighting factor development workshop resulted in the criteria "State of Technology Development" not being used in the technology scoring.

For each of the individual technologies identified within the literature review, data were acquired from a wide variety of sources focusing on the ranking criteria. Manufacturer's information and third party test results such as the NSF International (NSF) Standard 40 Protocol, EPA Environmental Technology Verification Program (ETV), or field and/or laboratory evaluations reported in the technical literature were utilized to develop the technology database. Some performance data were available only as manufacturer's claims, other data as a range of removal percentages from field installations, and some data included detailed analytical results with statistical ranges. Results were averaged because sufficient data was generally not available to distinguish between differences in scale, number of experiments and control of influent variability. Nitrogen effluent data were generally available while nitrogen influent data were not. The attributes of the performance consistency and performance reliability criteria were based on the type of treatment process used if quantitative data was unavailable. Construction cost was estimated for a newly installed treatment system in Suffolk County, and included primary treatment (i.e. septic tank) if necessary. Performance reliability data were available for a few systems for which frequency of maintenance visits recorded were available, and estimated for the remainder. Energy use data (kW-h/day or kW-h/year) were available for a few systems, and estimated for the others. Land area required, constructional complexity, operational complexity, local resources and climate resiliency data were very limited, so professional judgment was used to assign scores for individual criteria to the technology classifications. Assumptions used in the scoring process are footnoted below the criteria scoring tables that follow below.

5.3 Criteria Scores

The criteria were developed with the full knowledge that data for many of the criteria would be sparse and difficult to attain. Good engineering judgment and experience with various types of systems were used to develop technology ranking scores when data were not available. Two sets of criteria scores were developed 1) those for technologies with full scale testing data 2) emerging technologies with experimental (pilot/laboratory) testing data. For each technology classification, the criterion scores (Table 5.14) were multiplied by the weighting factor (Table 5.15) and summed to generate a total score.

5.3.1 Biological Processes Criteria Scores

A summary of the individual criterion scores for biological technology classifications is presented in Table 5.16. Technology classifications that lacked sufficient data to make a criteria ranking determination were left blank. A brief explanation of the technology classifications is provided below.

Single Sludge BNR

1. Suspended Growth: the microorganisms responsible for treatment are maintained in liquid suspension by appropriate mixing methods
    a. Extended Aeration
    b. Sequencing Batch Reactor (SBR)
2. Fixed Film: the microorganisms responsible for the conversion of organic material or nutrients are attached to an inert packing material.
    a. Porous media (textile, plastic, sand, expanded clay, etc.) biofilter with recycle
    b. Porous media (textile, plastic, sand, expanded clay, etc.) biofilter without recycle
    c. Peat biofilter
    d. Rotating biological contactor (RBC)
3. Integrated Fixed Film Activated Sludge (IFAS): technologies that combine both fixed film and suspended growth microbial communities.
    a. Low density biosupport media activated sludge
    b. Membrane bioreactor or Immersed membrane bioreactor (MBR or IMBR)

Two Sludge, Two Stage BNR

The two sludge, two-stage process cultivates two separate bacteria populations; one for nitrification and the other for denitrification. Any of the single sludge systems can be used for nitrification preceding the second stage denitrification biofilters, but the TN reduction performance of the denitrification biofilters will be directly dependent on the nitrification performance of the first stage, as most TKN will pass through the denitrification biofilters. Therefore, the two sludge, two stage systems criteria scoring only includes the heterotrophic or autotrophic denitrification biofilters, and for onsite wastewater treatment, the second stage denitrification biofilters were limited to submerged media biofilters utilizing reactive electron donor media for denitrification.

1. Heterotrophic denitrification submerged media biofilters
    a. Lignocellulosic media

2. Autotrophic denitrification submerged media biofilters

    a. Elemental sulfur media

Table 5.16: Criteria Scores for Biological Technology Classifications using Full Scale Test Data

Technology Classification	Criteria													
	1	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	
	Effluent TN Conc. (mg/L)	**Performance Consistency**	**Construction Costs ($1000)**	**CBOD/TSS Effluent Conc (mg/L)**	**Mechanical Reliability**	**Land Area Requirements**	**Restoration of Performance**	**Operation Complexity**	**Energy Req. (kW-h/yr)**	**Construction Complexity**	**Local Resources**	**Climate Resiliency**	**Stage of Technology Development**	**Total Score**
Weighting Factor	**12**	**11**	**10**	**8**	**7**	**7**	**7**	**6**	**4**	**3**	**2**	**1**	**0**	
Single Sludge BNR														
Suspended Growth														
Extended Aeration with recycle[1]	2	1	3	5	3	5	1	3	2	4	5	3	5	219
SBR[2]	3	2	1	4	2	3	1	3	2	2	3	3	5	183
Fixed Film														
Media with recycle[3]	2	4	2	5	4	3	4	3	3	3	5	3	5	257
Granular porous media with recycle[4]	2	4	3	5	4	3	4	4	4	3	4	3	5	275
Granular porous media without recycle[5]	1	4	3	5	5	4	4	5	5	3	4	3	4	287
Peat[6]	2	4	3	5	4	3	4	3	3	3	3	3	5	263
RBC[7]	3	4	3	5	3	5	4	3	4	4	3	3	5	289
Integrated Fixed Film Activated Sludge														
Low density biosupport media activated sludge[8]	2	2	1	4	3	3	2	2	2	3	3	3	5	182
IMBR[9]	2	3	1	5	3	5	2	2	1	2	3	3	5	208
Two Sludge, Two-Stage BNR (Second Stage Only, First Stage systems above)														
Heterotrophic Denitrification														
Lignocellulosic media[10]	4	4	2[12]	2	5	3	4	5	5	5	5	3	4	290
Autotrophic Denitrification														
Elemental sulfur media[11]	5	5	2[12]	3	5	3	4	5	5	5	3	3	4	317

1. Suffolk County (2015); NJ Pinelands Commission (2015); BCDHE (2016); Maryland Department of the Environment (2016); Roeder (2015); Singulair (2016); AquaKlear (2016); Biogreen (2016); Clearstream (2016); Rich (2007); Yelderman (2005); Hoot (2016); Nayadic (2016)

2. Suffolk County (2015); NJ Pinelands Commission (2015); Aquarobic (2016); Rich (2007); Ayres (1998); Ventura Regional Sanitation District (2001); Aquarobic (2008); Pavon (2008); Stead (2002); EPA (2009); OWNRS (1997).
3. NJ Pinelands Commission (2015); BCDHE (2016); Maryland Department of the Environment (2016); Roeder (2015); Suffolk County (2015); Hazen and Sawyer (2015b); Rich (2007); MASSTC (2001); MASSTC (2001) ; Ayres (1998); EPA (2009); Klargester (2006); OWNRS (1997); EPA (2004); Wren (2004); Loomis (2004); Ventura Regional Sanitation District (2001); Anderson & Otis (2000); EPA (2004); OSTP (2006); USEPA (2003); UCF (2009); Ursin (2013)
4. Suffolk County (2015); BCDHE (2016); Hazen and Sawyer (2015b); EPA (2004); Urynowicz (2007); Loudon (2004); Osesek (1994); Richardson (2004); Costa (2003); UCF (2009); Mancl and Peeples (1991)
5. Hazen and Sawyer (2015b); Smith et al. (2008); Philp (2006)
6. Roeder (2015); Mancl and Peeples (1991); EPA (2009); NPS Water Wastewater Systems (nd); Monson Geerts (2001); NSF (2006); OSTP (2006); EPA (2004)
7. Suffolk County (2015); NJ Pinelands Commission (2015); Ayres (1998); Rotordisk (1995); EPA (2009); NSF (2003) ; Klargester (2006); NPS Wastewater Systems (nd)
8. Suffolk County (2015); NJ Pinelands Commission (2015); BCDHE (2016); Maryland Department of the Environment (2016); Roeder (2015); EPA (2009); Biomax (2007); H2M (2013); NSF (2006); EPA (2009); Delta (2016); EPA (2009) (FAST); MASSTC (2001) (FAST); OWNRS (1997); OSTP (2006); Ventura Regional Sanitation District (2001); EPA (2009) (Jet); Multi-Flo (2000); Ursin (2013)
9. Suffolk County (2015); NJ Pinelands Commission (2015); BCDHE (2016); Maryland Department of the Environment (2016); Roeder (2015); Rich (2007); Microseptech (2008); Wistrom and Matsumoto (1999); Ventura Regional Sanitation District (2001); Busse (2016); Biomicrobics (2016); Huber (2016); Kubota (2016); Bord na Mona (2016)
10. Hazen and Sawyer (2015); Rich (2007); Dupois (2002); Lombardo (2005); Loomis (2007); Vallino (2007); UCF (2009); Shah (2007); Hagerty and Tayler (2007)
11. Hazen and Sawyer (2015); Smith et al. (2008)
12. Costs include both stages of the two sludge, two stage system.

Table 5.17: Criteria Scores for Biological Technology Classifications using Experimental Test Data

Technology Classification	Criteria													
	1	2	3	4	5	6	7	8	9	10	11	12	13	
	Effluent TN Conc. (mg/L)	Performance Consistency	Construction Costs ($1000)	CBOD/TSS Effluent Conc (mg/L)	Mechanical Reliability	Land Area Requirements	Restoration of Performance	Operation Complexity	Energy Req. (kW-h/yr)	Construction Complexity	Local Resources	Climate Resiliency	Stage of Technology Development	Total Score
Weighting Factor	**12**	**11**	**10**	**8**	**7**	**7**	**7**	**6**	**4**	**3**	**2**	**1**	**0**	
Two Sludge, Two-Stage BNR														
Heterotrophic Denitrification														
Tire chip/sulfur hybrid	5[1]			4									2	--
Paper/cardboard	5[2]			--									2	--
Ground penetrating carbon (GPC)	5[3]			5									2	--
Autotrophic Denitrification														
Pyrite media	3[4]			--									2	--

[1] Krayzelova et al. (2014)
[2] Healy et al. (2015); Healy et al. (2012)
[3] McGrath (2015)
[4] Kong et al. (2015)

The top ranked biological technology classifications were two sludge, two stage BNR employing heterotrophic or autotrophic denitrification. A potential concern associated with the use of elemental sulfur denitrification media is the effluent sulfate concentration. However, the first stage (nitrification) of the two stage system can be any of the single sludge technologies that achieve high nitrification levels (~95%). The top ranked single sludge technology was fixed film – rotating biological contactor. The fixed film technologies have the stability advantages that are inherent in fixed film processes.

5.3.2 Soil, Plant and Wetland Processes Criteria Scores

A summary of the individual criterion scores for natural treatment systems consisting of soil, plant and wetland processes is presented in Table 5.18. While the table encompasses the full range of possible systems contained in the classification, technology classifications that lacked sufficient data to make a criteria ranking determination were left blank.

Table 5.18: Criteria Scores for Soil, Plant and Wetland Processes using Full Scale Test Data

	Criteria													
	1	2	3	4	5	6	7	8	9	10	11	12	13	
	Effluent TN Conc. (mg/L)	Performance Consistency	Construction Costs ($1000)	CBOD/TSS Effluent Conc (mg/L)	Mechanical Reliability	Land Area Requirements	Restoration of Performance	Operation Complexity	Energy Req. (kW-h/yr)	Construction Complexity	Local Resources	Climate Resiliency	Stage of Technology Development	Total Score
Weighting Factor	**12**	**11**	**10**	**8**	**7**	**7**	**7**	**6**	**4**	**3**	**2**	**1**	**0**	
Soil Infiltration(STU)[1]														
With dosing	2	1	4	5	4	5	4	5	4	5	3	1	5	274
With drip dispersal	2	2	3	5	3	5	4	3	4	3	3	1	5	250
STU Modification[2]														
Modified for Nitrification/ Denitrification	4	4	3	4	5	4	4	4	5	3	3	1	4	305
Above with additional STU area required	4	4	2	4	4	1	4	4	5	3	3	1	4	267
Constructed Wetlands[3]														
Subsurface flow	3	3	2	1	5	1	3	3	5	3	3	1	4	214
Hybrid modified for enhanced nitrification/ denitrification	5	-	-	-	-	-	-	-	-	-	-	-	2	--

[1] Chen (2007); Masanuga (2007); Sato (2005); MacQuarrie (2001); Ayres (1998); Hazen and Sawyer (2015c)
[2] Robertson and Cherry (1995); Robertson, Blowes et al. (2000); Heufelder (2015); Chang, Wanielista et al. (2009); Hazen and Sawyer (2015c)
[3] UCF (2009); CSWRCB (2002); EPA (2000)

The top ranked soil, plant and wetland processes was the modified soil treatment unit for nitrification/denitrification.

5.3.3 Source Separation Classification Criteria Scores

In general, all urine source separation technologies can be considered “emerging”, especially in the United States, due to the fact that they have largely only been tested at the bench- and pilot-scale. However, pilot- and demonstration-scale installations of urine source separation are becoming more prevalent in the United States. Thus far, the focus has been on household collection and storage of source separated urine followed by intermittent truck transport to a central facility, e.g., agricultural site or centralized wastewater treatment plant.

A summary of the individual criterion scores for source separation technologies is presented in Table 5.19. While the table encompasses the full range of possible systems contained in the classification, technology classifications that lacked sufficient data to make a criteria ranking determination were left blank. A brief explanation of each technology classification is provided below. All urine source separation technology classifications require the installation of urine diverting toilets with dual piping, as well as a conventional septic system for non-urine wastewater (i.e., grey water and black water).

- Transport to WWTP: Onsite storage of source separated urine followed by annual collection and transport of source separated urine to a centralized wastewater treatment facility via septage truck.
- Direct land application: Onsite storage of source separated urine followed by annual collection and transport of source separated urine to agricultural areas for land application as a liquid fertilizer.
- Struvite precipitation: Transferal of dissolved nitrogen from the liquid phase (urine) to the solid phase via addition of a magnesium-rich input and subsequent precipitation of struvite ($MgNH_4PO_4 \bullet 6H_2O$). Struvite solids are separated from the bulk liquid and liquid is directed to the septic tank.
- Sorption/ion exchange: Transferal of dissolved nitrogen from the liquid phase (urine) to the solid phase via sorption and/or ion exchange onto a solid media (e.g., activated carbon, ion exchange resin, biochar, zeolites). Exhausted media can be used as a fertilizer/soil amendment, regenerated, or thrown away.
- Stripping + acid absorption: Transferal of volatile nitrogen compounds (ammonia) from a liquid phase (urine) to the gas phase via contact of the liquid with air. Ammonia emissions are known to have human and environmental health impacts, whereas ammonia-rich liquids can be used as fertilizer. Here, ammonia stripping is followed by an H_2SO_4 absorption unit for production of ammonia sulfate fertilizer.
- Nitrification + distillation: Stabilization of hydrolyzed urine via biological conversion of volatile ammonia/ammonium to oxidized nitrogen and subsequent complete nutrient recovery of a concentrated, dry solid by heating to induce evaporative loss of liquid.
- Membrane separation: The separation of targeted constituents from bulk liquid or other constituents for more efficient or safe recovery or disposal. Larger/repulsed compounds are rejected/retained by the membrane (e.g., urea, phosphate, sulfate, pharmaceuticals), while smaller/attracted compounds are passed (e.g., water, ammonia).
- Microbial fuel cells: Production of electricity from the breakdown of organic matter in source separated using the metabolism of microbes. The primary targeted constituent is organic matter; however, nitrogen is also removed via concurrent ammonia stripping and struvite precipitation.

Table 5.19: Criteria Scores for Urine Source Separation Technology Classifications

Technology Classification	Criteria													
	1	2	3	4	5	6	7	8	9	10	11	12	13	
	Effluent TN Conc. (mg/L)[1]	Performance Consistency	Construction Costs ($1000)	BOD/TSS Effluent Conc (mg/L)	Mechanical Reliability	Land Area Requirements[3]	Restoration of Performance	Operation Complexity	Energy Req. (kW-h/yr)	Construction Complexity	Local Resources[2]	Climate Resiliency	Stage of Technology Development	Total Score
Weighting Factor	**12**	**11**	**10**	**8**	**7**	**7**	**7**	**6**	**4**	**3**	**2**	**1**	**0**	
Transport to WWTP	2-3	5	4	4	-	5[3]	-	5	-	3	3	5	2-3	242
Direct land application	3-4	5	4	4	-	5[3]	-	5	-	3	1	3	3	248
Struvite precipitation	1	5	-	4	-	-	-	3	-	3	3	5	3	137
Sorption/ ion exchange	2-3	5	-	4	-	-	-	3	-	3	3	5	2	155
Stripping + acid absorption	2-3	5	-	4	-	-	-	3	-	3	3	3	3	153
Nitrification + distillation	3-4	5	-	4	-	-	-	1	-	1	3	3	2-3	147
Membrane separation	2	5	-	4	-	-	-	3	-	3	3	3	2	147
Microbial fuel cells	1-2	5	-	4	-	-	-	1	-	1	3	3	2-3	123

1. Assuming combined wastewater has a TN concentration of 63 mg/L and 75-85% of the TN load can be attributed to urine; all urine source separation technologies assumed to be coupled with a conventional onsite wastewater system that achieves 20% TN, <5 mg/L TSS, and <5 mg/L BOD removal in non-urine wastewater (Costa, Heufelder et al. 2002)
2. Acquisition of multiple source separating toilets may be a challenge, but units are available for purchase in the US; nearby farmland for "direct land application" of source separated urine may also be limited.
3. Pertains to land area required for storage of urine produced by a family of four in a year (600 gallons); does not include WWTP or agricultural land area

Within the urine source separation technology classification, the highest ranked approaches are direct land application and transport to WWTP. Either approach would entail installation of a household urine source separation system, onsite household urine storage, and truck transport of urine to a land application site or centralized wastewater treatment plant. The installation and testing of the "transport to WWTP" approach (Table 5.20) would provide immediate benefits and also help further inform whether the technical and social aspects of urine source separation lend themselves to onsite wastewater treatment.

Table 5.20: Major components of the recommended "transport to WWTP" urine source separation technology classification

Conventional Septic System	Household Urine Collection and Storage	Transport	Central Facility
• Conventional septic tank and soil treatment unit for non-urine municipal wastewater	• Urine source separating toilet(s) • Household urine-only piping from toilet(s) to household urine storage tank • Above- or in-ground urine storage tank • Urine storage tank high level alarm to schedule truck transport	• Conventional septic and urine hauling services	• Nearby central wastewater treatment plants • Urine allowance • Delivery schedule • Onsite storage

5.4 Nitrogen Reduction Technology Ranking Summary

The studies and data identified in the literature review were synthesized to develop matrices categorizing the major nitrogen reduction processes, stage of development, treatment effectiveness, operability, complexity, energy use, and other considerations. The technology classification ranking provides the basis from which to formulate recommendations for further testing. The recommended technologies for further full scale testing include two sludge, two stage BNR and modified soil treatment unit for nitrification/denitrification.

The first stage (nitrification) of the two sludge, two stage system can be any of the single sludge technologies that achieve high nitrification levels (~95%). The top ranked single sludge technology was fixed film – rotating biological contactor. The fixed film technologies have the stability advantages that are inherent in fixed film processes. The first stage biofilter can employ a variety of fixed film media as described in the literature review. The second stage of these systems would employ autotrophic denitrification or heterotrophic denitrification.

The modified soil treatment unit incorporates a vertically stacked approach of unsaturated porous media for nitrification overlaying autotrophic denitrification or heterotrophic denitrification media. This system is a passive technology and has the potential for low cost.

Urine source separation is an emerging nitrogen removal technology which is recommended for further pilot scale testing incorporating direct land application and transport to WWTP. The urine source separation approaches of direct land application and transport to WWTP entail installation of a household urine source separation system, onsite household urine storage, and truck transport of urine to a land application site or centralized wastewater treatment plant.

The ranking exercise also identified knowledge gaps and opportunities for the CCWT as described in the following Section.

6. Knowledge Gaps and Research Opportunities for OWTS

6.1 Biological Nitrogen Reducing OWTS

Proprietary single sludge nitrification/denitrification systems of many different configurations are available on the market for onsite wastewater treatment. Many are well tested and proven across the U.S. The processes involved are well understood because considerable research has been accomplished over the past 50 years related to municipal wastewater treatment utilizing the same unit processes. The primary difficulties with these systems are the lack of process control when used in individual home settings, and treatment effectiveness is not very high in many cases even when operating effectively. There are not many knowledge gaps relative to the treatment process or technologies for these systems.

There are several remaining knowledge gaps and research opportunities for the second stage (engineered denitrification) of two sludge, two stage biological systems. These are primarily related to optimizing design criteria for full scale systems as summarized below.

Nitrification and Denitrification media biofilters in general:

- Long term performance and reliability of the systems
- Range of hydraulic loading rates as related to Stage 2 nitrate reduction performance
- Cold temperature performance
- Organic and inorganic inhibitors
- Media longevity/efficacy
- Seasonal use (research on restoration of performance)
- Specifications for media including local suppliers, specific media designations, source
- Specifications for specific tank designations, source, materials, dimensions, and strength requirements
- Specifications for tank lids and covers that provide full and easy access to media within biofilters
- PPCP and pathogen removal effectiveness

Lignocellulosic Denitrification Biofilters:

- Material source and composition
- Nitrate reduction effectiveness of various local wood types/sources
- BOD release as related to various wood types
- Influence of Stage 2 influent (Stage 1 effluent) dissolved oxygen concentration as related to Stage 2 nitrate reduction performance

Sulfur Denitrification Biofilters:

- Susceptibility to clogging as referenced in prior Suffolk County work (unpublished) and at the Massachusetts Test Center which could be attributed to media particle size (granular powder vs pastille pellets), hydraulic loading rate, water quality chemistry, temperature etc.

- Alkalinity buffering media: various types related to effectiveness, dissolution rate and relative longevity.

Additional Media Options:

- Other reactive media could be investigated to compare to the lignocellulosic and sulfur media systems:
  - Pyrite
  - Iron
  - Other carbon source media
- Further demonstration (full scale) testing of denitrification media which has shown feasibility in pilot and lab-scale testing would provide additional commodities for the marketplace.

6.2 Soil, Plant and Wetland Processes Nitrogen Reducing OWTS

There are several remaining knowledge gaps and research opportunities for the modified soil treatment units incorporating engineered nitrification and denitrification. Many of these are related to optimizing design criteria for full scale systems as summarized below.
System design:

- Long term performance and reliability of the systems
- Range of hydraulic loading rates to nitrification media as related to nitrate reduction performance
- Cold temperature performance
- Organic and inorganic inhibitors
- Denitrification media longevity
- Specifications for denitrification media including local suppliers, specific media designations, source
- Optimized mixture percentages of denitrification media and support structure material to enhance hydraulic design, prevent subsidence, nitrate removal effectiveness and moisture content.
- Hydraulic design to ensure soil physics promote effluent passage into denitrification media mixture.
- Containment liners or use of soil texture/moisture retention to promote desired moisture content for denitrification conditions.
- Designs that can accommodate the replacement of denitrification media such as in-situ removable vessels/baskets.
- PPCP and pathogen removal effectiveness

6.3 Source Separation Nitrogen Reducing OWTS

The unique and nutrient-rich composition of urine, and the fact that nutrients are at once a resource and a contaminant depending on their fate suggest that urine source separation is an alternative onsite wastewater treatment approach that warrants consideration. However, additional information and technology advancement is required in order to make urine source separation widely applicable. Research opportunities span the design of the urine separating toilet itself to the final production of a urine-based fertilizer. Within Appendix A, Table A.1 includes parties involved in the practice of urine source separation with details on relevant experience and contact information.

Household collection of source separated urine followed by truck transport of urine to a centralized wastewater treatment plant shows promise as an onsite wastewater treatment approach because the majority of the nitrogen load (~75%) in municipal wastewater can be attributed to urine, yet urine is only a small fraction (~1%) of the volumetric flow. Thus, diversion of urine away from a conventional onsite wastewater system results in immediate diversion of nitrogen away from surrounding environment. The low volumetric production of urine also lends itself to reasonable urine storage tank sizing and transport frequencies. A 600 gallon storage tank is expected to be sufficient for one year's worth of urine produced by a family of four. The demonstration of household urine source separation coupled with a conventional septic system (for non-urine wastewater) and truck transport of urine to centralized wastewater treatment plants would fill current knowledge gaps related to:

Installation:

- Contractors would become familiar with recommended installation procedures for urine source separating toilets and the urine-only piping that conveys urine from the toilets to the onsite urine storage tank. In general, pipe material must be smooth and resistant to urine corrosiveness; bends and irregularities should be minimized due to the low flow and precipitation potential of urine; pipe gradient should be a minimum of 1%.

Transport:

- The identification of septic hauling services that are willing to pick up source separated urine from individual homes and deliver it to centralized wastewater treatment plants. Due to the fact that urine is being blended with combined wastewater at the wastewater treatment plant in this scenario, septic haulers may blend urine and conventional septic tank waste or have a truck that is dedicated to urine transport. If future scenarios are pursued in which source separated urine is taken to a central facility for pharmaceutical removal, nutrient recovery, or any other urine-only treatment process, then haulers would most likely require trucks solely dedicated to urine delivery to minimize contamination.

Delivery schedule:

- Demonstration of urine source separation with subsequent truck transport of urine to centralized wastewater treatment plants would require coordination with local wastewater treatment plants to determine the extent to which they are willing to accept urine deliveries, e.g., total volume, schedule, etc. Initial discussions and demonstration of the delivery process will help determine if any additional infrastructure or management strategies are needed in order to facilitate hauling of urine from households to the plant without impacting the plant's ongoing operations and effluent quality.

Conventional onsite wastewater system operation:

- If urine source separation is practiced at the household level, the majority of the nitrogen load in municipal wastewater is addressed, but the rest of the household's municipal wastestream must still be managed. Thus, all urine source separation technologies are assumed to be coupled with a conventional OWTS for treatment of greywater and blackwater. Installation and testing of these systems should involve monitoring of the conventional systems to determine the impacts of urine diversion on system operation and effluent quality.

Public acceptance:

- Public perceptions can play a significant role in the success or failure of alternative water and wastewater systems. Installation and testing of urine source separation in select households would provide the opportunity to collect feedback from multiple stakeholders, including homeowners with urine source separating systems, urine haulers, and wastewater treatment plant personnel. Purposeful collection and analysis of stakeholder feedback is expected to benefit the design and operation of future systems.

In addition urine source separation research related to fate and technology advancement provides research opportunities in order to make urine source separation widely applicable.

Fate:

- Overall, urine source separation research has tended to focus on nutrient recovery as opposed to nutrient removal in order to reap the environmental and economic benefits of producing a urine-based fertilizer. However, in some scenarios, nutrient removal from urine may be preferred over nutrient recovery, such as when it is desirable to remove nutrients from the watershed, but there is no local market for an alternative fertilizer or the risks of urine-based fertilizer are deemed prohibitive (e.g., potential pharmaceutical and/or pathogen contamination). Thus, one broad research opportunity is to view the area of urine source separation through the lens of nutrient removal and destruction/disposal, as opposed to nutrient recovery.

Treatment:

- The removal of nutrients from urine through the use of sorptive media is an attractive treatment approach due to the associated construction and operational simplicity, as well as the potential for using exhausted media as a soil amendment. However, the high concentration of nitrogen in urine make media selection a challenge, as it must have a capacity that is sufficient to remove nitrogen to a target level without requiring a prohibitive media footprint or replacement frequency. As a foundation for future efforts involving nutrient removal with various sorptive media, a review and assessment of previously conducted research is needed. This review should not only summarize previously tested media and associated nitrogen removal capacities in urine, but also equate these capacities to estimated exhaustion times using a range of reactor sizes. A minimum target media capacity can be quantified based on reasonable reactor sizing and media exchange frequencies. A review of previously conducted research would also facilitate a better understanding of whether sorptive media capacities tend to be higher when targeting nitrogen as urea (fresh urine) or ammonia/ammonium (hydrolyzed urine). This information speaks to the best suited placement of the sorption process within a urine source separation system, as well as the need for any pretreatment steps (e.g., to prevent urine hydrolysis during storage).

- Nitrogen recovery from urine via ammonia stripping and sulfuric acid adsorption has been demonstrated at the bench-, pilot-, and full-scale. This process benefits from the rapid conversion of urea to ammonia/ammonium during urine storage, the volatility of ammonia, and the common use of ammonium sulfate as fertilizer. However, the large volumes of sulfuric acid required for the absorption of volatilized ammonia can be a health and maintenance concern. Although the use of sulfuric acid lends itself to subsequent use as a fertilizer, one may investigate the use of other materials that are more appropriate for use at the household level. These materials may be investigated with the goal of nutrient recovery in mind (as is done with the use of sulfuric acid for the production of ammonium sulfate) or alternatively with the aim of ammonia capture and destruction/disposal.
- Lastly, as urine treatment technologies advance, demonstration-scale installations and regular monitoring of these systems is critical. Table 5.20 demonstrates that urine source separation systems are difficult to evaluate on a multi-criteria basis due to a current lack of demonstration-scale installations and associated monitoring data. Communities with septic systems requiring replacement and/or enhancement are a favorable setting in which to test urine source separation because it can be done in a phased manner (i.e., one house at a time) and some level of action is already required.

6.4 PPCPs Removal by OWTS

There are numerous knowledge gaps and research opportunities for nutrient, personal care product, pharmaceutical and pathogen removal research in the private sector, which is dependent upon the development of established regulatory standards for the treatment of these emerging contaminants. Manufacturer’s investment in research and fabrication of equipment will be driven largely by pending regulations or special design specifications for particular applications such as treatment of effluent from pharmaceutical manufacturing. Additionally, manufacturers will continue to design, manufacture and supply larger treatment equipment for centralized systems leaving applied research to major research universities who specialize in the treatment of emerging contaminants which include Arizona State University, Colorado School of Mines and others. Until specific regulations within the United States are mandated, scalable onsite systems to fully address this sector will be cost prohibitive and dependent upon the development of full scale treatment equipment which provide larger revenue streams for manufacturers.

In general, onsite treatment of PPCPs will benefit from further research directed toward the following focus:

- The separation of urine and feces at the source, prior to discharge to any onsite treatment system appears to be the most cost beneficial approach to removing PPCPs from household waste water, in addition to the benefits of reducing nutrient loadings as previously described. Accordingly, research directed toward ion exchange, electrolysis, activated carbon which either sequester or degrade.

- Ozone generation and UV in conjunction with biological nitrification/denitrification systems including sequencing batch reactors (SBR), submerged membranes (MBR) and moving bed biological reactors (MBBR). These activated sludge biological systems are already fully developed to meet stringent nitrogen and phosphorous requirements and experimental and an abundance of data regarding the biodegradation of PPCPs from these systems exist. Onsite treatment of PPCPs research could benefit from efforts directed toward understanding the relationship mechanisms between cost effective ozone and UV to reduce complex carbon compounds further so that they may be biodegraded by conventional activated sludge systems.
- Natural systems, which use select layered media strategies, could benefit from alternative approaches to configuration of the aerobic and anaerobic processes, ion exchange media selected for complex organic compounds and energy harvesting to power novel approaches to ozone and UV processes.

6.5 Collaboration with Patent Holders

The published patent grants and applications referenced as part of this report provide the initial information necessary to pursue further research opportunities and in most cases, will list the details of the technology, named inventor, and sponsoring organization or assignee if applicable. Typically, each patent also references similar inventions and research to describe the state of art which may have been used to develop the idea. A list of inventors, sponsoring organizations/assignees and summary of the technology opportunity are provided in Appendix A, Table A.2.

7. Summary and Recommendations

This technology assessment report was commissioned in order to assist the New York State Center for Clean Water Technology with recommendations and a roadmap for further academic research and implementation of nitrogen reducing onsite wastewater treatment systems. The ultimate goal for onsite systems development is to provide operationally simplistic wastewater treatment systems with the lowest cost per lbs. nitrogen removed and to further the development of these systems so they are readily accepted by regulatory agencies and the public. As described in Section 6, there are several knowledge gaps and opportunities to pursue in order to optimize the basic processes of source separation, nitrification, denitrification and solids removal. Our research suggests and we recommend that currently, two sludge, two stage BNR passive denitrification systems with dosing and recycle should be pursued at the full scale pilot stage, in climate and soil conditions native to Long Island. In order to improve the performance and reduce the cost of these full scale onsite systems, research should be directed toward:

- Locally sourced media substitutions used in unsaturated and saturated soil zones
- Media size, porosity and configuration in unsaturated layers to increase aeration and subsequently dissolved oxygen concentrations to optimize nitrification in porous media
- Similarly, media size, porosity and configuration of saturated layers to increase opportunities for denitrification
- Introduction of locally sourced media mixtures used to support the nitrification and denitrification process such as alkalinity to buffer pH or more sophisticated pre-treatment regimes which degrade complex organic compounds to resolve outstanding problems with PPCPs.
- Methods to easily access, monitor, and replace media
- Methods to easily access and monitor system parameters such as DO and NH3.
- Configurations which reduce footprint and depth of excavations which can be easily retrofitted with existing septic tank and leaching pool infrastructure.
- Combine urine nutrient and PPCP source separation, energy recovery and water reuse concepts with the modified soil treatment unit to provide a complete lifecycle solution to onsite wastewater treatment.

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Appendix A – Private Sector Contact Information

The following private sector entities identified in Table A-1 and patents listed in Table A-2 are either actively engaged in some aspect of PPCP research and treatment or have specific technologies related to onsite wastewater systems which could benefit from further applied research and serve as a collaborative partner for the Center for Clean Water Technology.

Table A-1: Summary of Potential Collaborators in the Private Sector

Entity	Contact Information	Topic Area
Adelante Consulting, Inc. (Pugo)	www.pugosystems.com (505) 866-5076	FF-media (with recycle)
Aquapoint (Bioclere)	aquapoint.com (508) 985-9050	FF-media (with recycle)
AquaO2 Wastewater Treatment Systems, Inc. (Aquarobic)	www.aqua-o2.com (540) 365-0154	SG-SBR
BB Innovation & Co (Dubbletten)	www.dubbletten.nu 46 (0) 380 - 42103	SS-urine
Bilfinger Water Technologies (Roevac)	www.water.bilfinger.com (574) 223 3980	SS-urine
Bioconcepts, Inc. (ReCip® RTS ~ 500 System)	(252) 249-1376	FF-media(with recycle)
Bio-Microbics, Inc (BioBarrier)	www.biomicrobics.com (913) 422-0707	IMBR/IFAS
Bord-na-mona, Environmental Products US Inc. (Puraflo)	www.anuainternational.com (336) 547-9338	FF-peat
Bord-na-Mona (PuraM)	www.bordnamona.ie +353 45 439000	IMBR
Busse Green Technologies	www.busse-gt.com (708)-204-3504	MBR, On-Site Treatment
Calgon Carbon	www.calgoncarbon.com; (412)-787-6700; (800)-4CARBON	Activated carbon, ion exchange, UV, Ozone, PPCP Treatment
Cape Cod Eco-Toilet Center	Hilde Maingay and Earle Barnhart capecodchemists@gmail.com; (508) 563- 3101	Urine SS
Clivus Multrum (Composting Toilets)	www.clivusmultrum.com (800) 425-4887	SS-urine
Composting Toilet Systems, Inc (Composting Toilets)	www.comtoilet.com (888) 786-4538	SS-urine

Table A-1: Summary of Potential Collaborators in the Private Sector

Entity	Contact Information	Topic Area
Consolidated Treatment Systems (Multi-flo)	www.consolidatedtreatment.com (937) 746-2727	IFAS
Cromaglass Corporation (Cromaglass)	www.septicsystem.com/brands/cromaglass.html (570) 326-3396	SG-SBR
Delta Fiberglass & Environmental Products, Inc (Delta Whitewater)	www.deltaenvironmental.com (225) 665-6162	SG-ExAir
Earthtek Environmental Systems (EnviroFilter C)	www.earthtekenvironmental.com (800) 934-5044	FF-media(with recycle)
EAWAG	Bastian Etter at bastain.etter@eawag.ch; Kai Udert at kai.udert@eawag.ch; Tove Larsen at tove.larsen@eawag.ch	Urine SS
EcoVita	Carol Steinfeld at (978) 318- 7033	Urine SS
Eliminite, Inc. (Eliminite)	www.eliminite.com (406) 581-1613	FF-media(with recycle)
General Electric (GE)	www.gewater.com	MBR, PPCP Treatment
Hoot Systems, LLC. (Hoot)	www.hootsystems.com (888) 878-HOOT	SG-ExAir
Incinolet (Incinerating Toilets)	www.incinolet.com (800) 527-5551	SS-urine
Innovative RUCK Systems Inc. (Ruck)	www.irucks.com (508) 548-3564	FF-media(with recycle)
International Wastewater Systems, Inc. (Model 6000)	www.sewageheatrecovery.com (604) 475-7710	SG-SBR
Ixon Watercare	www.miexresin.com; (877) 414-MIEX	Ion exchange, PPCP
Kingspan Environmental (Biodisc)	www.kingspanenv.com 0844 846 0500	FF-RBC
Kruger Inc./Veolia	http://www.krugerusa.com Chris Thomson, (919)-653-4562	FF-Media
MicroSepTec (Enviroserver)	www.microseptec.com (877) 4SEPTIC	IMBR
Norweco	www.norweco.com (419) 668-4471	SG-Extended Aeration, PPCP Treatment

Table A-1: Summary of Potential Collaborators in the Private Sector

Entity	Contact Information	Topic Area
NPS Wastewater Treatment Systems Limited (Biorotor)	www.npswastewater.com/ (604) 294-1661	FF-RBC
Orenco Systems, Inc (Advantex 20x)	www.orenco.com (800) 348-9843	FF-media(with recycle)
Ovivo	www.ovivo.com (512) 652-5805	MBR, PPCP Treatment
Planet Care, Inc. (Ecopure)	www.eco-purewastewatersystems.com (540) 980-2420	FF-peat
Quanics Incorporated (Aerocell biofilter)	www.quanics.net (502) 992-8200	FF-media(with recycle)
Rich Earth Institute	Kim Nace at kim@richearthinstitute.org, (802) 579-1857	Urine SS
SBR Wastewater Technologies	www.SBRww.com; (855) 391-2448	SBR, PPCP Treatment
SEPTITECH, INC. (SeptiTech)	www.septitech.com (207) 333-6940	FF-media(with recycle)
Sludgehammer Group, Ltd. (Sludgehammer Bio-Kinetic WWT System)	www.sludgehammer.net (231) 348-5866	SG-Extended Air
Smith and Loveless, Inc.	www.smithandloveless.com; (800) 898-9122	MBR, Small Scale Treatment
SoilAir	www.soilair.com; (860) 510-0730	Extended Aeration, On-Site Treatment
Spec Industries, Inc. (AIRR)	www.specind.biz (702) 434-9091	FF-media(with recycle)
Suez Treatment Solutions (Formerly Ozonia)	www.suez-environnement.com (201) 676-2241	Ozone, UV, PPCP Treatment
Trojan Technologies	www.trojanuv.com (888) 220-6118	UV, Ozone, PPCP Treatment
University of Florida	Dr. Treavor Boyer at thboyer@ufl.edu	Urine SS
University of Michigan	Krista Wigginton at kwigg@umich.edu	Urine SS
Uridan/ SANIT Chemie (Uridan Waterless Urinal)	www.uridan.de/en/meta/imprint.html +49 7131/90210-18	SS-urine
US Biochar Initiative	www.biochar-us.org; (406) 459-3486	PPCP Treatment
Waterloo Biofilter Systems, Inc. (Waterloo)	www. waterloo-biofilter.com (519) 856-0757	FF-media(with recycle)
Wedeco (Xylem)	www.xylem.com (914) 323-5700	MBR, UV, Ozone, PPCP Treatment
Xerolet International, LLC (Xerolet)	www.igreenbuild.com	SS-urine
Zoeller Pump Co. (Zoeller Fusion Series)	www.zoeller.com (502) 778-2731	SG-ExAir

Table A-2. Summary of Selected Patents

Patent #	Date of Publication	Title	Inventor(S)	Assignee
US 2016/0050916 A1	2/25/2016	Bio-Control Activity Surface	Victor Bellido-Gonzalez, Dermot Patrick Monaghan	Gencoa Ltd., Merseyside (GB)
US 2016/0039695 A1	2/11/2016	System And Method Of Treating Wastewater	William N. Carpenter, JR.	N/A
US 2016/0023932 A1	1/28/2016	Method For Deammonification Process Control Using PH, Specific Conductivity, Or Ammonia	Charles Bott, Stephanie Klaus	Hampton Roads Sanitation District, Virginia Beach, VA (US)
US 2016/0002079 A1	1/7/2016	Improved Fermentation Process And Products Useful For The Same	Atul Ambaji Nivargi	N/A
US 2015/0359915 A1	12/17/2015	Portable UV Devices, Systems And Methods Of Use And Manufacturing	Alexander Farren, Noah Bareket, Thomas Edgar Beard	N/A
US 9,187,342 B2	11/17/2015	Method For Removing Drugs From Waste Water Using Neutralized Bauxite Residue	Shannon L. Isovitsch Parks, David Iwig, John R. Smith, Jaw K. Fu, Rajat Ghosh	Alcoa Inc., Pittsburgh, PA (US)
US 2015/0239761 A1	8/27/2015	Recovery Of Resources From Waste Water	Daniel P. Smith	N/A
US 9,038,408 B2	5/26/2015	Wastewater Effluent To Geothermal Heating	Stephen A. Sabo	AK Industries, Inc., Plymouth, IN (US)
US 8,828,240 B1	9/9/2014	Residential Wastewater Purification System	Benjamin A. Schranze, Ronald Knepper	N/A
US 2014/0212219 A1	7/31/2014	Roll-Formed Conduit-Arch For Leach Field	E. Craig Jowett	Rowanwood IP Inc., Rockwood (CA)
US 8,758,613 B2	6/24/2014	Dynamic Anaerobic Aerobic (DANA) Reactor	Tamar Arbel, Nir Assulin, Antonius Johannes Hendrikus Hyacinthus Engelaar, Tammy Yalin	Aqwise-Wise Water Technologies Ltd, Harzliya (IL): Westt, Leeuwarden (NL)
US 8,652,329 B2	2/18/2014	Sewage Nitrate Removal By Free-Draining Asphyxiant Filtration And Carbon Addition	E. Craig Jowett	Rowanwood IP Inc., Rockwood, Ontario (CA)

Table A-2. Summary of Selected Patents

Patent #	Date of Publication	Title	Inventor(S)	Assignee
US 8,460,520 B2	6/11/2013	Electrochemical System And Method For The Treatment Of Water And Wastewater	David Rigby	N/A
US 2013/0126407 A1	5/23/2013	Leach Field System	David A. Potts	N/A
US 2013/0098840 A1	4/25/2013	Porous Composite Media For Removing Phosphorus From Water	Richard Helferich, Ramachandra R. Revur, Suvankar Sengupta, J. Richard Schorr	Metamateria Technologies, LLC, Columbus, OH (US)
US 8,419,858 B1	4/16/2013	Method And System For Removing Organic Compouds In A Closed Loop System	Frederick J. Haydock	Haydock Intellectual Properties, LLC, Murray, UT (US)
US 8,323,474 B2	12/4/2012	Electro-Chemical Water Processing Apparatus And Method Thereof	Chi-Jung Jeon, Jong-Sung Kim, Kwang-Su Kim, Sang-Ki Hong	Chi-Jung Jeon, Gyeonggi-do (KR)
US 8,318,008 B1	11/27/2012	Modular Individual Wastewater Nutrient Removal System	Steven M. Anderson	SepticNet, Inc., Butte, MT (US)
US 8,313,657 B1	11/20/2012	Method And System For Removal Of Ammonia From Wastewater By Electrolysis	Rick B. Spielman, Link E. Summers	N/A
US 2012/0234771 A1	9/20/2012	Solar Enclosure For Water Reuse	Anna Dyson, Jason Vollen, Mark Mistur, Peter Stark, Kristin Malone, Matt Gindlesparger	Rensselaer Polytechnic Institute, Troy, NY (US)
US 8,268,174 B2	9/18/2012	Wastewater Treatment Method And System With Ozonation For Microconstituent Removal	Dennis Livingston	Ovivo Luxembourg S.a.r.l., Munsbach (LU)
US 8,252,182 B1	8/28/2012	Subsurface Upflow Wetland System For Nutrient And Pathogen Removal In Wastewater Treatment Systems	Ni-Bin Chang, Martin P. Wanielista	University of Central Florida Research Foundation, Inc., Orlando, FL (US)
US 8,252,182 B1	8/28/2012	Subsurface Upflow Wetland System For Nutrient And Pathogen Removal In	Ni-Bin Chang, Martin P. Wanielista	University of Central Florida Research Foundation,

Table A-2. Summary of Selected Patents

Patent #	Date of Publication	Title	Inventor(S)	Assignee
		Wastewater Treatment Systems		Inc., Orlando, FL (US)
US 2012/0160706 A1	6/28/2012	Apparatus And Method For Electrochemical Treatment of Wastewater	Nicole A. Poirier, Valerie Leveille	Proterrgo Inc., Monthreal, QC (CA)
US 2012/0145611 A1	6/14/2012	Water Treatment System For Simultaneous Nitrification And Denitrification	Dean Smith, Ola Lysensteen, Cary Tope-McKay, Gary Gorian	N/A
US 8,191,716 B2	6/5/2012	Horizontal-Tube Sedimentation-Separation Apparatus	LiangchunZhang, Jianguo Zhang	Zhuhai 9 Tone Water Service Co., Ltd., Zhuhai (CN)
US 2012/0037512 A1	2/16/2012	Electrodes For Electrolysis Of Water	Maurice James Robertson	N/A
US 7,998,343 B2	8/16/2011	Wastewater Treatment System	D. Mark Aker, Dan A. Papczynski, Caleb Youker	Gast Manufacturing, Inc., Benton Harbor, MI (US)
US 7,927,484 B2	4/19/2011	Passive Underground Drainfield For Septic Tank Nutrient Removal Using Functionalized Green Filtration Media	Martin P. Wanielista, Ni-Bin Chang, Ammarin Makkeasorn	University Of Central Florida Research Foundation, Inc.
US 7,749,384 B2	7/6/2010	De-Nitrification Treatment System And Method	David W. Patton, Gerald Lee Lamb, Jamie Lee Miller	N/A
US 7,736,513 B2	6/15/2010	Liquid-Solid Fluidized Bed Waste Water Treatment System For Simultaneous Carbon, Nitrogen And Phosphorous Removal	Jingxu Zhu, George Nakhla, Yubo Cui	The University of Western Ontario, London, ON (CA)
US 2010/0135869 A1	6/3/2010	Ozone Generators	Lih-Ren Shiue, Masami Goto	Linxross, Inc., Tokyo (JP)
US 7,658,851 B2	2/9/2010	Method Of Growing Bacteria For Use In Wastewater Treatment	Douglas J. Nelson, Robert Rawson	Pseudonym Corporation, Oswego, NY (US)
US 2010/0018916 A1	1/28/2010	Horizontal-Tube Sedimentation-Separation Apparatus	LiangchunZhang, Jianguo Zhang	N/A
US 7,651,608 B2	1/26/2010	System For Denitrification Of Treated Water From	Jerry L. McKinney	N/A

Table A-2. Summary of Selected Patents

Patent #	Date of Publication	Title	Inventor(S)	Assignee
		Aerobic Wastewater Treatment Systems		
US 7,635,236 B2	12/22/2009	In Situ Remediation Of Inorganic Contaminants Using Stabilized Zero-Valent Iron Nanoparticles	Dongye Zhao, Yinhui Xu	Auburn University, Auburn, AL (US)
US 7,632,408 B1	12/15/2009	Passive Drain Field System For Wastewater Treatment And Associated Methods	Douglas G. Everson	Plastic Tubing Industries, Inc., Apopka, FL (US)
US 7,553,418 B2	6/30/2009	Method For Water Filtration	Boris M. Khudenko, Rocco M. Palazzolo, James R. Stafford	Khudenko Engineering, Inc., Atlanta, GA (US)
US 2009/0032451 A1	2/5/2009	Aeration-Less Water Treatment Apparatus	Masahiko Tsutsumi, Takumi Obara, Nobuyuki Ashikaga, Katsuya Yamamoto, Hiroshi Tamura	N/A
US 7,300,577 B1	11/27/2007	Wastewater Treatment And Dispersal System	Steven A. Branz	N/A
US 2007/0267346 A1	11/22/2007	Process For Autotrophic Perchlorate Reduction Using Elemental Sulfur And Mollusk Shells	Sukalyan Sengupta, Sarina Ergas, Klaus Nusslein, Ashish Sahu	N/A
US 7,022,235 B2	4/4/2006	Wastewater Biological Treatment System And Method Therefor	Alan F. Hassett	The White Oak Partnership, L.P., Berwyn, PA (US)
US 6,749,745 B2	6/15/2004	In-Pipe Wastewater Treatment System	E. Craig Jowett	N/A