Report to the Puget Sound Action Team

Nitrogen Reducing Technologies for Onsite Wastewater Treatment Systems – Report to the Puget Sound Action Team June 2005

Washington State Department of HEALTH

Prepared by Wastewater Management Program Environmental Public Health

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For more information or additional copies of this report:
Wastewater Management Program
Division of Environmental Public Health
Washington State Department of Health
PO Box 47824
Olympia, WA 98504-7824

360-236-3330
wastewatermgmt@doh.wa.gov

Umair Shah, MD, MPH
Secretary of Health

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Contents

Preface

The funding for this report is provided by a contract administered by the Puget Sound Action Team. The research and report on Nitrogen Reducing Technologies is part of a larger effort outlined in The Preliminary Assessment and Corrective Action (PACA) Plan to identify and remedy water quality issues in Lower Hood Canal.

The Preliminary Assessment and Corrective Action (PACA) Plan was developed through a collaborative and cooperative arrangement between the Puget Sound Action Team (Action Team), the state's partnership for Puget Sound, and the Hood Canal Coordinating Council (HCCC), the council of governments within the Hood Canal watershed.

According to Puget Sound Action Team's Web site, the objectives of the Preliminary Assessment and Corrective Action Plan are:

1. Identify and quantify nitrogen sources, using the best available data, the best professional judgment, and the amount and timing of nutrient materials that likely contributes to low dissolved oxygen in the canal's marine water;

2. Develop a corrective action and education plan that can be used by various partners to reduce human-influenced nutrient materials to the canal to the greatest extent possible.

3. Consider other options for improving the low dissolved oxygen situation by altering human activities and biological, physical and chemical processes that may affect a positive change in the dissolved oxygen levels.

The full text of the report is available on the Puget Sound Action Team website at http://www.psat.wa.gov/Programs/hood_canal.htm

The purpose of this report is to help on-site sewage industry members, regulators and concerned citizens understand the role of nitrogen in onsite wastewater systems and the nitrogen chemistry during wastewater treatment. It is also important for all parties to understand the complexities and limitations on the abilities of onsite sewage technologies to reduce this critical element from the waste stream.

Executive Summary

According to the Preliminary Assessment and Corrective Action Plan for Hood Canal, nitrogen from onsite sewage systems is one of the contributors of nitrogen to the Canal. The removal of nitrogen from sewage using an onsite wastewater treatment system involves natural biological processes. A variety of proprietary technologies have been developed for the purpose of enhancing these natural processes. However, none has been demonstrated to provide a simple, effective, and consistently reproducible effluent.

This report outlines impacts of nitrogen on the environment and health, and the biological and chemical processes that are involved in the transformation of nitrogen products as wastewater is treated and dispersed. It also discusses the technologies that are available to remove nitrogen from onsite wastewater treatment systems.

Removal of nitrogen from wastewater is a complex process, even for large wastewater treatment plants. Quality control of nitrogen removal processes from individual onsite wastewater systems is even more difficult to manage. Treatment systems that are most commonly used are relatively efficient in the removal of biological oxygen demand (BOD) and total suspended solids (TSS) from wastewater but provide less than optimal removal of nitrogen (10-30 %). Most of the nitrogen is released as nitrate (NO_3^-), which is highly mobile in the soil water.

In a conventional septic tank and drainfield system organic nitrogen in household wastes is transformed into ammonia products in the anaerobic conditions of the septic tank (ammonification). When these products exit the septic tank and encounter the aerobic conditions in the drainfield, the ammonia products are biochemically transformed primarily into nitrates (nitrification). These two steps, ammonification and nitrification, occur naturally in conventional systems. Transforming the highly mobile nitrate into nitrogen gas (denitrification) is the most difficult stage in the process and the step that adds the greatest complexity to the entire process. Because standard systems are inefficient in removing nitrate, additional treatment options need to be inserted into the treatment process.

A wide variety of public domain systems and proprietary devices have been promoted and used for denitrification. Significant issues remain on the treatment performance efficiencies for these systems. The variability in performance is due to the inter-relationship of numerous factors including:

- Fluctuating flow rates

- Variability in waste strengths
- Complexities in the biological and chemical treatment processes
- Temperatures
- pH and Alkalinity
- Inhibitory chemical compounds
- Increased complexity of the additional mechanical devices such as pumps, filters, timers, controllers, etc. that are added to the process.

Nitrogen removing technologies have been installed and tested at numerous Environmental Protection Agency (EPA) sponsored National Demonstration Sites around the country. The results of performance testing have been promising but quite variable. Research and development continue and should result in further improvement in the efficiency and reliability of treatment products.

In an effort to provide a national testing protocol that can be used to verify nitrogen removal performance, EPA and the National Sanitation Foundation (NSF) have developed the Environmental Technology Verification (ETV) protocol. This protocol is the only national protocol in existence. Six products have completed ETV testing with demonstrated total nitrogen removal efficiencies ranging from 51-64%.

The recently approved new State Board of Health rules for onsite sewage require that any systems to be used for nitrogen reduction demonstrate that their product's performance is verified through the EPA/NSF ETV Protocol. All of the products that have undergone ETV testing should be able to meet the Department of Health's proposed effluent standard of 20 milligrams per liter of total nitrogen.

Most of the tested systems, due to the sequential processes involved, are more appropriate for new construction or for complete system replacement. There are some systems, however, that are appropriate for use in retrofitting of existing systems.

Costs for nitrogen reducing systems are significant. They range from $4,000 for some systems used to retrofit existing systems to $11,000 or more for nitrogen reduction units more appropriate for new construction. These costs are in addition to the costs for septic tanks and dispersal units. It is estimated that total system costs could be approximately $20,000 per system.

The increased complexities of these systems require additional monitoring and maintenance by trained professionals. Experience has shown that monitoring is critical at system start-up, but most manufacturers still recommend quarterly or semi-annual visits to ensure that the treatment processes are working properly.

Implementing a nitrogen removal strategy is difficult and will require policy decisions by regional leaders. By necessity implementation would need to take place over a period of years, perhaps decades. Options may range from installing sanitary sewers to establishing standards for individual wastewater treatment systems. Specific policy decisions would need to deal with issues such as identifying areas where nitrogen is a contaminant of concern, developing standards for upgrades of existing systems (retrofitting) and developing standards for new construction.

Nitrogen Removing Technologies for Onsite Wastewater Treatment Systems (Condensed Version)

Purpose

This report summarizes nitrogen treatment options for onsite sewage systems. It is intended for homeowners, onsite sewage industry professionals and public policy makers. Therefore, it is a non-technical introduction to nitrogen and nitrogen removal processes related to onsite wastewater systems. Appendix A provides a detailed and technically oriented discussion as well as specific information on a variety of processes and products. References are provided to other sources of detailed information on this continually expanding aspect of wastewater treatment.

Introduction

The impetus for this report, which was funded by a contract from the Puget Sound Action Team (PSAT), came from the low dissolved oxygen problems in Hood Canal that were identified in the Preliminary Assessment and Corrective Action (PACA) Plan for Hood Canal.

The PACA Plan can be found at http://www.psat.wa.gov/Programs/hood_canal.htm.

The PACA Plan provides preliminary estimates of the quantities of nitrogen resulting from human activities reaching the marine waters of Hood Canal. Among the sources of nitrogen loading to Hood Canal identified in the plan, onsite wastewater systems are estimated to be a significant contributor to this problem. Reducing nitrogen inputs from onsite wastewater systems within the drainage basins of Hood Canal could have significant long term positive water quality impacts.

However, removal of nitrogen is a complex process. Adding a nitrogen removal process to existing onsite sewage systems requires detailed planning, design and operational oversight. Significant research has been conducted for nitrogen removal and a wide variety of products have been promoted as effective nitrogen removal processes. The test results for these systems have been quite variable. Understanding the basics of the process of nitrogen removal and the reasons for varying nitrogen removal efficiencies is as important as knowing what products and technologies are being promoted in the marketplace.

Nitrogen in the Environment

Nitrogen exists in many forms. It is a common element constituting 78% of the earth's atmosphere. In its gaseous state nitrogen is odorless, tasteless and inert. In another state, it is also an essential constituent of amino and nucleic acids, the building blocks of life for all living organisms.

Nitrogen is a constituent in human sewage. The principal forms of nitrogen with regard to onsite wastewater treatment and soil-groundwater interactions are organic nitrogen, ammonia/ammonium ion (NH_3/ NH_4^+), nitrogen gas (N_2), nitrite (NO_2^-), and nitrate (NO_3^-). Nitrate in particular, because of its mobility in groundwater, is the form of nitrogen that is the primary focus of nitrogen removal technology.

The Nitrogen Cycle (see Figure 1 in Appendix A) illustrates the interrelationship of the environment and nitrogen products. The transformation of nitrogen compounds occurs through several key mechanisms.

- *Nitrogen Fixation.*

Nitrogen fixation is the conversion of nitrogen gas into nitrogen compounds that can be assimilated by plants. Biological fixation is the most common, but fixation can also occur by lightning and through industrial processes:

Biological: Nitrogen gas → Organic Nitrogen

Lightning: Nitrogen gas → Nitrate

Industrial: Nitrogen gas → Nitrate and Ammonia/Ammonium ion

- *Ammonification*.

Ammonification is the biochemical degradation of organic nitrogen into ammonia or ammonium ion by bacteria that use organic carbon in building cell tissue. These are called heterotrophic bacteria. These bacteria can transform the nitrogen either in the presence of oxygen (aerobic conditions) or without oxygen (anaerobic conditions).

Within an onsite wastewater system, ammonification of organic nitrogen in the human waste stream occurs primarily within the anaerobic conditions of the septic tank. Some of the organic nitrogen, however, is not degraded and becomes part of the humus in the receiving soils.

- *Synthesis*.

Synthesis is the biochemical mechanism in which ammonium ion or nitrate is converted into plant protein (organic nitrogen):

Nitrogen fixation is a unique form of synthesis that can only be performed by nitrogen-fixing bacteria and algae:

- *Nitrification*.

Nitrification is the biological oxidation of ammonium ion to nitrate through a two-step process by two species of bacteria called *Nitrosomonas* and *Nitrobacter*. In the first step, ammonium ions are converted to nitrite by *Nitrosomonas sp*. The second step involves the conversion of nitrite to nitrate by *Nitrobacter sp.* Both these species are considered autotrophic bacteria because they use carbon dioxide (CO_2) as the source of carbon for building cell tissue

The two-step reaction is usually very rapid. Because of this it is rare to find nitrite levels higher than 1.0 mg/L in water. The nitrate formed by nitrification is, in the nitrogen cycle, used by plants as a nitrogen source (synthesis) or reduced to N_2 gas through the process of denitrification. Nitrate can, however, contaminate groundwater if it is not used for synthesis or reduced through denitrification.

- *Denitrification*.

Nitrate can be transformed to nitrogen gas under conditions where dissolved oxygen is absent (called anoxic conditions) by heterotrophic bacteria (those that use organic carbon for building cell tissue).

In order for denitrification to occur, it must happen without dissolved oxygen present. If dissolved oxygen is present, the organisms will use it rather than the nitrate bound oxygen in their metabolism. In this latter case, nitrogen in the form of nitrates would remain to pass into and through the soil, eventually ending up in groundwater.

Environmental Effects

Health Effects from Drinking Groundwater Contaminated with Nitrates

Contamination of groundwater with nitrates is a problem in many parts of the U.S. and has been widely documented. Human health concerns from nitrates in groundwater used as a drinking water source primarily focus on methemoglobinemia, however some studies suggest that nitrates may increase the risk of birth defects and development of certain cancers in adults.

Methemoglobinemia (Blue Baby Syndrome)

High nitrate levels in drinking water supplies can cause methemoglobinemia in infants, especially those less than six months old. After ingestion, nitrate is reduced to nitrite in the gut of the infant. The absorbed nitrite reacts with hemoglobin in the infant's blood, forming methemoglobin. Methemoglobin, unlike hemoglobin, cannot carry oxygen. As more of the blood hemoglobin is converted to methemoglobin, the oxygen-carrying capacity of the blood is significantly reduced. Reductions in blood oxygen concentrations can result in a bluish discoloration of the body, the "blue baby" syndrome. To reduce the chance of methemoglobinemia, the maximum contaminant level of nitrate in drinking water has been set at 10 mg/L by the US EPA.

Birth Defects

Epidemiological studies in Canada and South Australia have shown a statistically significant increase in congenital malformations associated with consumption of nitrate-rich well water. These studies, however, are considered to be too limited in scope to deduce a causal association between birth defects and nitrate ingestion. Experimental animal studies have not shown significant effects from elevated nitrate ingestion.

Cancer in Adults

In the human body nitrates consumed in drinking water can be converted to nitrites and then to nitrosamines, several forms of which have been classified as potential human carcinogens. While some scientific studies have shown an association between some types of cancers and nitrate intake in animals, a cause-effect relationship for risk of cancer has not yet been demonstrated conclusively.

Surface Water Pollution with Nitrogen

When nitrogen compounds are discharged to surface waters in high concentrations, several deleterious effects may occur, depending on the environmental conditions.

Eutrophication

Nitrogen is generally the limiting factor for algae growth in coastal waters. Thus, excess nitrogen, primarily in the form of nitrates, can cause the stimulation of plant growth, resulting in algal blooms or overgrowth of aquatic plants, which can have serious consequences for the receiving water such as odors, accumulation of unsightly biomass, dissolved oxygen depletion due to biomass decay, and loss of fish and shellfish. See the Preliminary Assessment and Corrective Action Plan for a more detailed discussion of this process as it specifically affects Hood Canal.

Oxygen Demand through Nitrification

The oxidation of organic nitrogen and ammonia/ammonium ion to nitrate through the process of nitrification can exert a significant oxygen demand on the receiving water, which is known as the nitrogenous biochemical oxygen demand (NBOD). The NBOD, similar to carbonaceous biochemical oxygen demand (CBOD) can have the effect of depleting oxygen from the water body. The rate of nitrification is dependent on several environmental factors, which include the population of nitrifying bacteria, temperature, alkalinity, and availability of dissolved oxygen.

Ammonia Toxicity to Aquatic Organisms

Nitrogen in the form of ammonia can cause acute toxicity to several species of fish. Many municipal wastewater treatment plants in the US are required to nitrify their effluent in order to avoid ammonia toxicity in receiving waters.

Eutrophication is the most prominent surface water effect, since nitrate, being mobile, is most likely to travel through soil to surface waters. Nitrogenous oxygen demand and ammonia toxicity are both processes that can adversely affect surface water, but the ability of onsite wastewater treatment systems to nitrify them suggest that neither of these processes is a significant factor for Hood Canal. Ammonia toxicity, in particular, is not a significant issue because of the relatively low volumes and concentrations that would be released from individual onsite systems.

Human-related Sources of Nitrogen to the Environment

Agricultural activities are significant contributors of nitrates to the groundwater in many regions of the United States. Nationally, nitrogen-based fertilizers are considered the most important source of nitrate contamination. Livestock and dairy operations also contribute to nitrate loading of soils and contamination of groundwater due to the mobility of nitrates through the soil.

The Preliminary Assessment and Corrective Action Plan for Hood Canal identified human sewage (onsite systems), stormwater runoff, fish carcass disposal, agricultural wastes, and forestry practices as the primary anthropogenic, or human-caused, nitrogen sources of Hood Canal contamination. These findings are consistent with other areas of the country. Examples of areas dealing with nitrate concerns, besides Hood Canal, include Chesapeake Bay in Massachusetts and the Florida Keys.

The concentration of total nitrogen in residential wastewater ranges between 40 and100 mg/liter. A typical family of four using a conventional septic system can be expected to generate 20 to 50 pounds of nitrogen per year. Ten to thirty percent of this nitrogen is trapped in the septic tank as part of the sludge/scum accumulation in the tank. The nitrogen remaining in the liquid waste is transformed to nitrate when the wastewater leaves the anaerobic conditions of the septic tank and percolates through the aerobic environment of the soil portions of the drainfield. Although there is some potential for denitrification as the wastewater moves through the soil, the majority of the nitrogen produced by the family remains as nitrate loading to the soil. Drainfields installed 2-3 feet deep in soils where there is little organic matter, are relatively inefficient at removing nitrogen.

Historically, the primary purpose of onsite systems was to dispose of the wastewater by getting it underground, so it was out of sight and smell of the residents. More recently, with increased understanding of and advancements in onsite sewage technology, the emphasis has been on providing a known level of treatment before the wastewater reaches the groundwater. Where the site and soil conditions are not adequate to provide this treatment, more complex and costly systems are used to achieve current treatment goals. However, the focus has been on removing and mitigating the effects of Carbonaceous Biological Oxygen Demand (CBOD), Total Suspended Solids (TSS), and pathogenic bacteria. Increased complexity requires more active monitoring and maintenance to assure that the system is able to provide the desired treatment over its useful life.

To reduce nitrate contamination of groundwater, public health and water pollution control agencies have limited the density of land use. Reducing the concentration of onsite systems combined with rainfall dilution of treated effluent can help to meet groundwater nitrogen standards. A significant increase in land development pressures has raised interest in finding onsite sewage technologies that will more-effectively remove nitrogen before effluent is applied to the soil.

Some of the complex systems for meeting the CBOD, TSS and pathogen treatment requirements may also improve the level of nitrogen removal. However, the research shows that the levels of removal are inconsistent and widely variable. See Table 2 in Appendix A for examples of nitrogen removal by various types of these more complex treatment technologies.

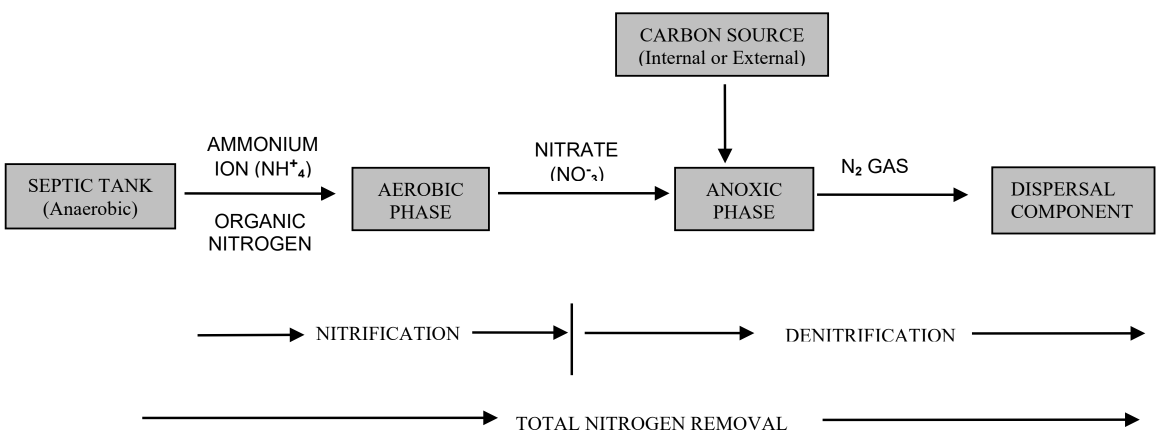
Treatment Processes

Treatment processes for nitrogen removal are generally premised on what is termed "sequential nitrification/denitrification". This process, when well-tuned, optimizes the natural biological processes using engineered systems. Although there are other possible processes, biological nitrification/denitrification is the only process that has been demonstrated to be economically and technically feasible for onsite nitrogen removal.

The first step in the sequence uses aerobic processes to transform the organic nitrogen and ammonia products in the septic tank effluent to nitrate. This is the nitrification step mentioned earlier in the discussion. A variety of treatment devices can be used to accomplish this aerobic process, such as sand or gravel filters or aerobic treatment units. For example, when septic tank effluent is applied at a low organic loading rate to deep, well aerated media, such as a two-foot deep, single pass sand filter, nitrification has been effectively accomplished. During this process, CBOD is also removed.

The second step requires shifting the process from an aerobic environment to an environment without dissolved oxygen (referred to as an anoxic process) where different species of bacteria can grow. These bacteria utilize the nitrate-bound oxygen formed in the first step to oxidize organic matter and in the process transform the nitrogen to gas. These bacteria also need organic carbon during the process in order to form new cell tissue. Inadequate supplies of organic carbon will limit the denitrification process. As will be discussed later, a carbon source must be provided for denitrification to occur.

Conceptually, the two-step process would look like this:



Biological Nitrification/Denitrification in Onsite Wastewater Systems

On paper, the process seems fairly straightforward. Unfortunately, real world operating conditions are more complicated. One of the most challenging tasks is to balance the environmental conditions needed to support the nitrifying bacteria with the different conditions needed by the bacteria that perform the denitrification process. Reconciling these differences makes nitrogen removal significantly more complex.

In addition, other inter-related factors affect the efficiency of the treatment process and engineering design must consider them in order to provide effective, consistent treatment.

These factors include:

- *Fluctuating Flow Rates*. The bacteria involved in both the aerobic and anoxic sequences can be adversely affected by either diminished flows (such as when the homeowners are on vacation), or by surge flows (such as large gatherings that cause peak flows). The problem of surge flows, for the most part, can be overcome by the use of timed dosing using programmable timers.

- *Fluctuating Waste Strengths*. Similar to waste flow impacts, varying waste strength can have an adverse impact on the bacterial colonies that keep the biological processes working.

- *Temperature*. Temperature variations can significantly affect the various bacteria involved in both the nitrification and denitrification steps. For instance, studies have shown that denitrification rates at 50◦ F (10◦ C) may be 20-40% lower than rates at 68◦ F (20◦ C). In order to compensate for this factor, longer detention times may be necessary in colder climates.

- *Alkalinity and pH*. The pH and alkalinity of the source water will have a dramatic effect on the rate of nitrification and denitrification. The optimum pH range for nitrification is 6.5-8.0. The biochemical process involved in nitrification consumes alkalinity and in areas with water sources that are low in alkalinity, nitrification will lower the pH to inhibitory levels for the nitrifying bacteria. Although the denitrification process produces some alkalinity, the net loss of alkalinity can be a limiting factor in areas with low alkalinity. Many water supplies in Western Washington, for example, have low alkalinity, near or less than 100 milligrams/liter, measured as calcium carbonate ($CaCO_3$). At these alkalinity levels, the ability to denitrify will be limited. See Appendix A, "pH and Alkalinity Effects" for a more in-depth discussion of the impacts of pH and alkalinity.

- *Inhibitory Compounds*. Because nitrogen transformation relies on bacterial processes, some chemicals can have immediate and serious impacts on the bacterial colonies living within the system. Nitrifying bacteria, in particular, are very susceptible to organic and inorganic inhibitors. Very small amounts of an inhibitor can kill these bacterial colonies and upset the nitrification process. Table 4 in Appendix A lists examples of inhibitors. Figure 6 in Appendix A illustrates the effects of a carpet cleaning solvent on the nitrification process in a recirculating sand filter.

Technologies for Removal of Nitrogen in Onsite Wastewater Systems

A wide variety of technologies provide enhanced nitrogen removal. Many utilize a combination of treatment processes. However, almost all of the processes include:

1. Aerobic and anoxic phases, and
2. Introduction of a carbon source during the denitrification step. This is done either by recirculating the nitrified wastewater back through the septic tank, which has high organic carbon content, or by adding an external carbon source to the denitrification unit.

Numerous non-proprietary products (those that are in the public domain) as well as proprietary products (those for which a patent exists) have been developed or considered for nitrogen reduction. In order to evaluate the efficiencies of these products, many have undergone or are currently involved in EPA sponsored test procedures. EPA and the National Sanitation Foundation (NSF) have collaborated on the development of the Environmental Technology Verification (ETV) protocol. The Barnstable County (Massachusetts) Department of Health and Environment conducts the ETV testing. The EPA/NSF ETV project is the only one of its kind and was established to facilitate the introduction of innovative or improved technologies through the performance verification process. Currently six products have completed the ETV testing protocol for denitrification. Table 9 of Appendix A.

In addition to the EPA/NSF ETV testing, EPA has funded demonstration projects in various areas of the country. These projects provide the opportunity for observing proprietary and non-proprietary products under field conditions to test their performance capabilities and identify operational issues. Table 10 in Appendix A provides a summary of selected proprietary products and their nitrogen removing performance.

For a more detailed discussion of the various technologies, see Part III, Examples of Onsite Nitrogen Removal Technologies in Appendix A. In addition, the California Water Resource Control Board has developed an extensive listing of products by type of system entitled, *Review of Technologies for the Onsite Treatment of Wastewater in California*. The full report can be accessed at http://www.waterboards.ca.gov/ab885/index.html.

Although not a treatment process, per se, dispersal of effluent into the soil using shallow trenches or subsurface drip distribution systems has the potential to further reduce total nitrogen levels. Both systems promote uptake of nitrate nitrogen by plant roots. In addition, by installing the system in the shallow soil where higher amounts of organic materials are present, denitrification may also take place. Studies of the use of shallow trenches have shown mixed results, with total nitrogen removal levels varying from 0-40%. Although it is difficult to quantify the amount of nitrogen uptake, coupling shallow trenches or subsurface drip distribution systems with other treatment technologies is an excellent way to enhance nitrogen removal.

Maintenance and Operational Oversight

Onsite wastewater treatment systems are clearly becoming more complex, even without nitrogen removal technologies being incorporated into their designs. The need for operational oversight and management is well recognized, although not successfully implemented on a consistent basis.

The additional complexity of nitrogen removal intensifies the demand for close system scrutiny. The variability in waste flows and waste characteristics, as well as the impacts of the other factors affecting nitrogen removal demand a high level of operational oversight, maintenance and periodic adjustment. For example, most of the ETV tested products recommend quarterly system checks to ensure that the system is in good operation condition.

Many manufacturers of nitrogen reducing products encourage the design of systems to serve multiple homes. This “clustering” can provide economy of scale for operation and maintenance. In addition, fluctuations in waste flows and waste strengths are leveled out by the multiple households served, thereby providing a more consistent and predictable waste product.

Since each individual system or cluster system is really a small wastewater treatment plant, a management entity is necessary to manage these systems. A method of collecting sufficient revenues to ensure the continuity of management for the life of the system needs to be addressed prior to the installation of these systems. This is especially true for cluster systems. Monitoring and servicing need to be provided by technicians trained for the specific type of system involved. Historically, homeowner associations have not exhibited strong success in managing community systems. Public Utility Districts or other public entities, if available and willing to serve as the management entity, have the capacity and legal authority to provide continuity of service.

Cost Considerations

It can be generally stated that adding nitrogen removal to a standard onsite system design will increase the cost of system construction. According to manufacturers' estimates, prices for ETV tested products range from $7500 -$11,000 per unit. Most of these products are intended for installation as a system component. Estimating the cost for a fully installed system is more difficult, since site conditions and regional installation costs vary significantly. However, most of these companies' project costs in the range of $20,000 for a complete system including septic tanks, nitrogen removal devices, and a dispersal component. Several manufacturers of ETV-tested products suggest that, where new development is planned, homes be clustered and sewage be piped to a shared, neighborhood treatment and dispersal facility. They contend that this type of development results in more consistent sewage treatment and is more economical for both users and the developer.

An exception to these costs, among ETV tested products, is the Bio-Microbics RetroFAST® device. It is intended for retrofitting into the second compartment of an existing septic tank. Costs are estimated to be $4000-$5000, assuming it can be placed in the existing septic tank.

The costs for non-ETV tested proprietary products appear to be very similar to the costs associated with the ETV tested products. Cost ranges vary widely depending on the particular device or combination of devices involved. Site conditions or site limitations also affect the total cost.

Not surprisingly, the additional capital and operating costs associated with adding nitrogen removal devices have severely limited their widespread use. Installations are mostly limited to areas where a community-wide determination has been made to increase the level of sewage treatment to protect a public water supply or other valued community resource.

Maintenance of individual systems can be a significant expense to the homeowner. Some studies have indicated annual operation and maintenance costs to be approximately $1500 per year. An economy of scale may result if maintenance personnel have several systems of the same type near each other, but this is difficult to establish. System owners may be able to reduce these costs by becoming trained by the manufacturer to undertake some routine monitoring and maintenance tasks.

Summary

The removal of nitrogen from sewage using an onsite wastewater treatment system involves natural biological processes. A variety of proprietary technologies have been developed for the purpose of enhancing these natural processes. However, none has been demonstrated to provide a simple, effective, and consistently reproducible effluent.

Review of research and field application data indicates that removal rates between 50-70% can be reached fairly consistently. However, attaining greater nitrogen reduction rates, or more-importantly, achieving consistently low nitrogen concentrations in the treated effluent, requires increasingly more complex treatment and greater oversight in order to keep all aspects of the nitrification/denitrification processes in balance.

Monitoring and maintenance are key elements in the successful, long-term operation of any onsite wastewater treatment system. However, they become even more critical when more complex devices are added to remove nitrogen.

When considering the use of nitrogen removal equipment, whether for an individual site or as a general community policy, the cost of the initial installation and the essential need for long term maintenance are factors that must be taken into account. These costs should be weighed against clearly identified and well-explained health and/or environmental benefits that are being sought.

Appendix A – Nitrogen Reducing Technologies (Full Version)

Report to the Puget Sound Action Team
June 2005

Part I: Nitrogen in the Environment

A. Chemistry of Nitrogen

Nitrogen, which in its pure gaseous form comprises 78% by volume of the earth's atmosphere, can exist in nine various forms in the environment due to seven possible oxidation states (WEF, 1998):

Nitrogen Compound	Formula	Oxidation State
Organic nitrogen	Organic-N	-3
Ammonia	NH_3	-3
Ammonium ion	NH_4^+	-3
Nitrogen gas	N_2	0
Nitrous oxide	N_2O	+1
Nitric oxide	NO	+2
Nitrite ion	NO_2^-	+3
Nitrogen dioxide	NO_2	+4
Nitrate ion	NO_3^-	+5

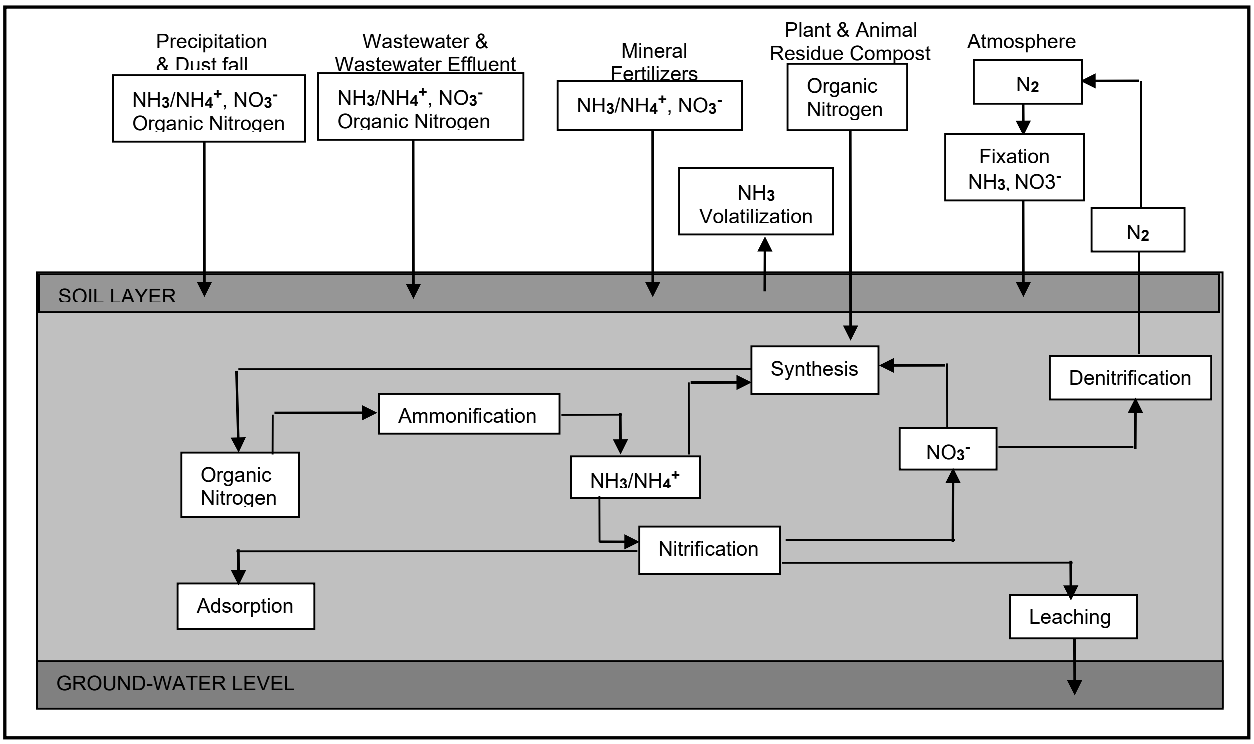
The principal forms of nitrogen of concern in onsite wastewater treatment and soil-groundwater interactions are Organic-N, ammonia/ammonium ion (NH_3/ NH_4^+), nitrogen gas (N_2), nitrite ($NO_2^{-)}$, and nitrate (NO_3^-) (Rittman & McCarty, 2001; Sawyer et al., 1994; US EPA, 1993).

B. The Nitrogen Cycle in Soil Groundwater Systems

As shown in Figure 1, transformation of the principal nitrogen compounds can occur through several key mechanisms in the environment: fixation, ammonification, synthesis, nitrification, and denitrification (US EPA, 1993).

Figure 1: The nitrogen cycle in soil and groundwater

Adapted from U.S EPA (1993)



Nitrogen Fixation.

Nitrogen fixation is the conversion of nitrogen gas into nitrogen compounds that can be assimilated by plants. Biological fixation is the most common, but fixation can also occur by lightning and through industrial processes:

Biological: $N_2 \rightarrow$ Organic-N

Lightning: $N_2 \rightarrow NO_3^-$

Industrial: $N_2 \rightarrow NO_3^-$; NH_3/ NH_4^+

Ammonification.

Ammonification is the biochemical degradation of organic-N into NH_3 or NH_4^+ by bacteria that use organic carbon as the carbon source in building cell tissue. These are called heterotrophic bacteria. These bacteria can transform nitrogen either in the presence of oxygen (aerobic conditions) or without oxygen (anaerobic conditions).

Organic-N + Microorganisms → NH_3/ NH_4^+

Some organic-N cannot be degraded and becomes part of the humus in soils.

Synthesis.

Synthesis is the biochemical mechanism in which ammonium ion or nitrate is converted into plant protein (Organic-N):

NH_4^+ + CO_2 + green plants + sunlight → Organic-N

NO_3^- + CO_2 + green plants + sunlight → Organic-N

Nitrogen fixation is also a unique form of synthesis that can only be performed by nitrogen-fixing bacteria and algae (WEF, 1998):

N-Fixing

Bacteria &Algae

N_2 → Organic-N

Nitrification.

Nitrification is the biological oxidation of NH_4^+ to NO_3^- through a two-step autotrophic process by the bacteria Nitrosomonas and Nitrobacter (Rittman and McCarty, 2001; Sawyer, et al., 1994):

Nitrosomonas

Step 1: $NH_4^+ + 3/2O_2 \rightarrow NO_2^- + 2H^+ + H_2O$

Nitrobacter

Step 2: $NO_2^- + 1/2O_2 \rightarrow NO_3^-$

The two-step reactions are usually very rapid and hence it is rare to find nitrite levels higher than 1.0 mg/L in water (Sawyer, et al., 1994). The nitrate formed by nitrification is, in the nitrogen cycle, used by plants as a nitrogen source (synthesis) or reduced to N_2 gas through the process of denitrification. Nitrate can, however, contaminate groundwater if it is not used for synthesis or reduced through denitrification as shown in Figure 1.

Denitrification.

NO_3^- can be reduced, under conditions where oxygen is absent (termed anoxic conditions), to nitrogen gas by heterotrophic bacteria (those that use organic carbon as a source of carbon for building cell tissue) as shown in the following unbalanced equation (US EPA, 1993):

$$NO_3^- + \text{Organic Matter} \xrightarrow{\textit{Heterotrophic Bacteria}} N_2 + CO_2 + OH^- + H_2O$$

The above equation is identical to the equation below for biological oxidation of organic matter except that it occurs without the presence of oxygen. In order for denitrification to occur, it must happen without oxygen present. If oxygen is present, the organisms will preferentially use the oxygen rather than the nitrogen to oxidize the organic matter. When this occurs, nitrates remain to pass into and through the soil, eventually ending up in the groundwater. Thus, it is very important that anoxic conditions exist in order that the nitrate ion (NO_3^-) will be used as the electron acceptor. It is also critical that a sufficient carbon source be available to serve as the electron donor in order for denitrification to occur.

$$O_2 + \text{Organic Matter} \xrightarrow{\textit{Heterotrophic Bacteria}} CO_2 + OH^- + H_2O$$

Autotrophic denitrification is also possible with either elemental sulfur or hydrogen gas used as the electron donor by autotrophic bacteria, but it is not a significant process in the treatment of wastewater.

C. Environmental Effects of Nitrogen Discharges

1. Health Effects from Groundwater Contamination with Nitrates

Contamination of groundwater with nitrates is a problem in many parts of the U.S. and has been widely documented (Bouchard, et al., 1992). Potential health concerns where contaminated groundwater is used as a drinking water source include methemoglobinemia, carcinogenesis, and birth defects.

Methemoglobinemia.

High nitrate levels in drinking water supplies can cause methemoglobinemia in infants, especially those less than six months old (Bouchard, et al., 1992). After ingestion, nitrate is reduced to nitrite in the gut of the infant. The absorbed nitrite reacts with hemoglobin in the blood, forming methemoglobin. Methemoglobin, unlike hemoglobin, cannot carry oxygen. As more of the blood hemoglobin is converted to methemoglobin, the oxygen-carrying capacity of the blood is significantly reduced. Reduction in blood oxygen concentrations can result in a bluish discoloration of the body, which is called "blue baby" syndrome or methemoglobinemia. To prevent methemoglobinemia, the maximum contaminant level of nitrate in drinking water has been set at 10 mg/L by the US EPA (Bouchard, et al., 1992).

Carcinogenesis.

High nitrate levels in drinking water could potentially have carcinogenic effects through the formation of nitrosamines. In the human body nitrates can be converted to nitrites and then to nitrosamines, several forms of which have been classified as potential human carcinogens (Bouchard, et al., 1992). While some scientific studies have shown a positive correlation between some types of cancers and nitrate intake in animals, a cause-effect relationship for risk of cancer has not yet been demonstrated conclusively.

Birth Defects.

Epidemiological studies in Canada and South Australia have shown a statistically significant increase in congenital malformations associated with nitrate-rich well water (Bouchard, et al., 1992). These studies, however, are considered to be too limited in scope to deduce a causal association between birth defects and nitrate ingestion. Experimental animal studies have not shown significant effects from elevated nitrate ingestion.

2. Surface Water Pollution with Nitrogen

When nitrogen compounds are discharged to surface waters in high concentrations, several deleterious effects may occur, depending on the environmental conditions.

Eutrophication.

Nitrogen is generally the limiting factor for algae growth in coastal waters (CENR, 2003; NRC, 2000). Excess nitrogen, normally in the form of nitrates, due to its soluble nature and mobility through soil, can cause the stimulation of growth, resulting in algal blooms or overgrowth of aquatic plants, which can have serious consequences for the receiving water such as odors, accumulation of unsightly biomass, dissolved oxygen depletion due to biomass decay, and mortality of fish and shellfish. See the Preliminary Assessment and Corrective Action Plan for a more detailed discussion of this process as it specifically affects Hood Canal.

Oxygen Demand through Nitrification.

The oxidation of organic nitrogen and ammonia/ammonium ion to nitrate through the process of nitrification can exert a significant oxygen demand on the receiving water, which is known as the nitrogenous biochemical oxygen demand (NBOD) (Metcalf and Eddy, 1991). The NBOD, like carbonaceous biochemical oxygen demand (CBOD), can have the same effect of depleting oxygen from a water body although it may not be exerted as rapidly. Although a different biochemical process, the effects of NBOD are similar to eutrophication. The rate of nitrification is dependent on several environmental factors, which include the population of nitrifying bacteria, temperature, alkalinity, and availability of dissolved oxygen.

Ammonia Toxicity to Aquatic Organisms.

Nitrogen in the form of ammonia can cause acute toxicity to several species of fish. Because the concentration of ammonia (NH_3-N) as opposed to ammonium ion (NH_4^+-N) is pH dependent, criteria for ambient water quality have been set for un-ionized ammonia as a function of pH and temperature (Sawyer, et al., 1994). Many municipal wastewater treatment plants in the US are required to nitrify their effluent in order to avoid ammonia toxicity in receiving waters.

3. Anthropogenic Sources of Nitrogen Discharges to Groundwater

Agricultural Activities.

Agricultural activities are a significant source of nitrate in groundwater. Nitrate can enter groundwater at elevated levels by excessive or inappropriate use of nitrogen-containing nutrient sources, which include commercial fertilizers and animal manures. Certain types of crops and cropping systems also affect nitrogen levels in soils.

Nitrogen fertilizer use increased five-fold during the period 1955-1988, and it is believed that misuse of nitrogen fertilizers is the most important source of nitrate contamination of groundwater in the US (Power and Schepers, 1989; Hallberg, 1989). Most nitrogen fertilizer is applied as anhydrous ammonia, urea, or as nitrate or ammonium salt. In an aerobic soil environment, much of the applied ammonia products can be transformed to nitrate ion (NO_3^-) which readily migrates to groundwater through most soil types as a result of its negative charge. Under anoxic conditions in the presence of a carbon source, however, nitrate can be reduced to atmospheric nitrogenous gases (N_2, NO and N_2O). Livestock and dairy practices that concentrate animals, such as feedlots, can also significantly contribute to nitrate contamination of groundwater if the animal wastes generated by the operation are not properly managed. (Bouchard, et al., 1992).

The types of crop and cropping system are also important in determining the potential for nitrate migration to groundwater (Bouchard, et al., 1992). Irrigated agriculture on sandy soils, and heavily fertilized, shallow-rooted crops, favor nitrate leaching. In animal production areas nitrogen contributions from manure and leguminous forages often results in significant nitrogen loading to groundwater (Nowak, et al., 1997).

Septic Tank-Soil Absorption Systems.

"Overall, from the available data, we calculate that nitrogen leached from onsite sewage systems is clearly the largest source entering Hood Canal. Although not precise, we estimate that sewage contributes between 33% and 84% of all anthropogenic nitrogen entering Hood Canal. The exact figure is likely to be in the middle of the range and will be improved as more data are gathered through the USGS focus studies and HCDOP."

-Preliminary Assessment and Corrective Action Plan

Contamination of groundwater with nitrates from septic tank-soil absorption systems is also a problem in many parts of the US. The build-up of nitrate in groundwater is one of the most significant long-term consequences of onsite wastewater disposal (Hantzsche and Finne-more, 1992). As an example, the annual nitrogen contribution for a family of four from a conventional septic system on a quarter acre lot would be approximately 50 lbs. per year (Hantzsche and Finnemore, 1992). The annual nitrogen requirement for a quarter acre of Bermuda grass, much of which may be supplied by fertilizer, is also about 50 lbs. per year (WEF, 2001). The problem, however, is that the nitrogen from septic tank-soil absorption systems is not uniformly distributed throughout a lawn and may be discharged at a depth below which plants can utilize it. Nitrogen primarily exists as Organic-N and ammonia products in septic tank effluent and is usually transformed into nitrate as the wastewater percolates through the soil column beneath the system's drainfield. Also, the nitrogen loading from high housing densities can greatly exceed any potential plant uptake of nitrogen even if the effluent were properly applied. (Gold and Sims, 2000; County of Butte, 1998; Hantzsche and Finnemore, 1992).

Part II: Theory, Design and Processes of Onsite Nitrogen Removal Systems

A. Processes for Biological Nitrogen Removal

1. Centralized Wastewater Treatment

Nitrogen removal through biological nitrification/denitrification, as practiced in centralized wastewater treatment, is generally classified as an advanced treatment process (Metcalf & Eddy, 1991). Detailed information on wastewater flows and characteristics is required for successful design, operation, and troubleshooting if nitrogen removal is to be successful. As a result, design and operational parameters have been widely published in order to advance knowledge and improve design and operation. These parameters include alkalinity requirements, organic loading rates necessary to achieve nitrification, and stoichiometric equivalencies for various reactions.

2. Onsite Wastewater Treatment Systems

Unfortunately, the same cannot be said of onsite nitrogen removal. Much of the published literature does not report data in terms of parameters that can be used to rigorously assess systems, compare them to other sites, and improve design and operation. As an example, the loading rates on single pass sand filter (SPSF) systems are almost exclusively expressed in terms of hydraulic loading rates. The most useful information in terms of nitrification, however, would be organic loading rates. A few studies such as Converse (1999) have reported on organic loading rates. Alkalinity concentrations are also very rarely monitored in onsite wastewater treatment studies but are fundamental in assessing the limits on nitrification.

What is needed are general design and operational parameters for the various onsite technologies so their performance can be adequately assessed and designs and operation improved.

Eventually onsite nitrogen removal technologies will have requirements similar to large-scale systems for their successful design and operation. It is unreasonable to assume that one particular design will work for all the ranges of wastewater flows and strengths, and environmental conditions. Design changes and operational adjustments will be required for the particular situation at hand... What is needed are general design and operational parameters for the various onsite technologies so their performance can be adequately assessed, and their designs and operation can be improved. Prudence dictates that the following information, at minimum, must be taken into account when assessing performance of onsite nitrogen removal technologies.

Wastewater Flows.

Wastewater flows from single family dwellings can range from 8 to 85 gallons per capita per day (gpcd), with an average of approximately 45 gpcd (Ayres Associates, 1993). Thus, depending on the number of occupants in a given dwelling, it is possible that the daily flowrate could range from 16 gallons per day (gpd) to 425 gpd if, for example, only two occupants discharge the lowest range of flows cited in the literature (8 gpcd) or five occupants the highest end of the range (85 gpcd). This range of flow rates could have a significant effect on nitrification/denitrification processes if they are not incorporated into the design or operational adjustments.

Time dosing with programmable timers can help manage peak flow rates better and thus produce a higher potential for nitrogen removal.

In addition, wastewater is generated by discrete events and typical wastewater hydrographs show that wastewater flow varies widely throughout a 24-hour period (US EPA, 1980). This wide fluctuation in flow rates can also have a significant effect on nitrogen removal processes. For example, the denitrification process can be adversely affected if there is an interruption in the recycling of nitrified effluent to mix with septic tank influent. Timed dosing with programmable timers can help manage peak flow rates better and thus produce a higher potential for nitrogen removal.

Wastewater Characteristics.

Oxygen demand of septic tank effluent has been reported to range from 7 to 480 mg/L as BOD_5 (Ayres Associates, 1993). If the wastewater is used as the carbon source, the variability in BOD_5 needs to be taken into consideration during design and operation in order to maximize nitrogen removal. High concentrations of BOD_5 in will inhibit nitrification while low concentrations will inhibit denitrification. Other wastewater characteristics, such as alkalinity, pH, BOD_5/TKN ratios, temperature, and existence of inhibitors, also need to be taken into consideration.

Technological Assessment and Design Considerations.

Table 1 summarizes the ranges of select wastewater constituents from septic tank effluent that have been cited in the literature. The wide variability emphasizes the importance of designing systems to the range of conditions that can be encountered.

Table 1: Ranges of Concentrations of Select Wastewater Constituents in Septic Tank Effluent

Constituent	Without Effluent Filter (mg/L)	With Effluent Filter (mg/L)
BOD_5	7-480	100-140
TKN	9-125	50-90
Alkalinity (as $CaCO_3$)	---	70-594

Adapted from Adolfson Associates (1999), Ayres Associates (1993), Converse (1999), Crites and Tchobanoglous (1998), and Oakley, et al. (1998).

B. Control of Nitrogen Discharges from Onsite Systems

Historically, nitrogen has not been a contaminant of concern in the design of onsite systems. The primary purpose of onsite systems was to dispose of the wastewater, getting it underground or at least out of sight/smell of the residents. As population increased and the science of wastewater treatment advanced it became clear that both treatment and dispersal were essential. However, even with advancements in treatment and disposal knowledge, the primary emphasis has been on mitigating the effects of BOD, TSS and bacterial/viral pathogens.

As the understanding of the impacts of nitrate groundwater contamination from septic tank soil absorption systems increased, public health and water pollution control agencies tried either to limit the number of onsite systems in a given area (establish minimum lot sizes) based on nitrogen loading to the soil, or by examining alternative onsite technologies that provide nitrogen removal (Ayres Associates,

Anti-degradation of Water Quality

State policy requires that discharges into receiving water shall not further degrade the existing water quality of the water body. In cases where natural conditions of receiving water are of lower quality than the criteria assigned, the natural condition shall constitute the water quality criteria. Similarly, when receiving waters are of higher quality than the criteria assigned, the existing water quality shall be protected.

Chapter 173-200 (030)
Washington Administrative Code

1998; California Regional Water Quality Control Board, 1997; Whitmeyer et al., 1991).

One method of quantifying nitrogen loadings to groundwater is based upon the measured factors of rainfall, aquifer recharge, septic system nitrogen loadings, and denitrification (Hantzsche and Finnemore, 1992). If the volume and total nitrogen concentration of wastewater applied over a development area can be estimated, along with the possible denitrification fraction, then the resultant concentration of nitrate in groundwater can be calculated when rainfall and recharge rates to the aquifer are known.

Because the maximum contaminant level (MCL) of nitrate in drinking water supplies is 10 mg/L, there is a tendency among regulatory agencies to take the stance that Total-N in effluents from onsite nitrogen removal systems should not be greater than 10 mg/L. The available performance data from decentralized onsite nitrogen removal systems show, however, that total nitrogen concentrations of 10 mg/L or less cannot be satisfied with any degree of reliability at the present time. This is not the case, however, with large-scale systems that are continuously monitored and receive regular operation and maintenance (US EPA, 1993). Thus, the protection of groundwater quality from onsite systems will likely continue to involve quantification of permissible nitrogen loading rates per unit area coupled with nitrogen removal technologies that will be required to satisfy some type of effluent performance standard.

C. Dynamics in Septic Tank Soil Absorption Systems

1. Wastewater Characteristics

The mass loading of nitrogen in domestic wastewater averages from 4 to 18 lbs. (1.8 to 8.1 kg) of Total-N per capita per year. Untreated domestic wastewater typically contains 20 to 85 mg/L Total-N, with the majority occurring as a mixture of ammonia and ammonium ion (12-50 mg/L) and organic nitrogen (8-35 mg/L) (Metcalf & Eddy, 1991). Because the carbon to nitrogen ratio of wastewater is typically on the order of 4:1 to 6:1, there will be excess nitrogen after secondary biological treatment (BOD removal) that cannot be assimilated by microorganisms.

It is for this reason that special attention must be given to nitrogen removal processes in wastewater engineering.

2. Septic Tanks

The removal of Total-N within standard septic tank and drainfield systems is on the order of 10%-30%, with the majority being removed in the septic tank as particulate matter through sedimentation or flotation processes. Because of the septic tank's anaerobic environment, nitrogen exists principally as organic nitrogen, ammonia and ammonium ion. The combination of these forms of nitrogen is referred to as Total Kjelldahl Nitrogen (TKN). Organic nitrogen is transformed to ammonia and ammonium ion via ammonification. Because the anaerobic environment in the septic tank is a dynamic environment, some of these ammonia products are reconverted back to organic nitrogen via bacterial cell growth. However, there will be a net increase of ammonia products in the septic tank effluent.

3. Subsurface Soil Absorption Trenches

Nitrogen can undergo several transformations within and below subsurface soil absorption trenches (Ayres Associates, 1993). These transformations include:

- adsorption of ammonium ion to soil particles in the soil;
- volatilization of ammonia in alkaline soils when the pH is above 8.0;
- nitrification and subsequent movement of nitrate towards the groundwater;
- biological uptake of both ammonia/ammonium and nitrate; and
- denitrification.

Within a well-designed and constructed subsurface absorption trench, diffusion of oxygen into the aerated soil layer above the water table (vadose zone) promotes the biological oxidation of ammonium ion to nitrate through biological nitrification. With adequate soil moisture and organic matter to serve as the carbon source within the soil column, nitrate can be reduced, under anoxic conditions, to nitrogen gas through heterotrophic denitrification. Although denitrification may be significant in some soils (Ayres Associates, 1993), in most instances there may not be sufficient organic substrate at a depth below the 'A' horizon (the shallow organic topsoil layer) to promote denitrification. Under these conditions nitrate can migrate downward into the groundwater aquifer, depending on soil moisture conditions (saturated or unsaturated flow).

The use of shallow trenches can also enhance the uptake of ammonium and nitrate by plant roots. The historical practice of constructing relatively deep subsurface soil absorption trenches (2 to 4 ft.) for septic tank effluents may often have the effect of diminishing denitrification potential and enhancing nitrate movement in the soil column.

D. Treatment Processes for Onsite Nitrogen Removal

1. Sequential Nitrification/Denitrification Process

Sequential nitrification/denitrification processes, which attempt to optimize natural biological processes through engineering, form the basis of all biological nitrogen removal technologies that have been used or proposed for onsite wastewater treatment. In these systems aerobic processes are first used to remove BOD and nitrify organic and ammonium nitrogen; anoxic processes are then used to reduce nitrate to nitrogen gas, either using the wastewater as a carbon source or an external carbon source. Figure 2 shows a conceptual model of biological nitrification and denitrification in onsite wastewater treatment. Although there are other possible processes, biological nitrification/denitrification is the only process that has been demonstrated to be feasible, both economically and technically, for onsite nitrogen removal (the same can be said for large-scale wastewater treatment plants) (Whitmeyer, et al., 1991). Table 2 gives a summary of onsite nitrogen removal systems that have been reported in the literature; these systems will be discussed in more detail in the section "Examples of Onsite Nitrogen Removal Technologies".

Figure 2: Biological Nitrification/Denitrification in Onsite Wastewater Treatment

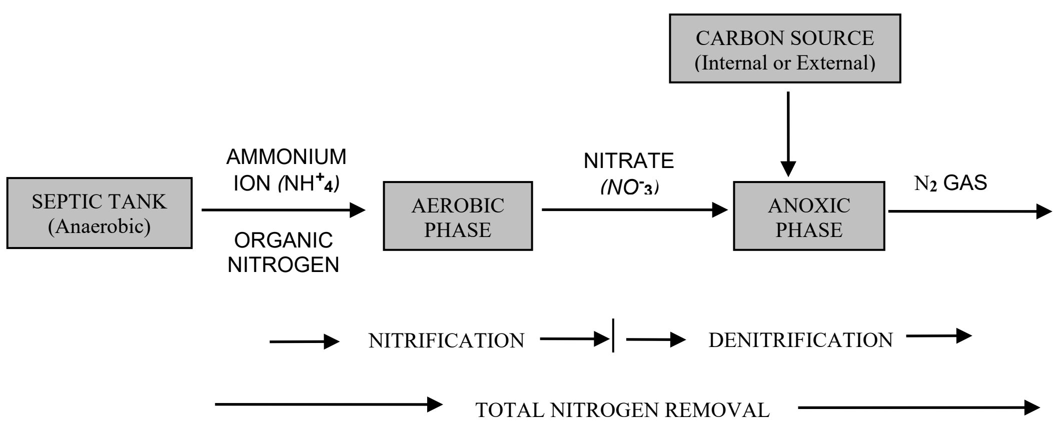


Figure 2: Biological Nitrification/Denitrification in Onsite Wastewater Treatment
Adapted from Gold, et al (1989)

2. Classification of Biological Nitrogen Removal Systems

Suspended-growth processes are biological treatment processes in which the microorganisms responsible for treatment are maintained in suspension within the liquid, usually through mechanical or diffused-air aeration. Attached-growth processes are those in which the microorganisms responsible for treatment are attached to an inert medium such as sand, gravel, or plastic media, and can include either submerged or nonsubmerged processes (Crites and Tchobanoglous, 1998; Metcalf & Eddy, 1991). Using the terminology of wastewater engineering, the systems outlined in Table 2 are categorized according to whether they are suspended growth or attached-growth processes.

Table 2: Examples of Onsite Biological Nitrogen Removal from the Literature

Technology Examples		Total-N Removal Efficiency, %	Effluent Total-N (mg/L)
Suspended Growth:			
	Aerobic units w/pulse aeration	25-61[1]	37-60[1]
	Sequencing batch reactor	60[2]	15.5[2]
Attached Growth			
	Single-Pass Sand Filters (SPSF)	8-50[3]	30-65[3]
	Recirculating Sand/Gravel Filters (RSF)	15-84[4]	10-47[4]
	Multi-Pass Textile Filters	14-38[5, 9]	9-83[5, 9]
	RSF w/Anoxic Filter	40-90[6]	7-23[6]
	RSF w/Anoxic Filter w/external carbon source	74-80[7]	10-13[7]
	RUCK system	29-54[8]	18-53[8]
	Nitrex	96[10]	2.2 [10]

[1]California Regional Water Quality Control Board (1997); Whitmeyer, et al., (1991).

[2]Ayres Associates (1998).

[3]Converse, (1999); Gold, et al., (1992); Loomis, et al., (2001); Nolte & Associates, (1992); Ronayne, et al., (1982).

[4]California Regional Water Quality Control Board (1997); Gold, et al. (1992); Loomis, et al. (2001); Nolte & Associates (1992); Oakley, et al. (1999); Piluk and Peters (1994); Ronayne, et al., (1982).

[5]Leverenz, et al. (2001).

[6]Ayres Associates (1998); Sandy, et al. (1988).

[7]Gold, et al. (1989).

[8]Brooks (1996); Gold, et al. (1989).

[9]Loomis, et al (2001)

[10]Rich, et al (2003)

E. Biological Nitrification

1. Process Chemistry

As mentioned above, nitrification is a two-step autotrophic process (nitrifiers use CO_2 instead of organic carbon as their carbon source for cell synthesis) for the conversion of ammonium ion (NH_4^{+}) to nitrate. During this energy yielding reaction some of the ammonium ion is synthesized into cell tissue giving the following overall oxidation and synthesis reaction (US EPA, 1993):

$$1.00NH_4^+ + 1.89O_2 + 0.08CO_2 \xrightarrow[\text{Bacteria}]{\text{Autotrophic}} 0.98NO_3^- + \underbrace{0.016C_5H_7O_2N}_{\text{new bacterial cells}} + 0.95H_2O + 1.98H^+$$

The above equation poses several key design constraints on nitrification systems. In mass terms, 4.32 mg of oxygen are required for each mg of ammonium ion oxidized, with the subsequent loss of 7.1 mg of alkalinity as calcium carbonate ($CaCO_3$) in the wastewater, and the synthesis of 0.1 mg of new bacterial cells. Stated yet another way, the oxidation of, for example, 20 mg/L of ammonium ion would require the consumption of 86.4 mg/L of dissolved oxygen, the destruction of 141.4 mg/L of alkalinity as $CaCO_3$, and the production of 2.6 mg/L of nitrifying organisms (US EPA, 1993).

Nitrification can thus exert a very high nitrogenous biochemical oxygen demand (NBOD) in addition to the carbonaceous BOD (CBOD). Using the above equation, a septic tank effluent of 40 mg/L ammonium ion would have a NBOD of about 184 mg/L in addition to the CBOD. This factor must be included in the design of nitrification systems to be sure there is sufficient dissolved oxygen (DO) within the system for nitrification to occur. Nitrification can also cause a significant drop in pH if there is not adequate buffering capacity in the wastewater.

2. Process Microbiology

Autotrophic organisms involved in nitrification (those that use carbon dioxide as a source of carbon for cell tissue formation) exhibit much lower growth rates than r heterotrophic bacteria (those that use organic carbon for cell tissue formation). As a result, the rate of nitrification is controlled by oxidation of CBOD by heterotrophic bacteria. As long as there is a high organic

(CBOD) loading to the system, the heterotrophic bacteria will dominate. Nitrification systems must thus be designed to allow sufficient detention time within the system for nitrifying bacteria to grow. Heterotrophic organisms can also play a key role in limiting oxygen transfer to nitrifying bacteria, especially in attached-growth systems (Rittman and McCarty, 2001; US EPA, 1993). After competition with heterotrophs, the rate of nitrification will be limited by the concentration of available ammonium ion in the system. Temperature, pH, and chemical inhibitors can also play a key role as discussed below.

Figure 3: Percent Nitrifiers vs. BOD/TKN Ration

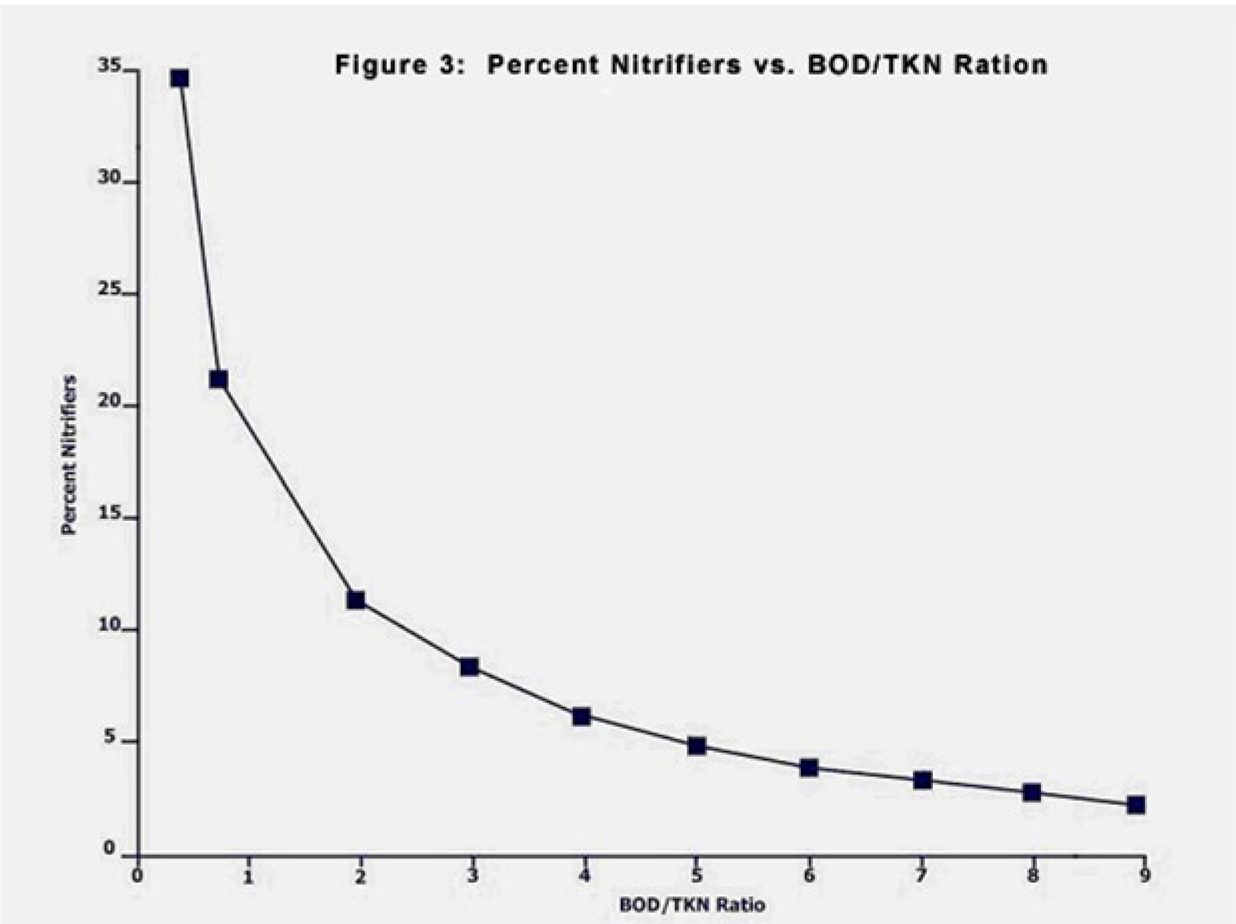


Figure 3 shows the relationship between the fraction of nitrifying organisms in suspended-growth wastewater treatment (activated sludge) and the BOD_5/TKN ratio. At low BOD_5/TKN ratios (0.5 to 3) the population of nitrifying bacteria is high, and nitrification should not be influenced by heterotrophic oxidation of CBOD (Metcalf & Eddy, 1991). This type of nitrification process is termed separate-stage nitrification. At higher BOD_5/TKN ratios, the fraction of nitrifying organisms in the system is much lower due to heterotrophic competition from oxidation of CBOD; this process is termed single-stage nitrification. Examples of single-stage and separate-stage nitrification are shown in Figure 4.

Figure 4: Single Stage and Separate Stage Nitrification

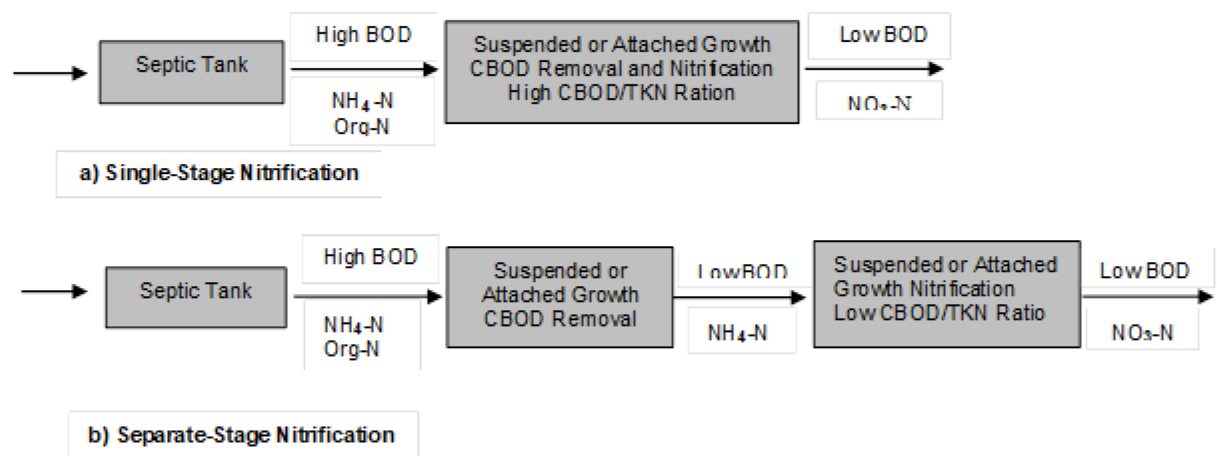


Figure 4: Single Stage and Separate Stage Nitrification

Separate-stage nitrification is highly desirable from the standpoint of process control and operation. Many onsite systems presently used or proposed for nitrogen removal, however, employ single-stage nitrification. This is because of the interest in reducing system size and footprint. Examples include aerobic treatment units with short hydraulic detention times and sand filters or media filters that are heavily loaded organically. Single-stage systems may require more rigorous process control to ensure adequate nitrification rates.

Another advantage of separate stage nitrification is that inhibitory effects can oftentimes be controlled by designing separate-stage nitrification systems (US EPA, 1993). Since heterotrophic bacteria are much more resilient than nitrifying bacteria, and because many of the inhibitory compounds are biodegradable organics, a separate-stage system first removes the biodegradable inhibitory compounds along with CBOD. The nitrifying organisms, which are in effect protected in the second stage, are then used to nitrify the low-CBOD, high ammonium effluent.

3. Dissolved Oxygen Requirements and Organic Loading Rates

Suspended Growth Systems

The concentration of dissolved oxygen (DO) has a significant effect on nitrification in wastewater treatment. Although much research has been performed, practical experience has shown that DO levels must be maintained at approximately 2.0 mg/L in suspended-growth (aerobic) systems, especially when ammonium ion loadings are expected to fluctuate widely (US EPA, 1993); this may or may not be the case in domestic onsite wastewater systems.

Attached-Growth Systems.

For attached-growth systems, which include both submerged and nonsubmerged processes (Crites and Tchobanoglous, 1998), DO levels must be maintained at levels that are at least 2.7 times greater than the ammonium ion concentrations in order to prevent oxygen transfer through the biofilm from limiting nitrification rates (US EPA, 1993). This is usually overcome in practice by using lower organic surface loadings than what would be normally applied for CBOD removal to allow for growth of nitrifying organisms; otherwise, the heterotrophic organisms will dominate the bacterial film within the attached-growth media. For trickling filters, for example, the organic loading rate for nitrification is only about 1/5 to 1/8 of the CBOD loading for CBOD removal (Metcalf & Eddy, 1991; US EPA, 1993). Recirculation of effluent through the attached growth media, and use of special media, such as trickling filter plastic media with high specific surface areas, is also used to lower organic surface loadings and to promote high oxygen transfer rates.

Unfortunately, organic loading rates for onsite attached-growth systems are not well defined even for CBOD removal, let alone nitrification (Crites and Tchobanoglous, 1998). The more commonly used hydraulic loading rates as cited in the literature show mixed results for nitrification. This is no doubt due, at least in part, to varying organic loading rates that were not taken into consideration since the $CBOD_5$ of septic tank effluent can vary greatly, ranging from less than 100 to 480 mg/L (Ayres Associates, 1993). Table 3 shows design organic loading rates for various attached-growth systems to achieve nitrification.

Table 3: Design Loading Rates for Attached Growth Systems to Achieve >85% Nitrification[1]

Process		Hydraulic Loading Rate, gpd/ft^2	Organic Loading Rate, lbs BOD/ft^2-day	State of Knowledge for Design
Trickling Filters[2]				
	Rock Media	30-900	0.04-0.12	Well Known

	Plastic Media	288-1700	0.10-0.25	Well Known
Sand Filters				
	Single-Pass	0.4-1.2	0.000135-0.002	Lesser Known[3]
	Recirculating	3-5	0.002-0.008	Lesser Known[3]
Textile Filters				
	Single-Pass	10	0.01	Lesser Known[3]
	Multi-Pass[4] (Partial Nitrification)	30	0.03	Lesser Known[3]

1. Adapted from Converse (1999); Crites and Tchobanaglous (1998); Leverenz, et al. (2001); Metcalf & Eddy (1991); and US EPA (1993).

2. The values for trickling filters given for both hydraulic and organic loadings are the ranges for low rate and high-rate filters. Rock filters were assumed to have a depth of 8 ft. and plastic filters a depth of 10 ft.

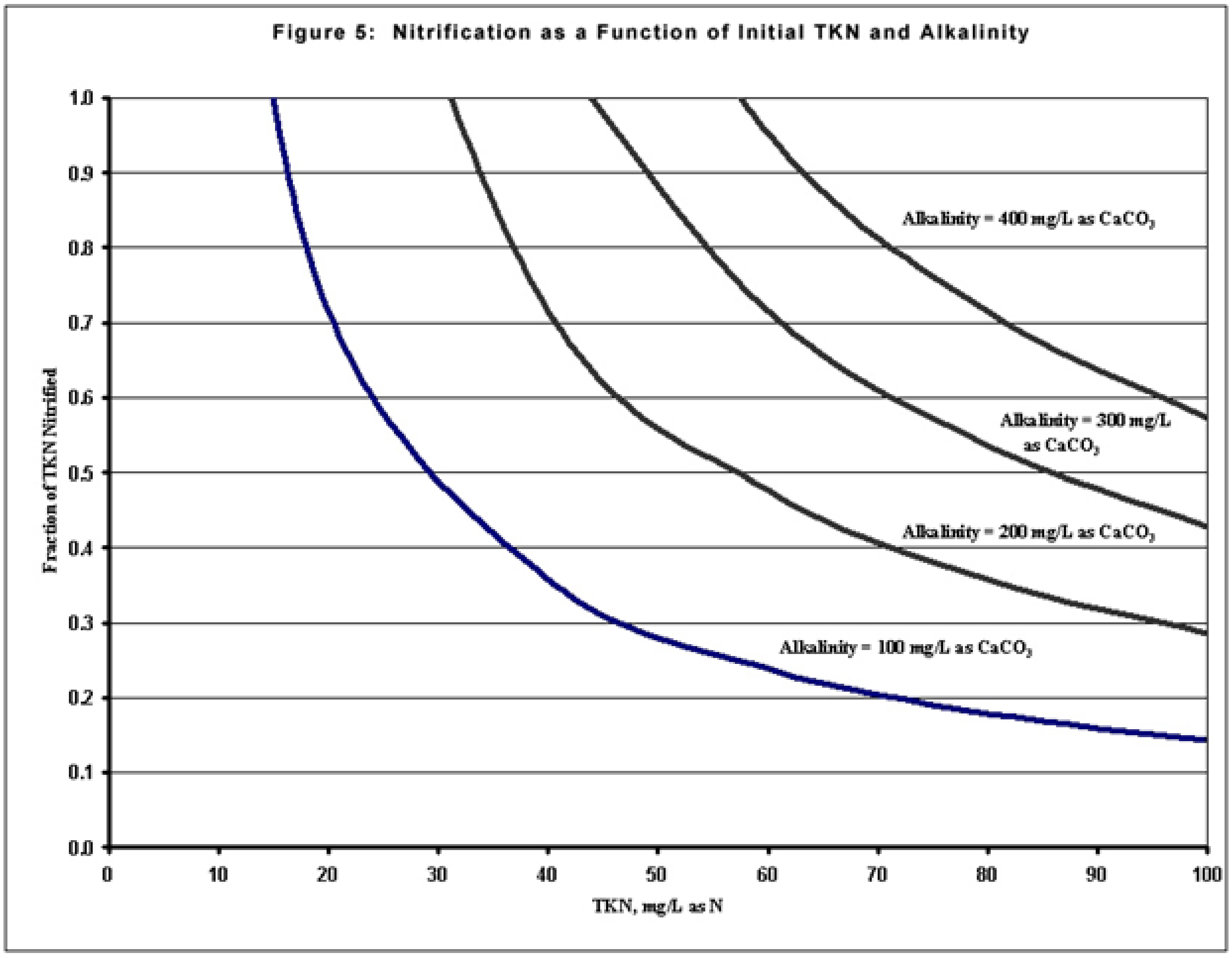
3. These systems have not traditionally been designed using organic loading rates to achieve nitrification. High strength wastes thus could affect nitrification performance.

4. At this organic loading rate only 59-76% nitrification was achieved (Leverenz, et al., 2001).

4. pH and Alkalinity Effects

The optimum pH range for nitrification is 6.5 to 8.0 (US EPA, 1993). Because nitrification consumes about 7.1 mg of alkalinity (as $CaCO_3$) for every mg of ammonium ion oxidized, there is a risk in low alkalinity wastewaters that nitrification will lower the pH to inhibitory levels. If, for example, it was desired to nitrify 40 mg/L of ammonium ion, approximately 284 mg/L as alkalinity would be required to maintain pH levels. This may be beyond the capabilities of some wastewaters derived from water sources that do not contain relatively high alkalinity. Figure 5 shows the theoretical relationship of the fraction of Total Kjedahl Nitrogen (TKN) that can be nitrified as a function of initial TKN and alkalinity in the wastewater.

Figure 5: Nitrification as a Function of Initial TKN and Alkalinity



5. Temperature Effects

Temperature has a significant effect on nitrification that must be taken into consideration for design (US EPA, 1993). In general, colder temperatures require longer cell residence times in suspended-growth systems and lower hydraulic loading rates in attached-growth systems due to slower growth rates of nitrifying bacteria.

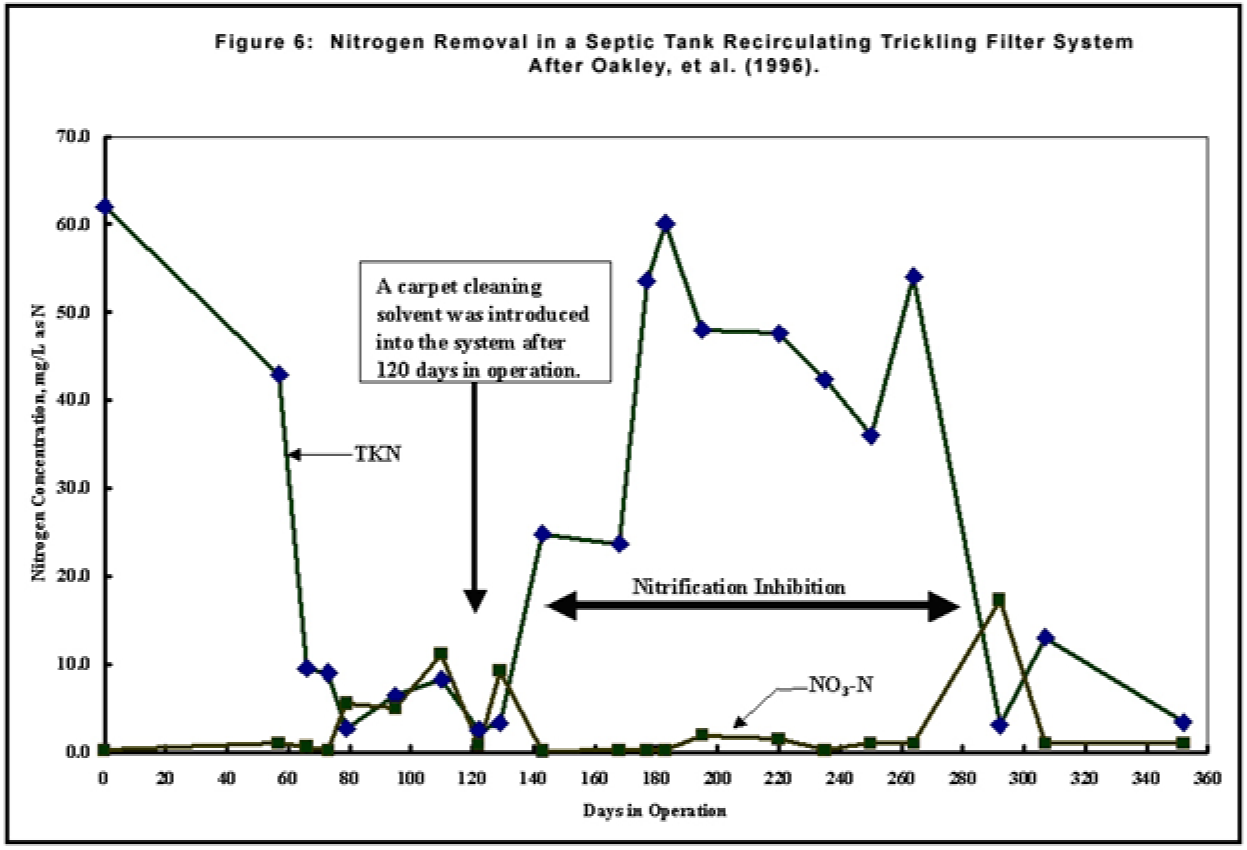
6. Effect of Inhibitors

Nitrifying bacteria are much more sensitive than heterotrophic bacteria and are susceptible to a wide range of organic and inorganic inhibitors as shown in Table 4. As has occurred in centralized wastewater treatment (US EPA, 1993), there is a need to establish a methodology for onsite wastewater systems for assessing the potential for, and occurrence of, nitrification inhibition. Figure 6 illustrates the effect of an inhibitor on nitrification in a septic tank/recirculating trickling filter system; in this particular case a carpet cleaning solvent that was flushed down the toilet contaminated the septic tank and destroyed the nitrifying bacterial population in the attached-growth media (Oakley, et al., 1996). If this system had not been continuously monitored, the effects of the inhibitor on nitrification would have passed unnoticed.

Table 4: Examples of Nitrification Inhibitors

Inorganic Compounds		Organic Compounds
Zinc	Thiocyanate	Acetone
Free Cyanide	Sodium cyanide	Carbon Disulfide
Perchlorate	Sodium azide	Chloroform
Copper	Hydrazine	Ethanol
Mercury	Sodium cyanate	Phenol
Chromium	Potassium chromate	Ethylenediamine
Nickel	Cadmium	Hexamethylene diamine
Silver	Arsenic	Aniline
Cobalt	Fluoride	Monoethanolamine
Lead	Free Ammonia	
Free Nitrous Acid		

Figure 6: Nitrogen Removal in a Septic Tank Recirculating Trickling Filter System



7. Summary of Nitrification Processes

Table 5 summarizes the various onsite technologies and their advantages and disadvantages for effective nitrification based on the factors discussed above. The available information suggests that an effective design strategy for nitrification in onsite systems would be to use attached-growth processes with relatively low organic loadings (compared to CBOD removal only) and deep, well-aerated media (such as a 2 foot deep Single Pass Sand Filter). This type of system would approach a separate-stage nitrification with its advantages while maintaining the cost and simplicity of a single-stage system. In this design the heterotrophic bacteria would grow in the upper levels and remove CBOD and inhibitory compounds; nitrifying bacteria would grow in the lower levels and would be protected both

An effective design strategy for nitrification in onsite systems would be to use attached-growth processes with relatively low organic loadings and deep, well-aerated media (such as a 2 ft. deep Single Pass Sand Filter).

from shock loadings and temperature extremes. A single pass sand filter, which is well known for its nitrification reliability, is an example of this design.

Table 5: Onsite Technologies for > 85% Nitrification

Process	Effectiveness	Onsite Status
Suspended Growth:		
Aerobic units	Potential	Insufficient design and performance data. Operation and maintenance unknown.
Attached Growth:		
Single-Pass Sand Filters (SPSF)	Proven	Widespread use. Need more design data for organic loadings for nitrification. Fair to good performance in cold climates.
Recirculating Sand Filters (RSF)	Proven	Widespread use. Need more design data for organic loadings for nitrification. Poorer performance in cold climates than SPSFs.
Single-Pass Textile Filters	Potential	Limited data to date. Probably similar to SPSF. Need design data for organic loadings for nitrification.
Multi-Pass Textile Filters	Potential	Limited data to date. Probably similar to RSF. Need design data for organic loadings for nitrification and performance in cold climates.

F. Biological Denitrification

1. Process Description

Denitrification is a biological process that uses either nitrate as the electron acceptor instead of oxygen to oxidize organic matter (heterotrophic denitrification) or inorganic matter such as sulfur or hydrogen (autotrophic denitrification). In the process nitrate is reduced to nitrogen gas. Both methods must occur under anoxic conditions (Rittmann and McCarty, 2001). Denitrifying bacteria, whether heterotrophic or autotrophic, are facultative aerobes. That is, they can shift between oxygen respiration and nitrate respiration. Because the principal biochemical pathway is a modification of aerobic pathways (i.e., NO_3^- is used as the electron acceptor instead of O_2), the denitrification process is said to occur under anoxic conditions as opposed to anaerobic conditions (where organisms that can only survive without oxygen would be present).

Autotrophic denitrification, which is more commonly used in water treatment rather than wastewater treatment, uses elemental sulfur or hydrogen gas as the electron donor. Some research (Shan and Zhang, 1998) used sulfur and limestone to denitrify septic tank effluent, but the process has never been put into practical use. Therefore, autotrophic denitrification will not be discussed in greater detail in this paper.

For heterotrophic denitrification, the carbon source can come from the original wastewater, bacterial cell material, or an external source such as methanol or acetate. The possible process configurations for heterotrophic denitrification are shown in Figure 7.

Figure 7: Heterotrophic Denitrification Configurations

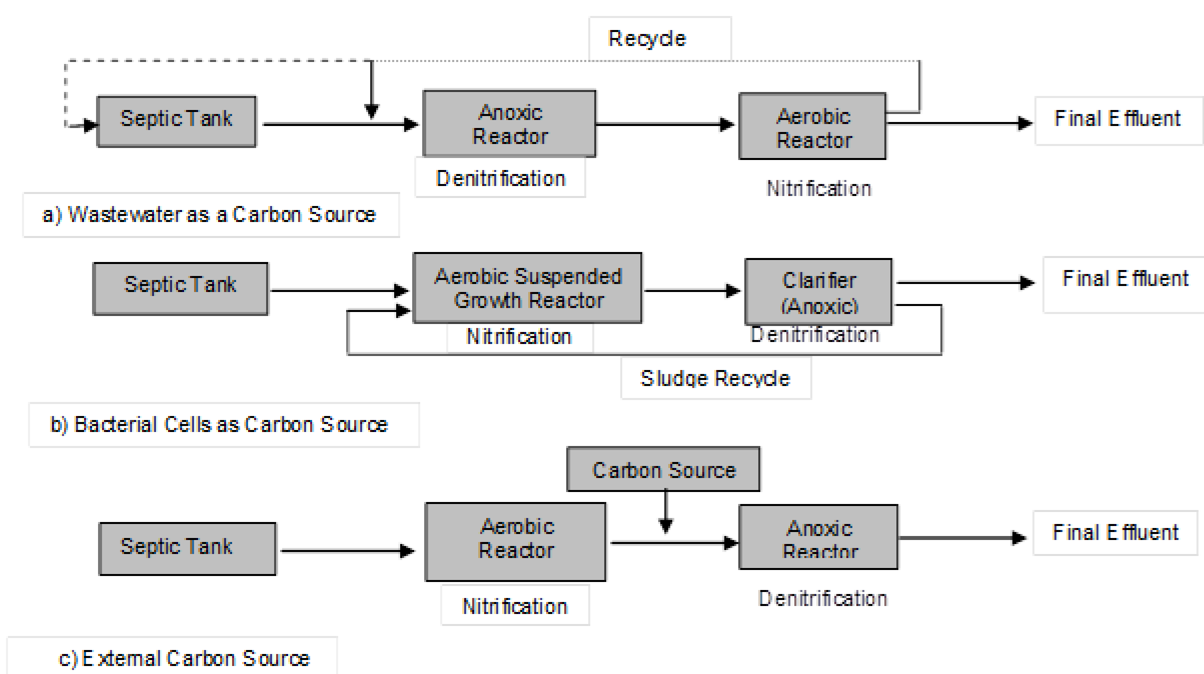


Figure 7: Heterotrophic Denitrification Configurations

2. Process Microbiology of Heterotrophic Denitrification

The differences between the autotrophic nitrifiers and the heterotrophic denitrifiers underscore the complexity of the design and operation of nitrification/denitrification systems. The nitrifiers are slow growers, sensitive to inhibitory compounds, and require low organic carbon concentrations with high dissolved oxygen concentrations; the denitrifiers, on the other hand, are fast growers, resilient to inhibitory compounds, and require high organic carbon concentrations with low or absent dissolved oxygen. The reconciliation of these differences is the greatest challenge to successful onsite nitrogen removal.

The differences between the autotrophic nitrifiers and the heterotrophic denitrifiers underscore the complexity of the design and operation of nitrification/denitrification systems. The reconciliation of these differences is the greatest challenge to successful onsite nitrogen removal.

The heterotrophic denitrifying bacteria are facultative aerobes that can use either oxygen or nitrate (under anoxic conditions) as an electron acceptor for the oxidation of organic matter. Denitrifiers are commonly found in nature and are omnipresent in wastewater. Denitrifying bacteria are very diverse and don't have the growth and competition problems associated with autotrophic nitrifying bacteria.

When an adequate carbon source is available, the principal problem associated with denitrification is the achievement of anoxic conditions. The dissolved oxygen concentration controls whether or not the denitrifying bacteria use nitrate or oxygen as the electron acceptor (US EPA, 1993). Dissolved oxygen must not be present above certain maximum levels, or the denitrifying bacteria will preferentially use it for oxidation of organic matter rather than nitrate. As a result, the design of anoxic zones is one of the most important factors in denitrification processes.

3. Wastewater as Carbon Source

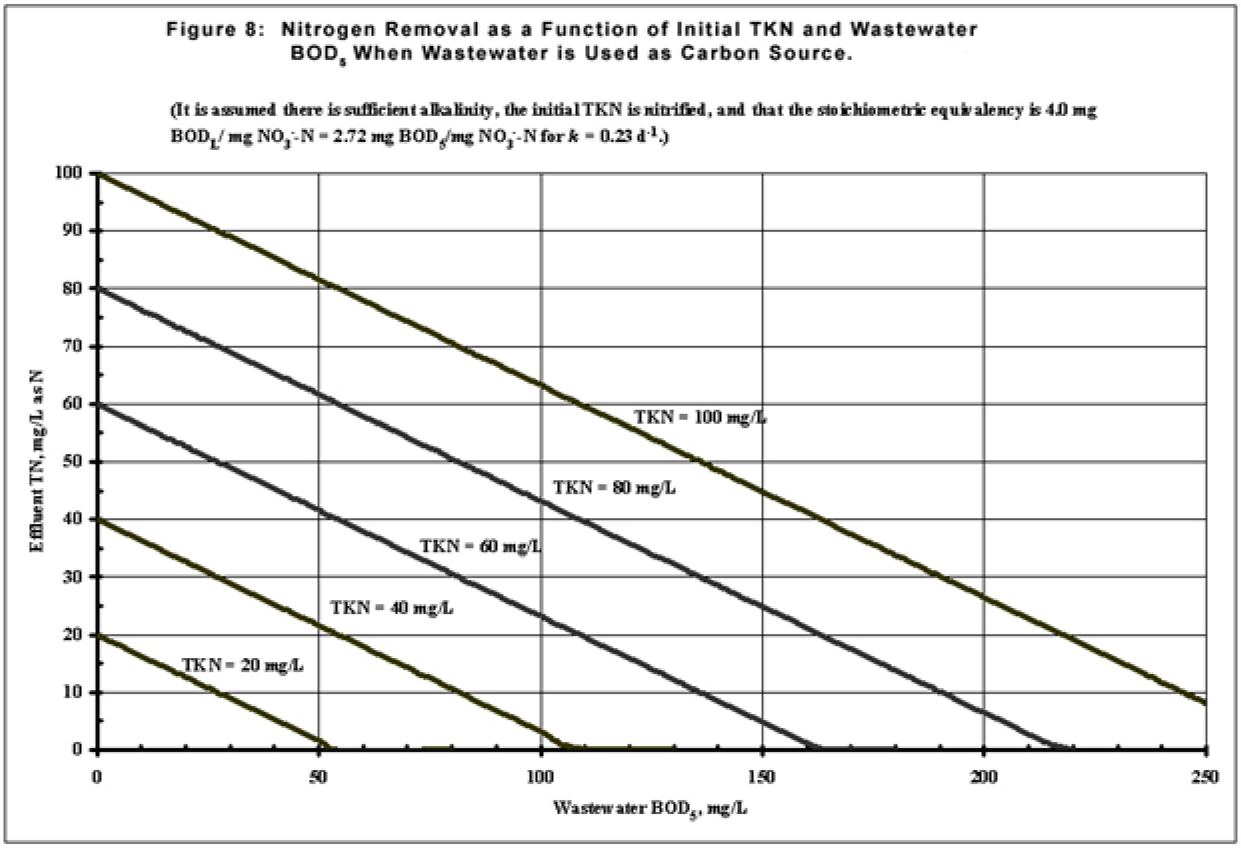
The following unbalanced equation illustrates the process when wastewater or bacterial cell material is used as the carbon source (US EPA, 1993):

$$\underbrace{COHNS}_{\text{organic matter}} + NO_3^- \xrightarrow{\text{Heterotrophic Bacteria}} N_2 + CO_2 + \underbrace{C_5H_7O_2N}_{\text{bacterial cells}} + OH^- + H_2O + \text{end products}$$

Theoretically, 2.86 mg of oxygen is equivalent to 1.0 mg of nitrate in the transfer of one electron (US EPA, 1993). Assuming that the wastewater itself will be the carbon source for denitrification, at least 2.86 mg of oxygen in the form of CBOD is thus required to reduce, through denitrification, 1.0 mg of nitrate to nitrogen gas (EPA, 1993). In practice, approximately 4.0 mg of oxygen is actually needed to reduce 1.0 mg of nitrate because a portion of the CBOD must be used for bacterial growth rather than the reduction of nitrate. Assuming an average of 175 mg/L BOD_5 in a septic tank effluent, the maximum possible removal of Total-N would thus be about 44 mg/L (e.g. 175 mg/L ÷4 = 43.75). Therefore, at least from a theoretical standpoint and assuming there are no problems with alkalinity or toxicity, there should normally be sufficient indigenous carbon in domestic wastewater from septic tanks to have almost complete nitrogen removal. The problem, of course, is being able to realize this potential through engineering design and appropriate operation and maintenance of biological systems. The variability in removal efficiencies shown in Table 2 underscores the difficulty in approaching the theoretical potential in practice.

Figure 8, which assumes the "rule of thumb" equivalency of 4.0 mg BOD/mg nitrate, shows total nitrogen removal as a function of initial TKN and wastewater BOD_5. In this figure it is assumed there is sufficient alkalinity for nitrification. It is obvious from Figure 8 that nitrogen removal by denitrification using wastewater as the carbon source is highly feasible for an initial TKN of 40 mg/L or less but becomes more problematic as the initial TKN increases in relation to BOD_5.

Figure 8: Nitrogen Removal as a Function of Initial TKN and Wastewater BOD, When Wastewater is Used as Carbon Source



In cases where there is insufficient CBOD left in the wastewater to serve as an electron donor for denitrification, an external carbon source must be supplied. Although there are many possibilities, methanol and acetate have been studied the most (Rittmann and McCarty, 2001; US EPA, 1993).

There are, however, few examples in the literature of an external carbon source being used for onsite denitrification. Although methanol has been studied extensively in centralized wastewater treatment plants, it is probably not a good choice for onsite systems because of its toxicity and potential for contaminating groundwater supplies. Gold, et al., (1989) reported on the use of both methanol and ethanol as an external carbon source in a recirculating sand filter system with an anoxic rock filter for denitrification. They noted that although the total nitrogen removal rate was as high as 80%, the use of the chemicals required operation and maintenance of the carbon source supply system, including an on-site storage facility, a metering pump mechanism, and supplying a diluted carbon source solution. They concluded that the external carbon source could probably best be handled by a wastewater management district or a private O & M contractor (Gold, et al., 1989).

4. pH and Alkalinity Effects

Theoretically, 3.57 mg of alkalinity as $CaCO_3$ is produced for each mg of nitrate reduced to nitrogen gas when the wastewater is used as the carbon source. Thus, denitrification can recover approximately half of the alkalinity lost in nitrification and can help overcome pH drops in low alkalinity waters. Because denitrifying organisms are heterotrophic, they normally will be affected by pH changes in the same way other heterotrophic bacteria are affected. Generally, it has been found that denitrification rates are depressed below pH 6.0 and above pH 8.0 (US EPA, 1993).

If an external carbon source is used, the alkalinity lost may or may not be recovered, depending on the chemistry of the carbon source. In the case of methanol, the alkalinity is recovered, while with acetic acid the quantity of acid added to the system is neutralized by the alkalinity produced and no alkalinity is recovered through denitrification.

5. Temperature Effects

The data from the literature suggest that denitrification rates can be significantly affected by temperature drops below 20° C (68° F), with the denitrification rate at 10 °C (50° F) ranging from 20% to 40% of the rate at 20°C (US EPA, 1993). It can be expected that this decrease is similar to that encountered for heterotrophic organisms removing CBOD and should be taken into consideration for designs in cold climates, where longer hydraulic retention times may be needed. There is also some evidence of a slight depression of denitrification rates at temperatures above 20-25 °C (US EPA, 1993).

6. Inhibitory Effects

In general, denitrifiers are much more resilient than nitrifying organisms. Denitrifiers most likely exhibit the same characteristics as heterotrophic bacteria for CBOD removal to inhibitory compounds.

7. Summary of Denitrification Processes

Table 6 summarizes the three processes for heterotrophic denitrification (which are shown in Figure 7) with their advantages and disadvantages for onsite nitrogen removal. In summary, organic carbon can be provided in the following ways:

- As an external carbon source to an anoxic reactor after nitrification;
- As an internal source in the form of bacterial cells through a sequential process of aerobic and anoxic zones;

- The influent wastewater can be used as the carbon source by recycling nitrified effluent to an anoxic reactor that precedes the aerobic nitrification reactor, operating alternating aerobic/anoxic zones on one reactor (sequencing batch reactor) or conveying the flow sequentially through alternating aerobic/anoxic zones (US EPA, 1993). Denitrification reactors can be designed as suspended-growth or attached-growth processes.

Table 6: Onsite Processes for Heterotrophic Denitrification

Process	Advantages	Disadvantages
External Carbon Source	High removal rates. Denitrification easily controlled.	Insufficient performance data for onsite systems. Operation and maintenance data lacking. Routine monitoring required. Alkalinity lost through nitrification may or may not be recovered, depending on the carbon source.
Wastewater as Carbon Source	Lower energy and chemical requirements. Fifty percent recovery of alkalinity lost through nitrification. Fifty percent reduction in O_2 requirements for CBOD removal.	Insufficient performance data. Process difficult to control. Routine monitoring required. Operation and maintenance data lacking.
Bacterial Cells as Carbon Source	Lower energy and chemical requirements.	Insufficient performance data. Process difficult to control. Routine monitoring required. Operation and maintenance data lacking.

Part III: Examples of Onsite Nitrogen Removal Technologies

There are a wide variety of technologies, many of which use a combination of treatment applications in order to maximize nutrient removal. Categorizing the various processes and treatment products varies widely and is complex and confusing to most persons. For instance, systems can be categorized as anoxic, anaerobic, suspended growth, attached growth, attached growth/suspended growth with each category having sub-categories.

Here is an example of how the California State Water Resource Control Board (2002) has categorized the various types of systems.

Categorizing Treatment Systems[1]

ANOXIC AND ANAEROBIC SYSTEMS

- Anoxic systems
- Anaerobic systems

TRICKLING BIOFILTERS (ATTACHED GROWTH AEROBIC TREATMENT SYSTEMS)

- Granular media trickling biofilters
- Organic media trickling biofilters
- Synthetic media trickling biofilters

SUSPENDED GROWTH AEROBIC TREATMENT SYSTEMS

- Complete mix reactors
- Sequencing batch reactors
- Membrane bioreactors

COMBINED SUSPENDED AND ATTACHED GROWTH AEROBIC TREATMENT SYSTEMS

- Continuous flow packed beds
- Continuous flow with suspended internal packing
- Rotating biological contactors
- Sequencing batch reactors

[1]Adapted from California State Water Resource Control Board (2002)

The California Water Resource Control Board's *Review of Technologies for the Onsite Treatment of Wastewater in California*, catalogs and describes the various products, by type of system. This report provides an extensive listing of products and is an excellent resource. The full report can be accessed at: http://www.waterboards.ca.gov/ab885/index.html.

Two onsite nitrogen removal technologies that have been widely discussed in the literature are suspended growth systems and attached growth systems. Table 2 summarizes each system's removal efficiency and effluent Total-N concentration as cited in the literature.

A. Suspended-Growth Systems

1. Aerobic Units with Pulse Aeration

These units are, in principle, extended aeration activated sludge systems in which aeration is periodically stopped or pulsed to promote denitrification. Operational data on these systems is lacking although nitrogen removal efficiencies have been reported to be only in the range of 25-35 percent (Whitmeyer, et al, 1991).

A recent study in Los Osos, California for one unit installed at a single-family residence showed removal efficiencies to range from 38-61 percent, with effluent Total-N concentrations ranging between 37 and 60 mg/L (California Regional Water Quality Control Board, 1997).

Figure 9: Process Diagram for Suspended Growth Systems: Pulse Aeration and Sequencing Batch Reactors with Aerobic/Anoxic Cycles

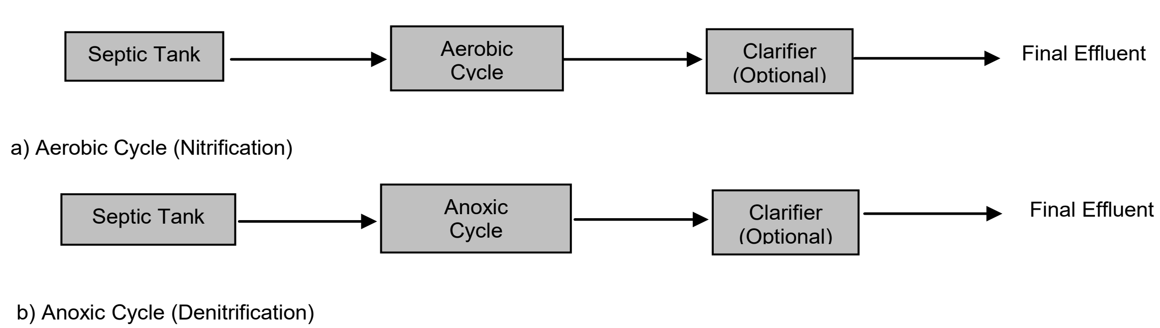


Figure 9: Process Diagram for Suspended Growth Systems: Pulse Aeration and Sequencing Batch Reactors with Aerobic/Anoxic Cycles

2. Sequencing Batch Reactor (SBR)

The SBR differs generally from aerobic units in that fill-and-draw, and alternating aerobic and anoxic cycles, are created within a single reactor. During the anoxic phase sedimentation takes place and the supernatant is pumped from the reactor. The carbon source is provided by both bacteria living in the reactor as well as influent wastewater. SBR technology has been demonstrated to be an excellent nitrogen control technology for large-scale systems (US EPA, 1993), but there is a scarcity of information for onsite systems.

One proprietary onsite system tested in Florida in an experimental facility exhibited an average of 60% Total-N removal, with effluent Total-N concentrations averaging 15.5 mg/L for an average influent TKN concentration of 38.4 mg/L (Ayres Associates, 1998). Unfortunately, no mention was made of alkalinity concentrations in this study. There was approximately 92% nitrification of influent TKN, so the system was apparently not nitrification limited. However, since the average influent BOD_5 of 170 mg/L (Ayres Associates, 1998), was ample as a carbon source to remove all of the total nitrogen, it can be assumed the system was not operating efficiently in terms of denitrification.

Sludge bulking in suspended-growth systems needs to be considered since all suspended growth processes require secondary sedimentation to remove the bacterial flocs. As denitrification is being promoted, rising nitrogen gas can create a potential problem with sludge bulking. This is also a problem with large-scale systems (Metcalf & Eddy, 1991). Whether this problem can be dealt with adequately in onsite systems has yet to be determined and it has not been discussed frequently in the literature.

B. Attached-Growth Systems

1. Single Pass Sand Filters (SPSF)

SPSF technology is the most studied of all proposed nitrogen removal technologies. The mechanism of nitrogen removal includes a combination of CBOD removal and nitrification within the sand medium at low organic loadings (low BOD_5/TKN ratio), and subsequent denitrification within anoxic microenvironments in the sand. Total-N removal rates with SPSFs have been quoted in the literature as ranging from 8% to 50% (Converse, 1999; Gold, et al., 1992; Loomis, et al., 2001; Nolte & Associates, 1992; Ronayne, et al., 1982).

The greatest advantage of SPSF technology is in the achievement of nitrification. The percentage of TKN nitrification in SPSF systems has been reported to range between 75% to 96% (Converse, 1999; Gold, et al., 1992; Nolte & Associates, 1992; Ronayne, et al., 1982). The results of the work by Converse (1999) and Gold, et al. (1992) suggest that nitrification rates in winter months do not change significantly from summer months in buried filters.

Unfortunately, as was discussed previously, there is a paucity of sound design data for nitrification based on organic loading rates. Most of the loading rates have been reported in terms of hydraulic loadings rather than organic loadings. Also, accurate data on measured loadings per unit area based on the type of distribution system used, as opposed to calculated loadings, are difficult to come by (Converse, 1999).

Assuming there is sufficient alkalinity for nitrification, it can be expected that SPSF systems will always be denitrification-limited due to the lack of availability of both a carbon source and anoxic conditions.

Figure 10: Process Diagrams for Attached Growth Systems

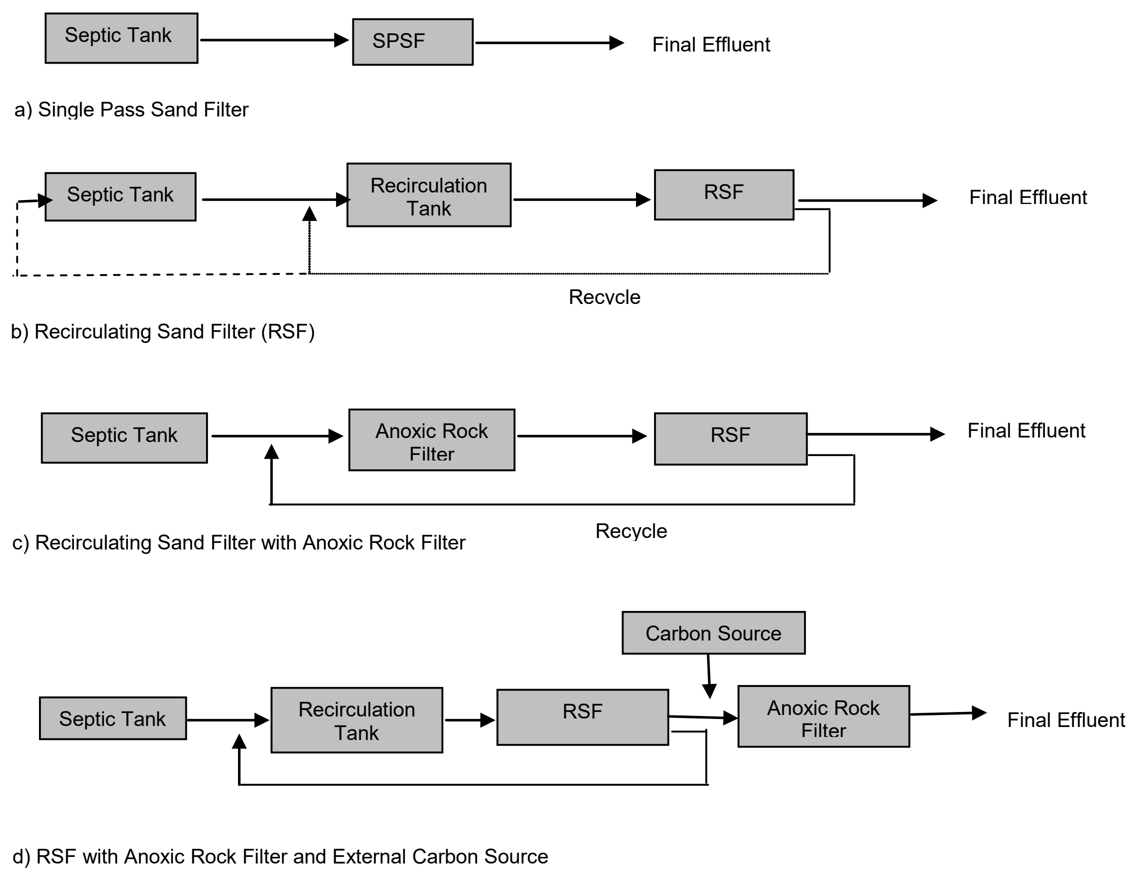


Figure 10: Process Diagrams for Attached Growth Systems

2. Recirculating Sand/Gravel Filters (RSF)

RSF technology is also very well studied in the literature. Total-N reduction has been reported to range from 15% to 84% (California Regional Water Quality Control Board, 1997; Gold, et al., 1992; Loomis, et al., 2001; Nolte & Associates, 1992; Oakley, et al., 1999; Piluk and Peters, 1994; Ronayne, et al., 1982). RSFs can achieve high nitrification rates and consistently higher denitrification rates than SPSFs. This is because the nitrified effluent can be recycled back to a recirculation tank where it mixes with wastewater from the septic tank, using the incoming wastewater as a carbon source.

As with SPSF systems, the organic loading rates for RSF systems are poorly defined in the literature. The available data suggest that organic loading rates that promote nitrification typically are in the range of 0.002-0.008 lbs. BOD_5/ft^2-day (Crites and Tchobanoglous, 1998). The extent of denitrification can be expected to vary widely since RSF systems have not typically been designed and operated specifically for nitrogen removal.

There is no doubt RSF performance could be significantly improved for nitrogen removal with design and operational changes. The recirculation tank is not generally configured to maximize the mixing of septic tank effluent with RSF effluent or to optimize the formation of anoxic conditions for denitrification. This may have the effect of maintaining permanent aerobic conditions in at least a part of the recirculation tank, especially if high recirculation ratios are used (e.g. to prevent drying of the filter bed during low flow periods).

A better design to enhance denitrification recycles the filter effluent to the inlet side of an anoxic recirculation tank, or, an anoxic rock filter, where it mixes with septic tank effluent as shown in Figure 10. The final effluent for discharge is then taken from the filter. This type of system has been termed "classical pre-denitrification" (Rittmann and McCarty, 2001). The rock filter fosters anoxic conditions by preventing hydraulic short-circuiting and allows denitrifying organisms to grow on the rock surfaces (Whitmeyer, et al., 1991). Systems using this type of design have been reported in the literature (Ayres Associates, 1998; Sandy, et al., 1988). While one system in Florida exhibited a mean Total-N removal of 40% with a mean effluent concentration of 23 mg/L Total-N (Ayres Associates, 1998), another study reported Total-N removals of 80 to 90% and effluent Total-N concentrations ranging from 7-10 mg/L (Sandy, et al., 1988). Sludge accumulation in the rock tank, however, can potentially cause serious operation and maintenance problems.

The optimization of the recirculation ratio for Total-N removal has to be done on a site-specific basis, which likely precludes its being done on many individual

Operational changes that could improve nitrogen removal include optimizing the recirculation ratio in order to minimize dissolved oxygen in the recirculation tank and maximize denitrification. The recirculation ratio for denitrification must be at least 4:1 or greater in order to remove a minimum of 80% of the nitrate (Rittmann and McCarty, 2001). Many RSFs in operation may be below this minimum since the range of recommended recirculation ratios is 3:1 to 5:1 (Crites and Tchobanoglous, 1998). Also, as mentioned above, very high recirculation ratios used to prevent filter drying during low-flow periods can inhibit denitrification because they cause high dissolved oxygen concentrations in the recirculation tank. The optimization of the recirculation ratio for total nitrogen removal has to be done on a site-specific basis, which is a serious operation concern for individual onsite systems.

The results of several studies have shown that nitrification rates in winter months in RSF systems in cold climates do change significantly from summer months (57% versus 84%) due to the fact that the RSF was exposed to surface temperatures (Gold, et al., 1992; Ronayne, et al., 1982). In cold climates it is thus recommended that the filter be covered, and the septic tank and recirculating tank be insulated (Loudon, et al., 1984).

3. Recirculating Sand/Gravel Filters with Anoxic Filter and External Carbon Source

This system is similar to the RSF above with an anoxic rock filter except the anoxic rock filter now follows the RSF and an external carbon source is added as shown in Figure 10. Part of the RSF effluent is recycled to the recirculation tank, and another part is discharged to the anoxic rock filter where the external carbon source is added. One detailed study in Rhode Island on pilot scale systems showed the following results (Table 7) for two different external carbon sources (Gold, et al., 1989):

Table 7: Nitrogen Removal Efficiencies with External Carbon Sources

External Carbon Source for RSF Anoxic Filter System	Mean Total-N % Removal	Mean Effluent Total-N, g/L
Methanol	74	13
Ethanol	80	10

As mentioned previously, the authors concluded that, because of chemical handling and operational requirements of the external carbon source, the system could probably best be handled by a wastewater management district or a private Operation & Maintenance contractor (Gold, et al., 1989).

4. Single Pass (SPTF) and Recirculating Textile Filters (RTF)

Textile filters are a relatively new technology. The design configurations and operational characteristics of single pass and recirculating (multiple pass) textile filters are essentially the same as for sand/gravel filters with one important exception: hydraulic and organic surface loading rates are much higher due to the specific surface area of the textile medium as shown in Table 8 below (Leverenz, et al., 2001; Crites and Tchobanoglous, 1998).

Thus, it is argued, for example, that RTFs require a much smaller footprint for equivalent attached-growth treatment, with as much as a 30-fold difference in area requirements and 850-fold difference in media weight from conventional SPSFs (Leverenz, et al., 2001).

Table 8: Hydraulic and Organic Loading Rates Sand Filters and Textile Filters

Media	Specific Surface Area, ft^2/ft^3	Hydraulic Loading Rate, gpd/ft^2	Organic Loading Rate, lbs. BOD_5/ft^2-d
Coarse Sand for RSF	387	3-5	0.002-0.008
Medium Sand for SPSF	2,100	1.25	0.000135-0.002
Textile Fabric for SPTF	5,000	10	0.01
Textile Fabric for RTF	5,000	30	0.03

One detailed study of RTFs showed that 83-95% nitrification occurred with as much as 14-29% total nitrogen removal at an organic loading rate of 0.01 lbs. BOD_5/ft^2-day, and that only 59-76% nitrification occurred with from 17-31% total nitrogen removal at an organic loading of 0.03 lbs. BOD_5/ft^2-day (Leverenz, et al., 2001). In this study alkalinity did not limit nitrification.

As with sand and gravel filters, more data are needed to adequately characterize textile media in terms of design and operational parameters for both nitrification and total nitrogen removal.

5. Peat Filters

Peat filters have been used in a manner similar to single pass sand filters, with similar hydraulic and organic loading rates (Crites and Tchobanoglous, 1998). The results of a few studies show that total nitrogen removal can be very high, with 80% removal and effluent total nitrogen concentrations less than 10 mg/L (Crites and Tchobanoglous, 1998). It can be assumed that the peat would serve as a carbon source for reduction of nitrate after nitrification has occurred in the filter. Only a few detailed design and operational data are available in the literature to adequately characterize the various peat media in terms of design and operational parameters for both nitrification and total nitrogen removal (Geerts, et al., 2001; Crites and Tchobanoglous, 1998).

6. RUCK System

The RUCK system, a version of which is shown schematically in Figure 10, is a proprietary system that uses source separation for nitrification and denitrification. Separate collection systems are designed for greywater and blackwater, with each having its own septic tank. The original system was configured as follows (Laak, 1986): The blackwater, originally defined as wastewater from toilets, showers and baths, was discharged to an SPSF for nitrification and then passed to an anoxic rock filter or tank. The greywater, defined as kitchen and laundry wastewater, passed from the septic tank directly to the anoxic rock filter or tank, where it served as the carbon source. There are newer configurations that have been used (Loomis, 2002), but there is a scarcity of information on them in the published literature.

While the RUCK system has often been cited as a potential technology for nitrogen removal, there is also a lack of performance data that have been published. While the process is intended to provide at least 80% total nitrogen removal (Laak, 1986), results from a few studies have shown much poorer removal rates of from 29-54% Total-N removal (Brooks, 1996; Gold, et al., 1989). One experimental design had nitrogen removal ability less than a conventional system and was withdrawn after one year of testing (Costa, 2002). The variability in nitrogen removal efficiency is no doubt due to the complexity of the system, the variability of the quality of greywater, and the need to adjust the operation to site-specific conditions, as is the case with a RSF system. The RUCK system is even more complicated than a RSF and likely requires significant adjustment to blackwater and greywater characteristics and site conditions.

7. Nitrex™

The Nitrex™ is a proprietary trickling biofilter developed at the University of Waterloo in Ontario, Canada. Nitrex™ is designed for denitrification and requires a nitrification process prior to the unit. The nitrification unit can be either a public domain process like a lined sand filter or there are a variety of proprietary products that would serve the same purpose. The unit is filled with a proprietary wood byproduct mixture that promotes nitrogen removal. Wastewater containing nitrate, such as nitrified wastewater is applied to the surface of the Nitrex filter. As the wastewater moves through the organic medium, microbial reduction of the nitrate nitrogen (denitrification) occurs. The bed must remain submerged for this to occur due to the anaerobic nature of this reaction. Typically, the units are single-pass and do not require pumping.

Results of testing have been encouraging, with reductions to levels of 2 mg/l reported (Rich, 2003). Unpublished testing data from the Massachusetts Septic System Testing Center (MASSTC) indicate slightly higher results (average of 5.4 mg/l; median of 4.2 mg/l) but still very good results (Heufelder, personal communication 2005).

8. Ion Exchange with Zeolites

Ammonium ions in wastewater can be preferentially removed by naturally occurring ion exchange materials called zeolites (Metcalf & Eddy, 1991). The selectivity of zeolite for the major ions in wastewater has been reported to be the following as reported by the California Regional Water Quality Control Board, (1997):

$K^+ > NH_4^+ > Ca^{++} > Mg^{++} > Na^+$

Only nitrogen in the form of ammonium ion can be removed in wastewater, and this must be done under anaerobic conditions in order to inhibit nitrification. The quantity of ammonium ion that can be removed depends on the zeolite bed volume and equilibrium kinetics.

Zeolite ion exchange filters have been used in several onsite wastewater experiments in California (California Regional Water Quality Control Board, 1997). The results showed that while ammonium ion could be removed (from 16.2% to 93.8% removal was reported), the filter performance was highly variable, and the filters required extensive maintenance for replacement or service of the zeolite. Indeed, ion exchange for ammonium ion removal has had limited application in centralized wastewater treatment because of the extensive pretreatment required and concerns about the useful life and regeneration of the zeolite (Metcalf & Eddy, 1991). The use of zeolite for onsite ammonium ion removal must therefore be considered to be in the experimental stage at the present time.

C. Shallow Trench and Subsurface Drip Distribution Systems

The use of either shallow trench or subsurface drip distribution (SDD) systems has been proposed as an alternative means to remove total nitrogen in the soil column (Ayres Associates, 1998; Crites and Tchobanoglous, 1998; Oakley, et al., 1998, 1999). Both systems have the potential to promote nitrogen uptake by plant roots if effluent is discharged directly within the root zone. There is also a potential that both systems, because they are installed within the shallow, more organic 'A' horizon of the soil, could promote denitrification if sufficient organic matter is present, and if anoxic conditions exist. The organic material can be either naturally present or could be added. This type of denitrification has been demonstrated with the use of a reactive porous media barrier using sawdust as a carbon source, which was used to denitrify nitrified septic tank effluents percolating through the soil column (Robertson and Cherry, 1995).

Coupling shallow trenches or SDD systems with other technologies is an excellent method to further reduce TN levels, even though the exact amount of the reduction produced by the SDD may not be easily quantifiable.

To date, the results on the use of shallow trenches or SDD systems for onsite nitrogen removal is mixed, with removal efficiencies of total nitrogen ranging from 0 to 40% (Ayres Associates, 1998; Bohrer and Converse, 2001; Oakley, et al., 1998, 1999). Balancing nitrogen loadings with plant uptake requires significant operational monitoring and adjustment. Denitrification, if it is desired, cannot be easily controlled within a trench system or the soil column as it can within a treatment reactor above ground. Monitoring of nitrogen removal in the soil column is also a significant problem since lysimeter systems have to be used, and they require some degree of sophistication in installation and sample collection (Oakley, et al., 1999). Coupling shallow trenches or SDD systems with other technologies, however, is an excellent method to further reduce total nitrogen levels, even though the exact amount of the reduction produced by the SDD may not be easily quantifiable. Reduction to levels meeting the Florida Keys TN effluent standards (10 mg/l TN) appeared to be achievable by combining various unit processes, including final disposal in a drip irrigation system (Ayers, 2000).

D. Testing and Certification Protocols

Currently local health jurisdictions may establish Areas of Special Concern and set standards for effluent total nitrogen (TN). However, this is not done on a consistent basis and no common standard among local health jurisdictions exists. The Department of Ecology has established groundwater standards, but they do not readily address the treatment processes necessary to achieve groundwater standards.

The Washington State Department of Health (DOH) is proposing, in the current draft rules

(Chapter 246-272A WAC Onsite Sewage Systems Draft) to the Washington State Board of Health, that standards be established for any proprietary products that are sold as nitrogen reducing technologies. In order to be registered in the State of Washington product manufacturers would have to verify that their product is capable of producing effluent TN equal to or less than 20 mg/L using the NSF/EPA Environmental Technology Verification program protocol (Protocol for the Verification of Residential Wastewater Treatment Technologies for Nutrient Reduction / EPA Environmental Technology Verification Program (November 2000)).

1. Environmental Technology Verification (ETV) Tested Products

The EPA created the Environmental Technology Verification (ETV) Program to facilitate deployment of innovative or improved environmental technologies through performance verification and dissemination of information. The goal of the ETV program is to further environmental protection by substantially accelerating the acceptance and use of improved and more cost-effective technologies. ETV seeks to achieve this goal by providing high quality, peer reviewed data on technology performance to those involved in the design, distribution, permitting, purchase, and use of environmental technologies.

NSF International (NSF) operates the Water Quality Protection Center (WQPC) under the U.S. Environmental Protection Agency's (EPA) Environmental Technology Verification (ETV) Program. The WQPC evaluates the performance of proprietary treatment systems for nitrogen removal for residential homes. The Barnstable County (Massachusetts) Department of Health and the Environment (BCDHE) perform the verification testing.

ETV works in partnership with recognized standards and testing organizations, stakeholder groups consisting of buyers, vendor organizations, and permitters, and the full participation of individual technology developers. The program evaluates the performance of innovative technologies by developing test plans that are responsive to the needs of stakeholders, conducting field or laboratory tests (as appropriate), collecting and analyzing data, and preparing peer reviewed reports. All evaluations are conducted in accordance with rigorous quality assurance protocols to ensure that data of known and verifiable quality are generated and that the results are defensible.

Although a number of USEPA supported demonstration projects are in existence around the country, the WQPC is the only site currently in operation that tests products under controlled conditions to compare product performances to a recognized standard. Their website can be reached at http://www.nsf.org/business/water_quality_protection_center.

The ETV Joint Verification Statements for each product can be found by clicking on the links to that product as shown in the description below. Each of the products is listed as nutrient reduction technologies for residential wastewater treatment by NSF/EPA.

Following is a brief description of each of the six products that have completed the ETV process for nitrogen reduction in domestic wastewaters from individual residential homes:

Table 9: Products that have completed the ETV process for nitrogen reduction in domestic wastewaters from individual residential homes (As of May, 2005)

System Name	Technology	Description of Process	Performance	Cost
Waterloo Biofilter® Model 4-Bedroom Waterloo Biofilter Systems, Inc.143 Dennis St.; PO Box 400 Rockwood, Ontario Canada, N0B 2k0 http://www.nsf.org/business/water_quality_protection_center/pdf/Waterloo-VS-SIGNED.pdf	Fixed film trickling filter.	The biofilter unit utilizes a patented lightweight open-cell foam that provides a large surface area. Settled wastewater from a primary septic tank is applied to the surface of the biofilter with a spray distribution system. The system can be set up using a single pass process (without any recirculation of biofilter treated effluent) or can utilize multi-pass configurations. The ETV testing results were generated by returning 50% of the biofilter effluent back to the primary compartment of the septic tank.	It averaged 62% removal of total nitrogen with an average total nitrogen effluent of 14 mg/l over the 13-month testing period. Earlier testing of this product in a single pass mode demonstrated that it could produce a 20-40% TN reduction.	$13,000-17,000 for total system installation. The Waterloo Biofilter® unit only would cost approximately $7,000.

System Name	Technology	Description of Process	Performance	Cost
Amphidrome™ Model Single Family System: F.R. Mahony & Associates, Inc. 273 Weymouth St. Rockland, MA 02370 http://www.nsf.org/business/water_quality_protection_center/pdf/Amphidrome_VS.pdf	Submerged growth sequencing batch reactor (SBR) in conjunction with an anoxic / equalization tank and a clear well tank for wastewater treatment	The bioreactor consists of a deep bed sand filter, which alternates between aerobic and anoxic treatment. The reactor operates similar to a biological aerated filter, except that the reactor changes from aerobic to anoxic conditions during sequential cycling of the unit. Air, supplied by a blower, is introduced at the bottom of the filter to enhance oxygen transfer.	It averaged 59% removal of total nitrogen with an average total nitrogen effluent of 15 mg/l over the 13-month testing period at MASSTC.	$7500 for unit only. The manufacturer estimates it would cost $12,000-15,000 for a total installation.
Septitech® Model 400 System: Septitech, Inc. 220 Lewiston Road Gray, Maine 04039 http://www.nsf.org/business/water_quality_protection_center/pdf/SeptiTech_VS.pdf	Two stage fixed film trickling filter using a patented highly permeable hydrophobic media.	Clarified septic tank effluent flows by gravity into the recirculation chamber of the SeptiTech unit. A submerged pump periodically sprays wastewater onto the attached growth process and the wastewater percolates through the patented packing material. Treated wastewater flows back into the recirculation chamber to mix with the contents. Treated water flows into a clarification chamber and is periodically discharged to disposal unit (drainfield, drip irrigation, etc.).	Averaged 64% removal of total nitrogen with an average total nitrogen effluent of 14 mg/l over the 12-month testing period at MASSTC.	$11,000 for Septitech unit includes shipping and installation. The manufacturer estimates that a total system with pressure distribution drainfield would cost approximately $20,000.

System Name	Technology	Description of Process	Performance	Cost
Bioclere™ Model 16/12: Aquapoint, Inc. 241 Duchanine Blvd New Bedford, MA 02745 http://www.nsf.org/business/water_quality_protection_center/pdf/Bioclere-VS-SIGNED.pdf	Fixed film trickling filter.	Septic tank effluent flows by gravity to the Bioclere clarifier unit from which it is sprayed or splashed onto the fixed film media. Treated effluent and sloughed biomass are returned to the clarifier unit. A recirculation pump in the clarifier periodically returns biomass to the primary tank. Oxygen is provided to the fixed film by a fan located on the top of the unit.	Averaged 57% removal of total nitrogen with an average total nitrogen effluent of 16 mg/l over the 13-month testing period at MASSTC.	$7500 for unit itself. Price for total system would need to include primary septic tank, Bioclere unit and disposal option, with costs in the range of $12,000 -15,000. The manufacturer recommends use in clusters to reduce per home costs and facilitate maintenance. Experience with 27 home cluster resulted in costs of $6800-8,000 per home.
Retrofast® 0.375 System: Bio-Microbics 8450 Cole Parkway Shawnee, KS 66227 http://www.nsf.org/business/water_quality_protection_center/pdf/Biomicrobics-FinalVerificationStatement.pdf	Submerged attached-growth treatment system, which is inserted as a retrofit device into the outlet side of new or existing septic tanks.	The RetroFAST® 0.375 System is inserted in the second compartment of the septic tank. Air is supplied to the fixed film honeycombed media of the unit by a remote blower. Alternate modes of operation include recirculation of nitrified wastewater to the primary settling chamber for denitrification. Intermittent use of the blower can also be programmed to reduce electricity use and to increase denitrification.	Averaged 51% removal of total nitrogen with an average total nitrogen effluent of 19 mg/l over the 13-month testing period at MASSTC.	Product and installation cost for the Retrofast® 0.375 System ranges is estimated to be from $4,000-5,500 depending on existing tankage. That cost includes the FAST unit, blower, blower housing and control panel. The local representative for Bio-Microbics units believes costs could be as low as $3,500 for multiple units.

System Name	Technology	Description of Process	Performance	Cost
Recip® RTS~500 System: Bioconcepts, Inc. P.O. Box 885 Oriental, NC 28571-0885 http://www.nsf.org/business/water_quality_protection_center/pdf/Bioconcepts_Verification_Statement.pdf	Fixed film filter	This is the newest product to complete ETV testing. It is a patented process developed by the Tennessee Valley Authority (TVA) and utilizes a fixed film filter medium contained in two adjacent, equally dimensioned cells. Timers on each of the two reciprocating pumps control the process.	Averaged 58% removal of total nitrogen with an average total nitrogen effluent of 15 mg/l over the 12-month testing period at MASSTC.	Very limited experience with this particular single-family unit. The unit built for ETV testing was a prototype. The cost per unit, by itself, is estimated to be $8,000-10,000. Cost of the septic tank and disposal unit would be extra and the cost would depend on site conditions. Conservatively, cost for a total system could be $11,000-15,000.

2. Other Proprietary Products

Although proprietary products, under the proposed regulations, would need to have their products verified under the ETV protocol before being allowed for use as nutrient reduction technology, there are a number of other products that have not undergone ETV testing but have been installed and their performance monitored at EPA-sponsored demonstration sites around the USA. The California Water Resource Control Board's *Review of Technologies for the Onsite Treatment of Wastewater in California*, catalogs and describes the various products, by type of system. The full report can be accessed at: http://www.waterboards.ca.gov/ab885/index.html

However, a brief summary of selected products and their performance gives readers an idea of the performance capabilities of a variety of proprietary technologies. Published data from the following EPA Demonstration Projects is included:

- LaPine, Oregon
- Rhode Island
- Florida Keys Onsite Wastewater Nutrient Reduction System (OWNRS)

Table 10: Performance Summary of Selected Proprietary Products Treated Effluent Total Nitrogen (TN) Levels (in mg/L)

System Name	La Pine, Oregon	Rhode Island	Florida Keys OWNRS
Orenco RTF (3 systems/site)	9.4, 26, 14	9, 69, 83	
Biokreisel (3 systems)	14, 12, 14		
Enviroserver (2 systems)	26, 40		
Nayadic (3 systems)	30, 43, 37		
Nitrex (2 systems)	2, 2.4		
FAST	***		10.97, 11.5*
IDEA BESTEP	***		15.46, **
Klargester RBC			12.52, 14.9*
Puraflow Peat	***	49	

* Phase II results with higher flows and equipment modifications (Ayers, 2000)

** Dropped from testing in Phase II due to lack manufacturer support (Ayers, 2000)

*** Study results have not yet been published.

Note: Since many of the influents to these products are a blend of septic tank influent and recirculated treated wastewater, no percentage reduction figures are shown.

As can be seen, there is a wide range of TN reduction performance, even among the same proprietary products. This variability is not unexpected since, as previously discussed, there is a wide range of flow rates and waste characteristics. In addition, there were various modifications in the application of the technologies as set up by the manufacturers at the demonstration sites.

These results provide encouraging data about the capabilities of many nitrogen reducing technologies. Although there is variability in the results, reductions from 34-98 % were demonstrated. The Nitrex system, in particular, has been shown to be very effective in removing nitrogen. These levels of reduction, if they can be translated from demonstration sites to in-the-field installations, can make a significant reduction in the amount of nitrogen discharged to ground and surface waters.

E. Cost Considerations

As reported in the literature, the capital costs of many onsite nitrogen removal systems have severely limited their widespread application. The costs of those systems designed specifically for nitrogen removal for a single family dwelling ranged from $5,800 to $11,300 in Wisconsin in 1991, $2,000 to $15,000 in California in 1997, and $7,872 to $17,414 in Florida in 1998 (Ayres Associates, 1998; California Regional Water Quality Control Board, 1997; Whitmeyer, et al., 1991).

Prices for the ETV tested products ranged from $7500 -$11,000 per unit according to manufacturer estimates. Costs for a fully installed system were more difficult to estimate since site conditions can vary. However, most of these companies estimated costs in the range of $20,000 for a complete system. The exception to these costs was the Bio-Microbics RetroFAST. It is intended for retrofitting into the second compartment of an existing septic tank and costs are estimated to range from $4000-5000, assuming it can be placed in the existing septic tank.

The costs for non-ETV tested proprietary products appear to be very similar to the costs associated with the ETV tested products. Cost ranges vary widely depending on the particular process or combination of processes involved. Site conditions or site limitations also affect estimates of the total cost.

Annual operation and maintenance (O&M) costs also need to be factored into decisions to install nitrogen-reducing technologies. Annual O&M costs of $1730-2841/year were estimated for innovative systems involved in the Florida Keys OWNRS Phase II Project (Ayers, 2000).

Manufacturers of ETV tested products varied in their responses to price savings for multiple units. Most recommended clustering of homes, with units sized to accommodate waste loading, as a better way to save money, as opposed to installing multiple separate systems. Costs for operation and maintenance were difficult to pin down since it would depend on the number of systems in the area and the economy of scale that could be attained.

For units other than the Bio-Microbics RetroFAST, manufacturers of these products stress that they would need to develop a market for their product and have trained local service providers before it would be economical to bring units into this region. Most also felt that clustering of systems was preferable, not only from a capital cost standpoint, but also from a point of treatment efficiency. Clustering of systems tends to provide a more consistent waste stream, both in terms of flow rates and wastewater characteristics.

F. Summary

There are any number of proprietary technologies and devices that are promoted for nitrogen removal, and it was beyond the scope of this paper to discuss all of them in detail. None to date have been found to offer a simple solution to the complex problem of biological nitrogen removal.

This is not surprising given the nature of biological nitrification/denitrification and the inherent variability of onsite wastewater flow rates and characteristics as well as so many factors (pH, temperature, alkalinity, etc.) that affect the process. Whereas removal of BOD and TSS is fairly straightforward, the processes for effective and consistent nitrogen removal are so inter-related that it is difficult to assure a consistently treated waste product.

However daunting this may seem, the recognition of the need to control and reduce nitrogen outputs has spawned multiple efforts to develop and perfect technologies that are more effective than the standard onsite systems. Additional research and testing continues but ample data has been generated to show that significant nitrogen reductions can be expected on an area-wide basis with the installation of site appropriate technology. Assuming the range of nitrogen reductions that can be expected and comparing that with those provided by a standard septic tank and drainfield system demonstrates the potential for a significant reduction in overall nitrogen inputs. According to the Preliminary Assessment and Corrective Action Plan for Hood Canal, an estimated 39-241 tons of nitrogen are added to Hood Canal from present day onsite systems. Even if removal rates of 50% are assumed that could lead to a reduction of 19-120 tons of nitrogen loading per year.

The variability in waste flows and waste characteristics dictate that site-specific solutions be developed. It is also clear that the complexity of the nitrogen removal process demands a significantly higher degree of operational oversight, maintenance, and periodic system adjustment. In order to reduce the impact of nitrate on our sensitive environments, such as Hood Canal, it is necessary to move beyond the relatively simple septic tank and gravity drainfield, which only removes 10 to 30% of the TN, even though more complex systems will be required. All these devices increase the opportunity for equipment malfunction that can either cause the system to fail or to operate at a reduced efficiency. Without sufficient O&M oversight, the initial (and significant) capital expenditure for these innovative technologies becomes a wasted investment for the homeowner and fails to address the nitrogen issues in Hood Canal.

In order to convince homeowners and policy makers that investment in these technologies will be worthwhile, they will need to be shown that not only will the technologies have a positive influence on the nitrogen levels and improve the water quality but also they will be operationally durable for extended years of operation as well as financially feasible to install and maintain. For instance, if a homeowner pays $20,000 for the installation of a product, pays $1500/year for O&M but the system only lasts for 10 years, his/her amortized cost is $3500/year. This could be a significant factor in the decision to install individual systems or to look for other solutions, such as centralized sewage treatment systems. Barnstable County, Massachusetts, which has similar problems in Cape Cod, has extensive experience with various options for dealing with nitrogen-rich embayments. One of the issues that communities in that area are struggling with is identifying the best long-term solution to the problem. Should they continue to require homeowners to invest in individual or cluster innovative technologies (where multiple homes are connected to one community system) when the science isn't clear that the installation of these systems will adequately reduce the nitrogen levels necessary to meet the treatment goals set for Cape Cod waters? Or should they withhold requiring those expenses by property owners and, instead, invest in larger, more efficient centralized wastewater treatment plants that have a proven record of effectively treating wastewater to high standards? (Wright-Pierce, 2004).

A similar decision based on scientific, engineering, geographic and economic factors faces policy makers dealing with Hood Canal.

Nitrogen removal has been widely successful in large-scale wastewater treatment plants as a result of continuous operator attention and chemical addition if necessary. Whether this type of success can be accomplished in onsite wastewater treatment remains to be seen.

Appendix B – Glossary

Aerobic:	Biological treatment processes that occur in the presence of oxygen.
Anaerobic:	Biological treatment processes that occur in the absence of oxygen.
Anoxic:	A lack of oxygen; the process in which nitrate nitrogen is converted biologically to nitrogen gas in the absence of oxygen.
Anthropogenic:	Caused by humans.
Autotrophic Bacteria:	Organisms that derive cell carbon solely from carbon dioxide.
Facultative Bacteria:	Organisms (either heterotrophic or autotrophic) that can shift between aerobic and anaerobic respiration.
Heterotrophic Bacteria:	Organisms that require organic carbon as the carbon source for the formation of cell tissue.
Total Kjeldahl Nitrogen (TKN):	TKN is the combination of organic nitrogen and ammonia nitrogen.
Vadose Zone:	Zone of aeration in the soil layer above the ground water level.

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