AN ABSTRACT OF THE THESIS OF

Jennifer L. Bushman for the degree of Master of Science in Bioresource Engineering presented on February 12, 1996. Title: Transport and Transformations of Nitrogen Compounds in Effluent From Sand Filter-Septic System Draintile Fields.

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Abstract approved: ______________________________

James A. Moore

A total of 44 intermittent sand filter-septic systems, in five counties of Western Oregon, were sampled over a three-month period during the summer of 1995. The sand filter systems varied in age from 36 months up to 167 months (3 to 13.9 years). Liquid samples were taken from the septic tank and distribution box. In addition, soil samples were taken adjacent to the disposal trench and away from the disposal field area (control). All samples were analyzed for Total Kjeldahl Nitrogen (TKN) and nitrate and nitrite. Nitrite was not detected in any of the samples. The average removal of total nitrogen (TKN + nitrate) through the filter was found to be 43%. Nitrate was determined to be the dominant form of nitrogen in the sand filter effluent making up 94% of the total nitrogen. The age of the system was found not to be a predictor of the system's performance. Once the effluent entered the disposal field, little if any transformation of nitrogen occurred at an average depth of 30 inches (76.2 cm).

# Transport and Transformations of Nitrogen Compounds in Effluent From Sand Filter-Septic System Draintile Fields

by

Jennifer L. Bushman

A THESIS

submitted to

Oregon State University

in partial fulfillment of
the requirements for the
degree of

Master of Science

Completed February 12, 1996
Commencement June 1996

Master of Science thesis of Jennifer L. Bushman presented on February 12, 1996

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I understand that my thesis will become part of the permanent collection of Oregon State University libraries. My signature below authorizes release of my thesis to any reader upon request.

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Jennifer L. Bushman, Author

## ACKNOWLEDGMENT

I would first like to thank Greg Farrell and his staff in the Roseburg office for their support and time given to this study. I would also like to recognize the Oregon Department of Environmental Quality for funding this study. The assistance of the four county offices who permit septic systems was greatly appreciated, and without their assistance, the study could not have gone forward.

Next, I would like to thank my major professor Dr. Jim Moore for his support and encouragement throughout my graduate schooling. I would also like to thank Dr. Anna Harding for her support and for not being upset with all my departmental changes. Thanks also go out to Dr. Ron Miner and Dr. Ward Carson for being on my committee.

I am very grateful for the time and statistical assistance I received from Dr. Scott Urquhart, he made SAS an enjoyable program to use, well, as much as possible.

Some people deserve special mention for their constant "harassment" and because they have always answered my stupid questions. Those people are Bob Schnekenburger, Penny Hatcher-Bangs, Elena Rudisill, Linda Hoyser, and June Rice.

Of course there are all of my BRE buddies: Steve Niswander, Jadene Kramer, Matt Boyd, Russ Faux, Billy Kim, Alireza Esteghlalian, and John Guida. Thanks for all the laughs, coffees, and hallway chats.

I wouldn't forget all of those non-BRE people who have listened to my woes, and cheered me on through the past two years. Holly Sherburne who will forever call me "Cheesehead", for reasons only she and I know. Joy Linn who, by example, taught me how to stand up for my political beliefs. Carol Adams just because we know each other so well, and still like each other. Heidi Sherburne, the little sister I never had, and Brian Bushman, the big brother I've always known. Last but not least, Susan Chan and Jim Turner who have remained my good friends despite the distance and effects of time.

Probably the most important people that need mention are my parents, all three of them! Mom, Jerry, and Dad thanks for all the love, support, encouragement, and money throughout the years, I think it has finally paid off.

I would like to dedicate this thesis to my
hero, good friend, and Mom
Dorothy Stanley.

Thanks for everything

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# INTRODUCTION

Over one hundred years ago, engineers and sanitarians were becoming aware of the need for sufficient household wastewater disposal systems (Burks and Minnis, 1994). This awareness facilitated the change of disposal systems, as time passed and cultures changed. At times, sewage was disposed outside the village, washed into nearby rivers, or dumped on city streets. Many cultures in India, Pakistan, and Crete were aware of the need for proper sewage disposal as far back as 2000 B.C. (Burks and Minnis, 1994). Ancient Romans also had systems utilizing running water for sewage disposal and Romans built one of the first underground sewage systems (Burks and Minnis, 1994). The Cloaca Maxima, or main drain, was a combination sewer for domestic wastewater and storm water runoff in Rome over 2500 years ago, and is still in use today.

In Ancient Rome, water used for the disposal of human waste was kept separate from water used for drinking. In the early 1700s during the Dark Ages, the practice of separating wastewater from potable water was abandoned. At the time, storm sewers existed, but the law prohibited the disposal of human waste directly into the storm drains. Eventually, due to the unsanitary practice of throwing waste out windows onto the street, it was discharged directly into the storm sewer. Water from sewers was not treated during this period but dumped directly into rivers, seas, or oceans (Fuhrman, 1984).

Gradually, due to a lack of anything better, communities in Britain and on the European continent began to use privy vaults, which were an odor nuisance and also posed disposal problems (Burks and Minnis, 1994). Poor sanitation conditions continued until the mid-1800s, when John Snow conducted an important epidemiological study. In 1854, Snow traced the outbreak of cholera to a city well that was being contaminated by privy vaults (Burks and Minnis, 1994). His finding heightened the public awareness of the need for a proper waste disposal system. In addition, conditions in many rivers were intolerable, and they grew worse as the population increased.

In the early 1800s the primary goal of a wastewater disposal system was to prevent chronic health problems, such as cholera, associated with human wastes. Early systems were also used for aesthetic reasons. It was not until 1873 that attention was directed to water pollution and how it might be prevented (Fuhrman, 1984).

During the mid-1800s, engineers in England and Germany developed some of the first treatment facilities, to separate solids and liquids (Burks and Minnis, 1994). The solids were stored in large structures and the liquids discharged into the rivers. These storage structures, or cesspools, were periodically cleaned and the slurry disposed on land. Land disposal, usually deposited on farmland, was practiced until the public protested the activity. Those concerned with the issue insisted that additional treatment be considered (Burks and Minnis, 1994). Accordingly, the septic tank, an anaerobic treatment cell was employed in the mid-1800s to treat large amounts of wastewater (Burks and Minnis, 1994). Continued interest in public health promoted research in the further treatment of wastewater to protect human health.

Mother Nature has been using sand filters since the beginning of time and man-made filters have been in use, at least in recorded history, as far back as 1868. Sir Edward Frankland, "the father of the trickling filter," experimented with columns of different sands and soils as he investigated treatment techniques. He passed raw sewage through the columns and analyzed the resulting effluent (Burks and Minnis, 1994). This led to the discovery of treatment properties of a column of sand and soil. The first use of a sand bed with a septic tank occured in 1893 at an Experimental Station in Lawrence, Massachusetts (Burks and Minnis, 1994). The treatment capabilities of sand filters have made them a popular means of sewage treatment.

There are many types of sand filters: intermittent sand filters, recirculating sand filters, in-trench sand filters, and bottomless filters. Between 1976 and 1992, more than 7000 intermittent sand filters (ISF) have been installed for use as primary treatment in single family homes in the western United States (Ball, 1992). Other areas of the U.S. have also incorporated sand filters of some type into the individual household septic systems. With the increased awareness of environmental issues, sand filters represent a promising means of producing a better quality of effluent.

ISFs were approved for use in the treatment of domestic wastewater in Oregon on January 1, 1980. An ISF is a "conventional filter with two feet or more of medium size sand media designed to chemically and biologically process septic tank or other treatment unit effluent from a pressure distribution system operated on an intermittent basis" (OAR, 1995). The ISF also physically filters the effluent. The systems were approved after an experimental research study was conducted by members of the Oregon Department of Environmental Quality (ODEQ). The study monitored the performance of ISFs by examining filters in various stages of life from 5 to 49 months old (Paeth and Ronayne, 1984). The effluent from the septic tank and from the ISF were analyzed for nine parameters: five-day biochemical oxygen demand ($BOD_5$), total nitrogen (TN), ammonia ($NH_3$), nitrate and nitrite ($NO_3$ and $NO_2$), Total Kjeldahl Nitrogen (TKN), suspended solids (SS), fecal coliforms (FC), and total coliforms (TC). The percent changes through the sand filter were 99, 47, 99, 99, 50, 97, 93, 99, and 99% respectively (Paeth and Ronayne, 1984). It was also found that the TN in the septic tank effluent consisted of less than 1% oxidized nitrogen (N), while the ISF effluent contained 96% of the TN as oxidized $NO_3$-N.

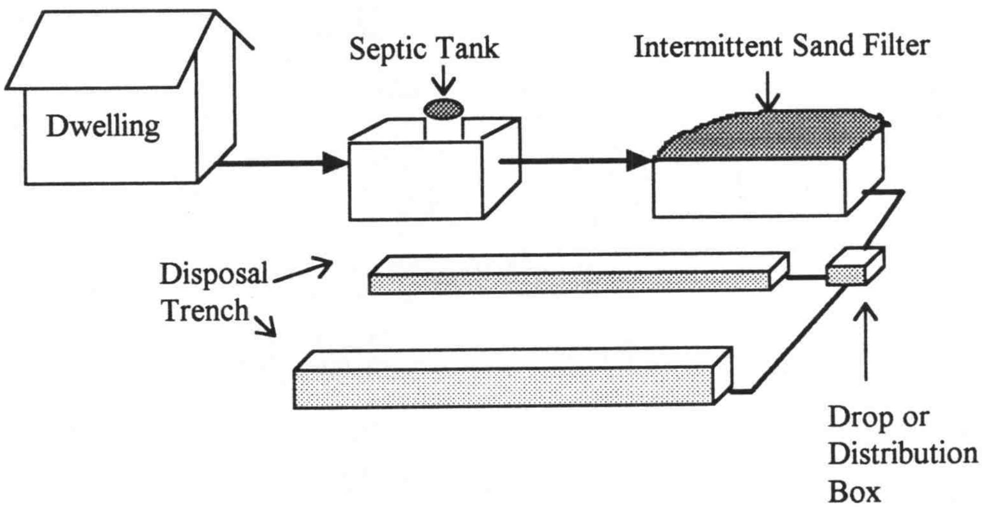
Based on the findings from the study by Paeth and Ronayne, ISF systems were allowed for installation on sites that would otherwise be declined a standard septic system. The percolation rate of soils present on these sites are generally too slow or too fast to allow the installation of a standard septic system. Because the ISF produces such a clean effluent, a shortened drain trench length is allowed. The length of the shortened trench is about half that of a standard system's. The primary purpose of an ISF is to allow sites, that would otherwise be declined a septic system, an alternative so a system can be installed. An incidental benefit of the ISF is its reduction of total nitrogen in effluent that passes through the filter.

The change in nitrogen form is an important indicator of the ISF's internal processing which is meant to be aerobic. Oxidation of ammonium and organic nitrogen to nitrate indicates a healthy aerobic and varied microbial environment. The microbial reduction of nitrate to nitrogen gas in anaeorobic microzones allows for the removal of the nitrogen from the system. A microbial community is the primary processing method in the ISF. Consequently, without the microbes, the removal of nutrients is likely to be minimal.

Many areas in Oregon have very fine grained soils with a high clay content, or coarse grained soils with a high sand content, either of which provide an unacceptable percolation rate for the installation of a standard septic system. An ISF allows many of these sites to meet state requirements. Other sites that may utilize an ISF to satisfy ODEQ requirements could have a geomorphic stucture such as a shallow hardpan or fractured rock, or insufficient depth to temporary or permanent groundwater.

As specified by the *Oregon Administrative Rules (OAR) for On-Site Sewage Disposal* (1995), the typical ISF in Oregon is used in conjunction with a septic tank and a subsurface absorption field (Figure 1).

Figure 1. Typical Layout of an Intermittent Sand Filter Septic System.




The ISF treats the effluent from the septic tank prior to discharge to a disposal field. For a single family residence, it is sized at 360 square feet (33.4 $m^2$) of sand surface to treat a peak sewage flow of 450 gallons (1703 L) per day (OAR, 1995). The design criteria is 1.25 gallons per square foot per day (50.9 L/$m^2$/day). Prior to April 1, 1995, the criteria was 1.23 gallons per square foot per day (50.1 L/$m^2$/day) (OAR, 1988). It should be noted that both post and pre-April 1, 1995 criteria are specified for comparison of the systems studied with present systems installed after April 1995. The systems in this study were built under the pre-April 1, 1995 criteria. The ISF may be above ground or subsurface and may be in one of many shapes, usually square, rectangular, or octahedron. The containment unit of the ISF is generally a concrete structure or a 30 mL PVC liner with earth berms. The bottom of the ISF contains collection piping, or an underdrain having a minimum diameter of four inches (three inches prior to April 1, 1995), surrounded by at least six inches (15.2 cm) of drain media. Drain media is defined by the Oregon Administrative Rules (1995) as

> [c]lean washed gravel, clean crushed rock, or other media approved by the Director's Designee, for the purpose of distributing effluent. When gravel or crushed rock[*] is used it shall have a minimum size of three quarters (3/4) inches[1.9 cm] and a maximum size of two and one-half (2-1/2) inches [6.4 cm]. The material shall be durable and inert so that it will maintain its integrity and not collapse or disintegrate with time and shall not be detrimental to the performance of the system (OAR, 1995).

*Note: crused rock is excluded from use in sand filters using a PVC liner.

Next is a layer of filter fabric, "a woven or spun-bonded sheet material used to impede or prevent the movement of sand, silt, and clay into drain media" (OAR, 1995). As of April 1, 1995, if pea gravel or underdrain media is used, the filter fabric is not necessary. A minimum of 24 inches (61 cm) of medium sand is installed over the filter fabric. Medium sand is "a mixture of sand with 100 percent passing the 3/8 inch sieve, 95 to 100 percent passing the No. 4 sieve, 80 to 100 percent passing the No. 8 sieve, 45 to 85 percent passing the No. 16 sieve, 15 to 60 percent passing the No. 30 sieve, 3 to 15 percent passing the No. 50 sieve, and 4 percent or less passing the No. 100 sieve" (OAR, 1995). The pressure distribution piping consists of a manifolded system using a minimum of one-half inch (1.3 cm) diameter 200 psi PVC pipe with one-eighth inch (0.32 cm) orifices facing up. It is laid on a minimum of three inches (7.6 cm) of drain media to separate it from the sand. Additional drain media covers the piping, then another layer of filter fabric covered by a minimum of six inches (15.2 cm) of soil no finer than loam. The pressure distribution system must be designed for a minimum of a five foot (152 cm) residual head at the most remote orifice to ensure adequate pressure to facilitate the even distribution of the liquid onto the filter surface. This also helps prevent clogging of the one-eighth inch (0.32 cm) orifices. Prior to April 1, 1995, double the amount of drain media and soil cover in the top layers were required (Figure 2 and 3).

Prior to April 1, 1995, the pressure distribution system required that the pressure laterals be spread on a mat of four foot (122 cm) centers and the one-eigth inch (0.32 cm) orifices a maximum of 24 inch (61 cm) centers. After April 1, 1995, the requirement was one orifice for every six square feet (0.56 $m^2$) of sand surface. This essentially gives you an appoximate 30 inch (76 cm) grid pattern for better distribution.

Water leaving an ISF is of sufficient quality to allow a shortened length of disposal trench compared to that of a conventional septic system. The trench length required is dependent on the soil conditions. The minimum length of a 2 foot (61 cm) wide disposal trench varies from 35 feet (10.7 m) per 150 gallons (568 L) projected daily sewage flow (PDSF) in gravel, sand, loamy sand and sandy loam soils to 75 feet (22.9 m)per 150 gallons (568 L) PDSF in high shrink-swell clays.

The ISF acts as a bioreactor as well as a physical filter. The individual grains of sand provide the necessary surface area for the bacteria to grow. In a properly functioning system, the sand allows the water to rapidly move through the ISF, thus helping to keep the ISF aerobic. Many types of microorganisms work together to treat the wastewater and digest the filtered solids, as well as keep the pathogenic microbial population small.

Excess water entering into the system may hydraulically overload the filter and soil disposal field. To minimize additions from precipitation, ISFs are gently sloped to facilitate runoff. In addition, planting of grass over the top is recommended to prevent erosion and to help take up excess water and nutrients. Deep rooting plants are not

Figure 2. Cross Sectional View of an Intermittent Sand Filter.

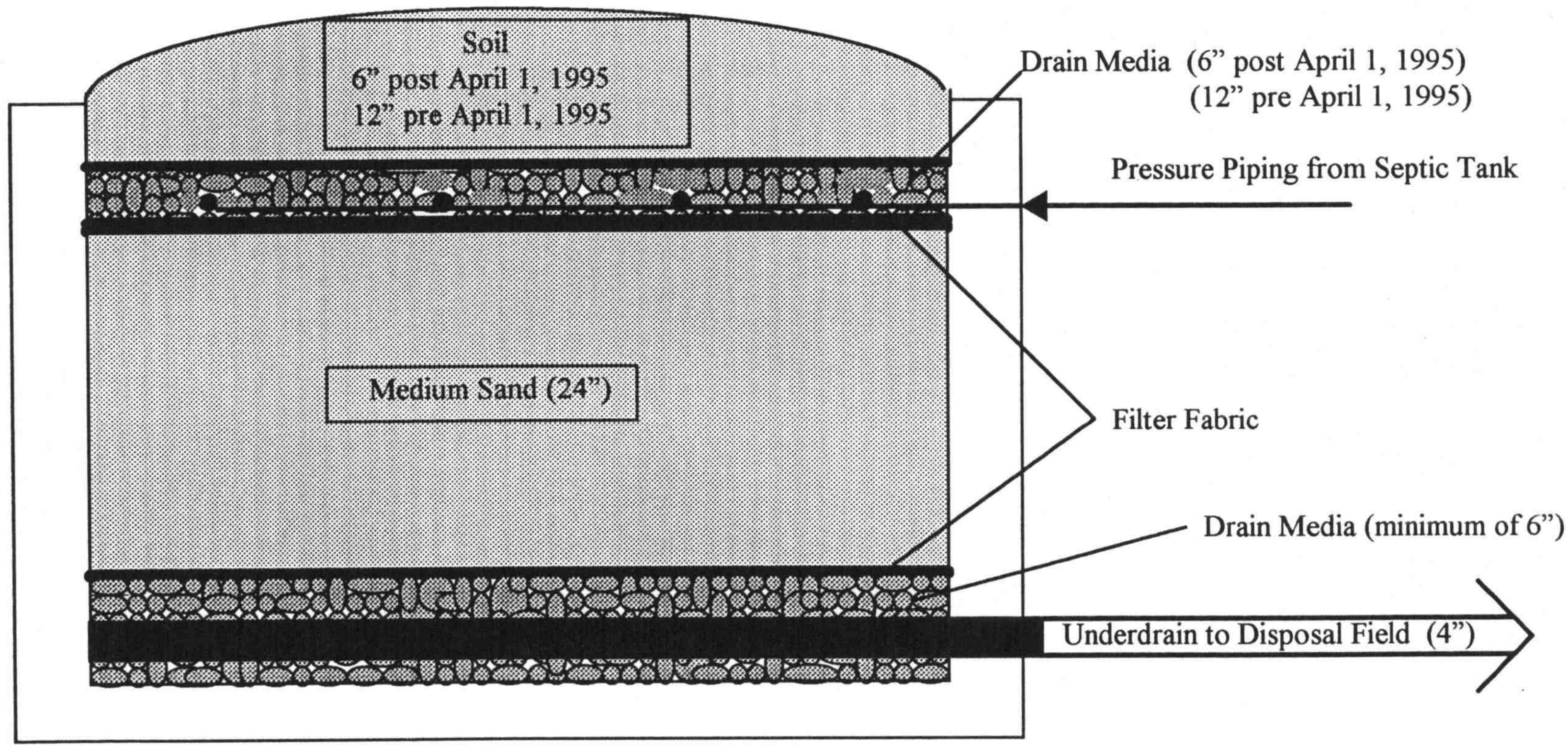
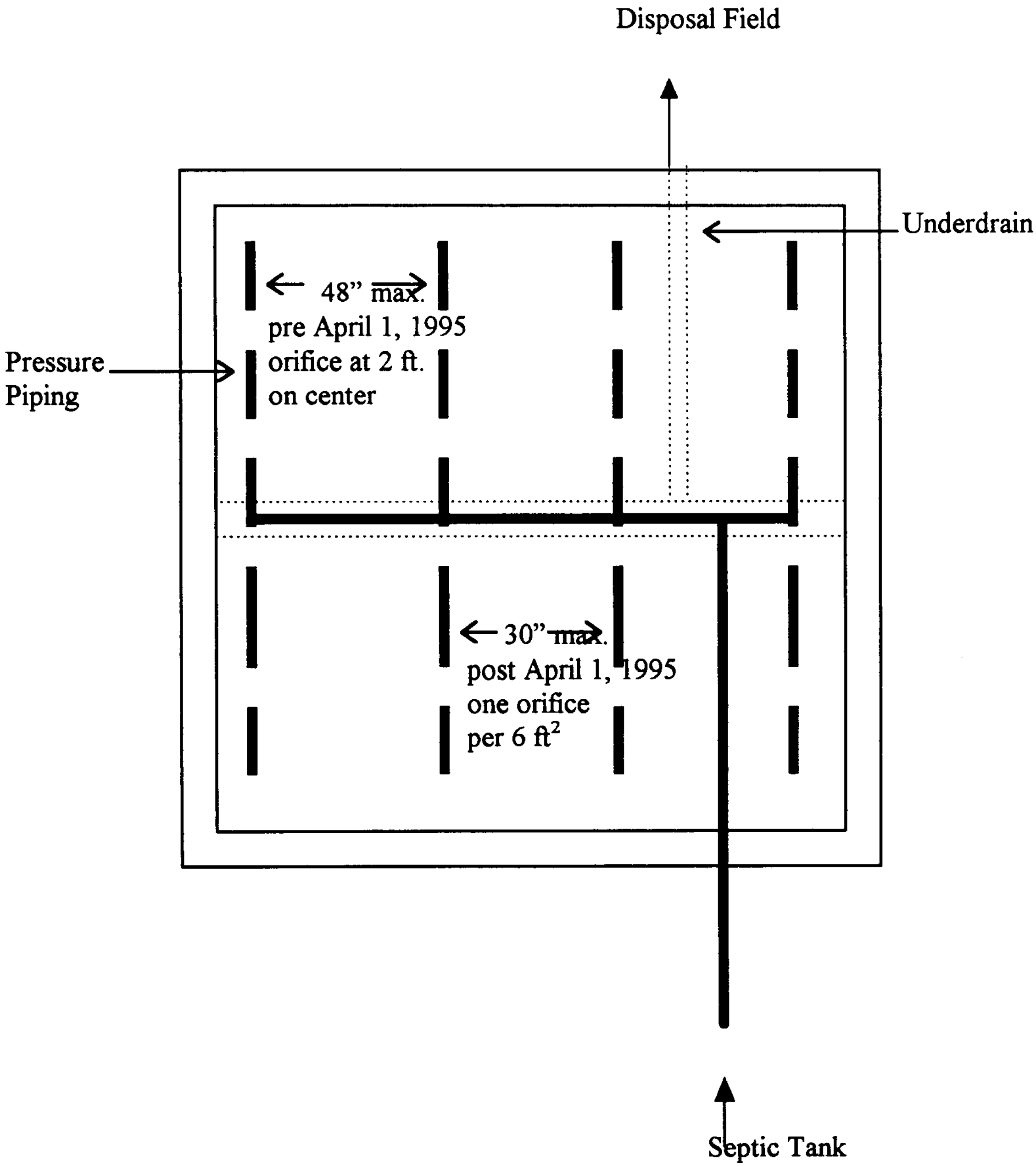


Figure 3. Top View of an Intermittent Sand Filter.



recommended on the ISF, as the roots can cause damage by penetrating the filter fabric (personal communication Ron Smith, Benton County Sanitarian, August 8, 1995).

To warrant continued usage, ISFs need to have a life expectancy of 20 years or more. One question of concern is whether or not the ISFs retain a high removal rate of total nitirogen over the years and what level of reduction can be expected after many years of use. This study will evaluate the justification for future use of ISF systems. If the ISF cannot maintain an acceptable nitrogen treatment level over time, improvements on the design and maintenance will have to be made, or a new technology created and placed in use.

# LITERATURE REVIEW

The disposal of human and household wastewater through on-site septic systems is a potential source of nitrate for groundwater. Between 25 and 30% of individual households in the United States have septic systems (Brown, et al., 1984; Burks and Minnis, 1994; Keeney, 1986; Lamb, et al., 1991; Starr and Sawhney, 1980). In addition, they are found in small rural establishments, such as resorts, motels, and restaurants. The nitrogen input from septic systems may be a relatively large and significant source of contamination to groundwater in many areas of the United States (Keeney, 1986; Perkins, 1984; Spalding and Exner, 1993; Yates, 1985). Many counties in Oregon have experience nitrate contaimination in groundwater. The contaimination was detected by the statewide groundwater monitoring program. The nitrate contaimination was found to be due to agricultural sources as well as septic systems. Two locations in particular were found to be due to septic systems, these locations were East Multnomah county and the city of LaPine (ODEQ, 1994; personal communication with Rodney Wick, Oregon Department of Environmental Quality, February 15, 1996). This may be due to improper siting of the system, or system failure. Once the effluent nitrogen leaves the disposal field, usually in the form of ammonia, it may be nitrified in the aerobic unsaturated zone of the soil. The predominant removal mechanism for nitrate is denitrification, which requires significant organic matter and an anaerobic environment. Unfortunately, most studies of septic systems have shown that the vadose zone below the systems are aerobic, therefore very little if any denitrification will take place (Starr and Sawhney, 1980; Walker et al., 1973; Whelan and Barrow, 1984). Nitrogen removal may be enhanced by the use of intermittent sand filters that oxidize a large proportion of the available nitrogen to nitrate, as well as removing approximately one-third of the TN through the process of nitrification and denitrification.

The density of septic systems in an area may be an indicator of the potential for pollution. If septic systems are dense, greater than 40 systems per square mile, then the potential for groundwater contamination from nitrate and fecal coliform bacteria is increased. The United States Environmental Protection Agency (USEPA) regards a density of 40 systems per square mile as an area having a potential for groundwater contamination (Pye, et al., 1983). In Oregon, the maximum density of septic systems is based on an area that is assumed to have nothing but sandy soil between the disposal trench and groundwater. This density is one site on a one-half acre lot. This density maximum is enforced in areas that are highly susceptable to groundwater contamination. In areas that are not as susceptable, lot sizes may be smaller, but sites smaller than one-half acre will often have trouble meeting all the setback requirements (OAR, 1995; personal communication Ron Smith, Benton County Sanitarian, February 20, 1996).

Areas in Colorado, Delaware, Massachusetts, New Mexico, New York, North Carolina, and Oregon have experienced groundwater contamination due, in part, to septic systems (Yates, 1985, personal communication with Greg Farrell, ODEQ Western Region On-Site Manager, 15 February 1996). Studies conducted in the these areas indicate high densities of septic systems as one of the sources of contamination (Yates, 1985). This source cannot be overlooked, especially when 90-95% of all people living in rural areas who are dependent on septic systems, use groundwater as their drinking water source (Bitton and Gerba, 1984).

## Health Effects

Elevated concentrations of nitrogen in natural systems may cause a variety of problems, including eutrophication, fish kills, and toxicity to aquatic plants by increasing the concentration of nutrients, such as nitrogen and phosphorus, and decreasing the dissolved oxygen concentration. One of the greatest concerns regarding elevated nitrate levels in drinking water is the threat to public health.

Cyanosis, "a bluish coloration in the skin and mucous membranes due to deficient levels of oxygen in the blood" (Parker, 1989), was first recognized by Hunter Comly in 1945. This condition is also known as blue-baby syndrome and methemoglobinemia. Ingested nitrate ($NO_3^-$) may be reduced biologically by microbes in the digestive system to nitrite ($NO_2^-$) (Goldsmith,1986; Hegesh and Shiloah, 1982; Johnson, et al., 1987; Shearer, et al., 1972; Shirley, 1975; USEPA, 1987; Walton, 1951; Wolff and Wasserman, 1972). Nitrate itself poses no health risk; the hazard occurs when it is reduced to nitrite. Nitrite is able to oxidize hemoglobin (Hb) to methemoglobin (metHb), which induces a high level of metHb in the blood stream.

> Hemoglobin is the red-colored pigment of the blood which binds molecules of oxygen and allows the blood to perform its vital function of transporting oxygen to the tissues of the body. In each hemoglobin molecule there is an iron atom in the ferrous or divalent state. The oxygen molecules carried by the hemoglobin are adsorbed and do not react with the iron atom directly. If the iron atom is oxidized to the ferric (trivalent) state, the hemoglobin changes color to brown and loses it capability to carry oxygen. This brown-colored derivative of hemoglobin is called met-hemoglobin, and to a very slight extent this occurs naturally and spontaneously. (Goldmith, 1986 pg )

Methemoglobinemia remains a potential threat to infants in rural America, despite the fact that nitrate has been tested in drinking water since 1945 (Lukens, 1987). Comley (1945) noted that infants drinking formula made with well water containing high nitrate concentrations turned blue, indicating a lack of oxygen. When the contaminated water was replaced with clean water, the symptoms disappear within a few hours to a few days. In the more severe cases, methylene blue was given to the babies to assist in the removal of the methemoglobin (Comly, 1945; Johnson, et al., 1987; Walton 1951; Cornblath and Hartman, 1948).

The metHb level in a healthy adult is generally less than 1% (Goldsmith, 1986; Hegesh and Shiloah, 1982). A level above 3% is defined as methemoglobinemia (USEPA, 1987). Persons older than six months of age have developed the enzyme metHb reductase, which converts the metHb to Hb before the levels in the blood become dangerous (Goldsmith 1986; Johnson, et al., 1987; Lukens 1987; Shearer, et al., 1972).

Infants younger than six months do not have fully developed immune systems and therefore do not have the ability to produce the metHb reductase enzyme. This makes them more susceptible to developing methemoglobinemia (Goldsmith 1986; Johnson, et al., 1987; Lukens 1987; Shearer, et al., 1972). Infants with a level of metHb above 10% may begin to show signs of cyanosis (Goldsmith, 1986; Walton 1951). A level above 60% metHb can trigger a coma or death (Johnson, et al., 1987; Wolff and Wasserman, 1972). The cases of methemoglobinemia in infants are often accompanied by acute diarrhea.

Studies have shown that infants with gastrointestinal disorders have a higher incidence of methemoglobinemia (Comly, 1945; Hegesh and Shiloah, 1982; Shearer, et. al, 1972). It is thought that the disturbed digestive tract environment is more conducive to the growth of the nitrate reducing bacteria *Escherichia coli*, salmonella and staphylococci (Shirley, 1975). The pH in an infant's stomach is lower than that in an older person's stomach and is a more suitable environment for the bacteria (Cornblath and Hartmann, 1948; Shirley, 1975; Walton, 1951; Goldsmith, 1986). Similarly, high concentrations of nitrate in drinking water, and a lack of the metHb reductase enzyme, make infants highly susceptible to developing methemoglobinemia. The problem is compounded by the fact that infants drink more water in relation to their body weight.

The U.S. Environmental Protection Agency (USEPA), in conjuction with the U.S. Public Health Service (USPHS), have set the maximum contaminant level (MCL) of nitrate in drinking water at 10 mg $NO_3$-N/ L as and nitrite at 1 mg $NO_2$-N/ L (USEPA, 1987). These values are based on evidence from case studies suggesting that nitrate concentrations of less than 10 mg $NO_3$-N/ L have not been associated with methemoglobinemia in infants (Comly, 1945; USEPA, 1987; Whitman, 1951). The USEPA health advisory for 70-kg adults and 10-kg children is 111 mg $NO_3$-N / L (USEPA, 1987); this is two orders of magnitude higher than the level for infants. To protect everyone's health, the drinking water standard is set for the more susceptible population.

Normally, nitrate and nitrite are absorbed through the digestive system into the blood stream and excreted by the kidneys. The inorganic nitrogen is able to distribute itself throughout various tissues, but does not bioaccumulate. Nitrate has been found in saliva and gastric secretions, the fluids where the nitrate reducing bacteria are found (USEPA, 1987). Once nitrate has been reduced to nitrite, the nitrite can react "in vivo with certain protein substrates (amines, amides, and urea) to form carcinogenic N-nitroso compounds" (Forman, 1987). These N-nitroso compounds are associated with gastric cancer, which according to studies is "probably the second most common fatal cancer in the world" (Forman, 1987).

A case-control study conducted in Colombia indicated a generally positive correlation between gastric cancer, nitrate content in drinking water, and nitrate excreted by the population (Cuello et al., 1976). The nitrosamines produced by the reaction of nitrite with protein substrates is thought to be the carcinogenic agent associated with nitrates. The risks from nitrate in drinking water requires continued study to establish or refute any association with cancer. The evidence to date is suggestive but not conclusive.

The greatest known and immediate health risk associated with nitrates and nitrites is infant exposure to contaminated drinking water. The long term carcinogenic effects of nitrate and nitrite needs further investigation before any causal connection can be made. At the present, nitrate may be avoided if a contaminated drinking water source is identified, by using bottled water for drinking and cooking.

**Nitrogen in Septic Systems**

The number of rural and community wells with nitrates is increasing. In 1990, only 2.4% of rural domestic wells and 1.2% of community water system wells contain nitrate in excess of the drinking water standard of 10 mg NO3-N/ L (USEPA, 1990). The 2.4% and 1.2% of rural and community wells, serve 4.5 million people including 66,000 infants under the age of twelve months (USEPA, 1990). These percentages could easily increase in the years to come since in 1990, 54.6% of rural wells and 50.9% of community water system wells contain detectable nitrate below the 10 mg NO3-N/L MCL (USEPA, 1990). The remaining wells, both rural and community, are below the detection limit for nitrate.

Contamination from commercial fertilizers, animal wastes, and septic systems may increase the levels of nitrate in groundwater (USEPA, 1990).

Previous studies of septic systems, both conventional and non-conventional, have found that the nitrogen in the septic tank effluent, predominantly ammonia, was almost fully nitrified when disposed of in unsaturated soil (Cochet et al., 1990; Gold et al., 1989; Starr and Sawhney, 1980; Walker et al. 73; Whelan and Barrow, 1984). The nitrification takes place in the aerobic soil environment just below the disposal field trench. In conventional systems without a sand filter, a biofilm, or crust, develops on the bottom of the trench, which promotes ponding. The unsaturated soil below the trench is well suited to nitrification.

Starr and Sawhney's (1980) study found that essentially all of the ammonia leaving the disposal trench was nitrified with the highest concentrations occurring at a depth of 60 cm. The study concluded there was little loss of the nitrate as it moved to the groundwater. A study by Whelan and Barrow (1984) found similar results. The ammonia from the septic tank effluent was nitrified in the unsaturated zone beneath the disposal trench. It was determined that once the effluent passed through the biofilm into the trench, the ammonia was nitrified. The nitrate concentration was found to be constant as it moved down the soil profile. A third study (Walker et al., 1973) also concluded that nearly all of the ammonia from the disposal trench was nitrified, but only below the trench and not adjacent to the trench. This study was conducted in sandy soils. Other studies have also found that near complete nitrification of ammonia is anticipated in a conventional septic system's disposal field (Cochet et al., 1990; Siegrist and Jenssen, 1989).

Other studies have found that some denitrification does occur as the nitrate moves through the soil profile (Ball, 1994; Eastburn and Ritter, 1984; Reneau, 1979). Eastburn and Ritter (1984) concluded that denitrification rates in conventional septic systems vary between 0 and 35%. The denitrification rate is essentially the removal rate of the nitrogen from the soil environment. Reneau (1979) estimated that the nitrate concentration decreased with depth, from 152 to 456 cm, with varying degrees of reduction depending on the season. Ball (1994) suggested that if sand filter effluent is disposed in the top 16 inches of soil, denitrification rates can be significant. In a general review of the literature, Siegrist and Jenssen (1989), determined that 20% of the nitrogen in a conventional septic system is removed in the subsurface absorption field. This removal is attributed to denitrification.

The requirements for dentrification are an anoxic environment, the presence of nitrate, and an available carbon source. Gold et al. (1989) suggested that in a properly functioning subsurface absorption field, denitrification will be limited by the lack of an anaerobic environment, as well as an insufficient carbon source. Eastburn and Ritter (1984) state that without an energy source, denitrification was unlikely, and organic matter in the soil is probably an unsatisfactory energy source for denitrification. A study of sandy soils by Walker et al. (1973) concluded that denitrification in a well aerated sandy subsoil, or in carbon deficient groundwater, was unlikely to occur.

Based on data from many studies mentioned above, the soil system cannot be relied upon for consistent nitrogen removal through denitrification. In some soils, denitrification may occur during the wet season, or even year round, but in most soils it seems that the denitrification of nitrate is not a significant source of nitrate removal. The lack of denitrification can be accounted for by the fact that codes require that standard systems are installed in well drained soils that have aerobic conditions even in wet months. Without the occurance of significant denitrification, the nitrate from the septic system enters the groundwater. A better means of nitrogen transformation is one that occurs before the disposal fields. This is one of the values of an intermittent sand filter.

Studies of sand filters have found total nitrogen removal rates of approximately 50% (Ball, 1994; Paeth and Ronayne, 1984). A study conducted in the late 1970's by the Oregon Department of Environmental Quality found a reduction of total nitrogen (TN) through intermittent sand filters of 47%, with 96% of the remaining nitrogen in the nitrate form. The study was repeated in Placer County, California, in the early 1990's, with similar findings of 40% removal of TN (Cagle and Johnson, 1994). The removal of the nitrogen through an ISF is most likely from a nitrification-denitrification process operating within the sand filter. Anaerobic microzones are thought to be the site of denitrification.

The available organic matter in the septic tank effluent serves as the energy source for the microbial reaction.

On-site ISF systems consist of a septic tank, ISF, and subsurface absorption area. The septic tank is a primary settling tank. The larger solids are separated from the liquid fraction by settling. The septic tank provides an anaerobic environment, permitting the conversion of organic nitrogen to ammonium. The ammonium enters the sand filter, an aerobic environment, and is oxidized to nitrate. Approximately 47% of the total nitrogen is removed as it passes through the filter, and the remaining nitrogen in the effluent appoaches 100% in the form of nitrate (Paeth and Ronayne, 1984). Most likely, the 47% of the TN that is removed is due to losses from nitrification-denitrification processing. The filter is kept aerobic by the natural diffusion of air into the filter as well as physical aeration by small organisms, such as nematodes.

The subsurface absorption field is predominantly aerobic supporting microbial processes such as nitrification. The soil contains enough air to keep the system oxic, but extended periods of saturation may cause anoxic zones to occur. Anaerobic microzones occur in the soil and in soil particles' crevices. These anaerobic microzones support anaerobic microbial processes, namely denitrification. In addition, Oregon rules allow the seasonal water table to come in contact with the ISF disposal trench by six inches on ground with a slope greater than 12% and a full 12 inches on flat ground. In the wet months of the year anoxic conditions will occur in the ISF disposal trench periodically.

A study examining the performance of disposal trenches receiving recirculating sand filter effluent found that denitrification of the nitrate was occurring within the trench. The effluent was found to be ponded within the trench due to the low permeability of the surrounding soils. The effluent was found to be one and one-half to three inches (3.8 to 7.6 cm) deep, with a low dissolved oxygen content and a high total organic carbon concentration. The appropriate conditions along with reductions of nitrate found in two separate trenches of 97 and 71% suggest the occurrence of denitrification (Wert and Paeth, 1985).

In order to maintain an acceptable level of performance form an ISF system, regular monitoring and maintenance on a yearly basis needs to be implemented. Systems need routine maintenance such as regular monitoring of the solids in the septic tank to determine when it needs pumping, cleaning of the pressure laterals, checking the controls for proper dosing cycles, cleaning of pump screens, and maintenance of mound vegetation. The maintenance of a system may greatly affect its performance. Systems that are not maintained may encounter clogging of the distribution systems affecting the distributing of the effluent on to the filter. Clogging may result in areas of the system being overloaded or short circuited through the system (OAR, 1988; OAR, 1995; personal communication with Greg Farrell, ODEQ Western Region On-Site Manager, January 10, 1996). If a system is not functioning as designed, the physical, chemical, and biological processes will be affected, reducing treatment efficiency.

**Intermittent Sand Filter Biology**

The microbiology of a sand filter is complex. Many types of organisms live in the filter, keeping it clean and preventing the accumulation of solids. Bacteria eat the organic matter; protozoa, and nematodes eat the bacteria, keeping the system aerated. The most prominent bacteria is the *Zooglea* genus. It has the desired ability to digest sewage making it important to the sewage treatment process (Calaway, 1957). *Flavobacterium* and *Bacillus* genera are other forms of bacteria that decompose organic nitrogen compounds and absorb or breakdown carbohydrates (Calaway, 1957). The *Alcaligenes* sp. bacteria are able to degrade the complex organic nitrogen compounds. This bacteria is found in the deeper layers of the filter and is dependent on other bacteria to start the digestion process (Calaway, 1957). Other forms of soil bacteria, such as *Nocardia* sp. and *Streptomyces* sp. digest humus minimizing clogging from the waste of other microbes (Calaway, 1957).

Ciliate protozoa are of the genus *Clopoda, Paramecium,* or *Clopidium* (Calaway, 1957). The flagellate *Peranema* is also present in the filter along with amoebae (Calaway, 1957). All of these protozoa help to keep the bacterial community from over populating and help keep the filter clear of solids.

The sand filter community also consists of metazoa, multicellular organisms. This group includes annelid worms, flatworms, nematodes, rotifers, water mites, insects, and insect larvae. By eating sludges and slimes that build up in the system, the metazoa work to keep the sand bed open and accessible to oxygen. The waste produced by the metazoa is eaten by the microbial community due to the waste's greater porosity, allowing easy access for bacteria and oxygen. The nematodes, water mites, and flatworms prey on oligochaets, keeping their population down and keeping the population in an active state (Calaway, 1957).

Of all the processes performed by the sand filter community, biological oxidation is doubtlessly the most important. Without the community, the removal of wastewater constituents would be reduced. The microbes use the organic material for an energy source and utilize some of the nutrients for their own cell growth. Biological oxidation and cell growth contribute to wastewater treatment as well as support a large and varied microbial population.

## Transport of Nitrogen in the Soil

The transport of nitrogen is affected by soil type and soil properties, such as porosity and cation exchange capacity. Moisture content, organic material content, and reduction-oxidation chemistry of the soil all interact with the nitrogen and influence its transformations. These properties are all interrelated and dependent on one another. The soil types that are composed of different percentages of silt, sand, and clay are an indicator of these properties. The soil type also gives an indication of the water holding capacity, and possibly indicates redox potentials. Redox potentials are a measure of the electrons available for reduction-oxidation reactions. A high redox potential indicates an oxidizing environment, while a low redox potential indicates a reducing environment. Soil properties are also dependent on the minerals in the soil.

The presence of water is important to the diffusion of nitrogen compounds within the soil. Diffusion is highly dependent on soil moisture (Reddy et al., 1980). Often the diffusion of molecules from one microenvironment to another is a limiting factor in the transformation process. The movement of ammonium ions into an aerobic zone and the oxidation of the ammonium is the limiting step in the nitrification-denitrification process (Reddy et al., 1980). Without water, the nitrogen compounds as well as all other nutrients have limited movement within the soil profile.

The process of infiltration may be slow, this is more likely to occur in soils that have a high clay content. In clay soils, the water must move through small pore spaces, which takes more time due to the cohesive and adhesive forces of water (Gilliam et al., 1977). The movement may also be quite rapid if the soil has a high sand content and is very moist. The coarse material allows for a greater interstitial space, which in turn allows for a more rapid downward movement of water (Kissel et al., 1974; Thomas and Phillips, 1979). Changes in a soil profile from one texture to another, often called a layer, may create an area of ponding, or of more rapid movement (Gilliam et al., 1977; Starr, et al., 1978). When a finer soil containing more clay is encountered under a coarse soil, ponding may occur. When water enters finer soil, where the infiltration rate is slower, the water coming into the space is greater than the water leaving the space. The ponding provides a potential anaerobic environment for denitrification (Gilliam et al., 1977).

The movement and transport of nitrogen from the surface to groundwater is mediated by water. Diffusion and advection of chemicals may take place in the soil water. The movement of water is dependent on the type of soil. Several factors affect water movement through the soil profile. These factors include the soil structure, soil texture, organic matter and moisture content. Of these, soil texture has perhaps the greatest affect on water movement. Water is a unique molecule in that it forms hydrogen bonds between other water molecules (cohesion) and with oxygen molecules from mineral sources. These bonds combine to form a film of water around soil particles. Sands with larger particles and larger interparticle spaces will conduct water quickly when compared to clays. Clays with a greater number of smaller particles and smaller openings between soil particles, have a much slower infiltration rate and are more likely to pond water.

Infiltration is the process of water movement through the soil column. Water is moved by the force of gravity and capillary action (Taylor et al., 1983). The infiltration rate is dependent on the soil type, the soil porosity, the soil permeability, the presence of any layering, and the moisture content of the soil (Black and Waring, 1976b; Gilliam et al., 1977; Taylor, et al., 1983). The rate of infiltration increases with increasing soil moisture. First the water is used to wet the soil and subsequent water flow is able to move through the soil profile (Taylor et al., 1983).

Infiltration may be very fast if fingers, or channels, exist in the soil to transport the water with less resistance (Keeney, 1986; Kissel et al., 1974). When soils go through transitions of wet and dry periods, fissures in the soil profile often emerge. The fissures are large cracks in the soil, extending many feet in depth and even to the water table itself. This is especially true of soils with a high shrink/swell clay content, although when saturated, the clay swells to close the fissure (Kissel et al., 1974). The fingers may extend to the groundwater, giving surface water a direct route. Rapid water movement may also be created by wormholes, or layers of rock that conduct the water along the rock surface (Keeney, 1986).

The rate of movement of nitrogen compounds is highly dependent on the water transport, but the rates are not exactly the same (Black and Waring, 1976b). Nitrogen compounds may be adsorbed on to the soil column, but the quantity adsorbed depends on the form and ionic charge of the nitrogen. Adsorption may cause the compounds to be slowed, relative to the water, when passing through a region of opposite charge (Black and Waring, 1976b). For ammonium, this is usually an area of high organic matter or clay and the amount of potential sites is called the cation exchange capacity. What little nitrate that is adsorbed will be adsorbed when there is low organic matter content. There are many factors associated with a zone of positive charge and nitrate adsorption. Soil minerals carrying a positive charge, pH, and the presence of other anions, all influence the adsorption of nitrate onto the soil column (Black and Waring, 1976a). Adsorption and movement are dependent on the concentration of nitrate present and are usually a negligible nitrogen loss mechanism (Black and Waring, 1976b).

The inorganic forms of nitrogen are often in an ionic state, having a positive or negative charge, and that makes water even more important to their transport. Ions are only stable while in a solution or adsorbed onto some other surface with an opposite charge, such as a humus or clay colloid. The ammonium or nitrate ions may be adsorbed onto oppositely charged soil sites or remain in solution and travel with the water. Due to the greater abundance of negatively charged soil particles or humus the nitrate has a tendency to remain in solution (Black and Waring, 1976b).

Transpiration and evaporation are additional factors affecting the movement of water and nitrogen compounds within the soil environment (Taylor et al., 1983). Transpiration, via plants, pulls the water from the soil, thus reducing the water's influence on the leaching process. In addition, evaporation from the soil surface may occur. Evaporation has the greatest impact during warm and windy times of the year (Taylor et al., 1983).

## Transformations of Nitrogen in the Soil

Transformations of nitrogen occur when the molecular form of the nitrogen changes. There may be a change in oxidation state or a change in one or more atoms of the molecule. Physical transformations do not change the molecular structure, but the physical state. There may be an adherence to a soil molecule or a change from an aqueous state to a gaseous one.

The adsorption of ammonium is one of many nitrogen transformations that may occur in the soil. The nitrogen element is transformed from one form to another by physical and biological processes. The transformations take the nitrogen from organic forms to inorganic forms and from one inorganic form to another. This entire process is called the nitrogen cycle (Appendix 1). Biologically mediated transformations include nitrification, ammonification, denitrification, immobilization, mineralization, and fixation. Physical processes are known as sorption and volatilization. Overall, the transformations of nitrogen keep it moving in the environment. Under the correct conditions, one form can be transformed to another. These processes all play an important role in the loss and movement of nitrogen in the soil and water.

The cycling of nitrogen involves many different chemical, biological, and physical transformations. These transformations are mediated by the presence of microbes and environmental conditions. The transport of nitrogen is strongly influenced by the movement of water within the soil profile and the presence of an ionic exchange capacity in the soil. Organic matter is another indicator of the processes occurring in the soil. The microbes require a carbon and energy source to sustain cell growth. Organic matter is often decomposed vegetation, animal feces, or decayed animal tissue which provide a source of available carbon. The carbon is used in cell growth and also a source of energy in anaerobic biological processes. Nitrogen cycling is also dependent on the cycling of oxygen within the soil environment. The presence or absence of oxygen creates aerobic and anaerobic zones, respectively, thus dictating the biological transformation most likely to occur in the zone.

## Physical and Chemical Transformations of Nitrogen

The chemistry of soil is influenced by the mineral makeup of the soil. Minerals such as sulfur, iron, and manganese, contribute to the chemical reactions occuring at different redox potentials. The presence or absence of these elements will influence, in part, the organisms that are able to live within the soil. Under highly reduced conditions, some of these elements may be transformed into chemicals that are toxic to the microbes and plants, inhibiting most activity in the soil. The content of the soil also dictate its ionic exchange capacity, which is a significant factor in the adsorption of nitrogen compounds to the soil. Soils with a high cation exchange capacity and large amounts of organic matter tend to reduce the loss of nitrogen applied to the soil due to the exchange of ammonium for calcium and magnesium ions (Fine et al., 1989; Reddy and Patrick, 1980). Ammonium ions have a strong affinity for the soil cation exchange sites; often very little ammonium is found in the overlying floodwaters (Reddy et al., 1980).

One physical transformation of nitrogen is sorption onto soil particles. Sorption includes adsorption and absorption. Adsorption is the process of a physical or chemical bond to the surface of the particle, and absorption is a physical trapping of a molecule within the particle (Fetter, 1993). The occurrence of sorption is dependent on the form of nitrogen, and its ionic charge. Most soils are negatively charged, so only positively charged molecules, such as ammonium, are sorbed. Negatively charged molecules, such as nitrate, are repelled by the like charge and tend to move freely through the soil.

The volatilization of ammonia is a physical transformation that converts the molecule from its liquid state to a gaseous state. In large part, this process is controlled by the pH of the soil solution. The pH dictates the ratio of ammonia to ammonium, which dictates the concentration of ammonia in solution. The amount of ammonia volatilized is dependent on Henry's Law coefficient for ammonia, which is $2.91 \times 10^{-4}$ atm $m^3$/mole at 20°C (Montgomery and Welkom, 1991). A high concentration of ammonia in the water will normally create a state of equilibrium with the air, causing the ammonia to volatilize. High concentrations are most likely to occur at pHs above 9.3. Below this pH level, non-volatile ammonium is the dominant form (Snoeyink and Jenkins, 1980).

## Biological Transformations of Nitrogen

For the most part, biological transformations are microbially mediated. Often referred to as assimilation, one process, immobilization, is the process of converting inorganic nitrogen to organic nitrogen. Inorganic forms of nitrogen are nitrate and ammonia, which contain no carbon. Organic forms of nitrogen contain a carbon atom in the molecule such as urea and proteins. Plants are able to assimilate nitrogen in the form of nitrate and ammonia. The nitrogen is taken up by the plant's roots and converted to amides within the plant. This "immobilizes" the nitrogen within the soil-plant-water environment, at least temporarily. Some species of plants have been found to remove as much as 0.20 g N per $m^2$ per day (Reddy, 1983). Microbes are also able to assimilate nitrogen into their cell material. All living organisims do this to some extent, or at least they are able to deal with the intake of inorganic nitrogen and convert it to organic forms and remove it from the body.

Often immobilization in plants is coupled with nitrogen fixation, the microbial conversion of nitrogen gas to ammonia. Nitrogen fixers belong to the families of *Azotobacteraceae, Rhizobiaceae, Bacillaceae,* and photosynthetic procaryotes, such as blue-green bacteria (Gaudy, 1988). The family *Azotobacteraceae* consists of free living aerobic microorganism found in soils and water, and *Rhizobiaceae* live symbiotically on roots of legumes (Gaudy, 1988). The conditions of nitrogen fixation is either aerobic or anaerobic, depending on the organism involved. There are relatively few organisms that are able to fix nitrogen gas, especially given its abundance in the atmosphere.

Nitrification is a microbial mediated two step process that converts ammonium to nitrite to nitrate. The process is completed by the bacterial families of *Nitrosomonas* and *Nitrobacter* (Gaudy, 1988). Nitrification is an aerobic process, requiring the presence of ammonium and a pH at or below 6.6 (Whitehead and Raistrick, 1993). Studies have shown that almost all of the ammonium is converted to nitrate within the first few centimeters of an aerobic media in the soil (Cochet et al., 1990). The movement of ammonium into an aerobic zone tends to be the limiting step in the nitrification and denitrification process. This rate of nitrification is dependent on the number of nitrifying bacteria and the abundance of oxygen (Reddy et al., 1980). The rate of ammonium oxidation is not dependent on the concentration of ammonium; it has zero order kinetics (Reddy et al., 1980). The presence of organic matter in great quantities may be detrimental to the process of nitrification, because the nitrifying bacteria cannot compete with microorganism better suited to the high carbon environment (Gaudy, 1988).

Denitrification is an anaerobic process in which nitrate is transformed to nitrogen gas. The presence of organic carbon, a high soil moisture content, and a high pH all contribute to favorable soil conditions for denitrification (Hantzsche and Finnemore, 1992). The microorganisms mediating this process are know as denitrifiers and belong to the genus *Pseudomonas* (Gaudy, 1988). For denitrification to occur, certain conditions must be present: anoxia, a supply of nitrate, and a supply of available carbon in a ratio of 1-3 to 1 (C to $NO_3$-N) (Cochet et al., 1990; Lamb et al., 1991). A lack of available carbon in soils, such as coastal sands, and not a lack of nitrate, tends to be the limiting factor in denitrification (Drury et al., 1991). Reddy and Patrick, (1984), found that the kinetics of the reaction changed, depending on the limiting factors. When carbon or nitrate is unlimited, the reaction is zero order; when one of the two is limited the reaction is first order; if both nitrogen and carbon are limited, the reaction has Michaelis-Menton kinetics (Reddy and Patrick, 1984). The redox potentials when denitrification takes place, are between 100 and 350 mV (Sikova and Keeney, 1976). Denitrification may increase the alkalinity and pH where it occurs (Andreoli et al., 1979).

Soil type can be a good predictor of the soil's water holding capacity. Soils containing a high percentage of sand are less likely to retain water compared to those with high clay content. Fine textured soils have been found to enhance denitrification (Keeney, 1986) due to a high hydraulic retention time, allowing biological processes to occur (Gilliam et al., 1978). The retention of water can cause anoxic conditions favorable to denitrification, as well as being necessary to a healthy environment for microbes. A lack of oxygen and a presence of available carbon may cause a reduction in the redox potential of the soil creating a reducing environment. Soil moisture and texture that add to the development of two distinct soil layers may enhance the nitrification-denitrification process by setting up aerobic and anaerobic sub-layers (Keeney, 1986).

The decay of organic matter by microbes converts the nitrogen from an organic form to an inorganic form. This process, known as mineralization, is sometimes referred to as ammonification. The rate of mineralization is dependent on the type of organic matter to be broken down and the concentration of nitrogen (Fine et al., 1989). The fraction of nitrogen mineralized is inversely proportional to the concentration of nitrogen in the soil system (Fine et al., 1989). Mineralization also occurs when fertilizers, in the form of urea, hydrolyze to ammonium (Andreoli et al., 1979). This process can be either aerobic or anaerobic.

# OBJECTIVES

The objectives of this study were to determine the long term removal transformations of nitrogen compounds through intermittent sand filters (ISF), and to determine if the disposal field effectively contributes to any further treatment of effluent nitrate.

# METHODS

A total of 44 intermittent sand filter septic systems in Western Oregon, were sampled over a 3 month period during the summer of 1995. Individual household intermittent sand filter (ISF) systems in five counties in Oregon were chosen to represent the western region of Oregon. The sites include soils from the Willamette Valley, the coast, the south western region, and the northern region. Sites with either high clay or sandy soils were emphasized because these soils are predominant when a standard system is unacceptable and an ISF offers an alternative. The sites are located in Benton and Lane counties in Oregon's Willamette Valley, Lincoln county on the Oregon coast, Clackamas county in northwest Oregon, and Douglas county in southwest Oregon. Each county had a total of 11, 12, 5, 8, and 8 sites sampled, respectively.

For selection of ISF systems, an eligible system had to be at least 36 months old. The process of site selection within a county was dependent on the number of intermittent sand filter (ISF) sites in that county. Benton and Lincoln counties, for example, had a small number of ISF sites, and therefore, all sites were contacted by letter. In counties with a large number of sites, such as Lane, Douglas and Clackamas, eligible sites from the appropriate county's computer database was used to identify the desired locales. From the computer list, every third ISF site was chosen as a potential area of study. Some sites were later eliminated due to a lack of sufficient information in the file that is, a lack of a certificate of satisfactory completion. Sites could be dropped from possible study if information on the current homeowner or a current phone number could not be obtained. Once it was determined that the site was at least 36 months old, and sufficient information was obtained, the site was added to the list for phone contact, and possible sampling.

Upon the completion of the list of potential sites, the appropriate government office sent an informational letter to homeowners (Appendix 2). The letter specified information concerning the purpose and objectives of the study and what could be expected if the homeowner agreed to participate. The researcher telephoned the homeowners two weeks after the letter was sent. At the time of contact, it was determined if the owner was willing to participate and if not, the homeowner was removed from the list. If the homeowner was willing to participate, information was obtained and entered on a questionnaire (Appendix 3) and a site visit was scheduled.

Prior to making a site visit, the pertinent information from the site, was photocopied from the county or state office file and attached to the homeowner information questionnaire. Relevent information included a site map with the layout of the system, soil information, and the certificate of satisfactory completion. The soil type was taken from the original site assessment.

The exact soil and wastewater samples taken from a site varied depending on the wishes of the homeowner. All homeowners agreed to have soil samples taken, as well as a liquid sample from the septic tank, if it was easily accessible. If the septic tank access port was covered, sampling depended on the homeowner's consent to have the area disturbed. The distribution box was also an optional site of wastewater sample collection.

The site map was used to locate the septic tank, ISF, and disposal field. Where available in the file, distances were used to facilitate the location of the distribution or drop box and trenches. A trench was chosen for sampling based on the site map and vegitation in the area, both gave an indication of which trench was recieveing effluent. Starting at approximately ten feet (3.0 m) from the end of the trench, a line perpendicular to the trenches was probed, and colored flags were used to mark the trench location. A soil probe, capable of reaching a depth of two feet (61 cm), was used to locate the trenches and distribution box. A trench was located when the probe hit gravel. When a probe indicated a trench, further probing was conducted to confirm the finding.

Once the trench was sufficiently flagged, and the end near the distribution box located, sample points were marked. Sample points were located two to three inches (5.1 to 7.6 cm) from the edge of the trench by probing until the edge was located. Most samples were within one inch (2.5 cm) of the trench. To facilitate obtaining a saturated soil sample, it was desirable to take the sample as close as possible to the side of the trench, without sampling inside the side wall. Sampling points were selected based on their distance from the distribution end of the trench. Points at approximately ten feet

(3.0 m) from the end of the trench, and then two more at approximately five foot (1.5 m) intervals, were flagged as sample points.

Prior to auguring next to the trench, a background, or control soil sample was taken in the yard, but away from the disposal field. Background levels of nitrate, Total Kjeldahl Nitrogen, and electrical conductivity were determined from the control sample. Soil samples were collected at a depth of approximately 30 inches (76.2 cm) using a three inch diameter auger.

All soil samples were collected and placed in one pint glass jars with Teflon lids and stored at temperatures ranging from 0 to 4°C. If a liquid was present in the sample hole, the soil sample was quickly removed to reduce any contamination from seepage. Liquid samples were obtained from holes using a hand pump. Liquid samples were collected in 500 mL polyethylene bottles, preserved with 12 drops of concentrated sulfuric acid ($H_2SO_4$), and stored at a temperature of 0-4°C. The location and depth of the hole was recorded. While in the field, the electrical conductivity (EC) of all soil samples was measured using a GLA Instant EC Salinity Drop Tester. This was compared to the EC of the background sample to give verification that the effluent was present in the soil sample. Due to its greater concentration of ionic compounds, such as ammonium and nitrate, the effluent sample was expected to have a higher EC, two to three times greater.

After the soil samples near the trench were taken, the distribution box was located using the probe. In general, the boxes were much more difficult to find; they were smaller and varied in depth. When located, the box was uncovered and the lid removed. When present, a liquid sample was taken from the influent. Otherwise it was taken from the standing wastewater in the box. This sample was placed in a 500 mL polyethylene bottle, preserved with 12 drops of $H_2SO_4$, and stored at a temperature of 0-4°C.

The last sample taken was of wastewater from the septic tank. A long handled dipper was used to collect a grab sample from the tank. A location near the pump was the preferred sampling point. A filter screen surrounded most pumps; consequently the effluent contained fewer solids, and a clear liquid sample was easier to obtain. The sample was collected and treated, using a method consistent with the other samples.

Samples were shipped to the Oregon Department of Environmental Quality (ODEQ) laboratory in Portland, Oregon. To keep the samples cold, in a range from 0 to 4 °C, the samples were shipped in a cooler full of ice. The samples were then handled by the ODEQ laboratory staff. All the samples were analyzed for Total Kjeldahl Nitrogen (TKN) and nitrate and nitrite ($NO_3$ and $NO_2$). The percent moisture was determined for all of the soil samples. Nitrate and nitrite analysis by the automated cadmium reduction method was used by the ODEQ laboratory, and conforms to US EPA-600 method 353.2 in *Methods for the Chemical Analysis of Water and Wastes*. The TKN was determined by the automated phenate method, conforming to US EPA-600 method 351.2 in *Methods for Chemical Analysis of Water and Wastes.* Due to the extraction process, the preparation of the soil samples added an extra step. The extraction methods were as follows:

Nitrate: Approximately 15 grams of the soil sample was extracted into 100 mL reverse osmosis (RO) water by stirring over a magnetic stirrer for one hour. The extract was centrifuged and then decanted and filtered through a glass fiber filter (Whatman 934-AH). The extracted sample was analyzed by an automated cadmium reduction method, as described in US EPA-600 method 353.2.

TKN: Some weight of soil, 0.10 to 0.50 grams, was weighed into a 75 mL standard digestion tube. The actual weight was accurately recorded to the nearest 1/100 gram. Twenty mL of RO/deionized water was added to the tube. The sample was then analyzed by the automated phenate method, as described in US EPA-600 method 351.2.

All results were reported as mg N per dry kg of soil. This value is calculated from the determination of soil moisture, reported as percent moisture by weight.

# RESULTS

Data were organized and imported into the statistics program using Microsoft Excel. Data were analyzed using the Statistical Analysis System, (SAS). All significance levels are based on a probability of the null hypothesis occurring by chance less than 5% of the time, or a p-value of 0.05. The 0.05 value is generally used within the scientific community as an acceptable level of statistical significance. All p-values below 0.05 are considered statistically significant; values larger than 0.05 are not statistically significant.

A total of 44 sites were sampled in five counties in Western Oregon. Of the 44 sites samples, only one had standing water on the disposal field and sand filter. This means that 98% of the systems were functioning properly hydraulically and therefore did not have standing water on the disposal field. Because 98% of the sites sampled appeared to be functioning without problems, the allowance of a shortened drain trench length for ISF effluent appears to be appropriate.

The average concentrations of nitrate and Total Kjeldahl Nitrogen (TKN) in the septic tank, distribution box, and soil samples as well as the average total nitrogen (TN) percent removal appear in Table 1. Nitrite was not detected in any of the samples and was therefore not included in any of the analyses or the calculation of total nitrogen. The number of sites sampled appear in Table 2. Total nitrogen is the sum of the nitrate concentration plus the TKN concentration. The average site specific percent removal of TN was 35% with a range of 6.0 to 91%. It should be noted that the system with a TN removal of 6.0% did not have standing water on the system and the nitrogen was predominantly in the nitrate form, yet little was removed. The percent removal of TN based on the average concentrations in the septic tank and distribution box was 43%. The difference between the 35% and 43% occurs because more septic tank samples were collected compared to the distribution box samples. The average removal of 43% was calculated using the average concentrations from all the septic tank samples and all the distribution box samples. The site specific average of 35% is based only on individual sites having both a septic tank and distribution box sample. The percent removal was calculated for each site then these values were averaged.

Table 1. Average Concentrations of Total Nitrogen (nitrate and TKN).

| Sample | n | Average | Standard Error |
|---|---|---|---|
| Septic Tank Nitrate* | 29 | 0.02 mg N/L | 0.01 |
| Septic Tank TKN* | 29 | 59.4 mg N/L | 4.6 |
| Distribution Box Nitrate* | 15 | 26.2 mg N/L | 5.2 |
| Distribution Box TKN* | 15 | 7.5 mg N/L | 3.2 |
| Drainfield Soil Nitrate** | 31 | 24.5 mg N/L | 3.5 |
| Drainfield Soil TKN | 40 | 558 mg N/kg dry soil | 30 |
| Background Nitrate | 32 | 8.8 mg N/L | 3.5 |
| Background TKN | 32 | 805 mg N/kg dry soil | 83 |
| Percent Removal of TN Based on Averages | 29/15 | 43% | |
| Average of Site Specific TN % Removal | 11 | 35% | |

*(Septic tank effluent is pre-ISF, Distribution box is post-ISF)

**(Drainfield Soil Nitrate = Trench Concentration - Background Concentration) Due to high background concentrations, some values were negative and were not used to determine the average.

Table 2. Number of Sites Sampled at Each Collection Point

| Sample Location | Number of Sites Sampled |
|---|---|
| Septic Tank | 29 |
| Distribution Box | 15 |
| Trench Soil, Hole 1 | 41 |
| Trench Soil, Hole 2 | 40 |
| Trench Soil, Hole 3 | 40 |
| Background Soil | 32 |

The nitrate concentration of the effluent leaving the disposal trench is the disposal trench soil nitrate concentration minus the background soil nitrate concentration. This calculation accounts for the concentration that would be present in the soil without the influence of the trench. After completing the calculation some of the trench concentration values were negative. These values were treated as zeros in the statisitical analyses since a negative concentrations cannot exist. The high background nitrate concentrations at these sites should not bias the analysis since site is a factor which will account for the between site variance. The negative values were not used to calculate the average concentration of disposal trench nitrate since it would bias this value. In addition, the nitrate concentrations are converted from mg $NO_3$-N/ kg dry soil to a mg $NO_3$-N/ L using the soil moisture content since it is assumed that all the nitrate is in the water.

Three soil samples were taken at each site and at different positions along the length of the first disposal trench. An analysis of covariance was used to determine if the position along the trench differed in nitrogen concentrations. Tables 3a and 3b show the summary statistics from the analysis, the important numbers to note are the p-values in the right hand column and the type III sum of squares (SS) in the bottom row. The p-value describes the statistical significance of the model or the factor (position, site), while the type III SS shows how much of the variance can be accounted for with the corresponding factor. The position along the trench was not statistically significant for either TKN or nitrate, $p = 0.286$ and $p = 0.148$ respectively. This means the concentration does not differ statistically with position, so all three positions are pooled and treated as one sample point for further analyses.

The site, or residence, was found to account for a large portion of the variance in the soil samples taken from the disposal trench (Table 3a and 3b, type III SS). In an analysis of covariance with site, trench position, and background concentration as factors, site accounted for 98% of the variance in the TKN data, and 98% in the nitrate data. The p-value in each analysis of covariance for both nitrate and TKN was 0.0001, the lowest output p-value in SAS. Consequently, both were statistically significant.

Table 3a TKN: Statistical Model Describing the Difference Between Sampling Locations Along Trench

| Source | DF | Sum of Squares | Mean Square | F-value | p-value |
|---|---|---|---|---|---|
| Model | 33 | 8389618 | 254231 | 5.1 | 0.0001 |
| Error | 62 | 3071773 | 49545 | | |
| Corrected Total | 95 | 11461390 | | | |
| | | Type III SS | Mean Square | F-value | p-value |
| Site | 30 | 7108224 | 236941 | 4.8 | 0.0001 |
| Position | 2 | 126494 | 63247 | 1.3 | 0.286 |

Table 3b $NO_3$: Statistical Model Describing the Difference Between Sampling Locations Along Trench

| Source | DF | Sum of Squares | Mean Square | F-value | p-value |
|---|---|---|---|---|---|
| Model | 42 | 75758 | 1804 | 5.75 | 0.0001 |
| Error | 76 | 23826 | 313 | | |
| Corrected Total | 118 | 99584 | | | |
| | | Type III SS | Mean Square | F-value | p-value |
| Site | 39 | 74735 | 1916 | 6.11 | 0.0001 |
| Position | 2 | 1227 | 613 | 1.96 | 0.148 |

A contrast statement in SAS, a type of analysis of variance, was used to compare the nitrate and TKN concentration from the disposal trench soil to that of the background soil concentration. The disposal trench nitrate concentration was found to be statistically different from that of the control samples, p-value = 0.0006. On average, the nitrate values in the trench are higher than the background samples, 24.5 compared to 8.8 mg $NO_3$-N/L. The nitrate concentrations in the background samples were higher than expected, but similar to background concentrations found in a study of at-grade on-site system in Wisconsin (Converse et al., 1991).

The TKN was also found to be statistically different between the trench and background samples, p-value = 0.0001. The TKN concentrations in the background samples were found to be higher than those found in the trench samples, 805 versus 558 mg TKN-N/kg dry soil. In both analyses, the trench and control concentrations were grouped by site because both were dependent on the site location. The grouping accounts for variation between sites.

The soil type present around an ISF system site was determined from the original site assessment. The different soil types, Table 4, were compared for any differences in the concentration of nitrogen (Table 5a and 5b). TKN concentration was found to be influenced by soil type (p=0.0001) (Table 6a). The difference between soils was due to the silt loam. This finding was based on one silt loam sample and therefore we can make no statements about how representative this data point may be.

The nitrate concentration was found to be influenced by the soil type as well as the site within the soil type (type III SS). Soil type was the predominant influence. The p-values for both soil type and site within soil type are 0.0001. Differences in nitrate concentration occur between the sandy loam and all other soil types ($p < 0.05$). The silt loam was found to differ from all other soil types ($p<0.05$), but this difference is based on one silt loam site and this conclusion should be considered with caution (Table 6b). The clay and clay loam were statistically different from each other as well ($p = 0.014$). All other soils were not statistically different from one another. Taken as a whole, most soils containing clay did not differ from one another ($p>0.05$).

Table 4. Number of Sites Within Each Soil Type.

| Soil Type | Number of sites |
|---|---|
| Sandy Loam | 5 |
| Silt | 1 |
| Silt Loam | 1 |
| Silt Clay | 15 |
| Silt Clay Loam | 3 |
| Clay Loam | 5 |
| Clay | 14 |

Table 5a. TKN: Statistical Model Assessing Differences in Soil Type.

| Source | DF | Sum of Squares | Mean Square | F-value | p-value |
|---|---|---|---|---|---|
| Model | 40 | 9598419 | 239960 | 5.05 | 0.0001 |
| Error | 80 | 3802733 | 47534 | | |
| Corrected Total | 120 | 13401152 | | | |
| | | Type III SS | Mean Square | F-value | p-value |
| Soil Type | 6 | 1849618 | 308270 | 6.49 | 0.0001 |
| Site (within Soil Type) | 34 | 7748800 | 227906 | 4.79 | 0.0001 |

Table 5b. $NO_3$: Statistical Model Assessing Differences in Soil Type.

| Source | DF | Sum of Squares | Mean Square | F-value | p-value |
|---|---|---|---|---|---|
| Model | 40 | 74531 | 1863 | 5.80 | 0.0001 |
| Error | 78 | 25052 | 321 | | |
| Corrected Total | 118 | 99584 | | | |
| | | Type III SS | Mean Square | F-value | p-value |
| Soil Type | 6 | 24204 | 4034 | 12.6 | 0.0001 |
| Site (with in Soil Type) | 34 | 46020 | 1354 | 4.21 | 0.0001 |

Table 6a. P-Values Explaining Differences in TKN Concentrations Between Soil Types

| ** | Sandy Loam | Silt | Silt Loam | Silt Clay | Silty Clay Loam | Clay Loam | Clay |
|---|---|---|---|---|---|---|---|
| Sandy Loam | | 0.341 | 0.0001 | 0.482 | 0.216 | 0.334 | 0.651 |
| Silt | 0.341 | | 0.0001 | 0.513 | 0.092 | 0.693 | 0.443 |
| Silt Loam | 0.0001 | 0.0001 | | 0.0001 | 0.0003 | 0.0001 | 0.0001 |
| Silt Clay | 0.482 | 0.513 | 0.0001 | | 0.066 | 0.637 | 0.758 |
| Silty Clay Loam | 0.216 | 0.092 | 0.0003 | 0.066 | | 0.051 | 0.097 |
| Clay Loam | 0.334 | 0.693 | 0.0001 | 0.637 | 0.051 | | 0.493 |
| Clay | 0.651 | 0.443 | 0.0001 | 0.758 | 0.097 | 0.493 | |

**A p-value less than 0.05 indicates a statistical difference between the nitrate concentrations of the two soils being compared.

Table 6b. P-Values Explaining Differences in Nitrate Concentration Between Soil Types.

| ** | Sandy Loam | Silt | Silt Loam | Silt Clay | Silty Clay Loam | Clay Loam | Clay |
|---|---|---|---|---|---|---|---|
| Sandy Loam | | 0.0007 | 0.0102 | 0.0001 | 0.0002 | 0.0018 | 0.0001 |
| Silt | 0.0007 | | 0.0001 | 0.418 | 0.632 | 0.106 | 0.677 |
| Silt Loam | 0.010 | 0.0001 | | 0.0001 | 0.0001 | 0.0001 | 0.0001 |
| Silt Clay | 0.0001 | 0.418 | 0.0001 | | 0.738 | 0.073 | 0.319 |
| Silty Clay Loam | 0.0002 | 0.632 | 0.0001 | 0.738 | | 0.155 | 0.845 |
| Clay Loam | 0.002 | 0.106 | 0.0001 | 0.073 | 0.155 | | 0.014 |
| Clay | 0.0001 | 0.677 | 0.0001 | 0.319 | 0.845 | 0.014 | |

**A p-value less than 0.05 indicates a statistical difference between the nitrate concentrations of the two soils being compared.

In order to determine if the nitrogen concentration in the trench soil samples differed from the nitrogen concentration in the distribution box, a contrast statement was used to compare the nitrogen concentration in the soil samples as one group to the nitrogen concentration in the distribution box as another group. The results of this analysis indicate that the nitrate concentration in the soil samples taken at an average of 30 inches below the ground surface adjacent to the trench did not differ from the nitrate concentrations of the effluent in the distribution box ($p = 0.240$).

When an analysis to determine any difference between the TKN in the disposal trench soil samples and the distribution box was performed, a statistical difference, $p = 0.0001$, was found. This indicates a difference between the two sample points. The difference is most likely due to the high background concentrations naturally occurring in the soil.

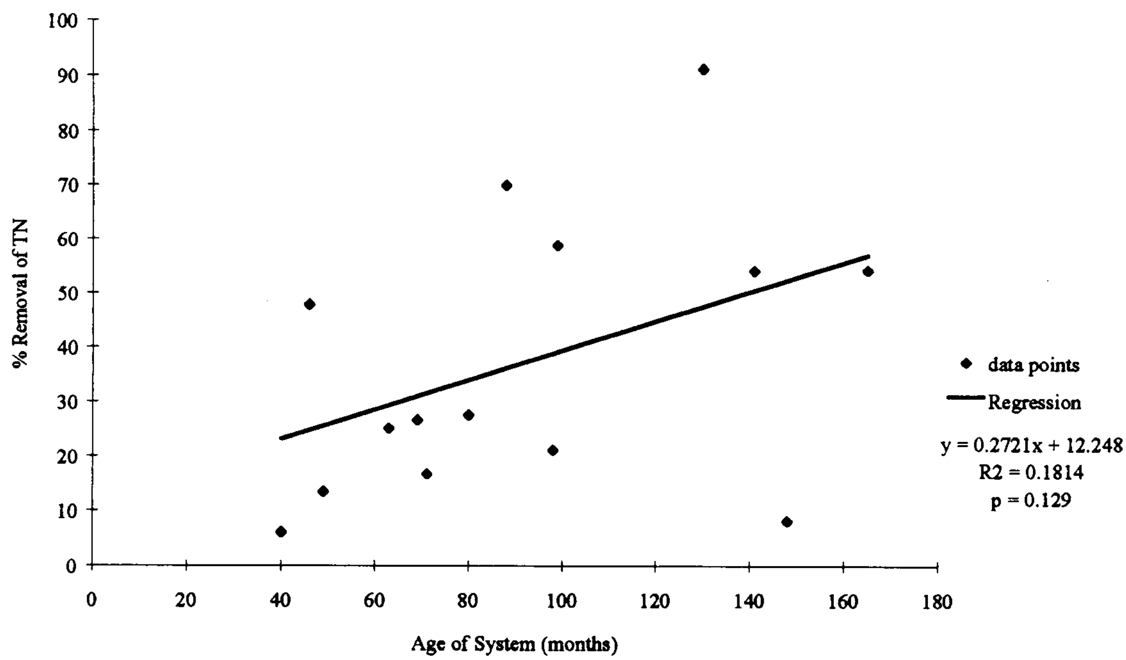
Two methods were used to develop an evaluation model describing the relationship between age of sand filter and percent of total nitrogen (TN) removed through the sand filter. These methods were the stepwise regression model and a linear regression. The stepwise method introduces one variable at a time into the model until the model with the "best fit" is found. Linear regression uses the variables chosen by the user to produce an equation that best represents the data trend.

When all the sites with both septic tank and distribution box data are used, both methods excluded all variables except age (i.e. number of people at the residence using the system, number of bathrooms in the home, septic tank TKN concentrations). Each method generated the same linear model, with only age as a dependent variable. The model (Table 7) produced by both methods was not statistically significant ($p = 0.129$), nor were the coefficients within the model. The individual system performances are plotted against the linear regression in Figure 4.

Table 7. Statistical Model Relating System Age to System Percent Removal of TN. (All Data, $n = 14$)

| Source | DF | Sum of Squares | Mean Square | F-value | p-value |
|---|---|---|---|---|---|
| Regression | 1 | 1557 | 1557 | 2.66 | 0.129 |
| Error | 12 | 7030 | 586 | | |
| Total | 13 | 8587 | | | |
| | Parameter Estimate | Standard Error | | F-value | p-value |
| Intercept | 12.2 | 16.6 | | 0.54 | 0.476 |
| Age | 0.27 | 0.17 | | 2.66 | 0.129 |

Figure 4. The Regression of Age Versus Percent Removal of Total Nitrogen (All Data, n = 14)



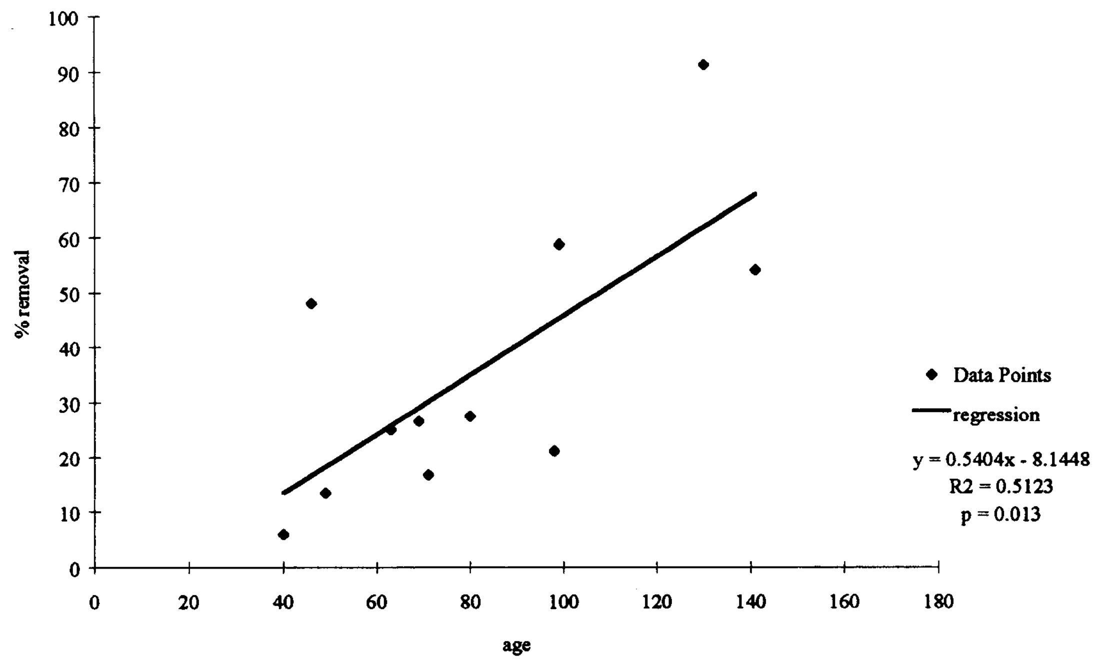

The ISF is designed to be an aerobic treatment unit thus the ammonia present should be nitrified, thus the TKN concentration in the distribution box should be low. In order to determine the average nitrogen transformations that take place in a properly operation system, it is desirerable to elimate all systems that are failing. In order to do this, a clear criteria was needed to define a failing system based on its performance. If a system is functioning properly, it should have nitrified greater than 90% of the total nitrogen leaving the filter. In a discussion with Greg Farrell, ODEQ Western Region On-Site Manger, Harold Ball and Terry Bound at ORENCO Systems Inc., Sutherlin, Oregon, it was decided that a system would be considered failing if the distribution box TKN concentration was greater than 5.0 mg TKN-N/ L. This value is based on the concentrations found in healthy systems from two previous studies (Cagle and Johnson, 1994; Paeth and Ronayne, 1984). The term "failing" will be used to denote the systems that had a TKN concentration greater than 5.0 mg TKN-N/ L in the distribution box.

The classification of failing systems resulted in the removal of three systems. The TKN values of the three systems defined as failing were 9.1, 11.0, and 45.0 mg TKN-N/L. The linear regression was rerun using only the systems that were classified as not failing, resulting in a statistically significant relationship (p = 0.013) (Table 8). The regression is plotted against the data in Figure 5.

Table 8. Statistical Model Relating System Age to System Percent Removal of TN. (Non-Failing Systems, n = 11)

| Source | DF | Sum of Squares | Mean Square | F-value | p-value |
|---|---|---|---|---|---|
| Regression | 1 | 3245 | 3245 | 9.5 | 0.013 |
| Error | 9 | 3090 | 343 | | |
| Total | 10 | 6335 | | | |
| | Parameter Estimate | Standard Error | Type II SS | F-value | p-value |
| Intercept | -8.15 | 15.22 | | -0.54 | 0.606 |
| Age | 0.54 | 0.18 | | 3.07 | 0.013 |

Figure 5. The Regression of Age Versus Percent Removal of Total Nitrogen
(All Non-Failing Systems, n = 11)




A statistical exercise was undertaken to determine if the concentration of TKN and nitrate in the septic tank could be predicted from different factors such as the age of the system, the number of people in the home, the number of bathrooms in the home, and the concentration of the other nitrogen compound (ie. is the septic tank nitrate concentration dependent on the TKN concentration). The results of this exercise were found to be statisitically significant, but did not make sense or appear to add to the overall knowledge of ISF systems. One of the variables in the model, number of bathrooms, resulted in a negative coefficient, this does not coincide with existing knowledge of factors affecting septic tank concentrations. As this analysis was conducted, but of little value to the objectives of this study, the analysis results have been included in Appendix 6.

# DISCUSSION

The soils associated with a septic tank - drainfield trench system are often thought to contribute to the treatment of the effluent by further reducing and oxidizing the nitrogen compounds. This was not substantiated in the study. The nitrate concentration in the soil samples from the disposal trench were not statistically different from the nitrate concentration in the distribution box. No difference could be found when the data were grouped by site, or by soil type. This would indicate that limited oxidation or reduction of the nitrate nitrogen occurs at the sampling depth of 30 inches, no matter what soil type is present. One possible explanation is a lack of sufficient organic matter (carbon) to facilitate the reduction of the nitrate nitrogen.

The sand filter effluent adds to the soil's nitrate content. The trench soil was found to be higher in nitrate than the surrounding soil, approximately three times as great. The higher concentration is expected since 94% of the nitrogen in the sand filter's effluent is in the nitrate form. This finding is consistent with an earlier study which found 96% of the nitrogen in the nitrate form (Paeth and Ronayne, 1984). The high conversion of organic nitrogen and ammonia to nitrate is an indication of a properly functioning ISF system. The average nitrate concentration in the distribution box, 26.2 mg $NO_3$-N/L, is also similar to that reported in earlier studies of sand filter systems (Cagle and Johnson, 1994; Paeth and Ronayne 1984).

The overall total nitrogen percent removal is only slightly less than what was previously found in the original ODEQ study (1984). The reduction based on the average TN was 43% in the current study, versus the original study's 47% removal of total nitrogen. The removal rate is less if one looks at the average of the site specific removals of TN. The difference may be accounted for in the length of time the sand filters were in operation, as well as homeowner knowledge of the study. It is possible that homeowners, aware of the 1984 study's monitoring, may have been more cautious in their use of a garbage disposal or just general use practices.

The Total Kjeldahl Nitrogen (TKN) in the trench soil samples was found to be statistically lower than the background samples. The utilization of the organic nitrogen, TKN, within the trench environment is not surprising. The trench environment is well suited to a large microbial community. The organic nitrogen within the soil is apparently used as either an energy source or a source for cell growth. Within the trench environment, the nitrogen will be eventually converted to the nitrate form. This is part of the natural cycling of nitrogen and has been substantiated by many studies (Cochet et al., 1990; Starr and Sawhney,1980; Walker et al., 1973; Whelan and Barrow,1984). It would appear that the organic forms of nitrogen within the trench have been broken down to a soluble form of nitrogen, such as nitrate, and moved with the water away from the sampling area near the trench. This would explain the lower TKN concentrations within the trench environment. Outside of the disposal field area, the environment is not as well suited to maintain a large and active microbial population. Knowledge of the TKN background concentration did not allow for the determination of what is entering the trench from the septic system effluent. Because the background sample was larger, it could not be subtracted from the trench concentration to give a value that is due only to the effluent from the trench.

Soil type was found to be an important factor in accounting for variance between samples' nitrate and TKN concentrations. In part, the soil type may explain the background levels of nitrate and TKN in the soil and the potential for different microbial processes. The sandy loam soils found in Lincoln County are more likely to have fast percolation rates, and, therefore, less opportunity for the establishment of anaerobic zones. The sandy loam soils were found to be different with respect to nitrate from all the other soils sampled. The concentration of nitrate in the trench samples was generally higher than that found in the other soils; yet, the background concentrations are not higher. The sandy loam environment is more likely to be aerobic than the soils containing clay, therefore less likely to have any denitrification occuring. This could account for the difference in nitrate concentrations.

A statistically significant relationship between age of the filter and filter performance based on the percent removal of TN could not be found using all the available data. A general linear trend could be fitted to the data, but not well. The average site specific percent removal of TN using all the data was 37%. If only the systems that were classified as not failing (TKN > 5.0 mg N/L) were used in the regression, a statistically significant relationship was found, and the average site specific TN removal is 35%. It should be noted that a large range of TN removals was found even in the systems that were classifed as not failing. A range of 6.0 to 91% was found. The system having a 6.0% removal had a high ISF effluent nitrate level and a TKN concentration less than 5.0 mg N/L. Of the systems that were classified as failing, one had a TN removal of 70%.

As the systems aged, the performance begins to scatter, this is seen in both regression analyses. Some systems achieved very high levels of total nitrogen removal (i.e. 91% removal in an 11 year old system), and some systems were classified as failing (i.e. 8% removal in a 13 year old system). In addition, the systems had a tendency to improve their removal of TN with age. The slope from the linear regression with only non-failing systems has a positive slope, indicating the increase in percent removal of TN with an increase in age.

In total, the concentrations of nitrate and TKN found in the septic tank effluent and sand filter effluent in this study were similar to that found in the original ODEQ (1984) study. Table 9 shows the results from this study contrasted with the results from the original study. The number that is different between the studies are the sand filter effluent's TKN. The TKN is either not being converted to other forms of nitrogen, which are then removed, or not physically filtered. This would indicate that some of the systems are not functioning as designed based on the criteria for failing systems of TKN > 5.0 mg TKN-N/ L in the distribution box.

Table 9. Comparison of Nitrogen Concentrations From the Original Study by Paeth and Ronayne in 1984 and This Study.

| | Paeth and Ronayne, 1984 | n | Bushman, 1995 | n |
|---|---|---|---|---|
| Septic Tank TKN* | 57.1 | 8 | 59.4 | 29 |
| Septic Tank $NO_3$* | 0.4 | 8 | 0.02 | 29 |
| Septic Tank TN* | 57.5 | 8 | 59.4 | 29 |
| Distribution Box TKN* | 1.7 | 7 | 7.5 | 15 |
| Distribution Box $NO_3$* | 29.1 | 7 | 26.2 | 15 |
| Distribution Box TN* | 30.3 | 7 | 33.7 | 15 |
| % Removal of TN Based on Averages | 47% | 8/7 | 43% | 29/15 |

* all concentrations are in mg of $NO_3$-N/ L or mg of TKN-N/ L

# CONCLUSION

This study proposed to determine the transformations of nitrogen compounds through intermittent sand filters (ISF), and to determine if the disposal field contributes to any further treatment of the effluent nitrate. During the summer of 1995, a total of 44 intermittent sand filter septic systems, in five counties of Western Oregon were sampled over a 3 month period. The sand filter systems varied in age from 36 months to 167 months (3 to 13.9 years). Liquid samples were taken from the septic tank (pre-ISF) and distribution box (post-ISF), as well as soil samples adjacent to the disposal trench and away from the disposal field area (background). All samples were analyzed for Total Kjeldahl Nitrogen and nitrate and nitrite. Nitrite was not detected in any of the samples.

The intermittent sand filter is a useful treatment unit for sites that would otherwise be denied a standard septic system. The average reduction of total nitrogen through an intermittent sand filter is 43%, with the remaining nitrogen predominantly in the form of nitrate. Some systems were found to have a high nitrogen removal rate (91%), and some had removal rates of nitrogen through the intermittent sand filter lower than the expected 47% from the 1984 ODEQ study. Once the nitrate enters the soil environment, there is only a slight chance that denitrification will occur to further reduce the nitrogen, this can be seen in the nitrogen removal from a standard septic system. The expected removal of nitrogen from a conventional disposal trench septic system is 20% (Siegrist and Jenssen, 1989), with a range of 0 to 35% (Eastburn and Ritter, 1984). Based on this range of expected nitrogen removal in the disposal trench, any further reduction of nitrogen after the intermittent sand filter is unpredictable for systems serving single family residents.

Sandy soils are the least likely to have any reduction of the nitrate due to a lack of sufficient anoxic zones and an available carbon source. The soils with a high clay content are more likely to develop anaerobic zones. The organic matter in clay soils, which is greater than that of sandy soils, may not be a sufficient energy source for the denitrifying microbes. The available carbon is supplied in the soil's organic matter or from the ISF effluent. Since the ISF is known to remove most of the organic matter, the ISF effluent is a small source of carbon. Therefore, less denitrification will take place in an ISF disposal trench compared to a standard disposal trench. The use of the sand filter will in most cases guarantee some removal of nitrogen.

The ISF systems were found to be functioning properly hydraulically. Only 1 of 44 sites sampled was failing hydraulically (standing water). This means that 98% of the sites were functioning without an apparent problem. The allowance of a shortened lenght of disposal trench appears to be validated.

The data indicate that a systems percent removal of TN increases with increased age. The linear regression which resulted from the non-failing sites' data, has a positive slope. The increased performance with age may be due to an enhanced microbial population, an increase in available organic matter for denitrification, or a development of anaerobic sites. As the ISF systems increase in age, there is also an increase in the variance between site performance. As the systems aged, some sites performed at a very high level, while others did not. When the systems are younger, less than 100 months, the TN removals have a greater tendency to be more similar in their performance.

The intermittent sand filter is a means of nitrogen transformation and reduction for septic systems. A given level of nitrogen loss through a sand filter can be expected, and may be sustained. It is unlikely that an ISF by itself will prevent nitrate contamination in groundwater, but it will reduce the total amount of nitrate entering the soil from septic systems. The continued use of ISF systems is warranted. The ISF systems have been found to remove an average of 35% total nitrogen through the filter, which is 35% better than a standard system.

One important aspect of this study is that it examines ISFs under actual operating and maintenence conditions imposed by the use of regular families. The systems in this study were not operated under experimental conditions as were the systems in the original ODEQ study. The results of this study are therefore more representative of ISF treatment from systems that have been installed and are currently in use in Oregon.

# RECOMMENDATIONS

One means of monitoring sand filter systems would be to sample effluent from the distribution box on an annual basis to determine the performance of the sand filter. The monitoring could sample and analyze the sand filter effluent for TKN to determine if the filter is aerobic. The TKN > 5.0 mg TKN-N/ L criteria could be used to determine if a filter is failing with respect to nitrogen transformations. If a filter was found to be failing, the septic tank should be checked to determine if pumping is needed, and the pressure manifold distribution piping may need to be cleaned to open clogged orifaces. Either of these may cause a build up of solids in the filter, or the creation of a biomat. A regular monitoring schedule would allow homeowners and regulatory agencies, to make decisions regarding the system based on its performance, and not just its appearance.

The removal of nitrogen through an ISF system would be greatly increased if a denitrification step was added after the ISF, before the disposal trench system. This study, and other studies, (Cagle and Johnson, 1994; Paeth and Ronayne, 1984) have shown that over 90% of the ISF effluent nitrogen is in the form of nitrate. A denitrification step, containing a sufficient carbon source and an anoxic environment, would decrease the nitrate concentration even further. One possible method that could be implimented is the use of shallow drainfields (Ball, 1995). The shallow drainfields take advantage of the soil's organic matter and is better suited to allow vegitation to utilized the available nitrogen. This method may be limited by the ability of the soil to sustain a high organic matter content. Another means of denitrification is to use a recirculating filter, or separate gray and black water, using the grey water after the filter as a carbon source.

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# APPENDICES

# APPENDIX 1:

## Reactions of Nitrogen and the Nitrogen Cycle

## A. REACTIONS OF NITROGEN AND DEFINITIONS

NITRIFICATION: The microbially mediated conversion of ammonium to nitrite to nitrate.

Step One: Nitrosomonas Bacteria (Ammonium to Nitrite)

$$2NH_4^+ + 3O_2 \longrightarrow 2NO_2^- + 2H_2O + 4H^+$$

Step Two: Nitrobacter Bacteria (Nitrite to Nitrate)

$$2NO_2^- + O_2 \longrightarrow 2NO_3^-$$

DENITRIFICATION: The microbially mediated conversion of nitrate to nitrogen gas.

$$C_6H_{12}O_6 + 4NO_3^- \longrightarrow 6CO_2 + H_2O + 2N_2$$

MINERALIZATION: Conversion of organic nitrogen to ammonium ($NH_4^+$)

IMMOBILIZATION: Conversion of ammonium to organic nitrogen.

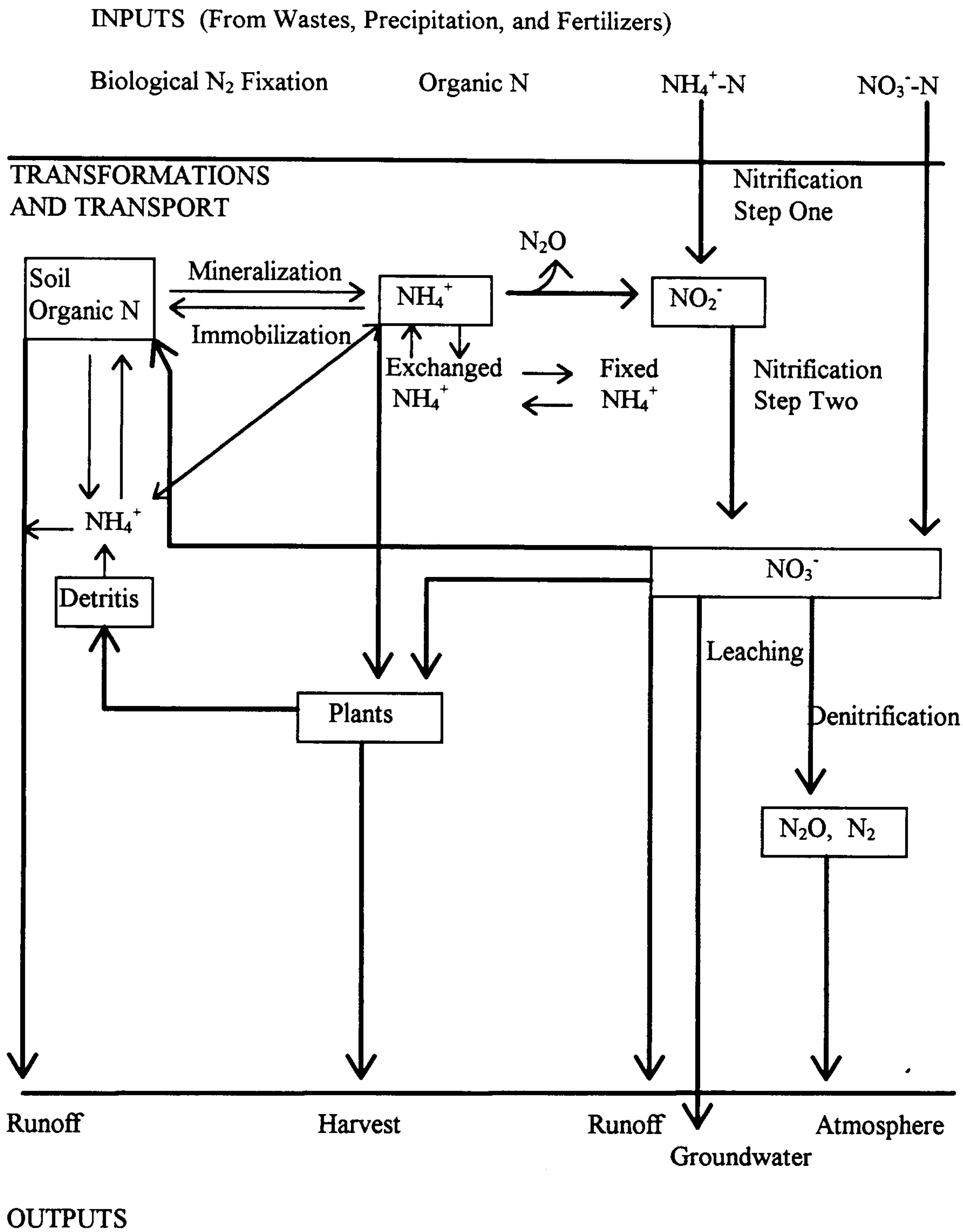
FIXATION: Conversion of nitrogen gas to ammonium, facilitated by microbes and plants.

$$N_2 \longrightarrow NH_4^+$$

EXCHANGE: The exchange of ammonia for other cations in the soil.

DETRITUS: Dead plant material.

## B. NITROGEN CYCLE



# APPENDIX 2:

## Sample of Informational Letter Sent to Homeowners

Dear Homeowner,

It has been brought to our attention that a study involving sand filters will be conducted this summer by a graduate student from Oregon State University. The student, Jennifer Bushman, will be working as an intern with the Department of Environmental Quality. The study will assess the effectiveness of the long term usage of sand filters.

Since their approval in 1980, sand filters have not been monitored for their continued usage in the field. This study will provide information on the transformations and removal of nitrogen through the system.

Jennifer will be calling to ask for your permission to come onto your property and take soil samples. The sampling will consist of 3 soil cores taken along the disposal field trenches. Tha sampling should not take more than a half day if all goes well. Great care will be taken to minimize the disturbance to your yard.

The participation in this study is purely voluntary, and no repercussions will result from any information regarding your site. And there will be no cost to you.

In addition, for those greatly interested and willing, a more intense sampling may be done. This would consist of taking samples from the septic tank and disposal field distribution box. This would mean a larger excavation of the property. Again, great care will be taken to minimize the disturbance, but inevitably some will result. The sampling would require gaining access to both the septic tank and distribution box. If both the septic tank access and distribution box are buried, a large enough hole will need to be dug to remove the lids and obtain samples. The soil and grass will be replaced from the approximately 3 ft by 3 ft holes. This will only be done on the sites that are willing. All others will only have 3 inch cores taken, resulting in very little disturbance.

Again we do encourage your voluntary participation.

Thank you for your time and consideration.

Sincerely,

County Officer

# APPENDIX 3:
# SAMPLE QUESTIONNAIRE

Site location ______

Site owner ______

Phone ______

Address ______

______

Soil type? ______

Size of septic tank ______

Length of drain line ______

When was the system installed? ______

______

How many people occupy the household? ______

for how long? ______

children? ______

______

How many bathrooms are in the house? ______

How many bedrooms are in the house? ______

Does the kitchen have a garbage grinder? ______

Have you encountered any problems with standing water on the drain tile field?

______

Have you encountered any problems with the sand filter?

______

How long have you lived in the house? ______

Have you ever used any type of additive? ______

if yes, what type? ______

why? ______

______

Has the property changed hands since that time?

______

What type of vegetation covers the system? ______

Additional Information:

# APPENDIX 4:

## Data

Legend:

1. TKN = Total Kjeldahl Nitrogen which is a measure of the organic nitrogen and ammonia
2. TN = total nitrogen (TKN + nitrate)
3. % removal = (TN of septic tank - TN of dist. box) divided by the TN of the septic tank
4. The numbers 1,2,and 3 indicate the position along the trench, the C indicates it is a background sample (ie. soil sample nitrate-2 concentration is the soil sample taken at the second position along the trench)
5. All concentrations are in mg/L
6. A value of 0 indicates a concentration below the detection limit. A value of **n** indicates no sample was taken.
7. The detection limit for nitrate is 0.02 mg $NO_3$-N/L, TKN is 0.2 mg TKN-N/L

Soil Type Key

| | |
|---|---|
| 1 | Sandy Loam |
| 2 | Silt |
| 3 | Silt Loam |
| 4 | Silt Clay |
| 5 | Silt Clay Loam |
| 6 | Clay Loam |
| 7 | Clay |

| House Number | age (mo) | soil type | Septic Tank nitrate | Septic Tank TKN | Distribution Box nitrate | Distribution Box TKN | Soil Samples (nitrate = trench - background) nitrate-1 | nitrate-2 | nitrate-3 | nitrate-C | TNK-1 |
|---|---|---|---|---|---|---|---|---|---|---|---|
| 1B | 142 | 4 | n | n | n | n | 33.28 | 24.19 | 20.06 | 7.00 | 180 |
| 2B | 165 | 6 | 0.02 | 30.00 | 2.70 | 11.00 | 8.92 | 7.02 | 17.70 | 2.60 | 360 |
| 3B | 47 | 7 | 0.00 | 25.00 | n | n | 3.63 | 4.04 | 2.37 | 1.50 | 410 |
| 4B | 63 | 7 | 0.00 | 41.00 | 28.00 | 2.70 | 46.90 | n | n | 1.50 | 140 |
| 5B | 130 | 5 | 0.00 | 64.00 | 5.50 | 0.00 | 44.83 | 5.73 | -2.60 | 2.60 | 970 |
| 6B | 108 | 4 | n | n | n | n | 7.90 | 26.92 | 19.05 | 7.00 | 250 |
| 7B | 73 | 7 | 0.00 | 48.00 | n | n | 16.49 | 1.10 | 15.28 | 1.50 | 540 |
| 8B | 156 | 4 | n | n | n | n | -5.62 | -5.14 | -2.79 | 7.00 | 1100 |
| 9B | 148 | 5 | 0.02 | 49.00 | 0.03 | 45.00 | n | n | n | 2.60 | n |
| 10B | 83 | 6 | n | n | n | n | 8.22 | 4.89 | 5.82 | 4.66 | 610 |
| 11B | 88 | 6 | 0.00 | 36.00 | 1.70 | 9.10 | 33.02 | 3.68 | 22.51 | 8.82 | 580 |
| 1L | 141 | 7 | 0.03 | 140.00 | 61.00 | 3.10 | 26.01 | 0.78 | -2.31 | 2.31 | 450 |
| 2L | 134 | 5 | n | n | n | n | 1.40 | 16.00 | 0.55 | 2.58 | 780 |
| 3L | 46 | 7 | 0.00 | 54.00 | 27.00 | 1.10 | 0.00 | 0.00 | 4.25 | 0.00 | 230 |
| 4L | 104 | 7 | 0.04 | 64.00 | n | n | 24.12 | 30.22 | 0.00 | 0.00 | 320 |
| 5L | 42 | 7 | n | n | n | n | 0.00 | 0.00 | 0.00 | 0.00 | 750 |
| 6L | 49 | 4 | 0.04 | 71.00 | n | n | 1.98 | -4.92 | -3.28 | 4.92 | 380 |
| 7L | 59 | 4 | 0.00 | 60.00 | n | n | 4.47 | 5.58 | -4.41 | 7.14 | 740 |
| 8L | 69 | 4 | 0.02 | 69.00 | 48.00 | 2.60 | 44.32 | 0.82 | 9.18 | 6.96 | 960 |
| 9L | 120 | 7 | 0.02 | 86.00 | n | n | -7.26 | -7.26 | -7.26 | 7.26 | 880 |
| 10L | 44 | 7 | n | n | n | n | 1.67 | -1.50 | -3.09 | 3.09 | 380 |
| 11L | 132 | 7 | n | n | 0.00 | 27.00 | 9.86 | 2.91 | 0.00 | 0.00 | 1000 |
| 12L | 51 | 4 | 0.15 | 61.00 | n | n | 0.00 | 1.82 | 0.00 | 0.00 | 710 |

| House Number | age (mo) | soil type | Septic Tank nitrate | Septic Tank TKN | Distribution Box nitrate | Distribution Box TKN | NIT-1 | NIT -2 | NIT-3 | NIT-C | TKN-1 |
|---|---|---|---|---|---|---|---|---|---|---|---|
| 1Li | 35 | 1 | n | n | n | n | 0.00 | 1.69 | n | 0.00 | 360 |
| 2Li | 145 | 1 | 0.04 | 100.00 | n | n | 94.37 | 159.39 | 90.29 | 6.90 | 120 |
| 3Li | 76 | 1 | 0.02 | 100.00 | n | n | 120.79 | 31.28 | 51.16 | 3.11 | 480 |
| 4Li | 140 | 1 | 0.00 | 26.00 | n | n | 0.00 | 2.78 | n | 0.00 | 130 |
| 5Li | 36 | 1 | n | n | n | n | 27.09 | 48.61 | 28.51 | 0.00 | 580 |
| 1C | 80 | 3 | 0.00 | 41.00 | 29.00 | 0.70 | 85.51 | 82.82 | 58.44 | 14.24 | 930 |
| 2C | 39 | 4 | 0.00 | 67.00 | n | n | 51.50 | 48.58 | 43.48 | 0.00 | 210 |
| 3C | 39 | 4 | n | n | n | n | -9.20 | 14.98 | 54.09 | 13.19 | 430 |
| 4C | 74 | 4 | 0.02 | 55.00 | n | n | 66.71 | 55.79 | -7.58 | 7.58 | 100 |
| 5C | 59 | 4 | 0.00 | 36.00 | n | n | -9.53 | -8.09 | -15.54 | 15.54 | 320 |
| 6C | 61 | 7 | n | n | n | n | -2.31 | 10.14 | -2.31 | 2.31 | 500 |
| 7C | 167 | 2 | n | n | n | n | 3.94 | 0.00 | 12.05 | 0.00 | 580 |
| 8C | 40 | 4 | 0.03 | 42.00 | 38.00 | 1.50 | n | n | n | 7.00 | n |
| 1D | 98 | 7 | 0.00 | 50.00 | 39.00 | 0.40 | n | n | n | 1.50 | n |
| 2D | 71 | 6 | 0.03 | 58.00 | 47.00 | 1.30 | 4.20 | 29.69 | 15.98 | 0.00 | 400 |
| 3D | 131 | 7 | n | n | n | n | -30.29 | 106.19 | -52.22 | 52.22 | 120 |
| 4D | 85 | 6 | 0.00 | 84.00 | n | n | 56.28 | 76.35 | 63.40 | 5.17 | 520 |
| 6D | 101 | 4 | 0.17 | 46.00 | n | n | -48.53 | -69.45 | -71.48 | 103.85 | 610 |
| 7D | 49 | 4 | 0.00 | 52.00 | 42.00 | 3.00 | 25.59 | 15.11 | 8.39 | 0.00 | 260 |
| 8D | 83 | 7 | n | n | n | n | 3.60 | 2.50 | 2.77 | 0.00 | 1040 |
| 9D | 99 | 4 | 0.05 | 67.00 | 24.00 | 3.60 | -2.21 | 1.44 | 4.62 | 14.66 | 1040 |

| House Number | TKN-2 | TKN-3 | TKN-C | nitrate-1 | nitrate-2 | Liquid Samples nitrate-3 | TNK-1 | TKN-2 | TKN-3 | % Removal of TN |
|---|---|---|---|---|---|---|---|---|---|---|
| 1B | 230 | 190 | n | n | n | n | n | n | n | n |
| 2B | 320 | 310 | n | n | n | n | n | n | n | 54.33 |
| 3B | 390 | 1000 | n | 0.76 | 0.12 | n | 0.60 | 5.50 | n | n |
| 4B | n | n | n | n | n | n | n | n | n | 25.12 |
| 5B | 560 | 610 | n | 49.00 | n | n | 3.60 | n | n | 91.41 |
| 6B | 290 | 280 | n | n | n | n | n | n | n | n |
| 7B | 650 | 770 | n | n | n | n | n | n | n | n |
| 8B | 480 | 900 | n | n | n | n | n | n | n | n |
| 9B | n | n | n | n | n | n | n | n | n | 8.10 |
| 10B | 380 | 390 | n | 38.00 | n | n | 1.40 | n | n | n |
| 11B | 930 | 600 | 1800 | n | n | n | n | n | n | 70.00 |
| 1L | 300 | 70 | 260 | n | n | n | n | n | n | 54.21 |
| 2L | 780 | 480 | 530 | 0.10 | 0.16 | 0.08 | 11.00 | 2.50 | 7.90 | n |
| 3L | 200 | 170 | 700 | 8.60 | n | n | 2.10 | n | n | 47.96 |
| 4L | 520 | 310 | 440 | n | n | n | n | n | n | n |
| 5L | 780 | 500 | 390 | n | n | n | n | n | n | n |
| 6L | 230 | 370 | 1800 | 55.00 | n | n | 0.80 | n | n | n |
| 7L | 1200 | 700 | 1200 | n | n | n | n | n | n | n |
| 8L | 550 | 1300 | 1300 | n | n | n | n | n | n | 26.67 |
| 9L | 320 | 430 | 1600 | n | n | n | n | n | n | n |
| 10L | 360 | 290 | 1700 | n | n | n | n | n | n | n |
| 11L | 1030 | 1200 | 710 | n | n | n | n | n | n | n |
| 12L | 290 | 340 | 400 | n | n | n | n | n | n | n |

| House | | | | | | Liquid Samples | | | | % Removal |
|---|---|---|---|---|---|---|---|---|---|---|
| Number | TKN-2 | TKN-3 | TKN- | nitrate-1 | nitrate-2 | nitrate-3 | TNK-1 | TKN-2 | TKN-3 | of TN |
| 1Li | 1500 | 1300 | 670 | n | n | n | n | n | n | n |
| 2Li | 840 | 1000 | 1200 | n | n | n | n | n | n | n |
| 3Li | 560 | 400 | 690 | n | n | n | n | n | n | n |
| 4Li | 370 | 220 | 540 | n | n | n | n | n | n | n |
| 5Li | 330 | 290 | 400 | n | n | 22.00 | n | n | 2.70 | n |
| 1C | 1600 | 1300 | 1300 | n | n | n | n | n | n | 27.56 |
| 2C | 230 | 240 | 420 | n | n | n | n | n | n | n |
| 3C | 410 | 380 | 500 | n | n | n | n | n | n | n |
| 4C | 250 | 600 | 450 | 2.00 | 2.70 | 1.80 | 43.00 | 43.00 | 40.00 | n |
| 5C | 320 | 640 | 400 | n | n | n | n | n | n | n |
| 6C | 430 | 370 | 500 | n | n | n | n | n | n | n |
| 7C | 390 | 330 | 180 | 2.00 | 0.14 | 0.07 | 5.20 | 14.00 | 38.00 | n |
| 8C | n | n | n | n | n | n | n | n | n | 6.02 |
| 1D | n | n | n | n | n | n | n | n | n | 21.20 |
| 2D | 540 | 420 | 750 | n | n | n | n | n | n | 16.77 |
| 3D | 920 | 470 | 560 | n | n | n | n | n | n | n |
| 4D | 460 | 500 | 550 | n | n | n | n | n | n | n |
| 6D | 490 | 600 | 780 | n | n | n | n | n | n | n |
| 7D | 360 | 340 | 640 | n | 40.00 | 39.00 | n | 3.00 | 1.40 | 13.46 |
| 8D | 1030 | 1300 | 1100 | n | n | n | n | n | n | n |
| 9D | 1200 | 1100 | 1300 | n | n | n | n | n | n | 58.84 |

# APPENDIX 5:

## Summary Statistics

| | *ST_NIT* | *ST_TKN* | *DB_NIT* | *DB_TKN* | *NIT_HOLE1* | *NIT_HOLE2* | *NIT_HOLE3* |
|---|---|---|---|---|---|---|---|
| Mean | 0.02* | 59.4 | 26.2 | 7.5 | 21.4 | 20.6 | 14.47 |
| Standard Error | 0.01 | 4.6 | 5.2 | 3.2 | 4.7 | 5.4 | 3.61 |
| Median | 0.02 | 55.0 | 28.0 | 2.7 | 6.2 | 4.5 | 3.51 |
| Mode | 0.00 | 41.0 | #N/A | #N/A | 0.0 | 0.0 | 0.00 |
| Standard Deviation | 0.04 | 24.9 | 20.1 | 12.5 | 29.7 | 34.1 | 22.26 |
| Sample Variance | 0.00 | 619.4 | 402.9 | 155.1 | 882.5 | 1163.2 | 495.53 |
| Kurtosis | 7.83 | 2.8 | -1.2 | 6.0 | 2.7 | 6.8 | 3.00 |
| Skewness | 2.76 | 1.4 | 0.0 | 2.5 | 1.7 | 2.5 | 1.86 |
| Range | 0.17 | 115.0 | 61.0 | 45.0 | 120.8 | 159.4 | 90.29 |
| Minimum | 0.00 | 25.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.00 |
| Maximum | 0.17 | 140.0 | 61.0 | 45.0 | 120.8 | 159.4 | 90.29 |
| Sum | 0.70 | 1722.0 | 392.9 | 112.1 | 856.6 | 823.0 | 549.96 |
| Count | 29.00 | 29.0 | 15.0 | 15.0 | 40.0 | 40.0 | 38.00 |
| Confidence Level(95.0%) | 0.02 | 9.5 | 11.1 | 6.9 | 9.5 | 10.9 | 7.32 |

* This value is essentially zero

| | *TKN_HOLE3* | *CONT_TKN* | *NIT_LIQ1* | *NIT_LIQ2* | *NIT_LIQ3* | *TKN_LIQ1* | *TKN_LIQ2* |
|---|---|---|---|---|---|---|---|
| Mean | 575.3 | 805.0 | 19.4 | 8.6 | 12.6 | 8.5 | 13.6 |
| Standard Error | 56.2 | 83.1 | 8.4 | 7.9 | 7.8 | 5.1 | 7.6 |
| Median | 450.0 | 655.0 | 5.3 | 0.2 | 1.8 | 2.9 | 5.5 |
| Mode | 1300.0 | 1300.0 | 2.0 | #N/A | #N/A | #N/A | #N/A |
| Standard Deviation | 355.4 | 469.9 | 23.7 | 17.6 | 17.4 | 14.4 | 17.1 |
| Sample Variance | 126302.5 | 220838.7 | 561.7 | 308.9 | 303.9 | 206.4 | 291.4 |
| Kurtosis | -0.2 | -0.4 | -1.7 | 4.9 | -0.5 | 6.7 | 3.5 |
| Skewness | 0.9 | 0.9 | 0.7 | 2.2 | 1.1 | 2.5 | 1.9 |
| Range | 1230.0 | 1620.0 | 54.9 | 39.9 | 38.9 | 42.4 | 40.5 |
| Minimum | 70.0 | 180.0 | 0.1 | 0.1 | 0.1 | 0.6 | 2.5 |
| Maximum | 1300.0 | 1800.0 | 55.0 | 40.0 | 39.0 | 43.0 | 43.0 |
| Sum | 23010.0 | 25760.0 | 155.5 | 43.1 | 63.0 | 67.7 | 68.0 |
| Count | 40.0 | 32.0 | 8.0 | 5.0 | 5.0 | 8.0 | 5.0 |
| Confidence Level(95.0%) | 113.7 | 169.4 | 19.8 | 21.8 | 21.6 | 12.0 | 21.2 |

| | CONT NIT | TKN HOLE1 | TKN HOLE2 |
|---|---|---|---|
| Mean | 8.8 | 523.2 | 575.5 |
| Standard Error | 3.5 | 46.0 | 56.2 |
| Median | 2.8 | 480.0 | 445.0 |
| Mode | 0.0 | 580.0 | 230.0 |
| Standard Deviation | 19.9 | 294.7 | 355.4 |
| Sample Variance | 396.3 | 86862.2 | 126333.1 |
| Kurtosis | 17.9 | -0.9 | 1.2 |
| Skewness | 4.1 | 0.4 | 1.4 |
| Range | 103.8 | 1000.0 | 1400.0 |
| Minimum | 0.0 | 100.0 | 200.0 |
| Maximum | 103.8 | 1100.0 | 1600.0 |
| Sum | 281.9 | 21450.0 | 23020.0 |
| Count | 32.0 | 41.0 | 40.0 |
| Confidence Level(95.0%) | 7.2 | 93.0 | 113.7 |

| | *LIQ_TKN3* | % REMOVAL of TN |
|---|---|---|
| Mean | 18.0 | 37.3 |
| Standard Error | 8.6 | 6.9 |
| Median | 7.9 | 27.1 |
| Mode | #N/A | #N/A |
| Standard Deviation | 19.3 | 25.7 |
| Sample Variance | 373.9 | 660.5 |
| Kurtosis | -3.2 | -0.4 |
| Skewness | 0.5 | 0.7 |
| Range | 38.6 | 85.4 |
| Minimum | 1.4 | 6.0 |
| Maximum | 40.0 | 91.4 |
| Sum | 90.0 | 521.7 |
| Count | 5.0 | 14.0 |
| Confidence Level(95.0%) | 24.0 | 14.8 |

# APPENDIX 6:

# Analysis of Septic Tank Nitrogen Concentrations

Analysis of Septic Tank Nitrogen Concentrations

Two different methods were used to develop a model to predict the concentrations of nitrate and TKN in septic tank. The methods were R-squared and stepwise regression. Both methods indicate that the best model to predict the septic tank TKN concentration involves the variables: number of people at the residence using the system and number of bathrooms at the residence. The model, with a p-value of 0.0003, is as follows;

$$\underset{(9.1)}{ST\_TKN = 64.3} + \underset{(1.7)}{5.3(PEOPLE)} - \underset{(3.0)}{8.7(BATH)}$$

(Standard Error of Coefficient)

The methods produced a model for septic tank nitrate concentration containing both septic tank TKN concentration and the number of bathrooms. The model has an overall p-value of 0.0008, and is as follows:

$$\underset{(0.02)}{ST\_NIT = -0.04} + \underset{(0.0002)}{0.0004(ST\_TKN)} + \underset{(0.005)}{0.02(BATH)}$$

(Standard Error of Coefficients)

The model for the nitrate concentration predicts very small values. Because the nitrate concentrations in the septic tank are essentially zero this model is not very useful.