# NITRIFICATION OF LANDFILL LEACHATE BY BIOFILM COLUMNS

by

Matthew M. Clabaugh

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APPROVED:

J.T. Novak, Chairman

C.D. Goldsmith

C.W. Randall

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Blacksburg, Virginia

**Keywords**: Landfill, Nitrification, Biofilm, Packing Media, Leachate, Recirculation

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**(ABSTRACT)**

Landfill leachate characteristics vary depending on the operation type of the landfill and the age of the landfill. At landfills operated as bioreactors, where leachate recirculation is practiced, leachate ammonia nitrogen concentrations may accumulate to extremely higher levels than during single pass leaching, thereby requiring treatment before final discharge to a receiving system (Onay, 1998). Usually several physical/chemical wastewater treatment technologies are used to treat the leachate. In most cases the COD and BOD are treated, and then nitrification is performed in a separate sophisticated ex situ system. The additional costs of these systems can be very high. The use of a readily available media for in situ nitrification should be considered a prime objective to avoid extra costs.

The possibility of removing ammonia nitrogen from bioreactor landfill leachate using trickling filter biofilm technology was studied in four laboratory scale reactors filled with four different types of packing media. The different packing media were examined to see which media is the most efficient at supporting ammonia removal biofilms. The highest efficiency was achieved by a packing media consisting of pine wood chips. The effects of varied concentration loading, varied hydraulic loading, and nitrification inhibitors were studied. Varied ammonia concentration did not have a huge impact on the ammonia removal rates (77-87%) in the reactor with pine wood media. The ammonia removal rates showed a strong dependence on hydraulic loading rate with the lowest loading rate producing the highest removal rates. Landfill leachate from the Middle Peninsula Landfill in Glens, Virginia was determined not to contain nitrifying inhibitors. Using a wood media filter chip and a low hydraulic loading rate was determined to be the best method to remove ammonia nitrogen from landfill bioreator leachate.

*Keywords*: bioreactor landfill, nitrification, nitrogen removal, biofilm, packing media, leachate

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# CHAPTER 1: INTRODUCTION

Land disposal of solid wastes has been practiced for centuries, dating back to prehistoric times. Municipal, industrial, agricultural, and urban activities produce huge amounts of wastes which require permanent disposal. Returning some of the solid wastes to the land is a practical approach for waste disposal. Because the human population rate increases every year, the solid waste generated increases each year (www.undp.org). As the amount of waste produced rapidly increases, space for permanent disposal becomes crucial. Since the production of solid waste is increasing much more rapidly than it degrades, land space for disposal has become more difficult and expensive to attain. There are several waste management options that can be used to reduce the amounts of waste requiring land disposal. Incineration of solid waste can be used but this is expensive and the emissions are of health concern. This is why landfills remain the major solid waste disposal option for most countries.

Solid waste in a landfill is degraded through aerobic and anaerobic processes. Stabilization of the wastes is a very complex and variable event due to the site-specific characteristics of each landfill. The degradation products generated from the stabilization process include leachate and gas. Landfill gas is generated due to the anaerobic biological degradation of organic material. Leachate is formed from the contact of water with refuse. The water, mainly from precipitation, dissolves soluble organics and inorganics including some toxic compounds if present in the landfill material.

A leachate stream can be compared to a complex wastewater stream with varying characteristics. Leachate characteristics not only vary because of the different kinds of waste present, but also vary according to the landfill age. Usually leachate from old landfills is rich in ammonia nitrogen due to the hydrolysis and fermentation of the nitrogenous fractions of the biodegradable wastes (Onay, 1998). Leachate from young landfills contains high dissolved solids content as well as high concentrations of organic matter compared to domestic wastewater (Reinhart, 1998).

Leachate is handled in two procedures by landfill operators, single pass leachate and recirculating leachate. For single pass leachate, the liquid stream is collected, stored in a lagoon or tank, and treated either on-site or off-site before discharge to a receiving system. Under the recirculation strategy, the leachate is collected and recirculated through the system by reintroducing the leachate into the landfill. Recirculation of leachate is practiced in two different types of landfills. These are the leachate recirculating landfill and the landfill bioreactor. A bioreactor is different from a recirculating landfill because a bioreactor is wetter. Landfills operated as bioreactors take water from ponds, biosolids, and other outside moisture sources and operate at high moisture contents. The main goal is to increase the moisture content inside the landfill to approximately 45%. These types of landfills result in more rapid and complete degradation of the solid waste and biological stabilization of the leachate. Compared with single pass leaching, landfill bioreactors provide more rapid, complete, and predictable conversion of readily degradable solid waste constituents, thereby enhancing the potential for gas recovery and utilization, diminishing management time, and reducing the potential for adverse health and environmental impacts, while increasing resource recovery and site reutilization opportunities (Pohland, 1995).

At landfills where leachate recirculation is practiced, leachate ammonia concentrations may accumulate to much higher levels than during conventional single pass leaching, thereby creating a leachate discharge problem (Onay, 1998). Leachates from bioreactor landfills have been known to have ammonia nitrogen concentrations to levels up to 5000 mg/l (Onay, 1995). This level is about 100 times greater than ammonia nitrogen levels usually found in municipal wastewater. This high level of ammonia can create numerous problems to the environment such as eutrophication of surface water. Other damaging impacts resulting from nitrogenous discharges include reduction of chlorine disinfection efficiency, an increase in the dissolved oxygen depletion in receiving waters, adverse public health effects, and a reduction in suitability for reuse (De Renzo, 1978). Due to the toxic effects that ammonia produces, the ammonium level must be treated to an acceptable level, <10 mg/l, before it is discharged (Welander et al., 1997).

This high level of ammonia and the other various components of landfill leachate make it very difficult to treat. There are many different landfill leachate treatment options. The options include complex and expensive events of exsitu physical-chemical and biological processes for the treatment of high- strength organics and inorganics, which include nitrogen. These separate treatment processes can result in large costs that could otherwise be profit.

Studies have shown that recirculation of leachate will produce stabilized leachates containing relatively low concentrations of degradable carbon compounds but high concentrations of ammonia (Knox, 1985). Since carbon compounds are being removed in situ, consideration has also been given to treating leachate ammonia in situ. The use of the landfill as a bioreactor for nitrification/denitrification should be considered a prime objective to avoid extra costs; especially since nitrification is a proven process to remove ammonia. The basis of this research is to examine the removal of ammonia nitrogen from bioreactor leachate.

The following objectives were developed to investigate this basis:

- Design and operation of lab-scale units in order to demonstrate the possibility of in situ nitrification at landfills operated as bioreactors.
- Evaluation of rubber chips, wood chips, synthetic plastic, and stable refuse as biofilm support media for nitrification.
- Examine landfill bioreactor leachate for nitrification inhibition.
- Evaluate different media for COD removal.

There is a substantial amount of information on nitrification of landfill leachate in the literature, but research that examines the use of biofilm support media, such as rubber and wood, at a landfill are not available. Since two biofilm support media (rubber and wood) are readily available at landfills, research in this area would be very valuable. The rubber media is obtained by shredding tires and wood chips would come from chipping wooden pallets. Oak and pine pallets are available.

Simulated filter unit reactors were designed and constructed to study the removal of ammonia from the leachate. Analysis of the results from these studies should provide a basis for full-scale design and operation of nitrification systems at landfills.

# CHAPTER 2: LITERATURE REVIEW

*Nitrification*

Nitrification is widely used to remove ammonia from wastewater by biological oxidation. Wastewaters containing high concentrations of ammonia create environmental problems because ammonia may be toxic to aquatic organisms and can cause fertilization of lakes and reservoirs which leads to algal growth and eutrophication (Forgie 1988, Welander 1998). Other damaging impacts resulting from discharges of ammonia include reduction of chlorine disinfection efficiency, an increase in the dissolved oxygen depletion in receiving waters, adverse public health effects, and a reduction in suitability for reuse (De Renzo, 1978). Due to the toxic effects that ammonia produces, the ammonium level must be treated to an acceptable level, <10 mg/l, before it is discharged (Welander et al., 1997). Nitrogen in wastewaters can be in the following forms: ammonia, ammonium, nitrite, and nitrate, and these forms originate from organic compounds, such as urea and proteins or their degradation products (Reynolds, 1996).

*Kinetics*

Conversion of nitrogen to the appropriate form for nitrogen removal is controlled by several biochemical reactions. These biochemical reactions are parts of the nitrogen cycle occurring in nature. In this cycle, bacteria convert organic and carbonaceous organic matter to ammonia. Continued aerobic biochemical reactions result in the oxidation of ammonia to nitrite, and then nitrite to nitrate. The overall biochemical process of oxidation of $NH_4^+$ to $NO_2^-$, then finally to $NO_3^-$ is known as nitrification. Nitrification is performed by the group of bacteria known as nitrifiers. The overall nitrification process is represented by the following equation:

$$\mathbf{NH_4^+ + 2O_2 \rightarrow NO_3^- + 2H^+ + H_2O}$$

The nitrifying process takes place in two steps and each step is carried out by a specific group of nitrifying organisms. The two microbes involved have been identified in many studies and are the aerobic autotrophic genera *Nitrosomonas* and *Nitrobacter* (Reynolds, 1996). The reactions are as follows:

$$2NH_4^+ + 3O_2 \rightarrow 2NO_2^- + 4H^+ + 2H_2O, \text{ Nitrosomonas}$$

$$2NO_2^- + O_2 \rightarrow 2NO_3^-, \text{ Nitrobacter}$$

Nitrosomonas performs the first step by oxidizing ammonium to nitrite. Nitrobacter completes the oxidation by converting the nitrite to nitrate. Since complete nitrification is a sequential reaction, treatment processes must be designed to produce an environment suitable for growth and survival of both groups of nitrifying bacteria (De Renzo, 1978).

*Environmental Requirements for Nitrification*

The most common, practical, and economical way to remove ammonia from a waste stream is to utilize nitrifying bacteria which are naturally present in the soil, freshwater, and saltwater. They are found wherever their required nutrients, ammonia, and oxygen exist. Nitrifiers are difficult to maintain because of their specific environmental requirements. The important environmental parameters that must be maintained for optimal performance of the nitrifiers include the correct pH range, a minimum dissolved oxygen concentration, the necessary temperature range, presence of ammonia, supply of micronutrients, and suitable hydraulic retention time (Rogers, 1983). Also for nitrification to occur, high organic concentrations (COD) and inhibitors, such as metals and specific organics, must be removed. The pH of the liquid must be kept in the range between 7.0-8.8, with the optimum nitrification rate being around 8.5 (USEPA, 1975). Nitrification produces $H_2CO_3$, so the pH drops; therefore, the pH must be maintained within the operating range often by adding base.

Liquid temperature should be maintained between 20-35 C for good activity (USEPA, 1975). Adequate aeration should keep the dissolved oxygen concentration at a minimum of 2 mg/l. Research shows that oxygen concentrations above 2.0 mg/l have little effect on prohibiting nitrification and it is seldom necessary to maintain the D.O. in excess of this value to get satisfactory nitrification; however, oxygen concentrations below 2.0 mg/l begin to have a strong effect (Grady, 1999). Chemical Oxygen Demand (COD) must be at levels that do not use all the available oxygen or create inhibitory conditions. COD must be removed because of competition between the heterotrophic and autotrophic bacteria. In biofilm systems, heterotrophic bacteria can grow faster than the nitrifying bacteria and out compete them for space (Grady, 1999), therefore readily degradable COD must be removed before nitrification occurs. Certain metals have been shown to inhibit biological activity, and should be analyzed initially. If nitrification does not occur during the treatment process, it will occur in the receiving system. This places an additional oxygen demand on the system and creates toxicity and eutrophication. Therefore, an efficient nitrification treatment process must be designed.

*Denitrification*

Denitrification is the biochemical conversion of nitrate to nitrogen gas. $N_2O$ can also form if denitrification is incomplete. This process uses the nitrate formed in nitrification and removes it from the system and is often the companion step to nitrification in the biological nitrogen removal process. Denitrification is also known as the final step in the removal of ammonia nitrogen from the system. The process is accomplished by the denitrifiers which include *Pseudomonas, Micrococcus, Archromobacter,* and *Bacillus* (Reynolds, 1996). Nitrification occurs in oxygen enriched environments, while denitrification occurs in environments without oxygen. In the absence of oxygen, the denitrifiers use nitrate as the final electron acceptor. A carbon source is needed for denitrification to occur, and usually methanol is added to the system to accomplish this. Since $NO_3^-$ has numerous harmful impacts when discharged to a receiving system, it is very important to remove it from the system. This is the reason denitrication is a very important step when managing nitrogen conversion for ultimate nitrogen removal.

*Comparison of Traditional Landfill, Leachate Recirculating Landfill,and Bioreactor Landfill*

The modern municipal solid waste (MSW) landfill has evolved into a sophisticated treatment and storage facility. Landfill bioreactors have emerged as one of the new generation methods of managing solid wastes (Pohland, 1995) and they are used to minimize environmental impacts while optimizing the degradation and stabilization processes. Many of the old sanitary landfills have been converted into bioreactor type landfills because of the many advantages that the bioreactors offer compared to the old systems. Recirculating landfills are often confused to be the same as bioreactor landfills, but this is not true. The major difference is that bioreactor landfills must operate at approximately 45% moisture content. Moisture from other sources, as well as leachate recirculation, drives the moisture content to high levels. Recirculating landfills do not operate at such high moisture content levels.

The new systems are operated and controlled to rapidly accelerate the biological stabilization of the stored waste. Leachate generation, collection, and in situ recirculation are what drive the bioreactor processes. The recirculation is what separates the two different types of landfills. As moisture accumulates and becomes more uniformly distributed with leachate recirculation, waste stabilization progresses sequentially through initial, transition, acid formation, methane fermentation, and final maturation phases (Pohland, 1995). Compared to traditional sanitary landfills, landfill bioreactors initiate and provide more rapid, complete, and predictable conversion of readily degradable solid waste products, therefore enhancing the potential for gas recovery and utilization, decreasing management time and process uncertainty, and reducing the potential for negative health and environmental impacts and attendant liabilities. The bioreactors also increase resource recovery and site reuse opportunities (Pohland, 1995).

*Leachate Production and Characteristics*

Rainfall is the main contributor to generation of leachate. The precipitation percolates through the waste and gains dissolved and suspended components from the biodegrading waste through several physical and chemical reactions. Other contributors to leachate generation include groundwater inflow, surface water runoff, and biological decomposition (Reinhart, 1998). Liquid fractions in the waste will also add to the leachate as well as moisture in the cover material. Moisture can be removed from the landfill by water consumed in the formation of landfill gas, water vapor removed in the landfill gas, and leachate leaking through the liner (Tchobanoglous, 1993). Since the short term leachate quantity depends heavily on precipitation, it is sometimes hard to predict. Long term leachate quantity is not as difficult to predict. Leachate quality is also difficult to predict because each landfill is unique and the wastes vary widely (Bagchi, 1990). The major factors that affect leachate quantity and quality are; the type of disposed waste, hydrogeolic and climactic conditions, the age of the landfill, the phase of waste decomposition occurring, and the chemical and physical properties of the precipitation (Bagchi, 1990). Leachate quantity and quality is site specific. In arid regions, leachate quantity can be zero, while in areas of wet climate, 100 % of precipitation can become leachate. Once the adsorptive capacity of the trash field capacity has been achieved, continuous leachate flow will occur.

The characteristics of leachate from landfills vary according to the operational stage of the landfill and the climatic features of the location of the landfill. Landfill leachates from old sites are usually highly contaminated with ammonia resulting from the hydrolysis and fermentation of nitrogen containing fractions of biodegradable refuse substrates (Carley and Mavinic, 1991). As stabilization of the waste proceeds, the accumulating concentration of ammonia is also influenced by washout as leachate is collected and removed for offsite treatment. However, in bioreactor landfills with leachate containment, collection, and in situ recirculation to accelerate decomposition of readily available organic fractions of the wastes, leachate ammonia nitrogen concentrations may accumulate to much higher levels when compared to traditional landfills (Onay, 1995). Recirculation of leachate will produce stabilized leachates containing relatively low concentrations of degradable carbon compounds but high concentrations of ammonia (Knox, 1985); therefore, COD and BOD will be removed, but ammonia concentrations will climb.

*Leachate Management Strategies*

There are two leachate management strategies used by modern landfills. These two processes are single pass leaching and leachate recirculation. Most traditional landfills use the single pass leaching strategy where the generated leachate is collected and treated to remove all the contaminants before it is discharged. There are several physical, chemical, and biological processes that can be used for treatment.

The recirculation management strategy includes leachate containment, collection, and recirculation. Using this strategy, the leachate that is produced is collected, and then redistributed back over the landfill. Recirculation turns the traditional landfill into a anaerobic bioreactor. There have been numerous studies which have proven the effectiveness of bioreactors (Reinhart,1998).

The fundamental process used for waste treatment in a bioreactor landfill is leachate recirculation (Reinhart, 1998). Recycling or recirculation of the leachate back to the landfill creates the perfect environment for rapid microbial decomposition of the biodegradable waste products. Not only does the system remain a storage facility for the solid waste, it also becomes a treatment system. The accelerated breakdown and stabilization of the waste can make the landfill a reusable system, and increase the operating life dramatically compared to the traditional landfill. This space that results from rapid stabilization can be used to store more solid waste instead of having to purchase more land. Laboratory and pilot scale studies have clearly demonstrated that operation of a landfill as a bioreactor accelerates waste degradation, provides in situ treatment of leachate, enhances gas production rates, and promotes rapid settling (Reinhart, 1998).

One of the most important factors that controls solid waste biodegradation is moisture content. This parameter can be controlled by leachate recirculation. Leachate recirculation optimizes environmental conditions within the landfill to initiate stabilization of the contents as well as treatment of the moisture flowing through the landfill. The numerous advantages of leachate recirculation include distribution of nutrients and enzymes, pH buffering, dilution and precipitation of inhibitory compounds, recycling and distribution of methanogens, liquid storage, and evaporation opportunites at low additional construction and operating cost (Reinhart, 1998). Not only does recirculation of the leachate accelerate rapid degradation, it also treats the leachate at no extra capital costs. It has been suggested that leachate recirculation can reduce the time required for landfill stabalization from several decades to two or three years (Pohland, 1995).

*Leachate Treatment*

Since leachate ammonia concentrations may accumulate to significantly higher levels compared to traditional single pass leachate and municpal wastewater, an ultimate leachate discharge problem may occur. Values of nitrogen in wastewater generally range from 15 to 50 mg/l, of which approximately 60 percent is ammonia nitrogen (USEPA, 1975), while landfill leachate contains 400 – 800 mg/l of ammonia nitrogen (Welander et. al, 1998).

Leachate that is collected and removed from a landfill must be managed with care. Some type of treatment, either at the landfill site or at a treatment plant offsite, must be performed. Treated leachate must meet the required regulatory limits for discharge to the environment as treated wastewater. There are many different landfill leachate treatment options. The options include complex and expensive routines of exsitu physical-chemical and biological processes for the treatment of organic and inorganic constituents.

A simple approach to managing leachate would be to discharge the leachate to a nearby sewage treatment plant. If the landfill had a sewer connection, the leachate could be directly discharged from the storage containers. Since most landfills are located in sparsely populated areas, sewer connections are not usually available. Therefore, leachate usually is hauled by tanker trucks to treatment facilities. Also, sewage treatment plants often refuse to treat landfill leachate because the leachate may contain high concentrations of inhibitory chemicals that might interfere with the facilities treatment process (Mulamoottil et al, 1999). If a landfill does not transport and treat the leachate offsite, a treatment facility can be constructed on site. There have been numerous studies of the various treatment alternatives for leachate from landfills. Processes that have been evaluated include biological treatment (aerated lagoons, activated sludge, anaerobic filters, and stabilization ponds), and physical-chemical processes such as adsorption, chemical oxidation, coagulation/precipitation, and reverse osmosis (Pohland, 1995). Other treatment options researched include trickling filters (Knox, 1985), and suspended-carrier biofilm processes (Welander et al, 1997). The types of constituents in leachate are different from typical domestic wastewater. Not only does leachate contain organic compounds that require biological treatment, it also contains inorganic dissolved solids (sodium, chloride, etc.) which cannot be removed by biological treatment (Reinhart, 1998).

The general acceptance of leachate recirculation within the regulatory community has resulted in the consideration of ultimate treatment of the leachate to remove nitrogen on site. There is much literature available on ammonia and nitrogen removal, but most of these deal with microbiology or wastewater treatment. There is some information on nitrification of landfill leachate, but research studies that examine the use of biofilm support media, such as rubber and wood, at a landfill are scarse. Research in this area would be very valuable since these two biofilm support media are readily available at landfills. The rubber media is obtained by shredding tires and wooden chips come from chipping wooden pallets. Oak and pine pallets are available.

Usually the organic fraction of the waste is treated and this is followed by nitirification in a sophisticated ex situ system. The cost of these off site treatment systems is very expensive (Onay, 1998). The use of the landfill as a bioreactor to initiate nitrification should be considered a prime objective to avoid the off site treatment costs. Only one study was found that addressed this issue. Onay (1995) researched the concept in a laboratory landfill column reactor. He found that reactor operation with internal leachate recircultation provided 95% nitrogen conversion. The reactor with single pass leachate provided 30 – 52% nitrification efficiency.

*Trickling Filters*

The term trickling filter refers to a wide range of attached growth biochemical operations in which the waste stream is introduced to fixed media in a packed tower. The waste stream is treated by microorganisms growing attached to the media type in the tower. There are several types of media that can be used including synthetic plastics and natural wooden products. Trickling filters are aerobic and used to treat biodegradable organic matter. They are also used to oxidize ammonia-N to nitrate-N in a process known as nitrification (Grady, 1999). Nitrification can be accomplished in a trickling filter used for COD and BOD removal, a process known as combined carbon oxidation and nitrification, or in a trickling filter receiving a waste stream that has already had the organic matter removed, a process called separate stage nitrification.

A typical trickling filter system consists of five main components: 1) the media bed, 2) the containment structure, 3) the dosing system, 4) the underdrain system, and 5) the ventilation system (Grady, 1999). The media bed is probably the most important component because it provides the surface where the microorganisms grow thus establishing a biofilm. There are several media type options which include rock, wood, and synthetic plastics in various shapes and sizes. Nitrification occurs by the same mechanism as it does in any other aerobic biochemical operation. Ammonia-N diffuses into the biofilm where part is utilized by the heterotrophs for biomass synthesis and the remainder is oxidized to nitrate-N by nitrifying bacteria (Grady, 1999). As previously stated, carbon oxidation must occur before nitrification can begin because the heterotrophs out compete the nitrifiers for space in the biofilm. This is the reason separate stage nitrification is preferred. The COD and BOD can be removed in a separate reactor and then the waste stream can be introduced into a nitrifying column where a greater level of ammonia removal is provided.

Factors affecting trickling filter performance include total organic loading, total surface loading, total Kjeldahl nitrogen loading, total hydraulic loading, media depth, temperature of the wastewater, and the media type (Grady, 1999). There are several design equations that exist for the estimation of performance of any given biofilm system. Because of the complexity of the physical and biological characteristics of trickling filters, efficiency equations for ammonia and organic matter removal are difficult to formulate.

The design of nitrifying fixed-film reactors could be carried out using a formula based on Eckenfelder and Ford (Viessman, 1985). The applicable equation is

$$S_e/S_o = EXP(-KD/W^n)$$

where,

$S_e$ = final or effluent ammonia concentration, mg/l

$S_o$ = influent ammonia concentration, mg/l

K = reaction rate constant for ammonia oxidation, $min^{-1}$

D = depth of filter, ft

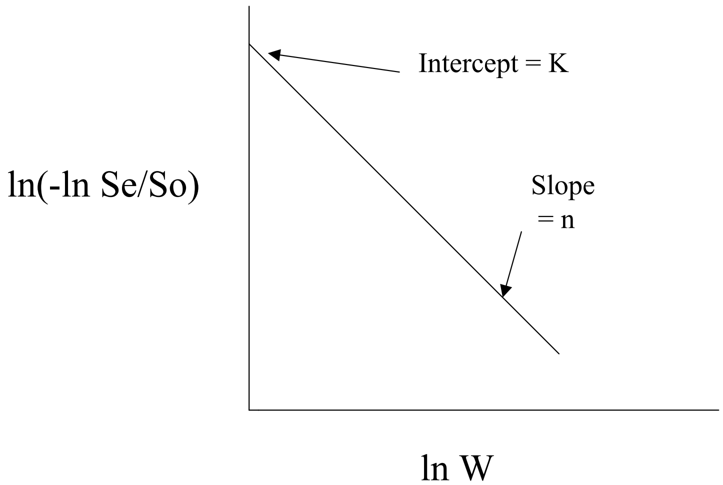
W = mass loading ($gpm/ft^2*mg/l$)

n = constant related to specific surface area and configuration of packing

The column volume, depth, and the surface area of the media remained constant throughout the study. Since D is constant, the equation simplifies to

$$S_e/S_o = EXP(-K/W^n)$$

The collected data can be plotted to determine the value of n and K for a specific media surface area.




Once these constants are determined, the equation can be used to select the depth of a filter unit that will be required to treat a specific amount of ammonia. Solve the equation for D and the amount of media needed to meet this requirement is provided in the filter unit.

*Future Issues*

Although there has been several studies on nitrification of landfill leachate (Welander 1997, Knox 1985, Onay 1995), there has been very little information compiled regarding rubber, wood, and other nitrifying biofilm support media. Since in situ treatment of landfill leachate, especially nitrification, has received little attention at either bench or full-scale, there is a need for information on their intrinsic treatability. Also, the effects of COD on nitrification rates of landfill leachate, and COD removal by the rubber and wood media still need to be examined.

## CHAPTER 3: METHODS AND MATERIALS

This research was performed to determine the nitrification kinetics for synthetic leachate and landfill leachate. Readily available biofilm support media such as wood chips and rubber tire chips were evaluated and compared to synthetic plastic biofilm media. The project consisted of five different phases. These were: 1) Evaluation of three types of media, wood chips, rubber chips, and plastic trickling filter media. In this phase the flowrate was constant and the ammonia concentration varied. 2) Evaluation of four media types, the same three as in in phase 1 and an additional media, stable refuse. The ammonia concentration was constant and the flow varied. 3) In this phase, a comparison was made between leachate and synthetic solution to determine if nitrification inhibition was present in the leachate. 4) In this phase, a comparison of oak and pine media was made to determine the most efficient bioflim support media. 5) A study of the kinetics of COD removal from leachate was conducted. Each of these experimental phases will be discussed.

*Experimental Setup*

The pilot study was performed using four aerobic downflow biofilm reactors to obtain nitrification. The reactors were constructed from 4 inch diameter PVC pipe with a length of 5 feet. The volume of the columns was 1.74 $ft^3$. Different biofilm support media were placed in each reactor. The four media evaluated were wood chips, rubber tire chips, stable solid waste, and synthetic plastic. A single pump with two heads was used to pump the test solutions to the top of the reactors where it was dispersed onto the media. The test system is shown in Figure 1.

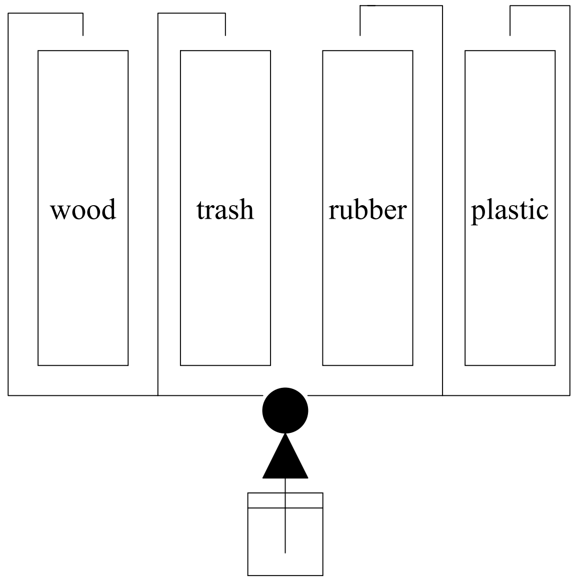


Figure 1. Experimental Setup of Biofilm Columns

The specific methods and materials used in each phase are discussed in the following material.

*Phase 1: 3 Media Evaluation, Constant Flow, Varied Ammonia Concentration*

The ammonia concentration was varied and ammonia removal measured during this phase. The three media evaluated were synthetic plastic packing from NSW, Roanoke, VA, shredded steel-belted rubber tires from a WMI landfill, and pine bark wood chips used in landscaping. A nitrification nutrient and mineral feed was prepared according to Gerhardt (1981). The feed contained water, magnesium sulfate, sodium bicarbonate, iron chloride, calcium chloride, potassium phosphate, and ammonium phosphate. The reactors were inoculated with nitrifying bacteria obtained from Alternative Natural Technologies, Inc. Each of the three columns were packed with a different media, and seeded with the organisms by pouring a mixture of the feed solution and nitrifiers on top of the media in each column. The nitrification feed solution was recirculated in each of the three columns for approximately two weeks and a nitrifying biofilm was allowed to become established in each of the columns.

Nitrate concentrations in the effluent were measured every other day and when a considerable amount of nitrate was present, the media were assumed to be supporting nitrifying bacteria and nitrification. The columns were then determined to be ready for testing.

A tap water solution with diammonium phosphate and sodium bicarbonate was introduced into each column at a constant flow rate (3.58 $m^3/m^2$/day). Eight different concentrations of diammonium phosphate were used to achieve ammonia-N concentrations of 22.4, 43.2, 47.6, 72.6, 93.2, 119.7, 176.0, and 248.4 mg/l. Each test run lasted for four hours. Samples were collected after the first hour, second hour, and fourth hour. Influent and effluent samples were tested for $NH_3$-N and pH. Analysis was performed using a Hach DR/2010 Spectrophotometer, and pH was measured on an Accumet pH meter with a Cole Parmer electrode. Ammonia-N was measured using the USEPA approved Salicylate Method (Standard Methods, 1989). Results from this phase were evaluated to determine which media is the most efficient at supporting ammonia removal biofilms.

*Phase 2: 4 Media Evaluation, Constant Concentration, Varied Flow*

Various hydraulic loading rates were evaluated during phase 2. Four biofilm support media were evaluated during this phase. The same three media from phase 1, plus stable, 35 year old solid waste from a WMI landfill were used. The $NH_3$-N concentration was held constant (40 ppm), while the flow going into the columns was varied. Four different flow rates were used: 52.0, 33.5, 20.0, and 10.7 ml/min (9.26, 5.93, 3.58, 1.85 $m^3$/day/$m^2$). The same test solution as in phase 1 was used and four hour test runs occurred. Samples were taken the first, second, and fourth hours after loading began. Influent and effluent samples were tested for $NH_3$-N, $NO_3$-N, pH, and flow. The $NH_3$-N was measured using the Salicylate Method (Standard Methods, 1989), $NO_3$-N was measured using the Cadmium Reduction Method (Standard Methods, 1989), pH was measured with the meter, and flow was measured using a graduated cylinder and stopwatch.

*Phase 3: Bioreactor Leachate versus Test Solution.*

Phase 3 was designed to determine if nitrification using a synthetic ammonia solution responded in a similar manner to landfill leachate. Landfill leachate might contain refractory organics or heavy metals that could inhibit nitrification. The same four media were used as in phase 2. Leachate from the Middle Peninsula recirculating landfill in Glens, Virginia was tested. The degradable Chemical Oxygen Demand (COD) in leachate must be stabilized or removed before nitrification can occur, so the leachate was recirculated in a biofilm reactor until the COD stabilized. The COD in the recirculated flow was periodically measured until the COD concentration remained constant. The leachate was introduced into the reactors, and the four hour nitrification periods were used after COD stabilization occurred. Influent and effluent samples were tested for $NH_3$-N, $NO_3$-N, pH, and flow. The data obtained in this phase was compared to the data resulting from phase 1 and 2 to determine if nitrification inhibition occurred when using bioreactor leachate.

*Phase 4: Oak vs Pine as Most Efficient Biofilm Support Media*

The same pine chips as in the first 3 phases and oak hardwood chips shredded from pallets at a WMI landfill were evaluated as support media during phase 3. These two media were used because in the first 3 phases, wood chips were proven to be the most efficient biofilm support media. This phase compared hardwood chips to pine chips. The test solution in phases 1 and 2 was used. Different concentrations and flows were studied. Again, test runs lasted 4 hours, and samples were obtained after the first, second, and fourth hours. Influent and effluent samples were tested for $NH_3$-N, $NO_3$-N, pH, and flow. The $NH_3$-N, $NO_3$-N, and pH were measured as before. The flow was measured using a graduated cylinder and stopwatch.

*Phase 5: Kinetics of Organic (COD) Removal from Leachate*

Since pine and hardwood chips can support nitrifying biofilms, it was also of interest to test the two support media for COD removal, especially since COD has to be stabilized for nitrification to occur. Two columns were packed with the media, one with pine, and one with hardwood. The Middle Penninsula recirculating landfill leachate was recirculated through the reactors and COD was monitored over time using the Reactor Digestion Method (Standard Methods, 1989).

*Landfill Leachate*

Leachate from the WMI operated Middle Peninsula leachate recirculation landfill was used in the experiments. This landfill is located in Glens, Virginia. The Middle Peninsula landfill began accepting waste on June 30, 1995 and is still in operation. The landfill is a traditionally mixed landfill receiving several different kinds of wastes including: municipal, industrial, construction, and demolition wastes. All the wastes received are non-hazardous. Leachate quantity varies with the seasons. The leachate collection system consists of 18 inches of #68 stone blanket above a Subtitle D liner with one central collection pipe, and a slope riser system with automatic pumps for dewatering the cells. The leachate is ultimately collected in a 250,000 gallon, above ground, bolted, steel tank. It is treated by on/off aeration in a 500,000 gallon tank of the same construction and equipped with a bottom mounted jet aeration mixer. The aeration cycle is one hour on and one hour off to achieve nitrification/denitrification. The jet mixer is powered by a 30 hp blower and a 30 hp pump (Goldsmith, 01).

*Ammonia loss pathways*

Due to the absortive nature of wood chips, some $NH_3$-N removal could be due to ion exchange in the wood chips. Solutions were run through the wood media reactors before nitrifying biofilms were established and influent and effluent samples were measured. The hydraulic loading flow of 3.58 $m^3/m^2$/day was used because it was the flow used in all phases of the study. The average influent $NH_3$-N concentration for the tests was 69.4 mg/l and the average effluent was 59.6 mg/l. Some portion of the ammonia is removed either by ion exchange or volatilization. Air stripping is not likely to be a major factor in the ammonia removal due to ammonia's low Henry's Law constant.

## CHAPTER 4: RESULTS AND DISCUSSION

*Phase 1*

The ammonia concentration (22.4 – 248.4 mg/l $NH_3$-N) was varied at a constant hydraulic loading 22 ml/min (3.92 $m^3$/day/$m^2$) in this phase. After development of a nitrifying biofilm (see Methods), the ammonia solution was introduced to the reactors and effluent samples were taken after the first, second, and fourth hours of operation. Figures 2 through 5 show the ammonia concentrations in each reactor at each sampling for four different influent ammonia concentrations.

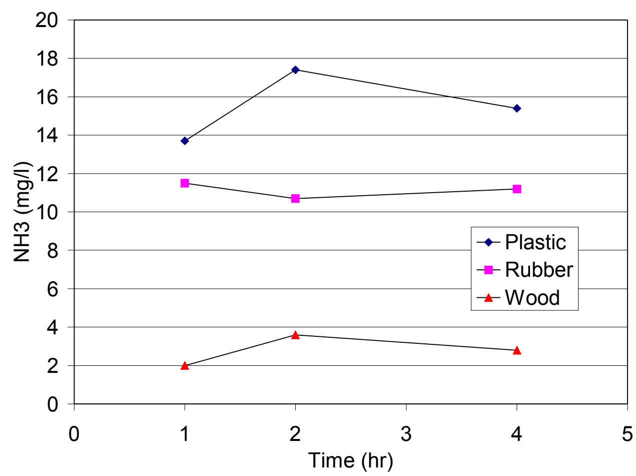



Figure 2. Change in ammonia concentration over time in a biofilm reactor at influent ammonia concentration of 22.4 mg/l.

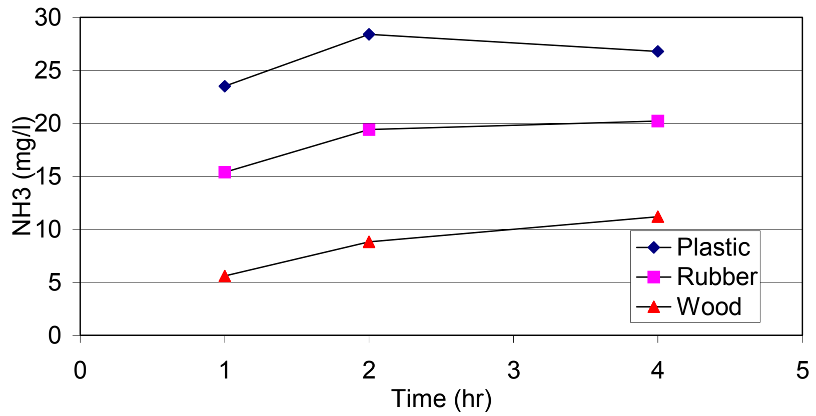



Figure 3. Change in ammonia concentration over time in a biofilm reactor at influent ammonia concentration of 48.3 mg/l.

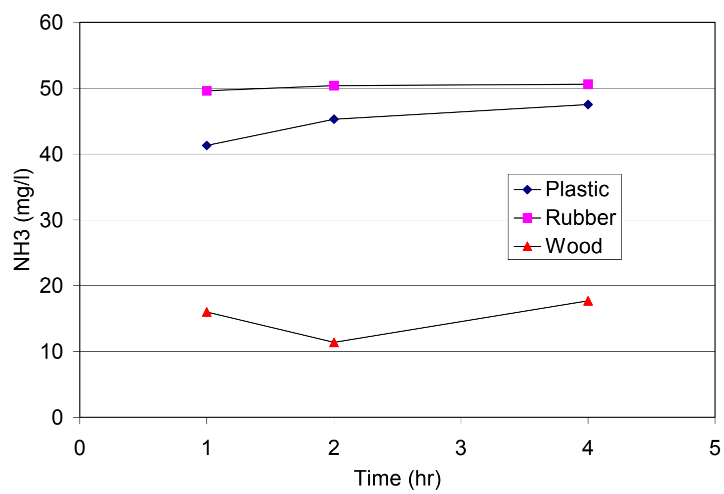



Figure 4. Change in ammonia concentration over time in a biofilm reactor at influent ammonia concentration of 72.6 mg/l.

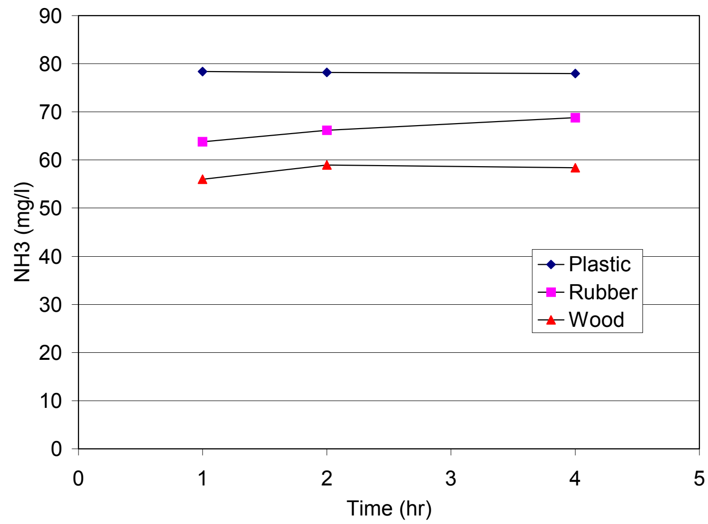



Figure 5. Change in ammonia concentration over time in a biofilm reactor at influent ammonia concentration of 93.2mg/l.

The trend seen in the four plots is that the rate of nitrification is relatively constant over the first four hours of operation, indicating that this approach is valid. Nitrification begins immediately because of the absence of organics in the test solution. The ammonia gradually increased above the first hour concentrations and obtained steady state somewhere between the second and fourth hour of treatment. The average influent and effluent ammonia concentrations from each entire test period were calculated and used in developing the equation for predicting ammonia removal. The average effluent concentration over the 4 hours for each media type were plotted for each $NH_3$-N concentration loading. In Figures 6 through Figure 8 the effects of varying ammonia loadings on ammonia removal are shown.

Different media types were examined to see which was most effective in supporting ammonia removal. The reactor containing the pine chips had the highest removal efficiency (77-87%), and the percentage removal was much higher for wood than for the rubber (20-62%) and plastic media (13-46%). Also the average percent removal for the wood media was much more consistent over the range of loadings (77 – 87 %) than the other two media (Table 1).

Table 1.  Percent Ammonia Removal for Different Media Types

| Media Type | Avg. Removal (%) |
|---|---|
| Plastic | 13-46 |
| Wood | 77-87 |
| Rubber | 20-62 |

The pine media reactor maintained approximately the same percent removal as the concentration of ammonia applied to the biofilters increased.  The rubber and plastic media reactors percent removal were lowered when the concentration of ammonia increased (Figure 6), suggesting that the media was near maximum capacity at the lowest applied ammonia concentration.

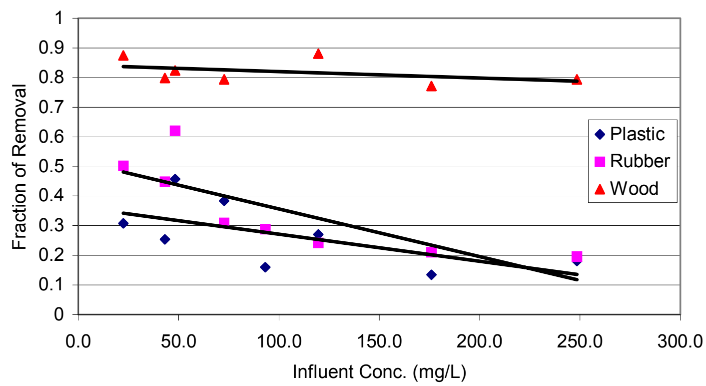



Figure 6.  Fraction of $NH_3$ removal in the biofilm reactors versus influent $NH_3$ concentration.

Figure 7 shows the amount of ammonia removed versus the influent concentration for the three types of media.  The amount of $NH_3$-N removed levels off for the plastic and rubber media as the influent $NH_3$-N concentration increases.  The pine media continues to remove the same percentage of $NH_3$-N compared with the influent.  The same trend can be seen in Figure 8, where the effluent $NH_3$-N concentration is plotted versus influent concentration.  The highest ammonia effluent concentration for the wood reactor is approximately 60 mg/l, while the rubber reactor produces effluent concentrations as high as 200 mg/l ammonia.

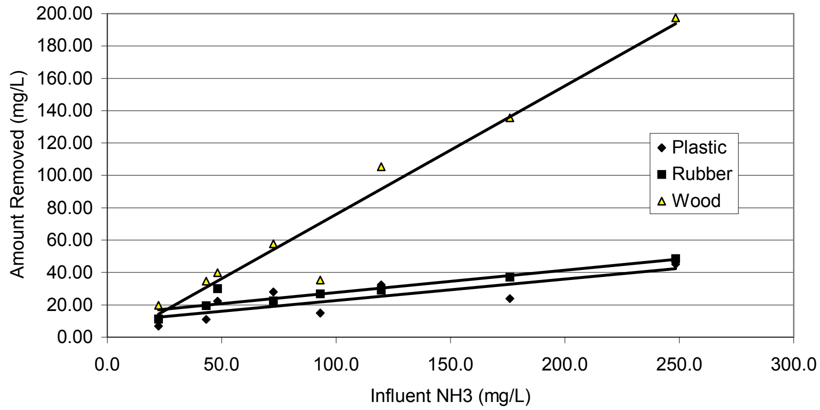



Figure 7. Amount of ammonia removed in biofilm reactors versus influent concentration.

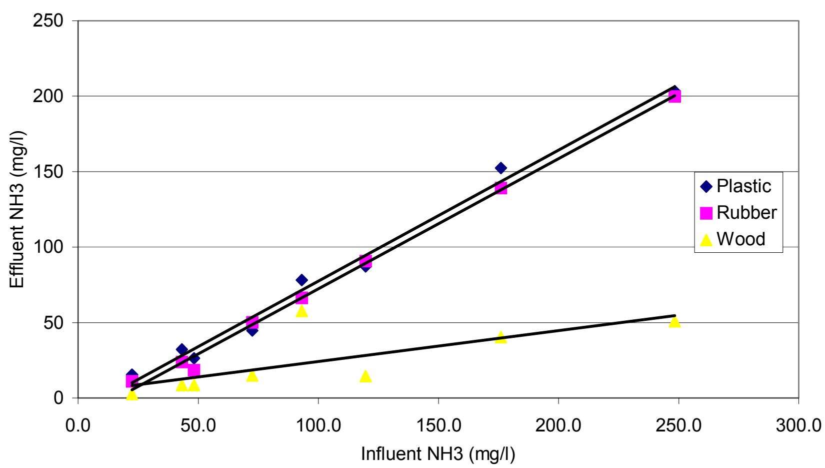



Figure 8. Ammonia effluent from biofilm reactors versus ammonia influent concentration.

The $K_s$ value for autotrophic nitrifying bacteria is reported to be approximately 1.0 mg N/l (Grady, 1999). Since all the ammonia concentrations ($S_s$) that were tested are much larger than the $K_s$ value, the Monod equation for the study can be simplified to a zero-order approximation. Under this condition, the specific growth rate is equal to the maximum specific growth rate. The nitrifiers in this study were consuming substrate and growing as fast as possible. This could help explain the linear increase in removal with loading.

The surface areas of the different packing media were very difficult to calculate due to the irregular and heterogeneous shapes. The wood chips had the most irregular shape, and this might have contributed to a larger surface area than the other two media. Due to the crevices and cracks in the wood, many more spaces for bacterial growth existed. This could be the reason for the higher removal rates from the wood chip reactor. Another advantage of the wood chip media is its ability to hold moisture, which would promote a better environment for bacterial growth.

The main objective of this phase was to see which media type would support the most efficient $NH_3$-N removal, and to examine the effects of varying the influent concentration. These issues are relevant to Waste Management, Inc. (WMI) because the bioreactor leachate $NH_3$-N concentrations will vary, and the reactor must be able to maintain efficiency under a variety of loadings. The media type evaluation is also very important to WMI because the rubber and wood chips are readily available on site. The most important conclusion from this phase of study is that the wooden pine media is an efficient media for nitrification, and can handle variations in $NH_3$-N loadings and still maintain its treatment efficiency.

*Phase 2*

Phase 2 analysis examined the effect of varied hydraulic loading rates on ammonia removal. In these studies, the $NH_3$-N concentration was held constant at approximately 40 mg/l for each run. Hydraulic loading rates control the detention time in the packed beds and the detention time has a significant impact on removal rates. The flow rates and ammonia removal rates are compared in Figure 9 through Figure 13. Figure 9 and 10 represent the ammonia concentrations versus time. In Figure 9, the highest flow rate was used and Figure 10 represents the lowest flow rate. The lower flow rate produces lower ammonia concentrations for each reactor. The lowest effluent ammonia is produced after the first hour and then gradually approaches steady state between the second and fourth hours as seen in phase 1. The effluent concentrations were averaged and used in the data analysis.

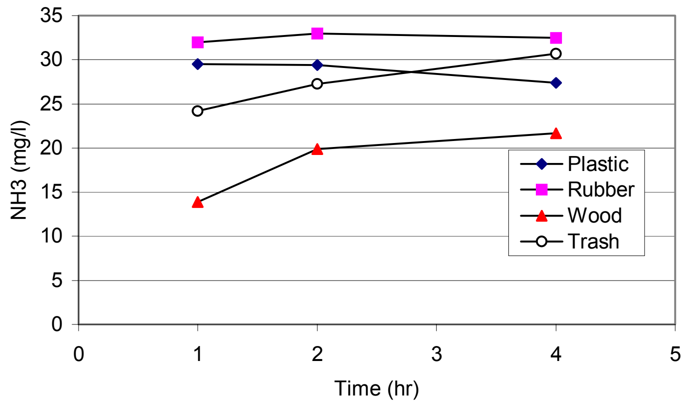



Figure 9. Ammonia concentration (influent 40 mg/l) in biofilm reactor versus operating time at the highest flow rate (9.26 $m^3/m^2$/day) tested.

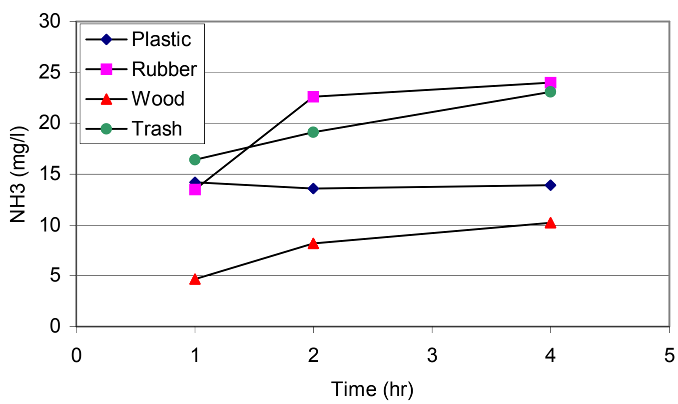



Figure 10. Ammonia concentration (influent 40 mg/l) in biofilm reactor versus operating time at the lowest flow rate (1.85 $m^3/m^2$/day) tested.

A new media type, stable landfill refuse, was introduced into this study phase, along with the other three media, wood chips, rubber chips, and plastic media. The different media types were compared to see which type is most efficient at handling the varied hydraulic loadings and removing $NH_3$-N. As expected, at higher flows there was less percentage removal of $NH_3$-N. The test solution has a shorter residence time in the treatment reactor due to the higher flow rates. Therefore, less ammonia is removed. As in phase 1, the pine chip media reactor was the most efficient in ammonia removal. The wood media reactor removed 100% of the incoming $NH_3$-N at the lowest flow tested 10.7 ml/min (1.85 $m^3$/day/$m^2$),.

As seen in Figure 11, as the flow rate increases, the removal fraction decreases. The effluent concentration, which is very important in discharge permits and regulations, goes up as the hydraulic loading increases as shown in Figure 12.

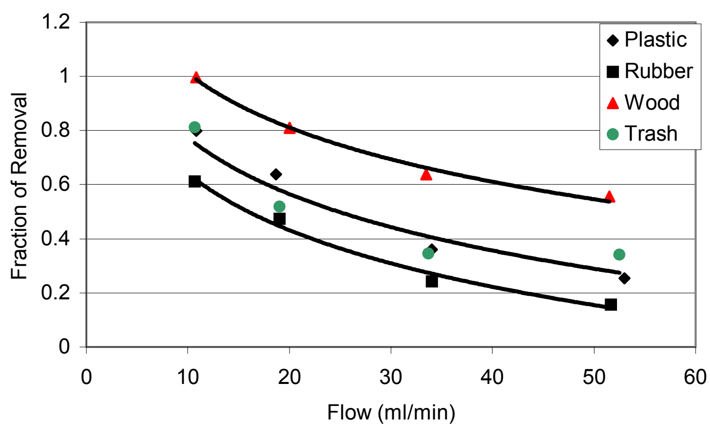



Figure 11. Fraction of ammonia removal in the biofilm reactors with varying flow rates.

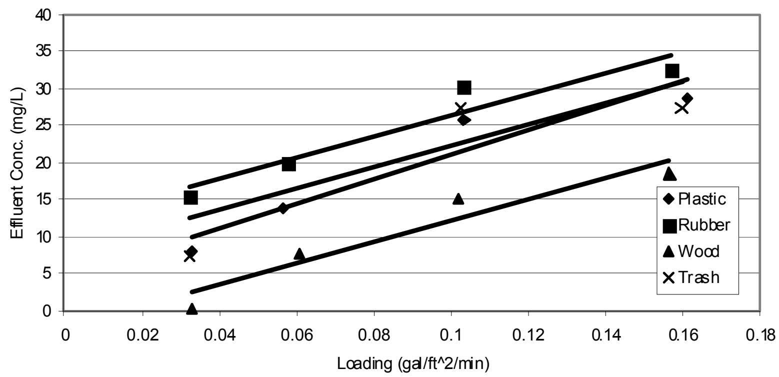



Figure 12. Effluent ammonia concentration from biofilm reactors versus hydraulic loading rates.

Nitrate was monitored in this phase of the study to evaluate the nitrification process and to provide a mass balance for influent ammonia. Even though pine wood has the highest removal rate of $NH_3$-N, it does not have the highest effluent $NO_3$-N concentration as seen in Figure 13. It was expected that $NO_3$-N would be stoichometrically related to the $NH_3$-N removed. In Figure 14, the ratio of $NO_3$-N produced to $NH_3$-N lost is shown. The expected ratio is less than 1 due to incorporation of some ammonia into cell mass. The plastic and rubber media reflect the expected results. For wood chips, the low production of nitrate relative to ammonia lost is probably due to denitrification. Deterioration and degradation of the wood chips is occurring and supplying a carbon source for denitrification. This might be benefitial since nitrogen loss occurs rapidly. However, it also implies that the wood chips will deteriorate and replacement material will need to be provided.

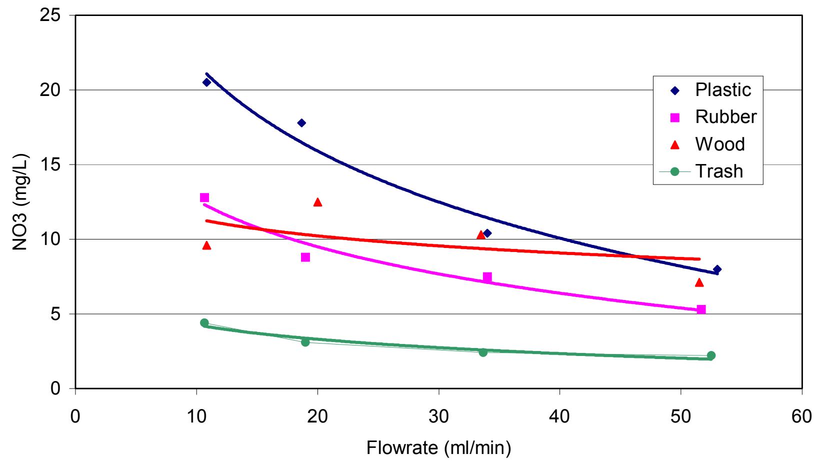



Figure 13. Nitrate produced in each biofilm reactor at different flows

Not only is wood a good media for nitrification, it could also be an effective media for denitrification. Since a thicker biofilm was visually observed in the pine wood reactor effluent, more $NH_3$-N could be used for assimilation by the microbes. Both organisms and portions of biofilm were present in the effluent. The nitrogen associated with these solids could also account for some of the nitrogen loss. The nitrate production drops with increase in flow rate as shown in Figure 13. This suggests a lower nitrification rate.

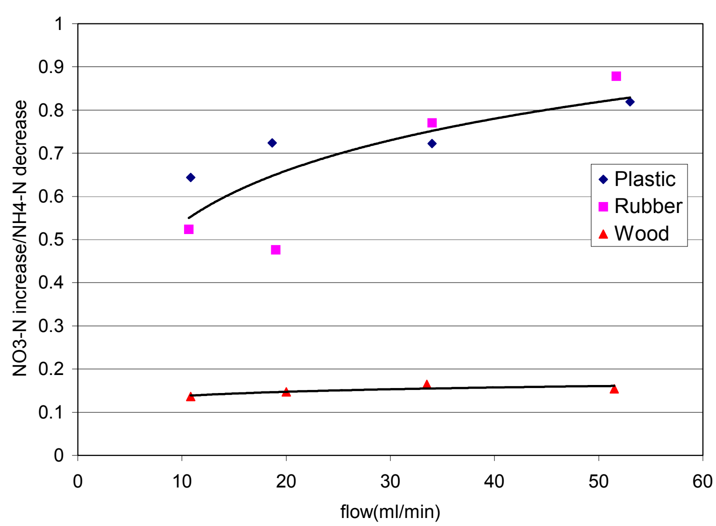



Figure 14. The amount of nitrate increase/ammonia decrease in each biofilm reactor at different flows.

Since phase 1 of the study did not use stable refuse as a media, the loadings (concentration x flow) from phase 1 and 2 for each media (wood, plastic, rubber) were plotted in Figure 15 to see how stable refuse compared with the media from phase 1. It can be seen in these figures that stable refuse was the second most efficient media with regard to $NH_3$-N removal.

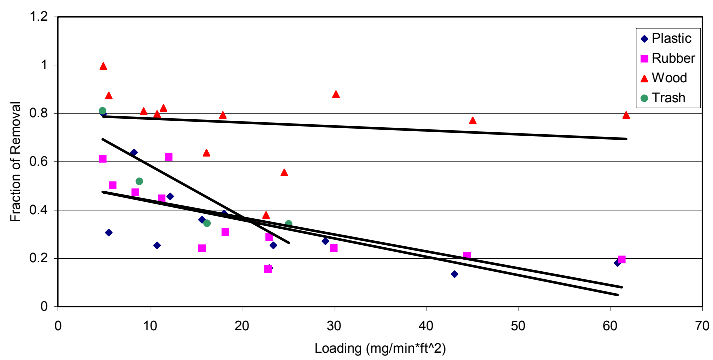



Figure 15. Fraction of ammonia removed in biofilm reactors at different loading rates.

The main finding from this study was that at lower hydraulic loading rates, the removal of $NH_3$-N was higher. The pine chips were again determined to be the most effective support media for $NH_3$-N removal. This study also shows that mass loading is the critical design parameter, not hydraulic loading (Figure 16). Mass loading can be increased by increasing either the ammonia concentration or flow rate.

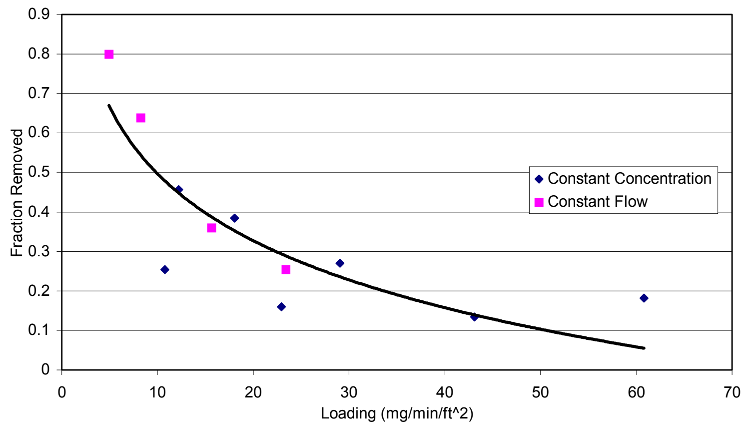



Figure 16. Fraction of ammonia removed in plastic biofilm reactor at different loading rates.

*Phase 3*

This area of study was designed to determine if nitrification inhibitors were present in landfill leachate. It has already been determined that nitrification will readily take place when using the test solution of DAP and tap water. It was of interest to know if bioreactor landfill leachate contained constituents that would disrupt the nitrification process.

Oxygen demand can act as an inhibitor to nitrification. It has been shown that COD must be removed before nitrification can proceed. Leachate from the Middle Peninsula landfill was recirculated through a pine wood media reactor to stabilize the COD. It took 4 days to stabilize the COD and initiate nitrification as seen in Figure 17. The leachate was used for analysis in the reactors once the COD was stabilized.

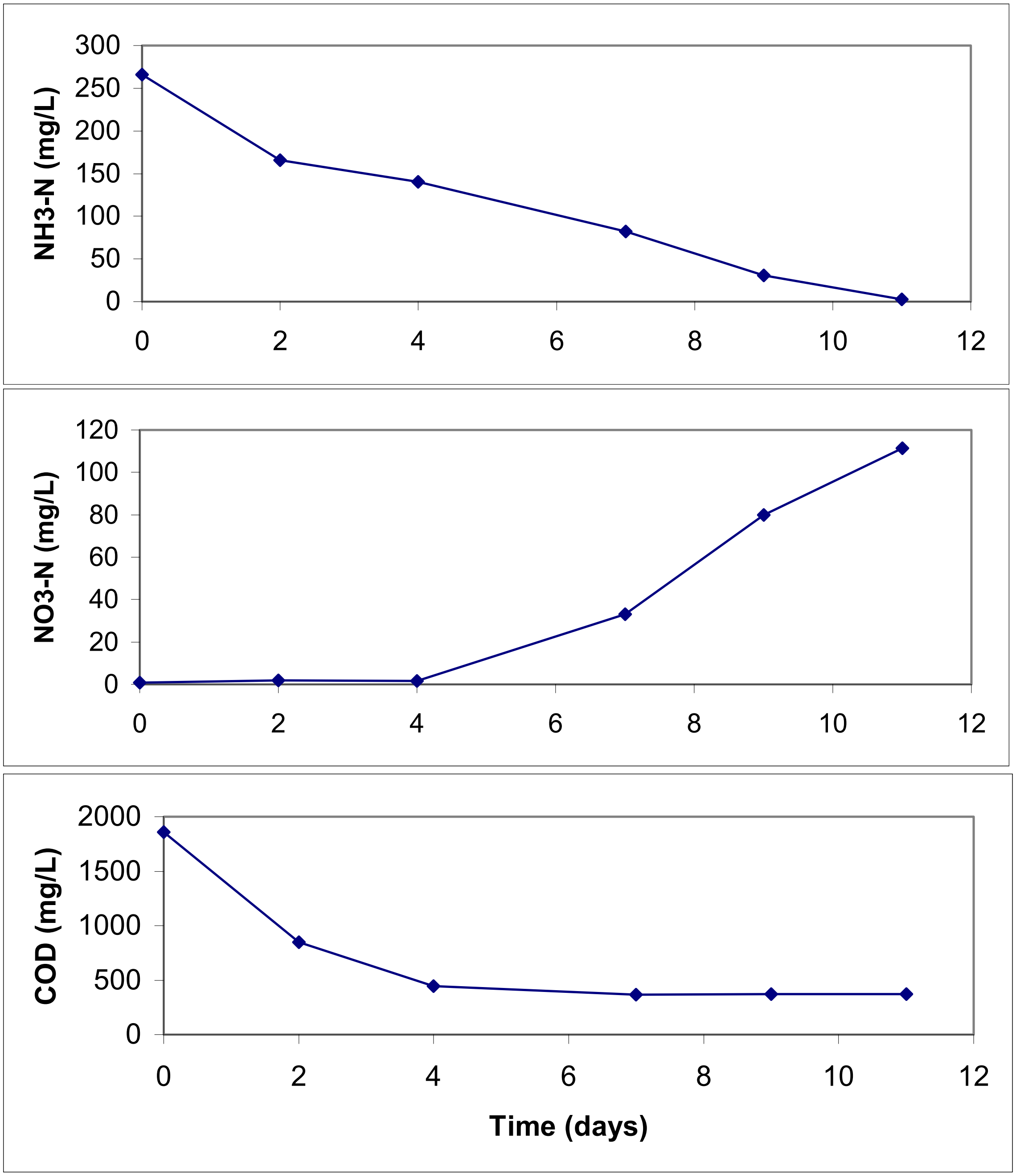



Figure 17. COD, nitrate, and ammonia concentrations in leachate effluent from the wood biofilm column over time.

The pine wood media was the only media used in this phase because it was already proven to be the most efficient at removing ammonia. Nitrification biofilm reactors were maintained by circulating a nitrification media while the COD was being removed in the other reactors. Average influent and effluent $NH_3$-N, $NO_3$-N, and $NO_2$-N concentrations from the 4 hour test runs were compared to the concentrations from the tap water solutions to see if any nitrification inhibition occurred. These results can be seen in Figure 18.

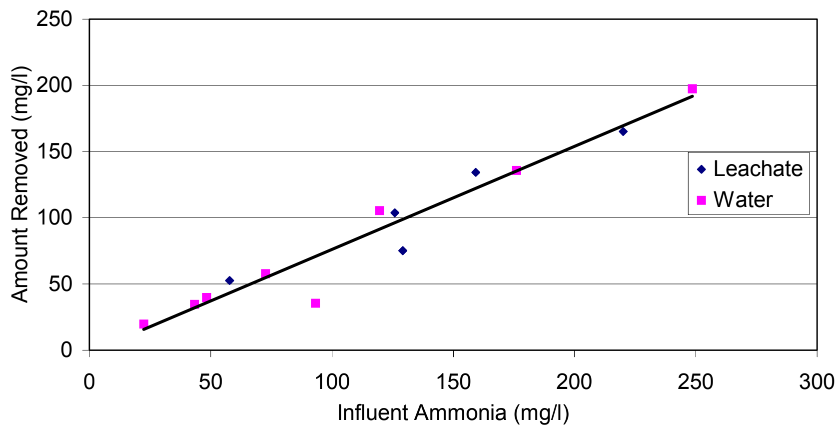



Figure 18. Comparison of Influent/Effluent from the biofilm columns for both leachate and water solution tests.

These data suggest that the leachate from the Middle Peninsula landfill does not contain constituents that will inhibit nitrification. Other leachates will require testing to insure that they are not inhibitory, but these data are promising.

*Phase 4*

Since wood chips were determined to be the best ammonia removal biofilm support media, it was of interest to determine if there was a difference in hardwood and pine wood chips. WMI has both pine and hardwood chips readily available. The DAP, bicarbonate, tapwater solution was used with varying $NH_3$-N loadings. Average influent and effluent samples from the four hour test runs were analyzed. Results from the comparison of these two media can be seen in Figure19. Figure 19 shows loading vs fraction of removal, and it can be seen that oak media support higher removal rates when compared to pine media. The $NH_3$-N removal rates for the oak reactor were not quite as high as for the pine media reactor in phase 1. However, during this phase when the two are directly compared, the oak provides a higher removal rate. The lower removal rates compared to phase 1 could be due to the temperature in the lab. Phases 1, 2 and 3 were performed in the summer, and phases 4 and 5 were performed in the winter. Temperature has a crucial effect on nitrification, and nitrification is much more efficient at warmer temperatures.

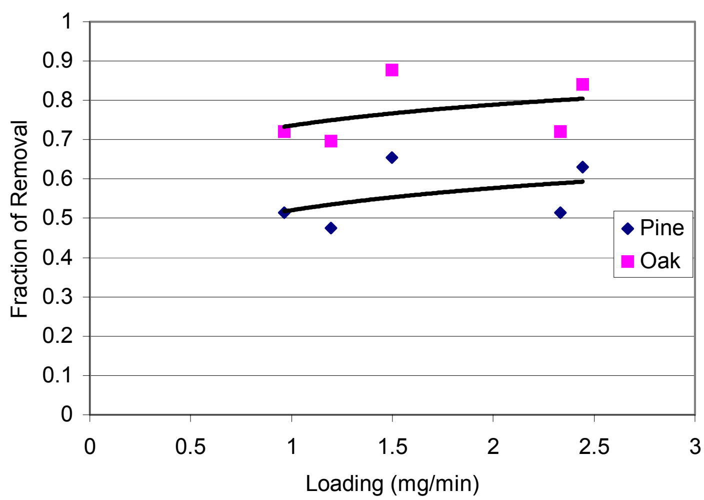



Figure 19. Comparison of fraction of ammonia removed versus loading rate with biofilm reactors containing oak chips and pine chips.

The difference in the removal rates of the oak and pine media could have been due to differences in surface area. The oak chips were more shredded, smaller, and offered more spaces for bacterial growth.

The important results from this study are that oak wood media is more efficient than pine wood media at supporting $NH_3$-N removal biofilms. Another important issue is degradation of the media. The oak media is much more resistant to degradation than the pine media. The pine media began to degrade after 5 weeks of loading. The oak chips were still very stable with no breakdown observed after 6 weeks.

*Phase 5*

Because of the need to remove or stabilize COD for initiation of nitrification, it was in interest to see if the wood chip filter unit could be used for COD removal. Landfill leachate was introduced into the reactors and recirculated. COD was monitored on a daily basis. The average influent COD concentration was 1282 mg/l and the average effluent COD concentration after four days was 1174 mg/l. This is very poor COD reduction. Although the wood chips work well for $NH_3$-N removal, wood chips do not seem to be an effective treatment process for the removal of COD. Reasons for this are unknown, but oxygen supply may be a factor. Additional studies are needed to determine design criteria for biodegradable COD in leachate using biofilms of natural material.

*Hydraulic Residence Time*

Another parameter that may have influenced the removal rates for the different media was hydraulic residence time (HRT). Due to the different physical characteristics of the different media, the hydraulic residence times were slightly different but probably not different enough to make a major difference in removal rates. Table 3 shows the different HRTs for each media. The refuse reactor's HRT was much higher than the other three media, but this still did not promote a significantly higher removal rate than the plastic and rubber media. Even though the refuse reactor had the lenghty HRT, the wood chip reactor was still superior. Due to the similarity in the HRTs for the different media, it can be concluded that the wood media's superior ammonia removability was most likely due to the high surface area of the wood chips, and the wood's ability to hold moisture.

Table 2. Hydraulic residence times for the different media.

| Media | Hydraulic Residence Time |
|---|---|
| Plastic | 55 s |
| Wood | 70 s |
| Rubber | 62 s |
| Trash | 18 min |

*Efficiency Equations*

Factors affecting trickling filter performance include total organic loading, total surface loading, total Kjeldahl nitrogen loading, total hydraulic loading, media depth, temperature of the wastewater, and the media type (Grady, 1999). There are several design equations that exist for the estimation of performance of any given biofilm system. Because of the complexity of the physical and biological characteristics of trickling filters, efficiency equations for ammonia and organic matter removal are difficult to formulate.

The design of nitrifying fixed-film reactors could be carried out using a formula based on Eckenfelder and Ford (Viessman, 1985). The applicable equation is

$$S_e/S_o = EXP(-KD/W^n)$$

where,

$S_e$ = final or effluent ammonia concentration, mg/l

$S_o$ = influent ammonia concentration, mg/l

K = reaction rate constant for ammonia oxidation, $min^{-1}$

D = depth of filter, ft

W = mass loading ($gpm/ft^2$*mg/l)

n = constant related to specific surface area and configuration of packing

The column volume, depth, and the surface area of the media remained constant throughout the study. Since D is constant, the equation simplifies to

$$S_e/S_o = EXP(-K/W^n)$$

The collected data can be plotted to determine the value of n and K for a specific media surface area.

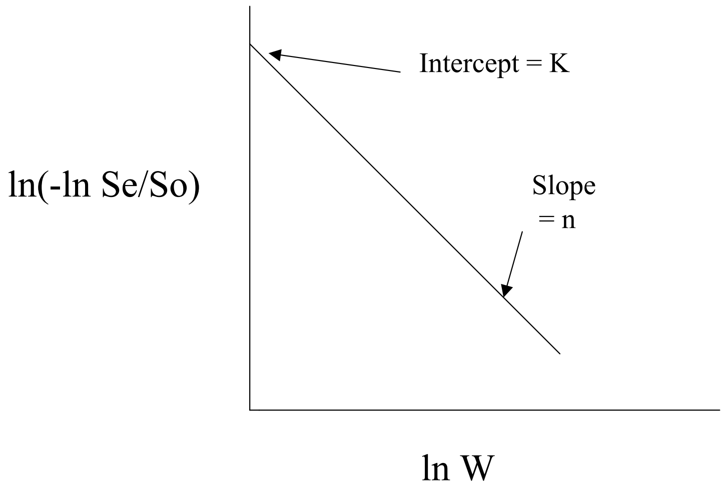



Figure 20 represents the procedure for determining n and K with this study's data. Once these constants are determined, the equation can be used to select the depth of a filter unit that will be required to treat a specific amount of ammonia. The constants for each media are given in Table 3. Solve the equation for D and the amount of media needed to meet this requirement is provided in the filter unit.

Table 3. K and n values for different media types for efficiency equations.

| Media | K value | n value |
| --- | --- | --- |
| Plastic | 0.07 | 0.70 |
| Rubber | 0.06 | 0.65 |
| Wood | 0.95 | 0.36 |
| Trash | 0.58 | 0.87 |

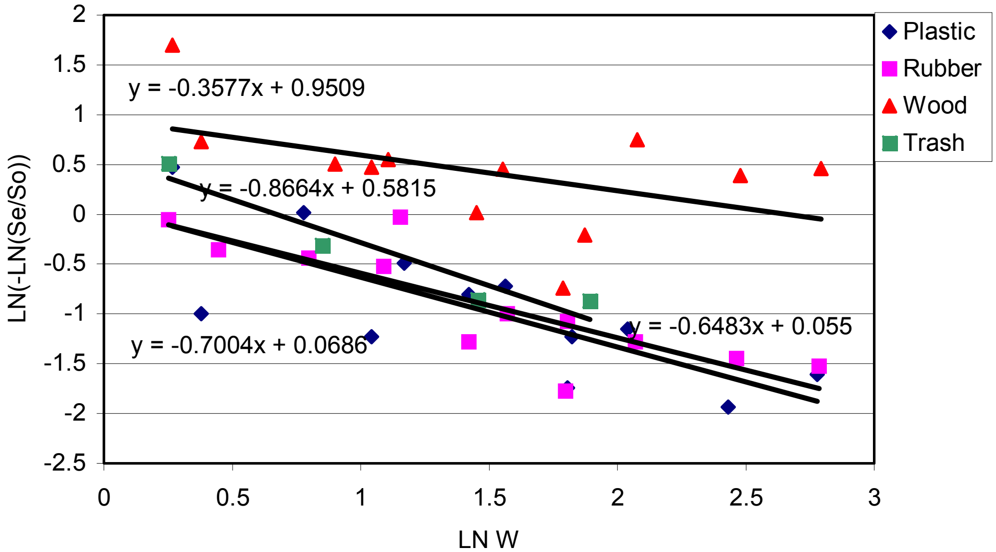



Figure 20. K and n determination for different media types for use in efficiency equations.

The K value determined in this analysis is the reaction coefficient at 20 ºC. The constant is corrected for temperature by the relationship:

$$K = K_{20}(1.035)^{T-20}$$

This is very important because temperature has a significant effect on nitrification.

*Filter Design*

Using the values obtained for the constants in the efficiency equations, treatment systems can be designed depending on the nitrification rate desired and the loading rate. The required ammonia removal rate, loading rate, K, and n can be used in the previous equation to design the systems. Figure 21 provides the required treatment depths for different nitrification efficiencies at different loadings. The designer can take the average loading to the system and choose the required depth for the filter unit off the plot. The effluent/influent ammonia concentration is plotted versus the loading rate in Figure 22. There are 3 different filter depths to choose from. This graph can be used to see how different filter depths affect the final effluent concentration at constant loadings. The effluent/influent ratio can be determined for specific loadings if the filter depth is known.

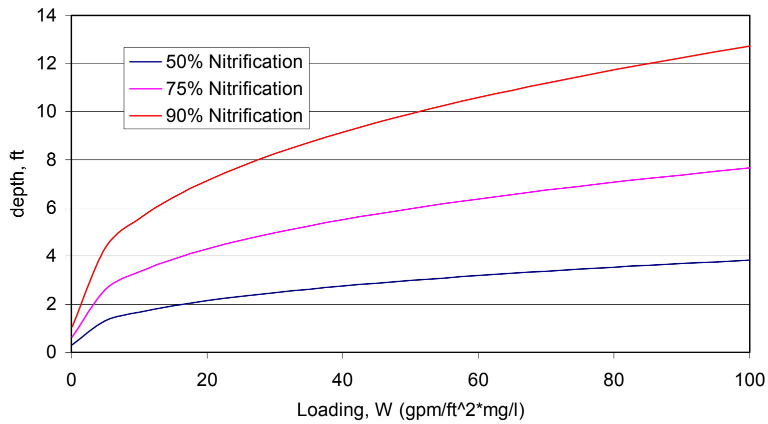



Figure 21. Oak wood chip filter depth needed for different nitrification rates at different loading rates.

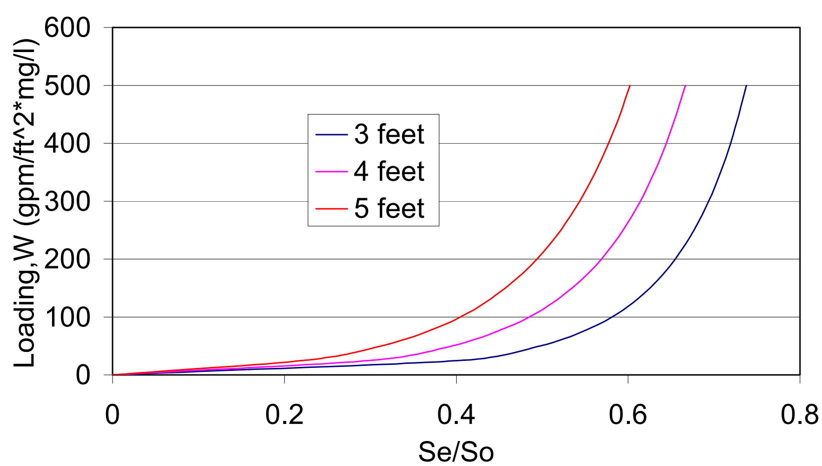



Figure 22. Loading rate vs effluent/influent ratio for different filter depths.

# CHAPTER 5: CONCLUSIONS

1. Utilizing a wood chip filter unit on top of a landfill to promote nitrification and ammonia removal in recirculated bioreactor leachate has been verified as an effective treatment option.
2. Wood chip media promotes average ammonia removal rates in the range of 77 – 87% for single pass leaching.
3. Hardwood chips would be a better packing media than pine chips because of their longer durability and slightly higher ammonia removal rate.
4. Landfill leachate from the Middle Peninsula leachate recirculating landfill in Glens, VA, contains no constituents which would inhibit nitrification.
5. Efficiency equations produced in this study can be used to size filter units.

# CHAPTER 6: THESIS SUMMARY

It is becoming popular to convert traditional landfills to bioreactor landfills because of the bioreactor's advantages, which include greater landfill capacity and more rapid waste stabilization. Though the bioreactor has many advantages over the traditional landfill, the recirculation of leachate results in very high ammonia nitrogen concentrations which create treatibility and disposal problems. The prices of the treatment technologies can become very expensive. One treatment option that has been proposed is to use a wood chip filter unit in the landfill to promote nitrification and ammonia removal in the recirculated leachate. This research has verified the effectiveness of this treatment option by utilizing lab-scale aerobic, downflow, biofilm reactors. Hardwood chips would be the better option over pine chips because of their longer durability and slightly higher ammonia removal rate. An incorporated zone for nitrification/denitrification could remove all unwanted constituents of the leachate in situ at very low costs since leachate recirculation is known to treat the leachate of COD, toxics, and metals (Pohland, 1995). It is recommended that full-scale studies be performed since no full-scale research has been documented on this filter unit treatment option. The data obtained from the full-scale studies would be suitable for use in the design of full-scale leachate treatment systems. Nitrogenous waste constituents would be completely attenuated if denitrification was incorporated into the system.

The data suggest that landfill leachate from the Middle Peninsula Landfill contains no inhibitors to nitrification and ammonia removal. It was a concern that recirculation might allow the leachate to dissolve inhibitory constituents but this was not the case in this specific system.

Chemical and biological oxygen demand must be removed or stabilized before nitrification can occur. Additional studies are needed to determine design criteria for biodegradable COD in leachate using biofilms on natural materials from landfills.

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## VITA

Matthew M. Clabaugh

The author received a B.S. in Environmental Science, Magna Cum Laude, from Virginia Tech in 1999. During the summer of 1999, he worked as an environmental technician with the Virginia Department of Environmental Quality. While pursuing his thesis at Virginia Tech, the author worked as both a research assistant and a teaching assistant