

EVALUATION OF LANDFILL LEACHATE TREATABILITY IN A MODIFIED
LUDZACK ETTINGER ACTIVATED SLUDGE SYSTEM

by

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(ABSTRACT)

The purpose of this study was to investigate treatment alternatives for a leachate from a municipal landfill in Roanoke County, Virginia. A continuous flow, laboratory scale, single sludge waste treatment system utilizing the Modified Ludzack Ettinger (MLE) nitrification/denitrification process with the addition of phosphorus and methanol was operated to determine its effectiveness in removing total nitrogen, organic compounds and metals from the leachate. The system was compared with the effectiveness of a totally aerobic activated sludge system operated in various configurations and at different mean cell residence times (MCRT's) with and without the addition of methanol. The ability of activated carbon to remove organics from the leachate was briefly examined.

The early results from this study indicated that the MLE process could successfully treat the leachate by removing up to 84% of the total nitrogen present while operating at

high MCRT's (eight days or greater). The MLE process removed as much as 56% of the total nitrogen at the lowest MCRT operated (1.5 days). By comparison, a maximum removal of only 30% was achieved for the completely aerobic activated sludge system operating under the same conditions.

The MLE process and the completely aerobic process were not effective in removing the organics from the leachate as evidenced by a leachate COD removal of only 2% to 5%. Limited studies with activated carbon proved successful in substantially reducing the COD from the effluent of the biological treatment systems.

The MLE process consistently reduced the iron concentration in the leachate from average level of 18.6 mg/l to levels below 1.0 mg/l.

Towards the end of this 584 day study there were indications that toxic agents were intermittently entering the leachate from the landfill as evidenced by two episodes of severe nitrification inhibition during operating periods when nitrification should have been essentially complete. This raises doubts concerning the ability of biological nitrification/denitrification to successfully treat the Dixie Caverns landfill leachate on a long term basis without special treatment for the removal of the unknown toxic components.

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Chapter I. INTRODUCTION

The long term ultimate disposal of solid municipal wastes has, in recent years, become a serious and major technical, political and emotional issue in many localities throughout the United States. During this year (1990) the Environmental Protection Agency (EPA) has estimated that 225 million tons of solid waste will be generated in the United States (1). The most economical method of disposing of these wastes is by landfilling, and approximately 90% of all solid wastes generated in the United States is disposed of in landfills (2). The problems involved with siting, permitting, operating, and closing landfills are extremely controversial and very highly publicized as can be seen by a brief review of any local newspaper. Although many problems can be mitigated by the proper design and operation of a sanitary landfill, problems such as blowing trash, offensive odors, gas and leachate production, noise, dust, rodents, insects and fires will exist to varying degrees at most landfills (1). Leachate collection, storage, treatment and disposal problems are expected to increase sharply in the state of Virginia as

more stringent regulations come into effect in the near future (25). It is probable that many leachate problems will result from "old" landfills, i.e., those no longer in use that have been idle for four or more years.

The Dixie Caverns Landfill is located in western Roanoke County in Virginia on approximately 60 acres of land. Refuse disposal operation at the landfill started in early 1965 and continued through July, 1976 (3). The landfill was utilized for both municipal and industrial wastes. Until 1968, the municipal wastes were dumped, burned and then covered with soil. During 1968, this disposal method was discontinued; and from then on, the waste was dumped, compacted and then covered (3).

In 1969, disposal operations began along a deep ravine on the southern boundary of the landfill through which a small natural stream flowed. Failure to install underdrains or piping to divert the natural water flow away from the refuse was the major cause for the leachate problems that still exist to this day (3).

Starting in October, 1971, Roanoke County attempted to obtain a permit from the state to operate the landfill as a solid waste disposal site in accordance with then newly issued State Health Department regulations (3). Initial

investigations by the state indicated severe surface water contamination. Several improvements were made by the county including the construction of a drainage cut-off ditch which drained to a holding pond at the lower end of the landfill. Both the ditch and the pond exist today and continue to function. However, these actions were insufficient to prevent the generation of or to properly control the leachate, and a long term landfill permit was never issued by the state (3). The lack of an operating permit caused the county to finally suspend landfilling operations in July, 1976.

More recent history of the landfill indicates that the leachate production and control problems have never been resolved. Personal observations starting in March, 1988 showed contaminated leachate from the covered fill area flowing into the holding pond from the drainage ditch. The holding pond being too small, the leachate was found to be either overflowing the pond's banks or draining out of the pond through a pipe. The leachate would then run off the landfill property and drain into a small stream which eventually joined the Roanoke River west of Roanoke City. Apparently prompted by complaints from local residents, the Virginia State Water Control Board began issuing a series of violations to Roanoke County for unpermitted discharges from the landfill (4).

In July, 1989, the county took action to prevent the leachate from flowing off the site. A large tank was installed to catch the pond overflow. On an almost daily basis, the leachate from the pond or the tank is pumped into a tank truck which transports the leachate to the Roanoke Country sewer system. This method of operation has continued to this day. Depending on how long the leachate continues to be produced, a more economical long term method of control might be for the county to pursue a permit to allow for the discharge of the leachate off-site after adequate on-site treatment.

During its operation, some areas of the landfill were utilized for the disposal of industrial wastes. An uncontrolled pile of flyash which contained high levels of lead and cadmium caused the EPA to place the landfill on the Superfund National Priorities List (5). However, this pile had no effect on the production or composition of the leachate in question discussed above as the areas where this pile was located did not drain through the municipal side of the landfill and to the pond.

The composition of leachates vary widely from landfill to landfill and depends on the age, hydrological factors, climatic factors, waste composition and design and operation

of the landfill (6,7,8,9,10,11,12,13,14,15,16,17,18). Even within a single landfill, the leachate can vary from location to location (19) and from one time period to the next (16). Generally the leachates from municipal landfills contain organic compounds, ammonia, alkalinity, hardness, and a variety of metals.

Characterization of the leachate from the Dixie Caverns landfill is presented in Table 4 in Chapter 4. The landfill has not operated for over ten years, and the leachate exhibits those characteristics of leachates from "old" landfills. These leachates typically possess low levels of BOD, COD and metals and relatively high levels of ammonia. Whatever organics are still in the leachate are usually non-biodegradable and typically consist of humic and fulvic like materials. This is compared with leachates from "new" or still operating landfills which can possess extremely high concentrations of organics, the majority of which are biodegradable volatile organic acids (11,13,14,15,16,17,18,20,21, 22).

Independent analysis of the organics in the leachate revealed the following priority pollutants (23):

1,4-Dichlorobenzene

1.60 ppb

Naphthalene	0.71 ppb
N-Nitroso-Diphenylamine	3.60 ppb
Chlorobenzene	10.0 ppb

A review of the characteristics presented in Table 4 indicates that the major problems resulting from the discharge of the Dixie Caverns leachate to a receiving water would be caused by the high ammonia-N concentration. Ammonia is toxic and can exert a high oxygen demand on receiving waters. The EPA criterion for freshwater aquatic life is 0.02 mg/l of un-ionized ammonia-N with the lowest reported lethal concentration of 0.2 mg/l for rainbow trout fry (24). At pH 7, which is approximately the highest pH of the Dixie Caverns leachate, the highest observed total ammonia-N concentration of 65 mg/l found in the leachate would result in an un-ionized ammonia-N concentration of about 0.26 mg/l. This concentration exceeds the EPA criterion and the lethal concentration reported above. In addition to being toxic, ammonia will consume dissolved oxygen in receiving waters when ammonia is oxidized to nitrate. Each mg of ammonia-N will utilize 4.57 mg of dissolved oxygen if fully oxidized (26). Depending on the dilution factor, the leachate from Dixie Caverns could exert a severe oxygen demand on the receiving waters. Also, nitrogen is an essential nutrient

for growth, and ammonia will stimulate aquatic growth which could lead to eutrophication and a decrease in the quality of the receiving waters (27).

A treatment system that either removes the ammonia-N from the leachate or converts it to nitrate would improve the quality of the leachate prior to discharging the leachate off site. However, although a nitrified leachate would be more preferable to one with a high amount of ammonia-N, these nitrates could also stimulate rapid aquatic growth and eutrophication (26). In addition, high concentrations of nitrates in drinking water can cause methemoglobinemia in infants, and because of this, the drinking water standard for nitrate-N is 10 mg/l (26). Therefore, removing the nitrate from the leachate would be a more desirable alternative.

As can be seen from Table 4, the only metal of relatively high concentration is iron. The compacted refuse in the landfill normally goes anaerobic very early in the life of the landfill (7,10,20). Under these conditions, the iron leaches from the refuse in a highly soluble reduced ferrous form (28). Once the leachate comes in contact with the atmosphere and is exposed to oxygen, the iron is oxidized to the ferric form and forms insoluble ferric oxides or hydroxides, which form reddish orange precipitates. This has been

observed on numerous occasions with the Dixie Caverns leachate. During periods when the leachate level in the holding pond was low, the leachate could be directly observed flowing from the drainage ditch into the pond. The leachate flowing into the pond was observed to be clear with no observable precipitate. As this leachate was aerated by agitation (for example, during transport of the sample back to the laboratory) a reddish orange precipitate would form. The leachate contained in the pond was always observed to be a reddish orange, and the reddish iron precipitate was observed wherever the leachate had previously overflowed from the pond. From an aesthetics point of view, the high iron concentrations in the leachate are not desirable; and the removal of iron should be a goal of any treatment process.

From Table 4, it is evident that the COD of the Dixie Caverns leachate should not be considered much of a problem because its concentration was always low and there was essentially no BOD associated with it. Its removal from the leachate, therefore, should not be a high priority during treatment.

Nitrogen can be removed from leachate by biological or by physical-chemical processes (24). Biological removal

processes have shown much promise for efficiently and economically removing nitrogen down to satisfactory levels (26,29,27,30,15,16). One such process that has been proven effective for nitrogen removal from municipal wastewaters is the Modified Ludzack-Ettinger (MLE) activated sludge process (15,30). Completely aerobic activated sludge systems only utilize nitrification to convert the ammonia-N and organic-N of the leachate into nitrate-N, thereby leaving most of the nitrogen in the effluent. The only nitrogen removal accomplished would occur with the removal of waste solids (27). The MLE process, however, includes both an aerobic zone for nitrification, and an anoxic zone for denitrification, a biological process which converts the nitrate-N produced from nitrification into gaseous nitrogen. This nitrogen gas leaves the system and becomes part of the atmosphere, thereby resulting in complete removal from the wastewater being treated without additional environmental impacts (30). Because of these potential benefits, it was decided to investigate the ability of the MLE system to treat the leachate from the Dixie Caverns Landfill.

Specifically, the objectives of this study were:

(1) to study the ability of the MLE system to biologically treat a leachate that exhibited the characteristics of

being from an "old" landfill, and compare the effectiveness of total nitrogen removal, organics removal and iron oxidation and removal with the effectiveness of a completely aerobic activated sludge system;

(2) to study the effect of the anoxic zone on nitrification in the MLE process by operating the system at low mean cell residence times and comparing the performance to that of the completely aerobic system;

(3) and, to study alternative methods such as activated carbon adsorption for the removal of bioresistant organics.

This investigation was part of an overall study whose goal was to develop information that could be used to formulate treatment procedures for landfill leachates based on their chemical characteristics. This study was conducted in cooperation with researchers at the University of British Columbia, and was sponsored by the Virginia Environmental Endowment.

Chapter II. LITERATURE REVIEW

There are a myriad of treatment alternatives for landfill leachates. In the past, physical, chemical, and biological treatment methods have been used with varying degrees of success. Alternative treatment methods such as recycling the leachate back through the landfill or piping or transporting the leachate to a municipal sewage treatment plant have also been attempted. This section presents a review of these treatment alternatives as specifically related to their applicability to the treatment of old leachates such as that discharged by the Dixie Caverns Landfill. First, alternative treatments such as land application and recycling of leachate will be examined. Physical-chemical treatment methods will be reviewed next followed by biological treatment methods. Also presented is a review of the Modified Ludzack Ettinger system and the nitrification/denitrification processes that are utilized by the system.

ALTERNATIVE TREATMENT OF LEACHATE

Land Application and Leachate Recycle

Simply collecting the leachate in a holding pond and spraying it on nearby land may be the least expensive method of treatment. Robinson (31) recommended this approach with the suggestion that the soil would purify the leachate by natural biological, physical (filtration and evaporation) and physical/chemical (ion exchange and adsorption) processes. In addition, the organics contained in the leachate could possibly help plant growth by acting as a fertilizer. Lema et al. (17), however, disagreed with this recommendation calling the spraying of leachate a noxious process which runs the risks of odors, contamination of ground and surface waters and plant toxicity problems. Lema et al. (17) also cited leachate irrigation research which showed that, even with the leachate diluted 5 to 20 times, plant growth using leachate was less than with conventional nutrient solutions. Personal observation of the Cloyd's Mountain Landfill in Pulaski County, Virginia, showed that the landfill operators were using this technique for leachate control with limited success as leachate could easily be observed flowing off the site at the bottom of the landfill. Land applying the Dixie Caverns leachate is probably not a good alternative. Besides

the problems discussed above, little land is available around the fill area for spraying. Spraying on the fill area would in essence be recycling which is discussed next.

A number of authors discuss the recycling of the leachate back through the landfill (6,10,14,15,17,21,32). Leachate recycle, accomplished by spray irrigation, can take advantage of the biological activity in the landfill and increase the rate at which stabilization of the landfill material occurs. This can reduce the required time period before a landfill site could be put to its ultimate use. Biological activity within the landfill would metabolize the organics in the fill and convert the organic nitrogen to ammonia. The mechanisms for ammonia removal from the leachate would be volatilization during spraying and by plant uptake. Any iron in the leachate would be oxidized during reaeration and would precipitate out. The volume of leachate would be reduced by evaporation and transpiration.

However, there can be problems with this type of treatment. Robinson et al. (14) did find that the COD of leachate draining from a recycled area of a landfill decreased at a faster rate than the COD of the leachate draining from a control area without recycle. However, they found that the rate of COD decrease was not as great as would be expected

based on the results of small scale studies. They also found that although some leachate volume was reduced by evaporation, much of the reduction in volume that occurred was due to storage of leachate in perched areas of the landfill. Lema et al. (17) also emphasized the need to carefully consider the site hydrology before recirculation is attempted to ensure that the groundwater is not in danger of contamination and that the recycled leachate is not stored in the landfill. In addition, even after recirculation, the final effluent may not yet be fit for discharge. Concentrations of ammonia, nonbiodegradable COD, chloride, and metals may still be high (14,17,21). Volatilization of ammonia could be very slow if the pH is low which causes essentially all the ammonia to be in the ionized form. Additional treatment, therefore, may be necessary prior to releasing the leachate off-site.

Recirculation of an old leachate such as that from the Dixie Caverns Landfill would probably not be advantageous. Based on the low BOD concentrations monitored in the leachate, it appears that the organics in the landfill are already stabilized. Given the age of the landfill, it is doubtful that much is known about the hydrology of the landfill or the surrounding area. Recirculation, however, may eventually shorten the ultimate time period during which leachate control at the landfill must be continued simply by washing the

contaminants out of the landfill at a faster rate.

Recycling appears to be best suited for "new" or operating landfills that have liners and leachate collection systems. Use of recycling would increase the rate of fill material decomposition and reduce the time period that leachate would have to be collected and treated.

Combining With Municipal Sewage

Piping or transporting landfill leachate to a sanitary sewer or, more directly, to a municipal sewage treatment plant has been recommended by several researchers (9,12,17). Lema et al.(17) actually proposed a benefit of combined treatment. Leachates normally contain an excess amount of nitrogen but little phosphorus. Municipal sewage typically has an excess amount of phosphorus. Therefore, by mixing the two wastes, phosphorus would not need to be added to the leachate, and phosphorus removal by the municipal plant would be improved. However, the combination of wastes would add substantially to the nitrogen load of the plant.

Lema et al., Metry et al.(9) and Boyle and Ham (12) all indicated that combined municipal/leachate is feasible only if the volume of leachate makes up less than 5% of the total sewage. This limitation appears to be based on the results of laboratory studies performed by Boyle and Ham (12) during

which they utilized extended aeration to treat combined sewage/leachate with the percentage of leachate ranging from 0 to 20%. They found that above 5%, system effluent BOD, effluent COD, oxygen uptake rate, solids produced, and the sludge volume index all substantially increased. Along with the 5% limitation, Boyle and Ham also recommended that the maximum organic loading of the combined sewage/leachate waste be less than 27 lb BOD/100 lb MLSS/day. The COD concentration of the leachate that they utilized was 10,820 mg/l, and the COD of the sewage was only 240 mg/l. Clearly, 5% of this leachate mixed in with the sewage would have a significant impact on the final mixture's strength and on the ability of the system to treat the combined waste. This would not be the case with an old, much weaker leachate and it could be combined with municipal sewage at a much higher percentage without causing significant degradation of treatment plant operation.

Kelly (33) found that using up to 16% leachate by volume did not create any significant problems although sludge production did increase substantially. However, the leachate used was much weaker than that used by Boyle and Ham. The COD of the leachate was 1178 mg/l, and the BOD concentration was 400 mg/l. At 16% the loading to the treatment plant was 22 lbs BOD₅/100 lbs MLVSS, below the maximum of 27 recom-

mended by Boyle and Ham. A percentage by volume cutoff should therefore not be used when attempting to determine whether to use combined treatment. Instead, the maximum loading recommended by Boyle and Ham is a more useful guide.

Lema et al.(17) mentioned several other potential problems with combined treatment such as corrosion, operational problems in the plant due to the precipitation of iron oxides, frothy scum and high effluent concentrations of ammonia. In addition, the high concentrations of metals may inhibit the biological activity of the sludge and may prohibit land application as an alternative for sludge disposal. During his study Kelly (33) found that the metals content in the sludge did not reach the levels which would prohibit its use for agricultural purposes.

An old leachate such as that from the Dixie Caverns Landfill appears to be an excellent candidate for combined treatment. As can be seen in Table 4, the leachate from Dixie Caverns is very weak compared to many other leachates, especially when compared to the strength of the leachates used by Boyle and Ham (12) and Kelly (33) in their combined treatment studies. In addition, the amount of leachate generated at the landfill, although it varies widely depending on rainfall, averages only about 20,000 gallons per day

(34). With a maximum BOD_5 concentration of about 10 mg/l, only 1.6 pounds of BOD_5 per day would be produced; an extremely low loading for virtually any municipal waste treatment plant. Indeed, to meet Boyle and Ham's recommended limit of 27 lb BOD_5 /100 lb MLSS/day, a one million gallon volume plant operating with a 3000 mg/l MLSS concentration would need to be fed with Dixie Caverns leachate at the rate of 86 million gallons per day.

In addition to little BOD, the metals concentrations in the Dixie Caverns leachate are low compared to many leachates reported in the literature. For example, the iron concentration in leachates has been found to be as high as 330 mg/l (35) and the manganese concentration as high as 170 mg/l (6). The maximum iron and manganese concentrations measured in the Dixie Caverns leachate were 30.5 and 1.2 mg/l respectively. These concentrations are probably a function of soil constituents, so low metals concentrations are likely at many sites. These low metals concentrations should not cause any toxicity problems in a municipal treatment plant.

As was discussed in Chapter 1, Roanoke County is currently controlling the Dixie Caverns leachate by collecting it on site and transporting it to the nearest sanitary sewer. The leachate eventually flows to the Roanoke Water Pollution Control Plant which is a 35 million gallons per day plant.

This combined treatment has apparently caused no observable effects (34).

PHYSICAL-CHEMICAL TREATMENT

A number of researchers have examined the ability of physical and/or chemical processes to treat municipal leachates. Examples of physical-chemical treatment processes include chemical addition (lime, ferric chloride, alum, ferrosulfate polymers, sodium hydroxide) followed by precipitation and sedimentation, activated carbon, breakpoint chlorination, ozonation, air stripping, ion exchange and reverse osmosis. Lema et al. (17) discusses several advantages of physical-chemical treatment which include immediate startup, easy automation, insensitivity to temperature changes, and the simplicity of the plant and materials required. These are outweighed by disadvantages such as the large amount of waste sludge produced and high operational and chemical costs. Lema et al. (17) recommended that such treatment be used; as pre or post treatment to complement biological treatment, to treat leachates from old landfills, or to remove a specific pollutant from the leachate. Metry and Cross (9) stated that physical-chemical treatment should be used prior to biological treatment to remove heavy metals or other inhibitory compounds. A few examples of physical-

chemical treatment are provided below.

Removal of Organics

Cook and Foree (6) performed numerous tests on leachate involving the use of physical-chemical treatment. During one test, activated carbon in a column was used as a final polishing step to treat the effluent from batch biological reactors treating leachate. The carbon reduced the COD concentration from about 300 mg/l down to 100 mg/l and also reduced color. Bleach was used to treat the effluent and was found to be effective in removing color but ineffective at COD removal. Cook and Foree also attempted to treat raw leachate with a COD concentration of 17,500 mg/l by chemical addition and precipitation/coagulation utilizing a number of different chemicals. In general, they found that this method of treatment was ineffective in reducing the COD of the leachate because most of the organics were soluble. They found varying degrees of color removal and good total suspended solids removal.

Randall and McClintock (19) added lime to raw leachate with a COD of 562 mg/l, aerated it for one hour, and then settled the resulting solution. Virtually no reduction of COD occurred. Using the same leachate, activated carbon was used in a jar test to lower the COD of the effluent from a

batch biological treatment system. Carbon at a concentration of one gram/liter mixed for 17 hours removed 30% of the residual COD.

Millot et al. (18) performed combined biological and physical-chemical treatment on leachate. They found that biological treatment was ineffective in removing organic compounds with molecular weights above 5000. However, they did find that coagulation/flocculation followed by sedimentation was effective in removing those high molecular weight compounds. Such combined treatment might not be effective for other leachates with different organic characteristics. For example, Gourdon et al. (36) found that in a leachate from an industrial landfill, 87% of the organic carbon that resisted combined anaerobic and aerobic biological treatment was contained in compounds with molecular weights less than 500. Only about 2.5% of the biologically resistant organic carbon was associated with compounds with molecular weights greater than 5000. In this case, coagulation/flocculation would probably not have substantially reduced the organics in the treated effluent and would not have been cost effective. However, activated carbon should be effective at removing the remaining organics (37).

Chain and Dewalle (11) summarized several experiments which evaluated the effectiveness of activated carbon in

removing organics from landfill leachates. Activated carbon removed from 34% to 94% of the organics in the leachate under a variety of conditions. The higher levels of removal occurred either when the carbon was used on effluent from biological systems treating leachate from young landfills or when used to treat raw leachate from old landfills. The carbon appeared to be the least efficient when treating raw leachate from young landfills. In particular, Chain and Dewalle (11) reported on a study of COD removal by activated carbon from an old, stabilized landfill, during which it was found that the carbon removed 85% of the COD. In a separate study, Chain and Dewalle (37) found that the major organic fraction removed by activated carbon were fulvic acid-like materials with molecular weights between 100-10,000. The authors found that the low molecular weight compounds (such as volatile organic acids found in young leachates) and the very high molecular weight humic-like materials were poorly removed by the carbon. The percentage of this fulvic-like molecular weight fraction contained in the leachate increases as the age of the landfill increases (11). It would be expected, therefore that as a landfill ages, activated carbon would be more effective at removing organics.

Powdered activated carbon has been combined with biological treatment to remove organics in leachates. Ying, et

al. (38) added powdered activated carbon directly to biological sequencing batch reactors treating a high strength leachate. This combined physical/biological treatment reduced the leachate TOC concentration from 3570 mg/l to a low of 121 mg/l. This was compared to an effluent TOC concentration of 286 mg/l from a control sequencing batch reactor (no carbon added). The authors also found that nitrification and denitrification continued to occur unaffected.

Chain and Dewalle (11) reported that reverse osmosis has been shown to remove anywhere from 56% to 98% of the organics from leachates.

Removal of Metals

Precipitation can be effective in removing metals from leachates. Randall and McClintock (19) added lime to raw leachate followed by aeration for one hour. They found that lime at 500 mg/l precipitated about 80% of the iron contained in the raw leachate, 92% of the manganese and 71% of the zinc.

Removal of Nitrogen

Few studies involving physical-chemical treatment of leachates with the goal of nitrogen removal have been conducted. Many of the treatment studies were only designed

with organics removal in mind due to the fact that for many leachates, the organic concentrations are very high. Most of the nitrogen in leachates is in the ammonia form. The basic physical-chemical methods that could be utilized for ammonia removal from leachates are air stripping, breakpoint chlorination and ion exchange (24).

All three methods have disadvantages. In order for air stripping to be effective, the pH must be raised to between 10.8 to 11.5 to convert the ammonium ion to the more volatile unionized ammonia (24). In addition, operation of a stripping tower is problematical at temperatures below 32F. Chlorination has the advantages of being highly efficient and insensitive to cold temperature. However, the addition of chlorine increases the dissolved solids concentration of the leachate and could lead to the production of trihalomethanes and other chlorinated organics unless pretreatment to remove organics is established. Trihalomethane formation could be a particular concern with leachates because the organic precursors for formation are typically humic and fulvic materials and these are usually contained in leachates (14,24). Ammonia removal by ion exchange may be the most complicated of the three methods primarily due to the problems involved with regeneration of the ion exchange medium (usually clinoptilolite). In addition, pretreatment of the leachate involv-

ing coagulation/precipitation and filtration would most likely be necessary before passing the leachate through an ion exchanger (24).

Keenan et al. (35) conducted a full scale test of leachate treatment during which air stripping of ammonia was conducted prior to the leachate entering biological treatment aeration tanks. An ammonia stripping lagoon followed a chemical/precipitation-clarification unit and took advantage of the clarification unit effluent's high pH. The purpose of the lagoon was to reduce the ammonia concentration to levels at which it would not be toxic to the activated sludge downstream of the lagoon. The lagoon reduced the ammonia-N concentration from an average of 1000 mg/l to an average of 422 mg/l; a reduction of 58%.

Criteria for Decision to Use Physical-Chemical Treatment

Chain and Dewalle (11) proposed a method by which the characteristics of the organics in the leachate and the age of the landfill can be used to determine whether physical-chemical or biological treatment would be more effective. For example, the COD to TOC ratio is greater than 2.8, the BOD to COD ratio is greater than 0.5 and the COD concentration is greater than 10,000 mg/l, i.e., typical of a landfill less than four years old, biological treatment will have a

good removal efficiency and physical-chemical treatments will generally have poor removal efficiencies. However, if the COD to TOC ratio is less than 2.0, the BOD to COD ratio is less than 0.1 and the COD is less than 500 mg/l, i.e., typical of a landfill greater than ten years old, biological treatment will not be efficient but physical-chemical processes such as reverse osmosis or activated carbon will have good organic removal efficiencies. For leachates that possess characteristics in between the two extremes discussed above, Chain and DeWalle stated that only reverse osmosis caused good organic removal efficiencies. All the other treatment processes, both physical-chemical and biological, were only rated as fair.

Applying Chain and Dewalle's criteria to the leachate from Dixie Caverns produces mixed results. The landfill has not been operating for at least 10 years, the COD concentration of the leachate (average = 90.2 mg/l) is less than 500 mg/l and $BOD/COD = 4.82/90.2 = .0534$ which is less than 0.1. Each of these criteria would point towards physical-chemical treatment. However, for the Dixie Caverns leachate $COD/TOC = 90.2/27.8 = 3.24$ which is much greater than 2.8 and would indicate that biological treatment would be more efficient. The 3.24 ratio is higher than all but one of eleven ratios for various leachates reported by Chain and Dewalle; many of

these leachates possessed substantial amounts of BOD and were treated very effectively by biological treatment. It is possible that, since both the COD and TOC concentrations are very low for the Dixie Caverns leachate, the ratio of these two parameters is somehow skewed. Considering only biodegradable organics, the BOD_5/TOC ratio = $4.82/27.8 = 0.173$ which is well below 2.

Using the BOD_5/TOC ratio instead of the COD/TOC ratio, all parameters of the Dixie Caverns leachate would indicate that some kind of physical-chemical process should be used if treatment of the leachate is required. The parameters also indicate that biological treatment would not be an effective treatment technique for this leachate. However, the characteristics that Chain and Dewalle recommended be examined in order to determine an effective treatment method only consider organic carbon. The need to remove ammonia from leachates was not considered by them. Considering the disadvantages involved with physical-chemical ammonia removal processes, biological treatment may be the most practical method for nitrogen removal from leachates even though the other characteristics of the leachate indicate that biological treatment would not be effective.

BIOLOGICAL REMOVAL OF TOTAL NITROGEN

Removal by Bacterial Uptake

For a number of leachates whose characteristics are provided in the literature, effective nitrogen removal can be achieved by bacterial uptake during COD treatment. However, the leachate would have to possess a sufficient amount of biodegradable COD and be close to or actually nitrogen limited. Leachates from young or currently operating landfills would make the best candidates for such treatment. Indeed, for some leachates nitrogen may have to be added as a nutrient so as to not limit organic metabolism.

Cook and Foree (6) used aerobic biological systems to treat a leachate with a BOD of 7,100 mg/l and a total nitrogen of 299 mg/l. This provides a BOD:N ratio of 23.7:1 which is slightly higher than the normally recommended 20:1 ratio (26). However, 270 mg/l of the total nitrogen was organic nitrogen. The authors decided to add 500 mg/l nitrogen as ammonium chloride to two of 4 identical treatment systems; the other two systems not receiving any nutrients. The effluent results indicated that the addition of nitrogen provided marginal improvements in COD/BOD removal but reduced the quality of the effluent from a total nitrogen point of view. The systems which treated only raw leachate reduced the COD from 15,800 mg/l to 350 mg/l and the BOD from 7,100

mg/l to 25 mg/l. The effluent total nitrogen from these systems averaged 18.7 mg/l of which 4.2 mg/l was nitrate nitrogen. The systems that had ammonia added to them reduced the COD and BOD to 300 and 11 mg/l respectively. The effluent total nitrogen from these systems was 52 mg/l of which 30 mg/l was nitrate nitrogen. These results clearly show that excess nitrogen was added. The excess ammonia that was not taken up was either lost due to volatilization, nitrified (hence, the higher effluent nitrates) or denitrified in the biological flocs. It is not clear whether or not the leachate was nitrogen limited at all. The total phosphorus concentration in the raw leachate was only 13 mg/l which provided a BOD:P ratio of 546:1 which is well above the 100:1 ratio recommended (26) indicating that the leachate was phosphorus limited. The authors added 100 mg/l of phosphorus to the same two systems that ammonia was added to; the other two systems were not provided any phosphorus (or nitrogen). It is possible that it was the additional phosphorus that caused the improvement in COD removal, not the additional nitrogen.

Uloth and Mavinic (8) performed a series of aerobic treatment experiments on leachate from a lysimeter with a BOD:N ratio of 33:1 and a BOD:P ratio of 183:1. Nitrogen and phosphorus were added to first provide a BOD:N:P ratio of 100:6.4:3.1. Addition of nutrients was later reduced to

provide a ratio of 100:5:1.3 with no apparent limiting effects. The best organics removal achieved during the study was a COD reduction from 48,000 to 386 mg/l and a BOD reduction from 36,000 to 27 mg/l.

Robinson and Maris (14) utilized aerobic treatment for a high strength leachate with a BOD of 11,900 mg/l and a total nitrogen concentration of 800 mg/l (BOD:N ratio of 14.9:1). The ammonia concentration in the leachate was 790 mg/l. At a mean cell residence time of 20 days, the BOD was reduced to 10 mg/l and the ammonia concentration was decreased to 1 mg/l by bacterial uptake.

Using a BOD:N ratio of the influent wastewater to determine the effectiveness of nitrogen removal by bacterial uptake is too simplified an approach. The nutrient requirements and the nutrient removal ability of a biological treatment system depends on the biological solids produced during the treatment process. In addition to the characteristics of the influent wastewater, the solids produced are also dependent on the mean cell residence time (MCRT). The lower the operating MCRT of the system, the higher the amount of solids produced. As the MCRT is increased, the amount of solids produced decreases (26). For example, Sherrard and Schroeder (60) found that, while treating a wastewater with a COD of 269 mg/l, 34 lbs of cells were produced per 100 lbs of COD

utilized at a two day MCRT. However, while treating the same wastewater, only 17 lbs of cells were produced per 100 pounds of COD utilized.

Stoichiometric considerations can be used to estimate the amount of nitrogen removal by uptake that can be expected at a given MCRT. Sherrard (61) developed balanced stoichiometric equations which represent the treatment of wastewater consisting of glucose, ammonia, and phosphate. Based on the equations provided by Sherrard, an activated sludge system treating 100 mg of COD as glucose would require (and therefore remove) 1.85 mg of nitrogen (a COD:N ratio of 100:1.85) while operating at an MCRT of 20 days. At an MCRT of 15 days, 2.2 mg of nitrogen would be required while treating the same 100 mg of COD (a ratio of 100:2.2). When reduced down to a 3 day MCRT, the system would require 3.45 mg of nitrogen (a ratio of 100:3.45). These numbers would have to be adjusted somewhat if glucose is not the predominant COD constituent of the wastewater and if only a portion of the COD is biodegradable.

Besides being practical for only a limited number of leachates, there is a disadvantage to relying on bacterial uptake to remove nitrogen from leachates. Using aerobic biological treatment to reduce the COD of the leachate (and to remove nitrogen) would result in a relatively large amount

of solids being produced that would have to be disposed of. For young or operating landfills with extremely high concentrations of biodegradable organics, an anaerobic biological process should probably be used in front of any aerobic system. Anaerobic processes have been shown to be effective in substantially reducing leachate COD concentrations (11). One of the advantages of the anaerobic process is that less biomass is produced per unit of substrate metabolized (26). This would result in much less nutrient uptake and less nitrogen removal from the leachate. Additional treatment for nitrogen removal may be necessary.

Nitrogen removal by biological uptake is not feasible for an old leachate like that from the Dixie Caverns landfill. The BOD:N ratio is 4.82:58.3 which is equivalent to 0.083:1 which indicates that nitrogen is substantially in excess.

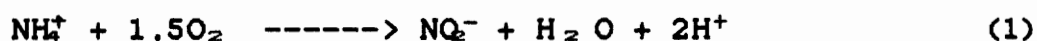
Nitrification and Denitrification

Nitrification and denitrification is a two stage process which completely converts ammonia to nitrogen gas which then escapes into the atmosphere. In the first stage (nitrification) *Nitrosomonas* bacteria oxidize the ammonia to nitrite. *Nitrobacter* then oxidize the nitrite to nitrate. In the

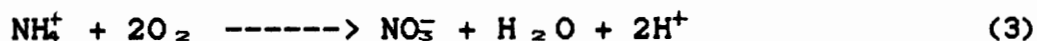
second stage (denitrification) heterotrophic bacteria utilize the nitrate as a terminal electron acceptor while metabolizing organic material. In the process the nitrate is reduced to nitrogen gas which escapes into the atmosphere. The nitrogen, therefore, is completely removed from the waste being treated. Both processes will be separately examined in more detail. Then several examples of research using nitrification/denitrification to remove nitrogen from leachates will be discussed. Last, a discussion of the Modified Ludzack Ettinger system will be provided.

Nitrification

Reactions 1 and 2 below present the two separate processes of nitrification as conducted by *Nitrosomonas* and *Nitrobacter* respectively (27).



Combining the two reactions results in the following:



Nitrite has rarely been observed to accumulate in municipal waste treatment systems (27). This indicates that the rate limiting step is the conversion of ammonia to nitrite by *Nitrosomonas*. Accordingly, most of the literature emphasizes factors that effect *Nitrosomonas*.

Both the *Nitrosomonas* and *Nitrobacter* genera of bacteria are chemoautotrophic. These bacteria obtain their carbon for cell growth from inorganic compounds such as carbon dioxide, bicarbonate and carbonate (27). Note from the equations above that aerobic conditions are necessary to provide oxygen for nitrification to occur. Barnes and Bliss (27) stated that 4.2 grams of oxygen are utilized for every gram of ammonia-nitrogen nitrified to nitrate-nitrogen. They provided a summary of research conducted to determine the effects of dissolved oxygen concentration on nitrification. The reported minimum dissolved oxygen concentrations required in order that the concentration would no longer have an effect on the nitrification rate varied widely from 1.0 to 7.5 mg/l. However, Barnes and Bliss concluded that it appeared that the minimum level required was higher for activated sludge plants than for pure cultures. A possible explanation for this observation is that, in an activated sludge plant where the nitrifiers represent only a small fraction of the total biomass, oxygen diffusion effects limit the amount of oxygen penetrating the biomass flocs. The nitrifiers inside the biomass flocs would therefore see little oxygen. This would require a higher dissolved oxygen concentration to maintain a given level of nitrifier activity. However, an activated

sludge plant treating a waste with little BOD may be able to sustain nitrification at lower dissolved oxygen levels (27). Benefield and Randall (26) recommended that for combined carbon oxidation/nitrification processes, the dissolved oxygen concentration should be at least 2 mg/l in order to eliminate any dissolved oxygen limiting effects. As the leachate from Dixie Caverns has little BOD, maintaining a sufficient level of dissolved oxygen for nitrification in an activated sludge reactor should not present any difficulty.

The nitrification reactions also indicate that hydrogen ions are produced during nitrification causing alkalinity to be destroyed. Theoretically, approximately 7.2 mg of alkalinity as calcium carbonate are destroyed for every 1 mg of ammonia-nitrogen nitrified (24). Sufficient alkalinity would need to be present to prevent a substantial drop in pH. The maximum TKN observed in the leachate from Dixie Caverns was 70.6 mg/l. If all this TKN was to be nitrified, 508 mg/l of alkalinity would be required. The minimum alkalinity measured in the leachate was 541 mg/l indicating that sufficient alkalinity would exist for nitrification.

Barnes and Bliss (27) indicated that the optimum temperature for nitrification is in the range of 30-36C with an

overall range of 4-50C over which nitrification can occur. Results from studies performed by Wild et al. (39) indicated that nitrification rates drop off quickly as the temperature is decreased from 30C. For example, the rates at 20C and 10C are 62% and 28% of the rate at 30C respectively. Buth (40) attempted to summarize numerous studies on the effects of low temperatures on nitrification. He reported that below 5C, very little nitrification occurred unless the influent ammonia concentrations were low. Between 5C and 10C, nitrification was often possible but with varying results. Above 10C, high rates of nitrification occurred in almost all cases.

The pH of the solution in the activated sludge reactor also effects the nitrification rate. Barnes and Bliss (27) provided the results of a number of experiments to determine the effects of pH on nitrification rates and concluded that the optimum pH lies in the range of 7.5 to 8.5. Wild et al. (39) found the optimum pH range to be 8.2-8.6 with less than 50% of the optimum nitrification activity occurring when the pH was below 7 or above 9.8.

The leachate from Dixie Caverns landfill has an average pH of 6.72. This would result in a nitrification rate of only 35% of the optimum rate based on Wild's results. This

could be a cause for concern for the Dixie Caverns leachate if nitrification treatment were to be utilized; however, there are two mitigating factors involved. Benefield and Randall (26) cited research that indicated that nitrifiers are capable of acclimating to lower than optimum pH's and reestablishing their maximum growth rates. In addition, with the high amounts of alkalinity contained in the Dixie Caverns leachate, the leachate is supersaturated with respect to inorganic carbon. Stripping of this excess carbon would occur in an aerated activated sludge reactor where nitrification takes place. Removing carbon dioxide reduces the carbonic acid concentration and should cause the pH to increase (28). For the Dixie Caverns leachate, this increase in pH would be towards the pH range in which higher nitrification rates would occur.

A number of researchers have attempted to determine the relationship between mean cell residence time (MCRT) and nitrification. Lawrence and McCarty (41) found that the minimum MCRT for nitrification at 20C was between 2 and 4 days. Benninger and Sherrard (42) found that nitrification began at a MCRT of 3 days and was complete at a MCRT of 10 days. Wuhrmann (29) stated that a 4 to 5 day MCRT will yield complete nitrification at 8-10C. Above a temperature of 18C,

2 to 3 days are sufficient for stable nitrification. Beckman et al. (43) found that a minimum MCRT of 3 days was required for nitrification to occur at 18C with nitrification being complete at 6 days. Balakrishnan and Eckenfelder (44) found that at 23C, a minimum MCRT of 3 to 4 days was required for obtaining good nitrification.

A number of researchers have found that the amount of nitrification decreased as the organic loading to combined carbon oxidation/nitrification treatment systems increased. As the nitrifiers are typically only a small fraction of the total biomass population in a combined treatment system, they will be distributed throughout the biomass flocs instead of only on the surface. On the inside of the flocs, the dissolved oxygen concentrations can be very low (26) which will inhibit the nitrifiers. Increasing the organic loading relative to the ammonia loading increases the concentration of the heterotrophs to the nitrifiers and increases the competition for limited amounts of oxygen in the flocs. This could result in a decrease in the amount of nitrification that occurs. Balakrishnan and Eckenfelder (44) varied the loading of domestic waste to combined carbon oxidation/nitrification treatment systems from 0.15 lb BOD/lb MLVSS/day to 0.88 lb BOD/lb MLVSS/day by increasing the flow

rate. Based on decreasing effluent nitrate concentrations and increasing effluent TKN concentrations, they found that nitrification decreased as the organic loading increased. They also made the same conclusion based on testing synthetic wastes at the same flow rate but with increasing BOD concentrations and constant ammonia concentration. In this case however, they did not provide sufficient data to support their conclusion. Effluent nitrate concentrations were provided, and they did decrease with increased loading. However, effluent TKN concentrations were not provided. It is possible that the decreased nitrate concentrations were due to increased ammonia uptake by bacteria thereby leaving less nitrogen available for nitrification. In addition, the same solids concentration was maintained in each treatment system. Therefore, the higher the loading to a given system, the more solids that were wasted from that system. This in effect caused the systems with the highest loading rates to have the lowest MCRT's. Indeed, the two systems with the highest loading rates had MCRT's of only 1.9 and 1.6 days which would be very marginal for nitrification at the operating temperature of 23C. The decrease in nitrates from the systems with the higher loading rates, therefore, could be due to a decrease in nitrification caused by the washing out of the nitrifiers and not due to high organic loading.

However, the results of their domestic waste experiment (the first study discussed) of their study appears to support the conclusion that increased organic loading caused a decreased in nitrification. The authors recommended that an organic loading factor of 0.3 or 0.4 should not be exceeded in order to obtain good nitrification.

Stove and Kincannon (47) studied the effects of varying the influent COD:TKN ratio on nitrification in single sludge nitrification systems. They concluded that nitrification decreases as the COD:TKN ratio increased.

Wang (16) utilized an intermittent aeration single sludge system to treat a leachate from an operating landfill. He found that the system achieved greater than 85% nitrification of the available ammonia when the influent BOD/TKN ratio was below 4.2:1. Nitrification, however, was inhibited by increasing BOD/TKN ratios. When the influent BOD/TKN ratio increased to 7.6:1, nitrification dropped off to only 50%.

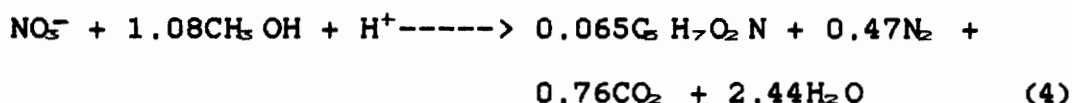
Not all researchers have found an organic loading/nitrification effect. Sawyer et al. (45) performed studies to determine the effects of varying influent BOD concentrations on nitrification. No effect was found. However, the highest BOD used was only 110 mg/l. Keenan et al. (46) performed studies of combined carbon oxidation/nitrification treatment of a landfill leachate. They

found no BOD or COD effect on nitrification even though the influent BOD and COD concentrations averaged 3564 and 6481 mg/l respectively. However, as the organic loading factors were not provided with the results, it is difficult to draw a conclusion from these results.

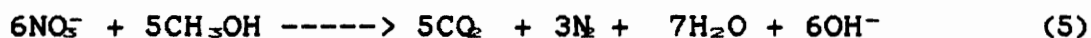
Nitrification of old leachates such as the Dixie Caverns leachate should not be effected by excessive organics loading. The maximum COD of the leachate measured was 107 mg/l. However, from the viewpoint of dissolved oxygen competition, the amount of biodegradable organics contained in the leachate is more critical. The BOD_5 concentration of the leachate is less than 10 mg/l with an average of 4.82 mg/l. With an average TKN concentration of 58.3 mg/l, the BOD/TKN ratio is a very low 0.083. Unless the organics in the leachate are toxic to nitrifiers, the organics should not interfere with nitrification.

Denitrification

Reaction (4) below presents the overall nitrate removal reaction that occurs during the denitrification as proposed by McCarty (48). This reaction considers both dissimilatory denitrification (the conversion of nitrate-nitrogen to nitrogen gas) and nitrogen assimilation (uptake of nitrate-nitrogen for cell synthesis).



Equation 5 presents an alternative reaction which considers only dissimilatory denitrification (49).



Under aerobic conditions, when organic matter is degraded, oxygen is utilized as the terminal electron acceptor. In the absence of oxygen, other compounds such as nitrate or sulfate can act as electron acceptors (27). For a waste water that has undergone nitrification, substantial amounts of nitrate should be present and which will act as a terminal electron acceptor in anoxic conditions (no dissolved oxygen present but nitrate present).

As can be seen from the reaction above, a source of biodegradable organics is needed to provide carbon for cell synthesis and electrons for energy. Many organic compounds such as acetic acid, citric acid, acetone, methanol, and glucose could be used. Raw sewage also normally contains sufficient biodegradable organics (27). Manoharan et al. (50) found that the addition of glucose as a BOD source during nitrification/denitrification treatment of a leachate caused unreliable nitrification and denitrification. The authors attributed this problem to the glucose creating an

environment in which facultative anaerobes compete with the denitrifiers for the glucose. The authors found no similar problems when they switched from glucose as a carbon source to methanol as a carbon source. The facultative competition consideration is supported by the findings of Wilderer et al. (51). They found that during denitrification of wastewater, nitrite, an intermediate of denitrification, was found to accumulate when fermentation conditions were introduced. The accumulation of nitrite was believed to be caused by facultative anaerobes which were capable of reducing nitrate to nitrite but could not reduce the nitrite any further. These facultative anaerobes were found to proliferate at the expense of the true denitrifiers. If carbon addition is necessary to fuel denitrification, the carbon source should therefore be carefully chosen. Barnes and Bliss suggested that methanol was the preferred choice (27).

As can be observed from equation 5, alkalinity is produced during denitrification. For each mg of nitrate-nitrogen denitrified, 3.57 mg of alkalinity as calcium carbonate are produced. This is half the amount of alkalinity destroyed for each mg of ammonia-nitrogen nitrified (52).

Stensel et al. (53) stated that the minimum MCRT required for denitrification is about 0.5 days for a tempera-

ture range of 20C to 30C and recommended that at 20C, an MCRT of four days should be used for complete denitrification. Stensel reported that at 10C, denitrification rates decreased substantially. Lawrence and McCarty (41) reported that the minimum MCRT for denitrification is about two to four days.

Barnes and Bliss (27) stated that the optimum pH for denitrification is between 6.5 and 7.5. As can be seen from equation 4, the process of denitrification increases the pH by reducing the hydrogen ion concentration.

Barnes and Bliss (27) reported that denitrification occurs with the temperature range of 0C to 50C with the optimum rates at 35-50C.

Barring any toxic effects of the leachate, denitrification of the Dixie Caverns leachate combined with nitrification appears to be a feasible process for total nitrogen removal from the leachate. However, due to the fact that the leachate contains little BOD, an additional carbon source would have to be added to fuel denitrification.

Nitrification/Denitrification Leachate Treatment Systems

A number of studies have been performed to evaluate the ability of single sludge nitrification/denitrification systems to treat landfill leachates. Several recent studies will be discussed.

Dedhar (15) performed a biological nitrification/denitrification treatment study on a leachate with characteristics similar to the Dixie Caverns leachate in that it had a high ammonia-nitrogen concentration (122-288 mg/l) and a low BOD₅ concentration (6-24 mg/l). Dedhar used a Modified Ludzack Ettinger system (to be discussed in more detail below) which is a two reactor, continuous flow single sludge system. Anoxic conditions were established in the first reactor for denitrification. The second reactor was aerated for nitrification. Nitrates were recycled back to the anoxic tank with the return activated sludge. The system was operated at a 15 day MCRT. At an influent flowrate of 10 liters per day, the maximum nitrogen loading rate was about 3000 mg/day. Nitrification in this system was virtually complete with the effluent ammonia-nitrogen concentration less than 1 mg/l. Complete denitrification was also achieved during several periods of the study. Due to the low BOD concentration in the leachate, an additional carbon source in

the form of glucose was added to fuel denitrification.

Randall and McClintock (19) used sequencing batch reactors to evaluate biological nitrification/denitrification treatment of a leachate from an operating landfill. The leachate exhibited characteristics of a "young" leachate in that it possessed a high BOD₅ concentration of 365 mg/l and a relatively low TKN concentration of 40.3 mg/l. The most effective sequence for total nitrogen removal was found to be fill with leachate, aerate the resulting mixed liquor for eight hours, mix for four hours without aeration, aerate for one hour, settle for one hour and then decant. The first aeration period provided for nitrification and BOD removal. The mixing without aeration period established anoxic conditions for denitrification and total nitrogen removal. As the previous aeration period would remove biodegradable organics, an additional carbon source would be necessary during the anoxic period. This carbon source was provided by the addition of extra leachate to the reactor at the beginning of the anoxic period. The second aerobic period was a polishing step to remove any BOD left over from the anoxic period and to nitrify any ammonia left over from the anoxic period. This sequence removed 72% of the influent nitrogen from the leachate and produced an effluent TKN of 9.9 mg/l, an ef-

fluent ammonia-N of 6.4 mg/l and an effluent nitrate concentration of 1.2 mg/l. The benefits of denitrification were clearly demonstrated by the fact that a completely aerobic sequencing batch reactor removed only 61% of the influent nitrogen.

Wang (16) used a single sludge biological nitrification/denitrification system to treat a leachate from an operating landfill with a TKN of concentration of 29 mg/l and a BOD_5 concentration which varied widely from 10 to 185 mg/l. Rather than two reactors for separate nitrification and denitrification, Wang used one reactor with intermittent aeration (two hours on; two hours off). During the aeration period, nitrification and BOD removal took place. During the unaerated period, anoxic conditions were established for denitrification. With a flowrate of 6 liters/day, the influent nitrogen loading rate was 174 mg/day and the hydraulic retention time was 12 hours. The system was operated with a MCRT of 15 days and at a constant temperature of 20°C. The total nitrogen removed varied widely from a low of about 25% to a high of about 88%. As was stated earlier, Wang found that nitrification decreased as the BOD/TKN ratio increased. Denitrification varied widely and was not complete until glucose addition was begun as an additional carbon source.

In general, Wang demonstrated that the intermittent anoxic/aerobic system was effective in removing total nitrogen from the leachate. Under optimum conditions, the system removed an average of 75% of the total nitrogen. This was compared to only 30% nitrogen removal by a completely aerobic control system.

The Modified Ludzack Ettinger System

Numerous system configurations for biological nitrification/denitrification have been proposed or utilized (6,24,27). These include single sludge, double and triple sludge and oxidation ditch systems. All of these systems have at least one aerobic zone for nitrification and at least one anoxic zone for denitrification. One such system, the Modified Ludzack Ettinger (MLE) system is presented in Figure 1. Ludzack and Ettinger (30) found that modifying the operation of a completely aerobic activated sludge treatment system resulted in lower effluent nitrogen concentrations. They demonstrated that recycling the effluent mixed liquor to and reducing the aeration in the influent zone of the aeration basin forced the reduction of oxidized nitrogen (nitrate) returning in the recycled mixed liquor while organics in the feed were being metabolized. The resulting nitrogen would be stripped from the tank by the aeration portion of

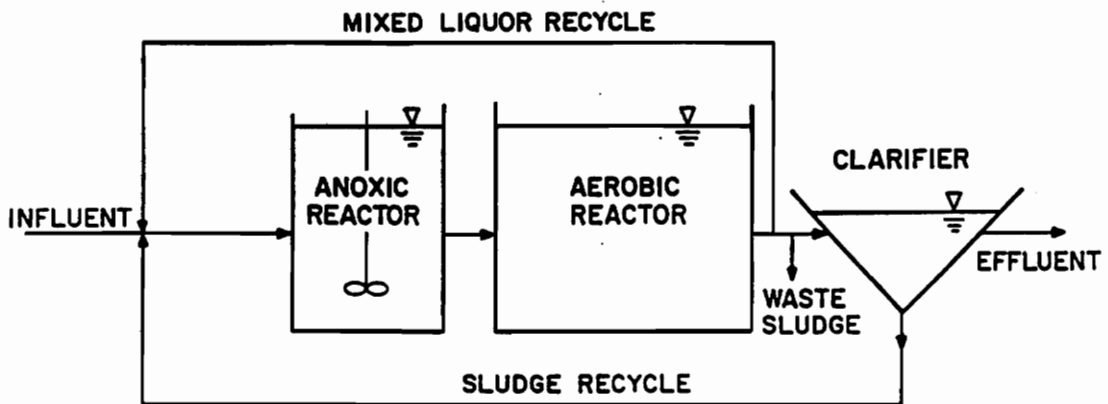


Figure 1. Diagram of the Modified Ludzack Ettinger Process

the basin. The authors stated such a system would produce superior effluents from a BOD and total nitrogen standpoint at a lower air requirement. The two zones can either be contained in the same basin (and should be separated by baffles to minimize the effects of backmixing) or in separate basins.

Reviewing Figure 1, the nitrogen content (excluding the organic nitrogen contained in the biomass) in both the solids and mixed liquor recycle lines is practically all in the form of nitrates if nitrification in the aerobic zone is complete (52). These nitrates are mixed with the untreated influent, and in the anoxic reactor nitrate respiration and cell synthesis occurs utilizing the influent BOD as the carbon and electron source and the recycled nitrates as the terminal electron acceptor in place of dissolved oxygen. The effluent of the anoxic reactor contains unutilized BOD and the TKN from the influent. In the aerobic zone nitrification, aerobic BOD respiration and cell synthesis occur.

The amount of nitrate-nitrogen denitrified in the anoxic reactor is equal to the effluent nitrate concentration times the ratio of the total recycle flowrate to the influent flowrate (52). Therefore, increasing either or both recycle flowrates will increase the nitrate loading back to the anoxic tank, increase the amount of denitrification and lower

the effluent nitrate and total nitrogen concentration. Typically, 70-80% of the influent nitrogen could be removed from a wastewater (24) although operational realities usually result in removals of around 50%. Using the BOD of the wastewater to fuel denitrification in the first reactor reduces the amount of oxygen required in the aerated zone because a portion of the organics in the wastewater have already been metabolized prior to the wastewater reaching the aerated zone (27). The addition of an anoxic zone also should improve the settling characteristics of the sludge (27) and will result in less sludge production than in a completely aerobic system (55).

The system utilized by Dedhar (15) at the University of British Columbia to treat a leachate (results discussed earlier) was a MLE system. The only difference between Dedhar's system and the system shown in Figure 1 was that there was no mixed liquor recycle. Nitrates were recycled back to the head of the plant by means of the solids recycle. Mavinic et al. (54) also at the University of British Columbia used this system to treat leachate from an "old" landfill with a TKN concentration of 196 mg/l and a BOD concentration of 25 mg/l. At an MCRT of 10 days, a temperature of 20C, and a nominal hydraulic retention time of about 36 hours, nitrification was virtually complete with the effluent ammonia-N

concentration consistently below 1.0 mg/l. The researchers also varied the sludge recycle flowrate to determine the optimum recycle rate. At a solids recycle ratio of 4:1, good nitrification and denitrification were observed and the effluent nitrate-N concentration averaged 45 mg/l. Increasing the solids recycle to 6:1 decreased the effluent nitrate-N down to 30 mg/l. Any further increase in the recycle flowrate (eventually up to 10:1) resulted in erratic nitrification and denitrification and in poor effluent suspended solids concentration. The effluent nitrate-N concentrations varied wildly from 20 to 70 mg/l. This led Mavinic to recommend that 6:1 should be the maximum recycle ratios that should be utilized.

The MLE system should be effective in treating the leachate from the Dixie Caverns landfill unless the leachate proves toxic to the biomass. However, a major advantage of the MLE system, placing the anoxic zone first to utilize the wastewater's BOD in order to fuel denitrification would not be realized while treating the leachate. The leachate has little BOD and a carbon source would have to be added for denitrification. The anoxic section should still be placed first. The following aerobic zone would strip out the nitrogen formed in the anoxic section and also metabolize any

biodegradable BOD that is not utilized in the anoxic zone.

SUMMARY

There are numerous methods available for the treatment of municipal landfill leachates. These treatment methods include piping or transporting the leachate to operating waste treatment plants, land application or recirculation, physical-chemical treatment and biological treatment. Most of the early research performed on leachate treatment emphasized organics removal. Because of this much of the literature provides treatment recommendations based only on organics removal. Biological treatment is recommended for leachates from young or operating landfills typically containing high concentrations of biodegradable organics. If the COD concentration is extremely high, the biological treatment system should probably include an anaerobic process. Leachates from old landfills typically contain low concentrations of organics, a high percentage of which are nonbiodegradable. For such leachates, the literature recommends that physical and/or chemical methods be used to remove organics with activated carbon adsorption and reverse osmosis probably being the most effective. However, these leachates can contain very high concentrations of ammonia; and the litera-

ture normally does not take the need to remove ammonia and total nitrogen into account when recommending treatment methods. Even though physical-chemical methods have been recommended for old landfill leachates, biological treatment could be a viable treatment alternative if nitrogen removal is necessary.

Biological nitrogen removal can be achieved by two basic methods. For leachates with high organic concentrations, nitrogen removal can occur by biomass uptake during cell synthesis. For old leachates or for leachates with low organic concentrations, biological nitrification/-denitrification will remove nitrogen. Since the characteristics of landfill leachates vary widely and since nitrogen removal from leachates has not received much attention until recently, more research and experience with nitrification/denitrification treatment of leachates is required.

The Modified Ludzack Ettinger Process (MLE) is a single sludge anoxic/aerobic process which utilizes nitrification and denitrification to remove nitrogen. Dedhar and Mavinic at the University of British Columbia utilized this basic process to successfully remove nitrogen from "old" leachates.

The Dixie Caverns Landfill in Roanoke County, Virginia is an old landfill and the leachate from the landfill exhibits those characteristics that are typical of leachate

from old landfills with a low biodegradable organics concentration and a high ammonia concentration. An evaluation of the ability of the MLE system to treat Dixie Caverns leachate should provide valuable leachate treatment experience applicable to landfills throughout Virginia.

Chapter III. MATERIALS AND METHODS

The research conducted during this study was essentially divided into two parts. One part concentrated on the ability of the Modified Ludzack Ettinger (MLE) process to treat leachate that exhibited the characteristics of being produced from an old landfill; emphasizing total nitrogen removal. The second part of this research studied the ability of the MLE process to nitrify the ammonia-N in the leachate as compared to a completely aerobic activated sludge system while both systems were operated at several different low Mean Cell Residence Times (MCRTs).

A description of the experiment is presented below.

LABORATORY APPROACH AND APPARATUS

Two laboratory treatment systems were designed, con-

structured and operated during this study. The first system, which will be called the Experimental System, is shown in Figure 2. The Experimental System utilized the MLE process which is a single sludge nitrification/denitrification system. In the aerobic reactor (the second tank), the ammonia-N and organic TKN are converted to nitrate by nitrification. The nitrate are recycled back to the anoxic reactor (the first tank) through the sludge return line and the mixed liquor recycle line. In the anoxic reactor (unaerated), nitrate formed in the aerobic tank is used as a terminal electron acceptor during biochemical oxygen demand (BOD) removal and is converted to nitrogen gas. The nitrogen gas leaves the treatment system through the surface of both tanks thereby being removed from the waste water. The effluent from the Experimental System should be low in ammonia-N, TKN and nitrate-N concentrations if it operates satisfactorily.

The second system termed the Control System, was operated as a single sludge completely aerobic activated sludge system and is shown in Figure 3. In this system, the ammonia-N and organic TKN are converted to nitrate in the aerobic tank by nitrification. However, unlike the Experimental System, little denitrification should occur as there is no anoxic zone. The effluent from this system,

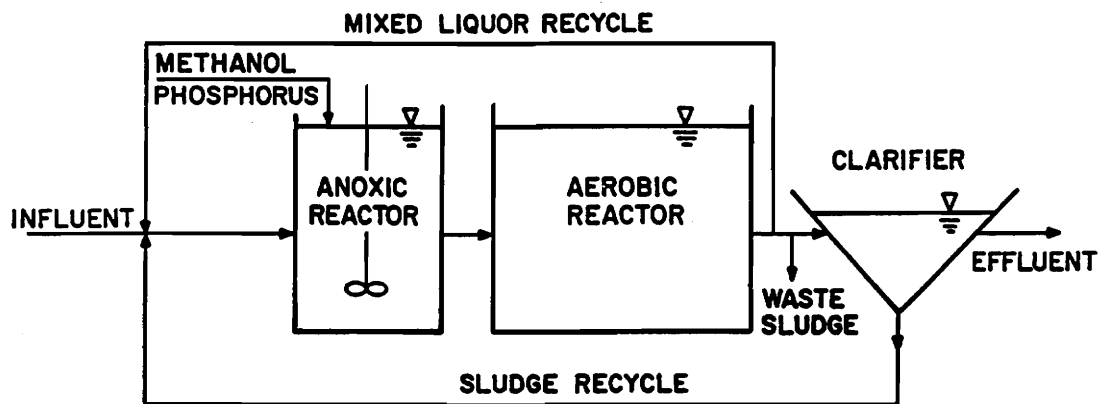


Figure 2. Diagram of the Experimental System

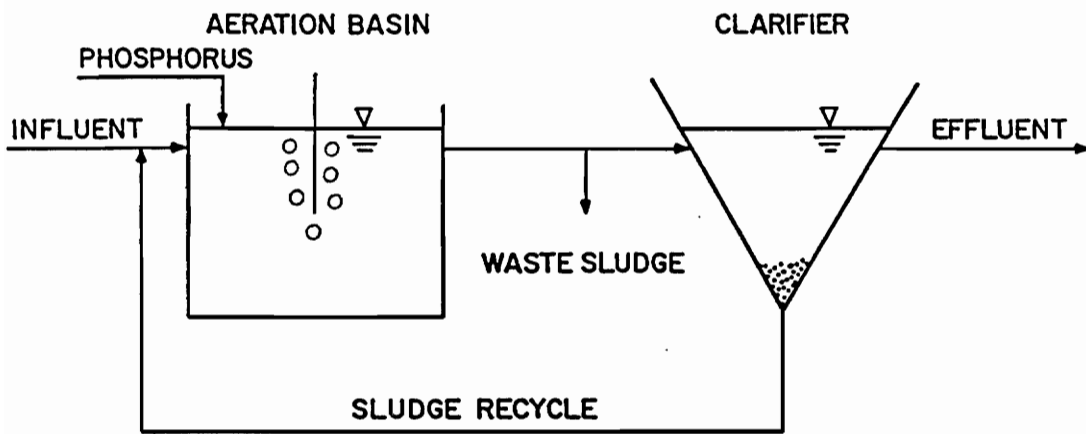


Figure 3. Diagram of the Control System

therefore, should be low in ammonia-N and TKN but high in nitrate-N. This system served as a control system and was utilized to compare nitrification rates with those of the Experimental System.

Both systems were used to determine the ability of an activated sludge treatment process to remove nitrogen, degrade organics and to remove metals contained in the Dixie Caverns Landfill leachate.

Each reactor in both systems consisted of a round plastic tank. The raw leachate was continuously pumped from a Nalgene carboy through Tygon tubing to each system. In the Experimental system, mixed liquor from the anoxic reactor was gravity fed to the aerobic reactor through a tube connecting both tanks several inches above the bottom of each tank. Mixed liquor from both systems was gravity fed from the aerobic reactors into clarifiers where the suspended solids were settled and pumped back to the front of the treatment systems. In the Experimental System, a nitrate recycle line was used to pump mixed liquor directly from the aerobic reactor back to the anoxic reactor. In the aerobic reactors of both systems, air was continuously bubbled through a porous stone diffuser to maintain sufficient dissolved oxygen concentrations and to keep the reactors well mixed. An

electric motor driven paddle was used to provide adequate mixing in the Experimental System's anoxic reactor. The rate of mixing was the lowest possible to minimize the entrainment of air while at the same time keeping the solids suspended. Electric motor driven scrapers operated at one revolution per minute were used in the clarifiers to assist in the collection, settling and recycle of the sludge. The operating parameters utilized during the early phase of the study are listed in Table 1.

Because the leachate was very low in BOD, methanol was added to the anoxic reactor of the Experimental System to fuel denitrification. From reaction 4 in Chapter 2, 1.08 moles of methanol are stoichiometrically needed to denitrify one mole of nitrate-N. For an average TKN in the influent leachate of about 60 mg/l, about 150 mg of methanol per liter of leachate would be required assuming that all the nitrogen would first be nitrified, and then denitrified. It was decided to add 167 mg of methanol per liter of leachate to ensure that the lack of methanol would not limit denitrification. This methanol was fed directly into the Experimental System's anoxic reactor as a water solution by a low flow pump. Methanol was initially added to the Control system, but this addition was discontinued before the beginning of

Table 1. Operating Parameters of Treatment Systems

Parameter	Experimental	Control
Flow Rate (liters/day)	6	6
Aerated Volume (liters)	3	3
Anoxic Volume (liters)	1.5	NA
Clarifier Volume (liters)	0.5	0.5
Hydraulic Retention Time (days)	0.75	0.5
Sludge Recycle Ratio	1:1	1:1
Anoxic Recycle Ratio	2:1	NA
Methanol Addition (mg/l)	167	NA
Phosphorus Addition (mg/l)	8.75	8.75

sampling and was not reinstituted until later in the study.

As the leachate contained essentially no phosphorus, orthophosphate in the form of potassium phosphate was added to both the Experimental and Control systems as a nutrient. The recommended BOD_5 to phosphorus ratio is usually 100 to 1 (26). However, prior leachate treatment experience indicated that much of the phosphorus added to leachate treatment systems co-precipitated with the metals that typically exist in high concentrations in leachates and is apparently not available as a nutrient (16,54). Mavinic et al. (54) recommended that a minimum concentration of 0.5 mg/l of soluble orthophosphate be maintained in activated sludge reactors in order to prevent phosphorus limitation of the biological processes involved. To overcome any co-precipitation effects, it was decided to add five times the normal amount of phosphorus (a BOD_5 :P ratio of 100:5 instead of 100:1). The 167 mg/l of methanol being added represented a COD of 250 mg/l and a BOD_5 of approximately 175 mg/l (assuming that BOD_5 is 0.7 COD). With a 100:5 BOD_5 :p ratio, 8.75 mg of orthophosphorus per liter of leachate was required. The phosphorus was added in the same water solution as the methanol.

SYSTEM STARTUP

The original seed of microorganisms for this study was obtained from the Blacksburg-V.P.I. Sanitation Authority's Stroubles Creek Plant. System startup in accordance with the descriptions provided in Figures 2 and 3 and Table 1 was achieved utilizing the seed microorganisms on March 31, 1988 (day 1 of the study). Throughout the study, changes to Mean Cell Residence Time (MCRT) and changes to the physical configurations of the systems were made and will be briefly discussed.

At first, both systems were operated at an aerobic MCRT of 15 days. The aerobic sludge age only considered the solids concentration and volume of the aerobic reactors in each system. Therefore, the aerobic MCRT of the Control System was also the total system MCRT. For the Experimental System, the total system MCRT was approximately 50% greater than the aerobic MCRT due to the presence of the anoxic reactor. The aerobic sludge age was reduced to eight days on June 11, 1988 (day 73), to three days on June 29, 1988 (day 91), and to two days on July 24, 1988 (day 116). During the two day MCRT operating period, the clarifiers were modified. On July 27, 1988, the 0.5 liter clarifiers were replaced with

1.0 liter clarifiers with the intention of improving settling. After several days of sampling, the new clarifiers were replaced with the original 0.5 liter clarifiers on August 1, 1988.

On September 23, 1988 (day 177), both systems were converted to chemostats. The clarifiers and sludge recycle lines were removed. Without the solids recycle, the MCRT of the systems became the same as the hydraulic retention time. On September 23, the leachate influent flow rate was decreased to 1.5 liters per day to provide a hydraulic retention time and an aerobic MCRT of two days. After completing sampling at two days, the leachate flow rate was increased on October 8, 1988 to three liters per day in order to establish an MCRT of one day.

On November 11, 1988 (day 227), chemostat operation was discontinued. The Experimental System was put back into its original configuration as was shown in Figure 2 by reinstalling the clarifier and sludge recycle line. The Control System, however, was not put back into its original configuration. Instead, it was modified to be identical to the Experimental System with the exception that it was completely aerobic. A 1.5 liter reactor was placed in front of the original 3 liter aerobic reactor. Unlike the anoxic reactor

of the Experimental System, this reactor was aerated making the total aerated volume of the Control System equal to 4.5 liters. A mixed liquor recycle line was also added to the Control System and methanol addition was begun at the same rate as the Experimental System. The intent was to make the Control system identical to the Experimental system in size, physical configuration, and influent feed but with aeration of both reactors instead of maintaining an anoxic state in the first one. Thus, the aerobic volume was 4.5 liters in the the Control System versus 3 liters for the Experimental System.

At first, both systems were operated at a total system MCRT of three days. The MCRT of both systems were reduced to 2.5 days on February 13, 1989 (day 321), to 2 days on March 13, 1989 (day 349), and then to 1.5 days on May 4, 1989 (day 401).

On June 8, 1989 (day 436), the Control System was modified to be identical to the Experimental System by converting it to an MLE process. This was accomplished by turning off the aeration in the first 1.5 liter tank, thereby turning it into an anoxic reactor. At first both systems were operated at a 4 day MCRT, but the MCRT was increased to 8 days on July 2, 1989 (day 460). On August 5, 1989 (day 494), the mixed liquor recycle flowrate of both systems was doubled from 12

liters per day (twice the leachate flow rate) to 24 liters per day (4 times the leachate flow rate). The addition of methanol and phosphorus was increased to 1.63 grams of methanol and 86 mg of phosphorus per day to ensure that the biological processes were not limited by a lack of nutrients.

The study was discontinued on day 584.

DAILY PROTOCOL

Solids were wasted directly from the aerobic reactors of both systems daily to maintain the desired MCRT. The sides of the reactors and clarifiers were scraped and the tubing squeezed daily to reduce attached growth effects.

LEACHATE COLLECTION AND STORAGE

Leachate from the Dixie Caverns Landfill was obtained every two or three weeks during the study. During the early stages of the study, leachate from the landfill was obtained from the leachate holding pond at the lower end of the landfill. The leachate was pumped through a garden hose with the

use of a Cole-Parmer Series 7090 "Guzzler" hand pump into Nalgene carboys. In the summer of 1989, Roanoke County installed a holding tank below the pond; and from the beginning of September, 1989, the leachate was collected from the tank. The leachate filled carboys were immediately transported to the laboratory and stored at either 4C or 20C. The leachate stored in 20C was given priority when feeding the treatment systems. Once this portion of the collected leachate was used, the quantity stored at 4C was then fed to the systems.

SAMPLING PROCEDURE

The leachate and treatment system parameters monitored during the study are listed in Tables 2 and 3. Sampling of the treatment systems at each different MCRT and physical configuration was begun after steady state conditions were reached. Several hundred milliliters of clarified effluent were collected from each treatment system. During chemostat operation (no clarifiers), system effluents were collected from the effluent lines of each system. Samples from the anoxic zones were taken either directly from the anoxic reactors with a beaker or from the tube between the anoxic

Table 2. Influent and Effluent Parameters Monitored During Study

Parameter	Influent Leachate (unfiltered)	Effluent (filtered)
Total Kjeldahl Nitrogen	x	x
Ammonia Nitrogen	x	x
Nitrate and Nitrite Nitrogen	x	x
pH	x	x
Chemical Oxygen Demand	x	x
5-Day Biochemical Oxygen Demand	x*	x*
Total Organic Carbon	x*	x*
Alkalinity	x	x*
Metals (Fe, Mn, Ni, Zn, Cu, Cr, Cd and Pb)	x*	x*
Total Phosphorus	x*	
Soluble Orthophosphorus		x*
Suspended and Volatile Solids	x*	x
* Measured infrequently		

Table 3. Reactor Parameters Monitored During Study

Parameter	Anoxic Reactor	Aerobic Reactor
Mixed Liquor Suspended Solids	x	x
Mixed Liquor Volatile Susp. Solids	x	x
Chemical Oxygen Demand	x	
Nitrate and Nitrite Nitrogen	x	
pH	x	x
Dissolved Oxygen	x*	x*
* Measured Infrequently		

and the aerobic reactors. Filtered samples were obtained by drawing the samples through glass microfiber filters with a pore size of 1.5 μm . In addition, samples for soluble metals and phosphorus were filtered through membrane filters with a 0.45 μm pore size.

Analyses for alkalinity, pH and BOD_5 were performed immediately upon taking the sample. If not analyzed immediately, the samples for nitrates, nitrites, and phosphorus were frozen and stored. The samples for the determination of metals were preserved with HNO_3 and stored at 4°C. All other samples were stored, if not analyzed immediately, by acidifying with H_2SO_4 to a pH of two or below and stored at 4°C.

ANALYTICAL TECHNIQUES

The laboratory analyses of the parameters monitored during the study were performed as follows:

Total Kjeldahl Nitrogen (TKN)

The leachate and system effluent TKN concentrations were determined in accordance with the semi-micro-Kjeldahl method as described in Test 420 B in Standard Methods for the Examination of Water and Wastewater (56). The samples were first

digested in 100 ml Kjeldahl flasks. The digested samples were then distilled into 20 ml of indicating boric acid solution. This solution was then titrated with 0.02 N standard sulfuric acid solution in accordance with Test 420 B.

Ammonia Nitrogen ($\text{NH}_3\text{-N}$)

The leachate and system effluent ammonia-N concentrations were determined in accordance with Test 417 A and D in Standard Methods (56). The pH of the samples was adjusted to 9.5 and distilled into 20 ml of indicating boric acid solution. The resulting solution was then titrated with 0.02 N standard sulfuric acid solution.

Nitrate and Nitrite Nitrogen ($\text{NO}_3\text{-N}$ and $\text{NO}_2\text{-N}$)

The leachate, effluent and anoxic zone nitrate and nitrite nitrogen concentrations were determined with the use of a Dionex 2010i Ion Chromatograph in accordance with Test 429 in Standard Methods (56).

pH

The leachate, effluent and anoxic zone pH's were determined with the use of a Fisher Accumant pH Meter Model 610 A.

Chemical Oxygen Demand (COD)

The leachate, effluent and anoxic zone COD concentrations were determined using the closed reflux titrimetric method in accordance with Test 508 B in Standard Methods (56).

Biochemical Oxygen Demand (BOD₅)

The leachate and effluent BOD₅ concentrations were determined in accordance with Test 507 in Standard Methods (56). Seed microorganisms for the tests were obtained from the Experimental System effluent. As nitrifiers were well established in the Experimental System, 2-chloro-6-(trichloromethyl)pyridine was added to the dilution water to inhibit nitrification. A YSI Model 57 oxygen meter was used to determine the initial and final dissolved oxygen concentrations.

Total Organic Carbon (TOC)

The leachate and effluent TOC concentrations were determined by the persulfate-ultraviolet oxidation method in accordance with Test 505 B in Standard Methods (56). A Horiba PIR-2000 Total Organic analyzer was utilized to perform this test.

Alkalinity

The leachate and effluent alkalinity concentrations were determined by potentiometric titration to a preestablished pH with 0.02 N standard sulfuric acid solution in accordance with Test 403 in Standard Methods (56). As the alkalinity of leachate was normally well above 500 mg/l as CaCO_3 , the samples were titrated down to a pH of 4.3 in accordance with Test 403.

Metals

The leachate and effluent metals concentrations (cadmium, copper, chromium, iron, manganese, nickel, lead and zinc) were digested in accordance with the procedure for total recoverable metals contained in the Environmental Protection Agency's Methods for Chemical Analysis of Water and Wastes (57). The digested samples were then analyzed with either a Perkin-Elmer 703 Atomic Absorption spectrophotometer or a Perkin-Elmer 5100 PC Atomic Adsorption spectrophotometer.

Total Phosphorus

Several times early in the study the leachate was monitored for total phosphorus by the ascorbic acid method in accordance with Test 424 F of Standards Methods (56) after

persulfate digestion in accordance with Test 424 C.

Soluble Orthophosphorus

The soluble orthophosphorus concentrations in the system effluents were determined by first filtering the effluents through a 0.45 micron membrane filter and analyzing the filtrate using the Dionex 2010i Ion Chromatograph.

Suspended and Volatile Solids

The leachate, mixed liquor and effluent suspended and volatile suspended solids were determined in accordance with Test 209 of Standard Methods (56). The samples were filtered through Whatman glass microfiber filters (pore size 1.5 μ m) held in place by a Buchner funnel. A Mettler Instrument Corporation Model H10 analytical balance was used for solids measurements.

Daphnia Magna Toxicity Testing

Due to inhibition of nitrification that occurred during the study, a limited amount of daphnia toxicity testing was conducted on a number of batches of the leachate. *Daphnia magna* were cultured in EPA synthetic hardwater following the recommendations outlined in Peltier and Weber (58). A modi-

fication of the effluent toxicity test described in Peltier and Weber was used to monitor relative toxic effects of the different batches of leachate. For each dilution the leachate was diluted with EPA synthetic hardwater on a volume percentage basis. *Daphnia magna* neonates (less than 24 hours old) were transferred to a beaker of dilution water. They were rinsed by carefully withdrawing culture water and food particles with a pipette and replacing the volume with fresh dilution water. Ten neonates were distributed to each of the beakers containing the diluted leachate test solutions. Duplicates were run for each dilution. Cool white fluorescent bulbs provided 50-60 lux with a photoperiod of 16 hours on, 8 hours off. The beakers were incubated for 48 hours at 20C. The number of dead at 48 hours was then recorded. Death was defined as no movement of internal organs as viewed through a 2x magnifying lens with the aid of a light box.

ANALYSIS

Mean Cell Residence Time (MCRT)

The MCRT's of both systems were maintained by wasting solids directly from the reactors. The amount of mixed liquor volume wasted to maintain the desired MCRT was deter-

mined from the following equation:

$$Q_w = \{X*V - Q*X_e*MCRT\} / \{MCRT*(X - X_e)\}$$

where: Q_w = Volume of mixed liquor wasted each day, l/day

X = Mixed liquor volatile suspended solids concentration, mg/l

V = Reactor Volume, l

Q = Influent flowrate (6 l/day)

X_e = Effluent volatile suspended solids concentration, mg/l

During the first part of the study, the systems were controlled by aerobic MCRT's. The aerobic MCRT was determined using the equation above but with only the aerobic volume and the aerobic reactor mixed liquor volatile suspended solids concentrations included.

Percent Nitrogen Removal (%N-Rem.)

The percent of nitrogen removed from the leachate by each system was determined by using the following equation:

$$\%N\text{-Rem.} = (TN_{in} - TN_{out}) * 100\% / TN_{in}$$

where: TN_{in} = Total influent nitrogen = $TKN_{in} + NO_{xin}$

TKN_{in} = Influent TKN, mg/l

NO_{xin} = Influent nitrate-N + nitrite-N, mg/l

$$\text{TNout} = \text{Total filtered effluent nitrogen} = \text{TKNout} + \text{NOxout}$$

$$\text{TKNout} = \text{Filtered effluent TKN, mg/l}$$

$$\text{NOxout} = \text{Effluent nitrate-N} + \text{nitrite-N, mg/l}$$

Amount of Nitrogen Nitrified (N-Nitrified)

The amount of nitrogen nitrified in mg/day was determined using the following equation:

$$\text{N-Nitrified} = (\text{TKNin} - \text{TKNout}) * Q - N_w$$

Where: N_w = Nitrogen wasted, mg/day = $N_s * (Q_w * X + (Q - Q_w) * X_e)$

$$N_s = \text{Nitrogen in volatile solids (10.4\%)}$$

The amount of nitrogen nitrified per day per liter of aerated volume in mg/l-day was determined by dividing the amount of nitrogen nitrified in mg/day by the aerated volume of each system.

Percent Nitrified (% Nitrified)

The percent of nitrification was determined by the following equation:

$$\% \text{ Nitrified} = \text{N-Nitrified} * 100\% / \text{TNa}$$

Where: TNa = Total nitrogen available for nitrification

$$\text{TNa} = \text{TKNin} * Q - (\text{TKNout} - \text{NH}_3\text{-Nout}) * Q - N_w$$

Nitrification Rate

The nitrification rate in 1/day was determined by using

the following equation:

$$\text{Nitrification Rate} = \text{N-Nitrified} / X * V_a$$

Where: V_a = Aerobic volume

Amount of Nitrogen Denitrified (N-Denitrified)

The amount of nitrogen denitrified in mg/day was determined by using the following equation:

$$\text{N-Denitrified} = \text{N-Nitrified} + \text{NO}_{\text{xin}} * Q - \text{NO}_{\text{xout}} * Q$$

The amount of nitrogen denitrified per day per liter was determined by dividing the amount denitrified in mg/day by the volume of the anoxic tank (1.5 liters) of the Experimental System and the total volume (3 or 4.5 liters) of the completely aerobic Control System.

Denitrification Rate

The denitrification rate in 1/day was determined by the following equation:

$$\text{Denitrification Rate} = \text{N-Denitrified} / X * V$$

Where: V = the anoxic volume of the Experimental System or the total volume of the Control System

Percent Denitrified (% Denitrified)

The percentage of nitrate denitrified was determined by using the following equation:

% Denitrified = N-Denitrified*100%/Total NOx-N available

Where: Total NOx-N available = N-Nitrified + NOx-Nin*Q

Statistical t-test

Hypothesis testing on two means was conducted in order to determine if the average effluent ammonia-N concentrations from the Experimental and Control Systems during several periods of operation were significantly different. T-tests were performed in accordance with the method described in An Introduction to Statistical Methods and Data Analysis (62). The following equation was used to determine the test statistic (t-obs):

$$t\text{-obs} = (y_1 - y_2) / Sp \sqrt{(1/n + 1/n)}$$

Where: y_1 = Average Control System effluent ammonia-N concentration

y_2 = Average Experimental System effluent ammonia-N concentration

Sp = Pooled standard deviation

n = Number of observations

Two sided tests were conducted at a .05 significance level. A t-test was also used for one period of the study to determine if the amount of nitrogen nitrified by the systems was different. In this case the above equation was used, and y_1 and y_2 were the amounts nitrified by the Control and Ex-

perimental Systems respectively. A t-test was also conducted to determine if the effluent nitrate concentration from the Control System decreased when the mixed liquor flowrate was doubled on day 494.

CHAPTER IV. RESULTS

This research had two major goals. The first goal was to determine the treatability of leachate that possesses the characteristics of being produced from an "old" landfill by an activated sludge waste treatment system utilizing the Modified Ludzack Ettinger (MLE) process. The emphasis of this treatment was on total nitrogen removal as the leachate treated was low in biodegradable organics but high in ammonia. The second goal was to examine the effects of the anoxic zone on nitrification in the MLE process. This was done by comparing the nitrification capability of the MLE process with a completely aerobic activated sludge system operating as a control. This comparison was accomplished by reducing the Mean Cell Residence Times (MCRT) of both systems until their nitrification capacities were exceeded. As the results obtained during this study support both goals simultaneously, the results are not separated by which goal they

emphasize but are instead presented chronologically and by the constituent of the leachate (ammonia-N, organics, etc.) being evaluated for treatment.

CHARACTERIZATION OF THE LEACHATE

Characterization of the Dixie Caverns Landfill leachate was performed every time new leachate was collected. The results are presented in Table 4.

NITROGEN REMOVAL

Day 1 to Day 177

During this time period, both systems were operated at the same aerobic mean cell residence times (MCRT's). The Experimental System was operated as a Modified Ludzack Ettinger (MLE) system and the Control System was a three liter completely aerobic activated sludge system. Both systems were operated in accordance with the parameters listed in Table 1 in Chapter 3.

From day 1 until day 73, both systems were operated at a 15 day aerobic MCRT. On day 73, the aerobic MCRT was reduced

Table 4. Characteristics of Dixie Caverns Leachate

Parameter	N*	Average Concentration**	Range
COD	24	90.2	71.4-106.8
BOD ₅	13	4.82	0.86-9.41
TOC	4	27.8	26.5-29.3
TKN	24	58.3	53.4-70.6
Ammonia-N	24	56.3	47.0-64.6
Alkalinity	24	578	541-634
Phosphorus	4	not detectable	
Iron	12	18.6	9.40-30.5
Manganese	12	0.84	0.58-1.20
Zinc	12	0.076	0.039-0.14
Nickel	12	0.024	0.009-0.07
Copper	12	0.016	0.006-0.03
Cadmium	12	0.0067	0.0002-0.06
Chromium	12	0.0024	0.0014-0.004
Lead	12	0.023	0.002-0.20
pH	21	6.72	6.48-7.02

* Number of observations

** All values expressed as mg/l, except for pH

to 8 days; on day 91 it was reduced to 3 days; and on 116 it was reduced to 2 days. Operation at a 2 day MCRT was continued until day 177.

The day-by-day experimental data collected during this period of the study are contained in Appendix B. Data collection after startup began on day 59 of operation. Table B-1 contains influent and effluent TKN's; Table B-2 presents influent and effluent ammonia-N's; Table B-3 presents influent leachate nitrate concentrations; Table B-4 contains effluent nitrate and nitrite concentrations; Table B-5 presents anoxic zone nitrate and nitrite concentrations; B-6 presents suspended and volatile suspended solids concentrations; and B-7 presents effluent suspended and volatile suspended solids concentrations.

Figure 4 presents the effluent ammonia-N concentrations of both systems from day 60 to day 116. As can be seen, the effluent ammonia-N concentrations were near zero until day 99. From day 99 onward the Control System concentration started to increase. Except for some instability around day 100, nitrification in the Experimental System remained essentially complete.

Tables 5 and 6 summarize the nitrogen removal results obtained during this period of the study for the Experimental and Control systems. The values presented in the tables

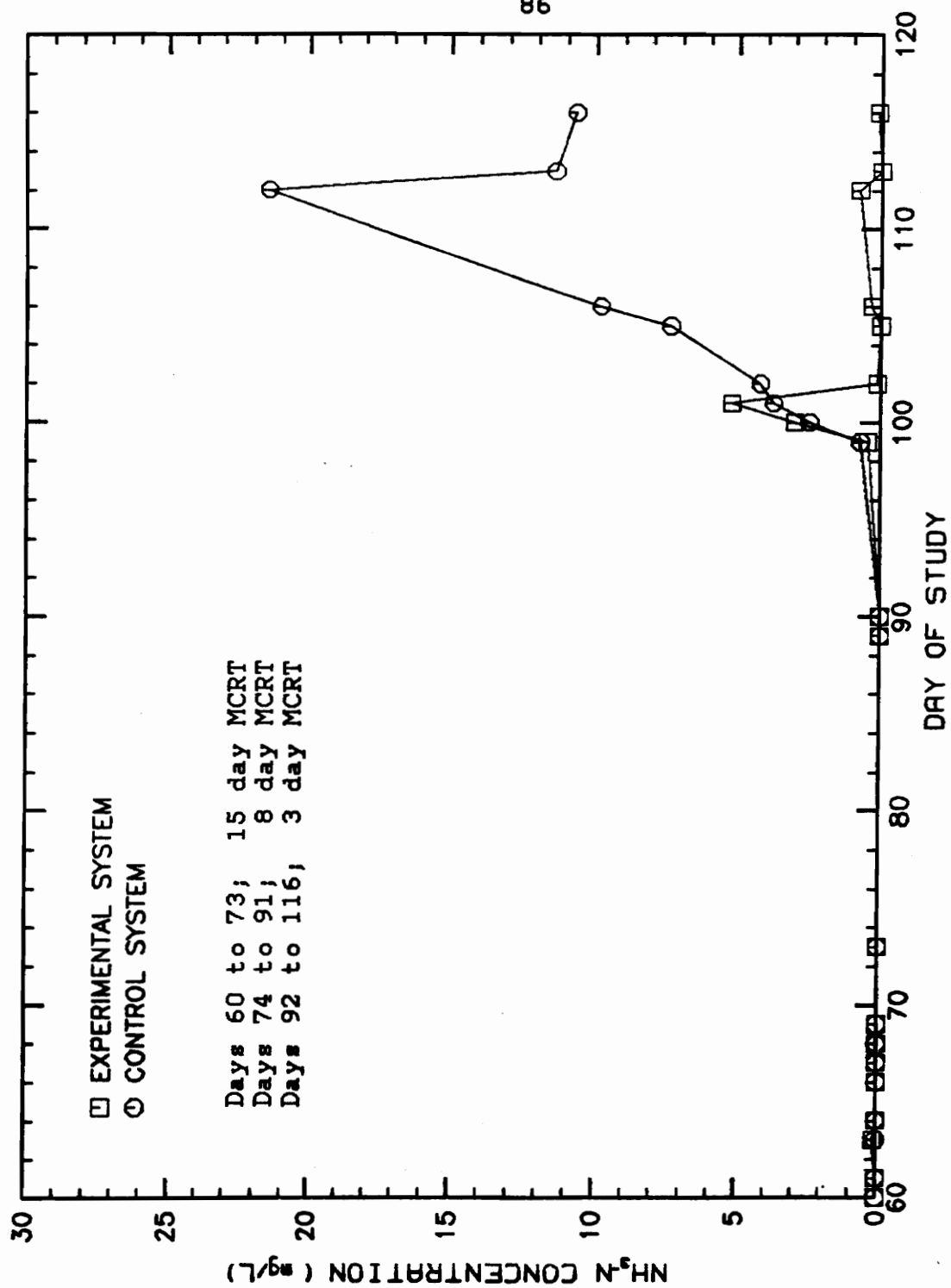


Figure 4. Effluent Ammonia Concentrations; Days 60 to 120

Table 5. Nitrogen Removal Results; Experimental System*
Days 59 to 177

MCRT (days)	TKN inf.	TKN eff.	Ammonia-N inf.	Ammonia-N eff.	NOx-N inf.	NOx-N eff.	Percent N Removal
15	55.9	2.52	50.6	0.01	2.40	9.67	79.1
8	55.3	2.73	50.2	0.00	0.41	8.20	80.4
3	56.9	4.7	51.3	1.10	0.37	12.4	70.1
2@	58.4	4.43	52.1	0.48	0.34	17.6	62.5
2#	60.6	25.4	56.1	20.6	0.18	4.56	50.7

* All concentrations in mg/l

@ Operation with 1.0 liter clarifier

Operation with the original 0.5 liter clarifier

Table 6. Nitrogen Removal Results; Control System*
Days 59 to 177

MCRT (days)	TKN inf.	TKN eff.	Ammonia-N inf.	Ammonia-N eff.	NOx-N inf.	NOx-N eff.	Percent N Removal
15	55.9	0.10	50.6	0.00	2.40	51.1	12.2
8	55.3	0.43	50.2	0.00	0.41	43.1	21.9
3	56.9	8.82	51.3	7.94	0.37	38.1	18.1
2@	58.4	2.62	52.1	1.42	0.34	46.3	16.7
2#	60.6	33.6	56.1	28.3	0.18	21.8	8.85

* All concentrations in mg/l

@ Operation with 1.0 liter clarifier

Operation with original 0.5 liter clarifier

represent the means of the individual days' results. As can be seen, both systems nitrified well at the 15 and 8 day MCRT's as evidenced by a lack of ammonia in the effluent of both systems. Ammonia began breaking through during the three day MCRT operation indicating that the nitrification capacity of both systems was exceeded. However, as shown in Figure 4, the effluent ammonia-N concentrations from the Experimental System during the 3 day MCRT period of operation were low with the exception of days 100 and 101. Nitrification, therefore, although somewhat inconsistent, had not yet been consistently limited in the Experimental System.

The two day aerobic MCRT data is split into two groups. On day 119, the original 0.5 liter clarifiers were replaced with 1.0 liter clarifiers with the intention of achieving better settling. Data was taken from days 121 to 124, and the results are presented in Figure 5 and as the first 2 day MCRT line in Tables 5 and 6. As can be seen, nitrification actually improved as evidenced by lower effluent ammonia-N concentrations from both systems. This decrease in ammonia-N concentrations could have been due to continuing biological action of the bacteria in the clarifiers. The doubling of the clarifier volume and detention time either allowed for more complete nitrification to occur or allowed for the better capture and recycle of nitrifiers. In order to be

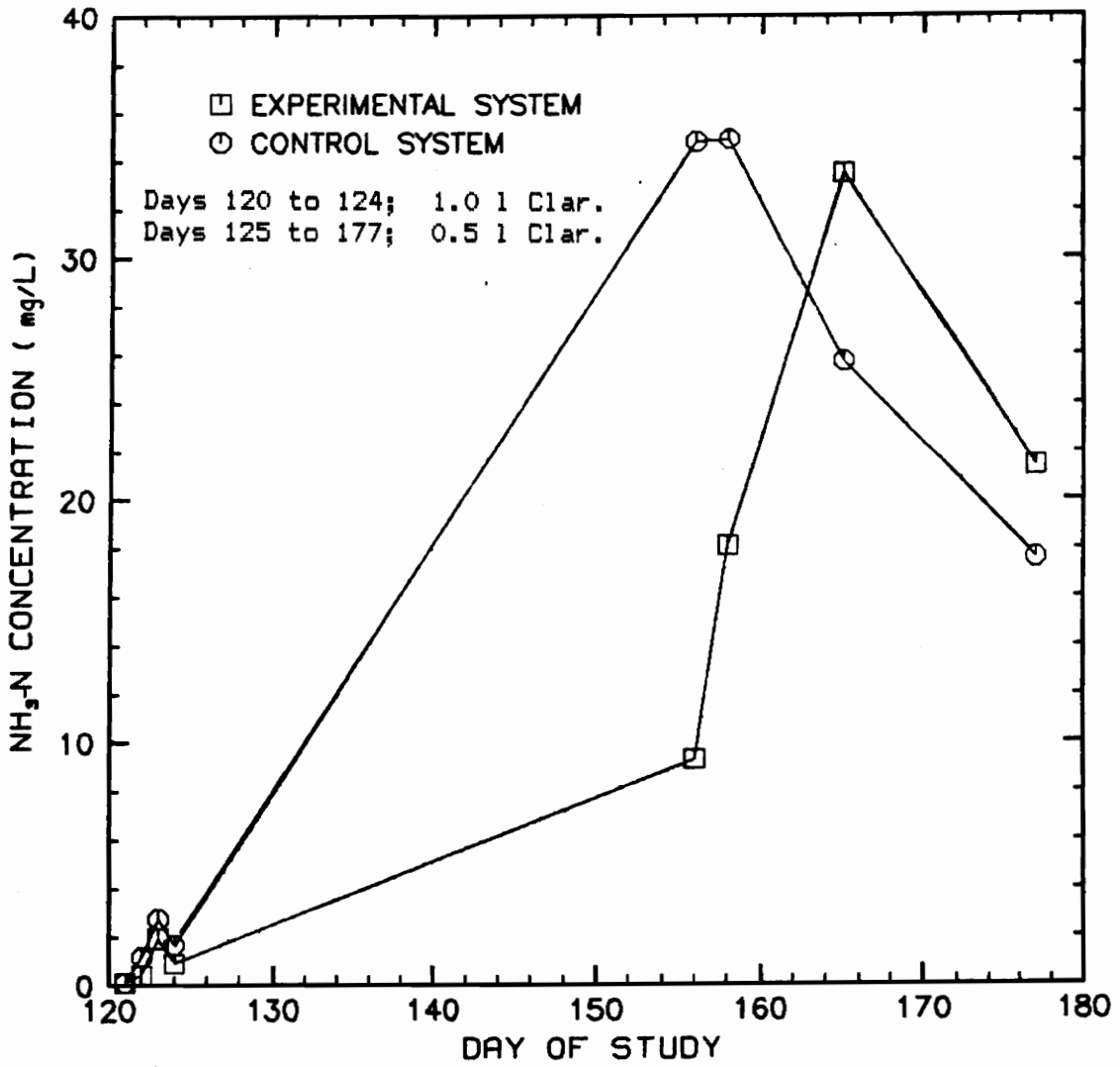


Figure 5. Effluent Ammonia Concentrations; Days 120 to 180

able to compare results with previous data, the original 0.5 liter clarifiers were reinstalled, and operation at a 2 day MCRT was continued from days 156 to 177. These data are also presented in Figure 5 and as the second 2 day MCRT line in Tables 5 and 6. Obviously, the nitrification capacities of both systems had been exceeded under these conditions as evidenced by the average effluent ammonia-N concentrations of 20.6 and 28.3 mg/l for the Experimental and Control systems respectively.

A review of Tables 5 and 6 shows that, at MCRT's of three days and below, the average effluent ammonia-N concentrations from the Experimental System were lower than the concentrations from the Control System. A t-test was performed comparing the effluent ammonia-N concentrations of the two systems at three day and two day (with the 0.5 liter clarifiers) MCRT's. For the three day MCRT period, at a significance level of 0.05, the results of the test showed that the average Experimental System ammonia-N concentration was lower than that of the Control System ($t\text{-obs} = 3.12$; $t\text{-crit} = 2.12$). The standard deviations for the effluent ammonia-N concentrations from the Experimental and Control systems were 1.81 and 6.31 respectively. For the two day MCRT period, the t-test showed that the effluent ammonia-N concentrations from the two systems were not significantly

different ($t\text{-obs} = 0.72$; $t\text{-crit} = 2.45$). This was due to the scatter of the data. As can be seen from Figure 5 from days 150 to 180, the ammonia-N concentrations varied widely. The standard deviations for the Experimental and Control Systems were 10 and 18.8 respectively. Appendix A contains an example of the performance of the t-test.

Denitrification can best be examined by monitoring the nitrate and nitrite concentrations in the anoxic zone of the Experimental system. If these concentrations are low, denitrification is working properly; and the denitrification capacity of the anoxic zone has not been exceeded. Higher levels of nitrates or nitrites indicate that denitrification has begun to fail, or the denitrification capacity of the zone is being exceeded. Table B-5 in the Appendix presents the results from the anoxic zone. These data are summarized in Table 7. An anoxic zone had not yet been added to the Control System and is therefore not considered in the table.

**Table 7. Anoxic Zone Nitrate and Nitrite Concentrations;
Experimental System; Days 59 to 177**

MCRT (days)	Nitrate-N Concentration (mg/l)	Nitrite-N Concentration (mg/l)
15	0.46	0.00
8	0.34	0.00
3	2.24	0.00
2@	7.89	0.50
2#	0.18	0.42
@ 1.0 liter clarifier operation . # 0.5 liter clarifier operation		

Denitrification was essentially complete at 15 and 8 day MCRT's, but the anoxic zone's capacity was exceeded during the 3 and 2@ day MCRT's as evidenced by anoxic zone nitrate-N concentrations of 2.24 and 7.89 mg/l. Denitrification appeared to improve during the 2# day MCRT. However, nitrification during this period was poor, and a smaller amount of nitrates were recycled back to the anoxic zone (see Table 5; effluent NO_x-N concentration of 4.56 mg/l). Hence the nitrate loading to the anoxic zone was lower than normal and at a level at which the anoxic zone could handle.

Day 177 to Day 220

On September 23, 1988 (day 177) chemostat operation was begun. This was done to facilitate operation at low MCRT's. From day 177 until 193, the aerobic MCRT and hydraulic retention time was two days; and from day 193 until day 220, the aerobic MCRT and hydraulic retention time was one day. Figure 6 presents the daily effluent ammonia-N concentrations during this time period. Tables 8 and 9 summarize the nitrogen removal results for this period. The effluent ammonia-N concentrations for both systems at both MCRT's were unusually low when compared to the concentrations during the 2# day MCRT period of operation prior to conversion to chemostats. The nitrification capacity for both systems should have

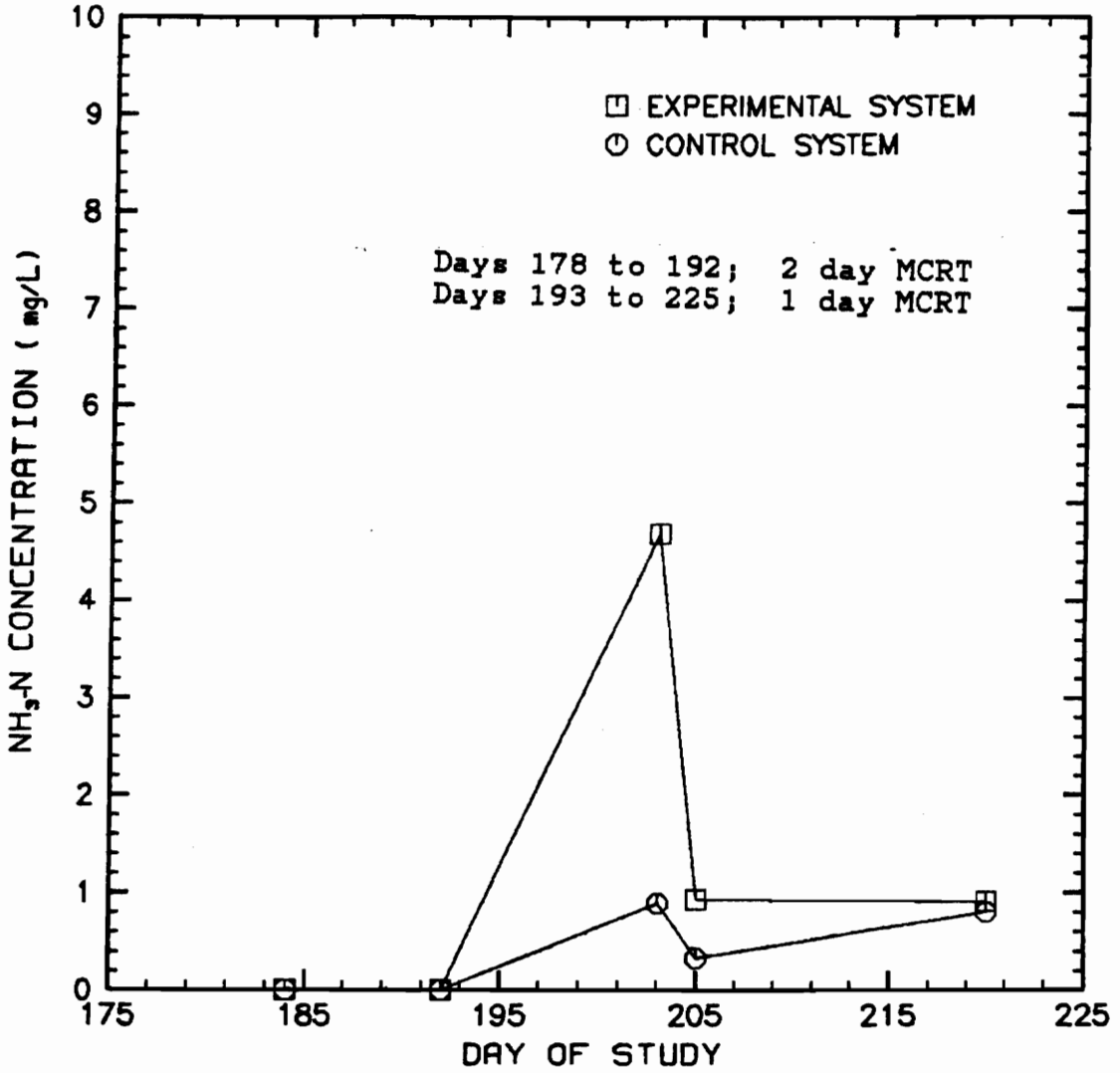


Figure 6. Effluent Ammonia Concentrations; Chemostat Operation

**Table 8. Nitrogen Removal Results; Experimental System
Chemostat Operation (mg/l)**

MCRT (days)	TKN		Ammonia-N		NOx-N		Percent-N Removal
	inf.	eff.	inf.	eff.	inf.	eff.	
2	62.4	3.35	56.7	0.00	0.11	11.2	76.7
1	61.7	5.31	55.9	2.18	0.34	20.8	57.9

**Table 9. Nitrogen Removal Results; Control System
Chemostat Operation (mg/l)**

MCRT (days)	TKN		Ammonia-N		NOx-N		Percent-N Removal
	inf.	eff.	inf.	eff.	inf.	eff.	
2	62.4	0.67	56.7	0.00	0.11	52.7	14.6
1	61.7	1.46	55.9	0.67	0.34	54.6	9.6

**Table 10. Anoxic Zone Nitrate and Nitrite Concentrations;
Experimental System; Chemostat Operation**

Day	MCRT (days)	Nitrate-N Concentration (mg/l)	Nitrite-N Concentration (mg/l)
184	2	1.85	7.12
192	2	0.39	0.00
203	1	6.70	6.18
205	1	4.95	9.64
220	1	8.65	1.63

continued to be exceeded during chemostat operation because of the low MCRT's at which the systems were operated. Since nitrification was virtually complete for both systems during this period, the actual MCRT of the systems must have been higher than the assumed MCRT. Table 10 presents the Experimental system anoxic zone nitrate and nitrite concentrations during chemostat operation. The first sample had a high nitrite concentration. However, denitrification appeared to be complete by day 192 when the second sample was taken as evidenced by the low nitrate and nitrite concentrations. This complete denitrification at a two day MCRT is inconsistent with results obtained earlier when the denitrification capacity of the anoxic zone was exceeded during 3 day MCRT operation. The denitrification capacity of the anoxic zone was obviously exceeded during the one day MCRT operation.

Day 221 to Day 428

Chemostat operation was discontinued on day 227. The Experimental System was placed back into its original configuration by reinstalling the clarifier and the sludge recycle line. The clarifier and the recycle line were also reinstalled on the Control System. In addition, a completely

aerated 1.5 liter tank was added to the front of the Control System; and a mixed liquor recycle line was also installed. The Control System, therefore, was identical to the Experimental System with the exception that the Control System consisted of 4.5 liters of aerated volume whereas the Experimental System consisted of 1.5 liters of anoxic volume and 3 liters of aerated volume. The influent feed to the Control System was also made identical to that of the Experimental System by feeding methanol to the Control System at the same rate as for the Experimental System. Operation of both systems at a three day total system MCRT was begun.

Starting on day 234 and continuing through day 238, a series of samples were collected. The measured effluent ammonia-N concentrations are shown in Figure 7. As can be seen, nitrification in the Control System was poor between days 234 and 238 as indicated by high ammonia-N concentrations. The Experimental System was nitrifying well as evidenced by the low effluent ammonia-N concentrations.

On day 238, approximately one liter of mixed liquor was transferred from the Experimental System to the Control System with the intention of seeding the Control System with nitrifiers. Starting on day 248 and running until day 320, sampling of both systems while still operating at a three day MCRT was conducted. Figure 7 presents the resulting effluent

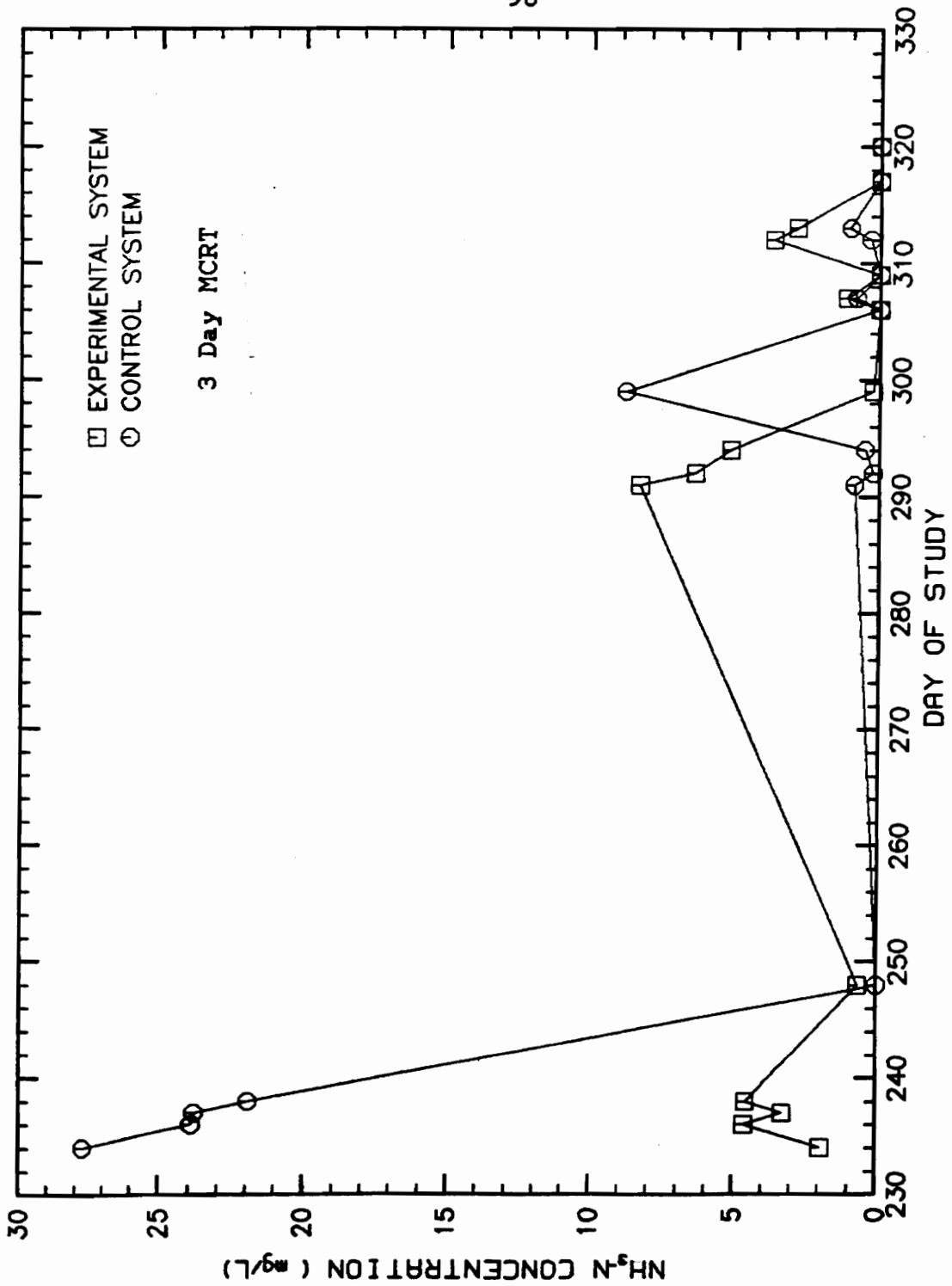


Figure 7. Effluent Ammonia Concentrations; Days 230 to 330

ammonia-N concentrations. Nitrification in the Control System had recovered, and nitrification in both systems was complete on many of the days.

Starting on day 321, the MCRT of both systems was gradually decreased. The systems were operated at a 2.5 day MCRT from days 321 to 349; a two day MCRT from days 350 to 394; and a 1.5 day MCRT from days 395 to 428. The effluent ammonia-N concentrations for this period are presented in Figure 8. The nitrogen removal results for days 221 to 428 are presented in Tables 11 and 12 for the Experimental and Control Systems respectively.

As can be seen from the individual day's results (Figure 8), it was not until the MCRT was reduced to two days that ammonia-N was consistently found in the effluent of the Experimental System. The MCRT had to be reduced down to 1.5 days before ammonia-N was regularly found in the Control System effluent. Therefore, the nitrification capacity of the Experimental System was exceeded at a two day MCRT; and the nitrification capacity of the Control System was exceeded at a MCRT of 1.5 days. The 30% nitrogen removal obtained by the Control System during 2.5 day MCRT (Table 12) was the highest percent removal achieved by the completely aerobic Control System during the entire study.

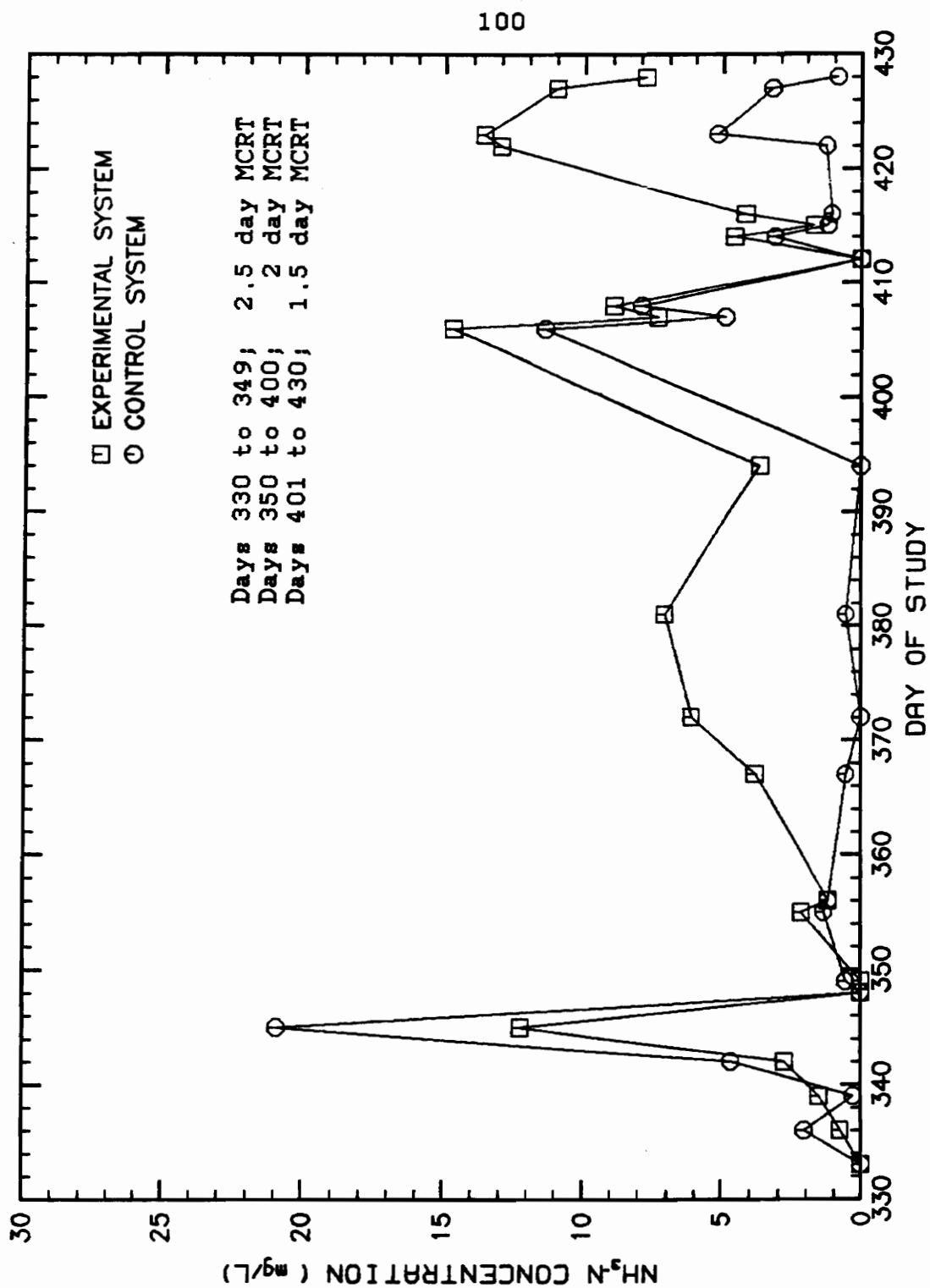


Figure 8. Effluent Ammonia Concentrations; Days 330 to 430

**Table 11. Nitrogen Removal Results; Experimental System
Days 221 to 428**

MCRT*	TKN		Ammonia-N		NOx-N		Percent-N Removal
	inf.	eff.	inf.	eff.	inf.	eff.	
3	60.2	6.21	54.2	2.68	0.64	14.3	66.3
2.5	67.3	6.32	62.5	2.47	0.79	14.8	68.1
2	65.1	7.29	61.4	4.02	0.88	13.5	68.5
1.5	61.1	11.7	56.6	7.91	0.15	15.3	55.9

* Concentrations in mg/l; MCRT in days

**Table 12. Nitrogen Removal Results; Control System
Days 221 to 428**

MCRT*	TKN		Ammonia-N		NOx-N		Percent-N Removal
	inf.	eff.	inf.	eff.	inf.	eff.	
3**	60.2	28.5	54.2	24.3	0.64	17.1	25.0
3	64.0	3.32	59.0	1.05	0.47	43.8	26.9
2.5	67.3	6.86	62.5	4.06	0.79	40.8	30.0
2	65.1	2.33	61.4	0.61	0.88	48.9	22.4
1.5	61.1	4.92	56.6	3.68	0.15	43.0	21.8

* Concentrations in mg/l; MCRT in days

** Operation prior to transfer of mixed liquor from Experimental to Control System

Table 13 presents the nitrate and nitrite concentrations in the anoxic zone of the Experimental System and in the 1.5 liter aerated tank of the Control System. The nitrite concentrations are given in parentheses. Denitrification in the Experimental System was not complete at any MCRT as evidenced by the presence of nitrates and nitrites in the anoxic reactor. The concentrations in the Control System reactor indicate that nitrification was not complete in this tank but continued in the second aerated tank. This is evident because the total nitrate and nitrite concentrations in the first tank (from 30 to 40 mg/l) were lower than the effluent total nitrate and nitrite concentrations for the same MCRT operating period (from 41 to 49 mg/l; see Table 12).

**Table 13. First Stage Nitrate(Nitrite) Concentrations
Days 221 to 428**

MCRT (days)	Nitrate(Nitrite) Concentration*	
	Experimental**	Control***
3	2.28(1.12)	33.3(1.72)
2.5	1.58(1.11)	30.2(2.23)
2	1.66(1.29)	36.2(3.90)
1.5	2.24(1.33)	27.9(2.28)

* Concentrations in mg/l
 ** Anoxic zone
 *** Aerobic zone

Day 436 to Day 554

On day 428, the Control System was converted from a completely aerobic activated sludge system to an MLE system identical to the Experimental System by suspending aeration in the first 1.5 liter tank. Operation of both systems at an MCRT of four days was also begun. The intention was to utilize the Experimental System for metals toxicity testing with the Control System acting as a MLE control. At a four day MCRT, it was expected that nitrification in both systems would have been essentially complete based on the previous results presented in this chapter. The effluent ammonia-N concentrations were monitored starting on day 441 and are presented on Figure 9. As can be seen, the effluent ammonia-N concentrations for both systems, although zero at the beginning of the period, increased substantially. Based on previous experience during this study, this was completely unexpected.

On day 460 the MCRT of both systems was increased to eight days. As the results in Figure 9 show, this appeared to improve nitrification as the effluent ammonia-N concentrations of both systems decreased down to zero mg/l by day 465. However, starting with day 468, the Control System effluent ammonia-N concentration was on the rise again. The Experimental System during this time period was being used for

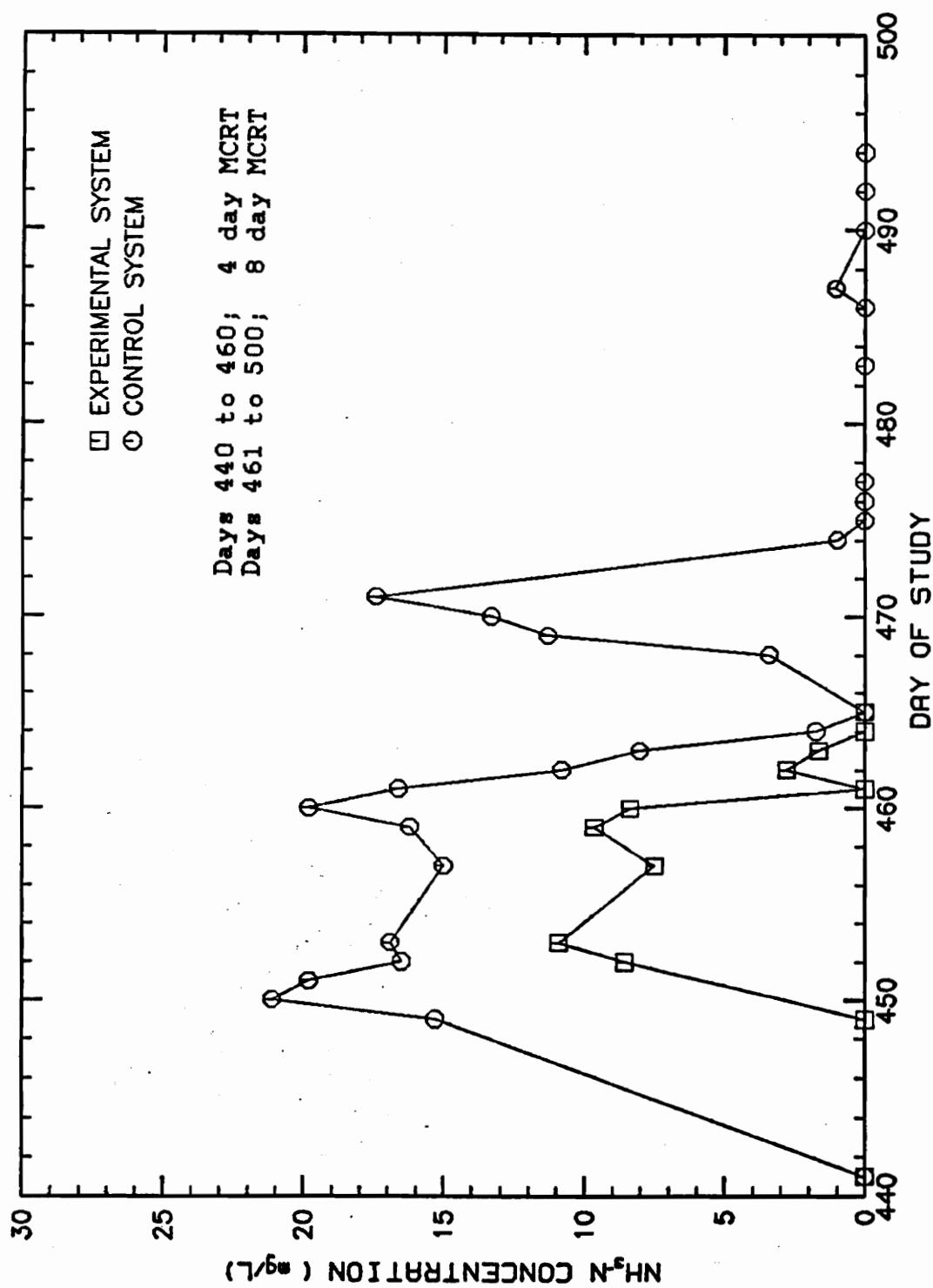


Figure 9. Effluent Ammonia Concentrations; Days 440 to 500

metal toxicity testing and, therefore, is not shown. The effluent ammonia-N concentration of the Control System peaked on day 471, and by day 475, had decreased to zero. This decrease had coincided with a new quantity of leachate, the feeding of which began on day 472. Upon reaching zero, the effluent ammonia-N concentration remained essentially zero through day 554 (Figure 9 only shows the data up to day 500).

On day 494, the Control System mixed liquor recycle flowrate was doubled from 12 liters/day (2Q) to 24 liters/day (4Q). This was done to increase the loading of nitrates to the anoxic reactor. Anoxic reactor and system effluent nitrate and nitrite concentrations were monitored before and after the flowrate change to determine if the anoxic zone could denitrify the higher nitrate loading and, therefore, lower the effluent total nitrogen concentration. The methanol feed was increased from one to 1.63 grams/day, and phosphorus addition was increased from 52.5 mg/day to 86 mg/day so that these nutrients would not be limiting. The results of the 4Q operating period (days 494 to 553) were compared with the results from the day 474 to day 483 time period during which the mixed liquor recycle was 2Q and the Control System was operating properly and nitrifying completely.

Figure 10 presents the effluent NO_x-N concentrations

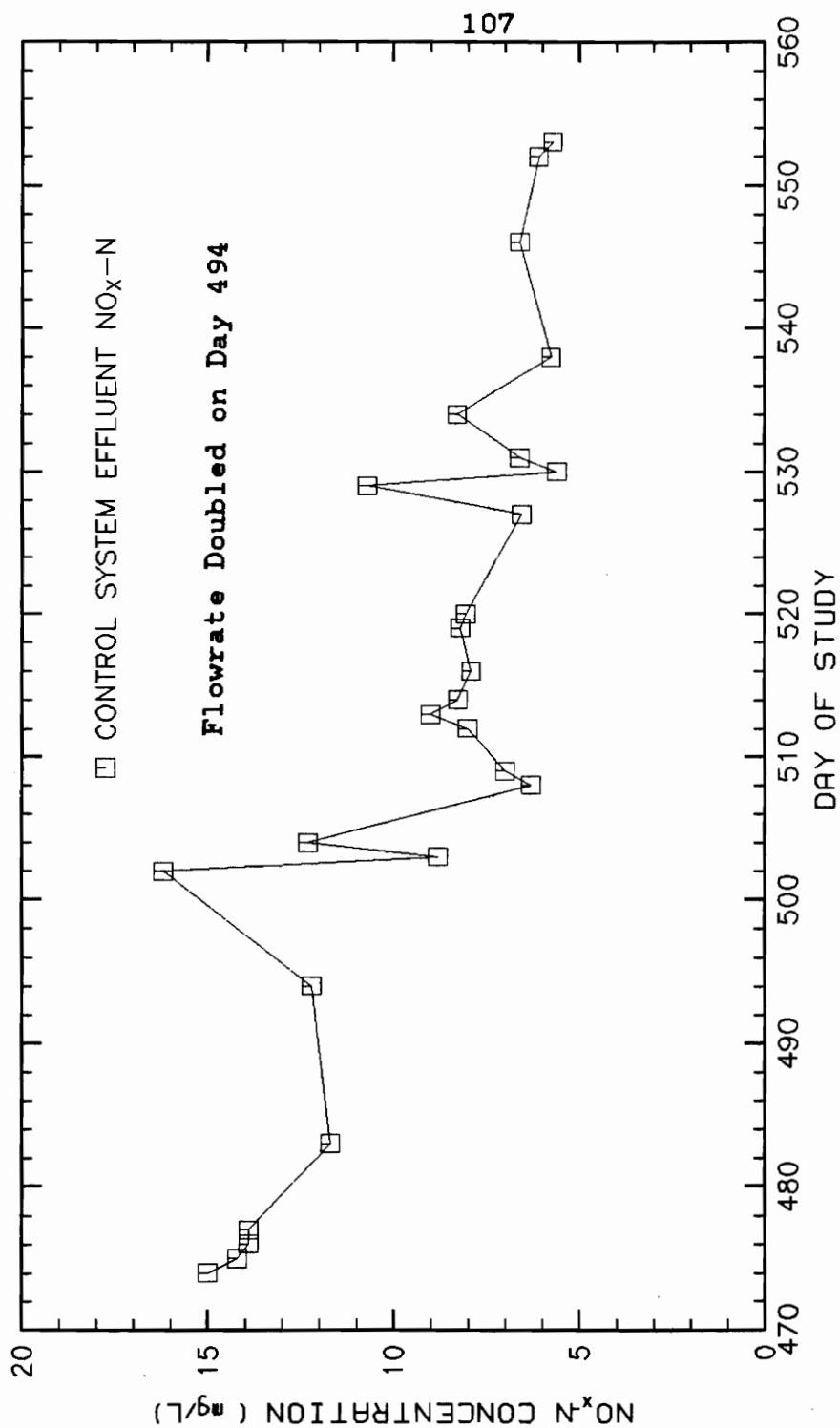


Figure 10. Effluent Nitrate Concentrations; Effect of Doubling Mixed Liquor Flowrate

during the 2Q time period (days 474 to 483) and the 4Q period (days 494 to 553). As can be seen, the effluent NO_x-N concentrations gradually decreased from a high of approximately 14 mg/l to a low of about 6 mg/l. Using the results during the 4Q period from days 516 to 553, the nitrogen removal results are listed in Table 14. A t-test performed on the two average effluent nitrate concentrations (13.7 mg/l vrs. 7.18 mg/l) concluded that the effluent nitrate from the 4Q period was lower than the nitrate from the 2Q period ($t\text{-obs} = 7.61$; $t\text{-crit} = 2.131$; significance level = .05). Clearly, increasing the nitrate recycle flowrate was effective in reducing the total effluent nitrogen and increasing the percent nitrogen removed. A review of Table B-5 for this time period shows that neither nitrates nor nitrites were found in the anoxic reactor during the 4Q period which indicates that denitrification was complete. The 83.7% nitrogen removal during the 4Q period was the highest percent removal obtained by a MLE process during the entire study.

Table 14. Nitrogen Removal Results; Effect of Increasing Nitrate Recycle Flowrate; Control System; MLE Mode

Recycle Flow	TKN inf.	TKN eff.	Ammonia-N inf.	Ammonia-N eff.	NOx-N inf.	NOx-N eff.	Percent-N Removal
12	57.9	3.03	50.5	0.20	0.00	13.7	71.1
24	58.9	2.59	56.8	0.00	1.21	7.18	83.7

* All parameters in mg/l except for Recycle Flow which is in liters/day and for Percent Removal

Day 555 to Day 584

From day 555 until day 573, the Control System continued to operate as a MLE process with a 4Q recycle flow rate. Figure 11 presents the Control System effluent concentrations during this time period. As can be seen, nitrification continued to be complete. From day 465 until day 554, the Experimental System had been utilized for metal toxicity testing during which nitrification had essentially been destroyed (results not presented in this thesis). On day 556, the mixed liquor in the Experimental System was completely discarded and replaced with two liters of mixed liquor from the Control System with the intention of re-establishing nitrification in the Experimental System. On day 559, another liter of mixed liquor was transferred from the Control System to the Experimental System. Figure 11 also contains the effluent ammonia-N concentrations for the Experimental System. As can be seen, although fluctuating, nitrification recovered; the effluent ammonia-N concentration decreased to zero mg/l by day 573.

Beginning with day 572, powdered activated carbon (PAC) was added on a daily basis to the Control System to reduce the effluent COD (to be discussed later). Figure 11 shows that, starting with day 575, nitrification began to steadily

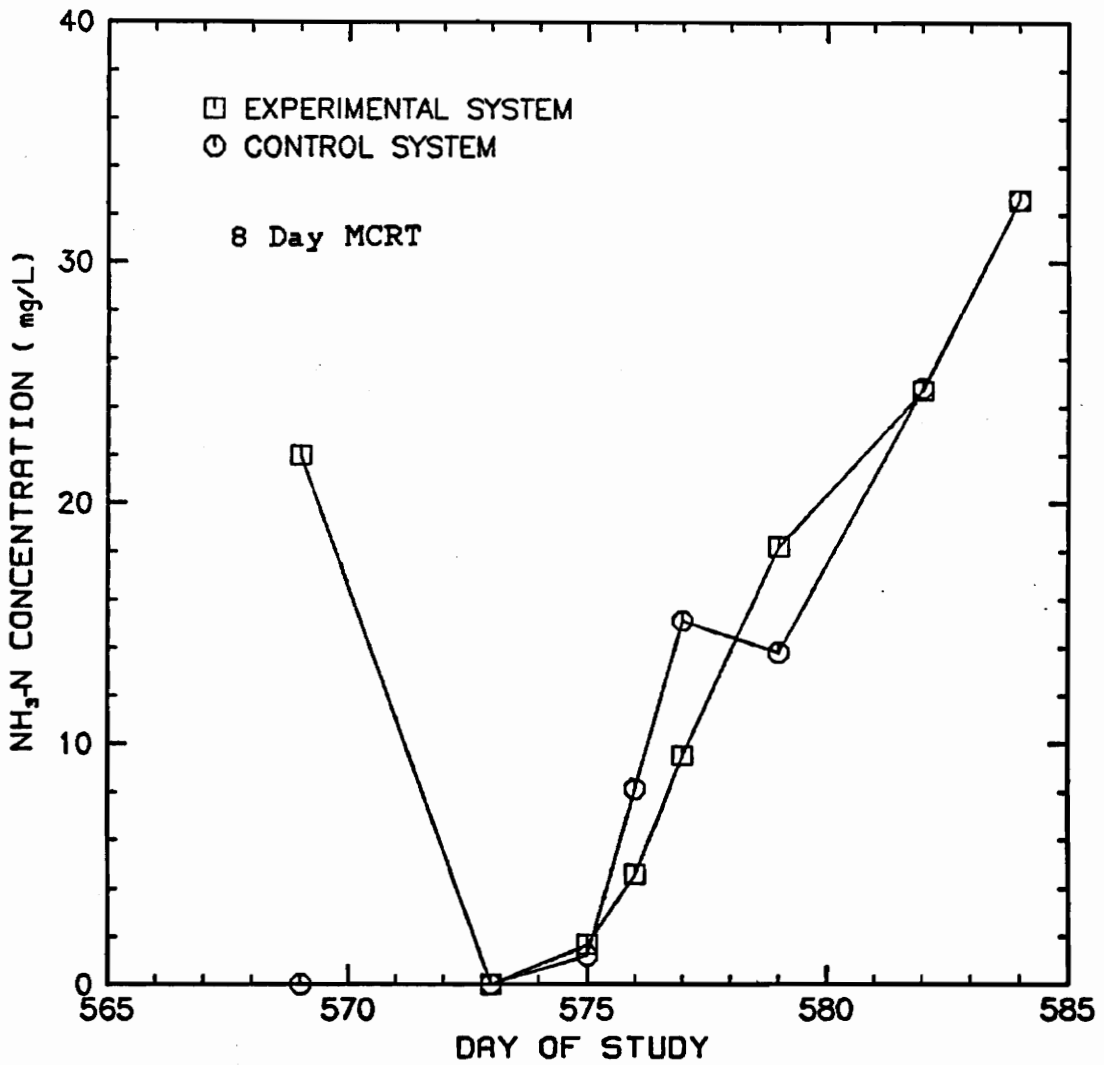


Figure 11. Effluent Ammonia Concentrations; Days 555 to 585

degrade in both systems. During this time period, the Experimental System was in effect acting as a control system as no carbon was being added to it. If the carbon was the cause of the loss of nitrification in the Control System, the effluent ammonia-N concentrations of the Experimental System would have remained low as nitrification had been re-established in that system. It is concluded that the addition of PAC was not the cause of the loss of nitrification since nitrification was lost in both systems. Foaming was also observed on the surface of the aeration tanks of both systems indicating solids destruction. The study was terminated on day 584 when the effluent ammonia-N concentrations from both systems reached a high of 32.6 mg/l.

DAPHNIA TOXICITY TESTS RESULTS

During the two periods of nitrification inhibition discussed above, it was decided to perform daphnia toxicity testing on a number of leachate batches that the systems were treating during the inhibition periods. For each batch of leachate tested, 100% leachate and one dilution (75% or 80%) were tested. The results presented in Table 15 represent the average percent mortality of the duplicate tests for each

dilution. The date and day number when each batch of leachate was collected is also provided. As can be seen, the leachate was relatively toxic to the daphnia during days 436 to 472, was mildly toxic during days 526 to 538 (9/6/89 leachate) and relatively toxic from days 561 to day 584. As Figures 9 and 11 show, the periods of inhibition occurred from days 449 to 474 and from days 575 to 584. The two periods of nitrification inhibition occurred when the leachate being treated was most toxic to the daphnia.

Table 15. Daphnia Toxicity Testing Results

Leachate Batch	Day #	% Mortality	
		75% Dilution	100% Dilution
6/8/89	436	30*	70
6/26/89	454	78*	100
7/14/89	472	0.0	0.0
7/31/89	489	0.0	0.0
9/6/89	526	20	60
10/11/89	561	20	95
10/31/89	581	20	80
* 80% Dilution			

ORGANICS REMOVAL

Organics Removal By Biodegradation

The COD, BOD₅, and TOC influent and effluent concentrations measured throughout the study are presented in Tables B-8, B-9, and B-10, respectively, in Appendix B. The BOD₅ concentrations of the leachate were consistently less than 10 mg/l. Due to these low levels, the monitoring of BOD's was discontinued early in the study. TOC's were also only monitored early to confirm that the leachate COD was due to organic compounds. As was shown in Table 1, the average COD concentration of the leachate throughout the study was 90.2 mg/l, and the average TOC concentration was 27.8 mg/l.

During the first 226 days of the study, methanol was not fed to the Control System. Therefore, the Control System effluent COD during this period indicated the biodegradability (or lack thereof) of the leachate organics. Methanol was added to the Experimental System, and the possibility that methanol was a component of the Experimental System's effluent COD, although extremely unlikely, could not be discounted. In theory, therefore, the Experimental System should not be used to determine the biodegradability of the organics in the leachate.

Table 16 presents the influent and effluent COD averages for both systems at the various operating conditions prior to the beginning of methanol addition to the Control System.

The results show that little of the leachate's COD was biodegradable in either treatment system. During the 15, 8, and 3 day MCRT operating periods, the Control System removed only 2.8% of the influent COD whereas the Experimental System removed an average of only 4.8%. The low biodegradability of this leachate was expected based on the low BOD concentrations and the age of the landfill. The data in Table 16 also indicate that methanol was probably not breaking through the Experimental System as evidenced by effluent COD concentrations as low as the Control System's effluent COD concentrations.

The COD removal capabilities of both systems continued to be poor throughout the study (see Table B-8 in Appendix B).

Table 16. Influent and Effluent COD Concentrations

MCRT (days)	Influent (mg/l)	Control Effluent (mg/l)	Experimental Effluent (mg/l)
15	74.5	70.3	71.8
8	85.4	85.0	88.1
3	87.2	86.6	80.6
2*	88.7	102	83.5
2**	85.4	90.7	91.3
2@	93.9	79.0	90.1
1@	93.1	88.0	82.5

* Operation with 1.0 liter clarifiers

** Operation with 0.5 liter clarifiers

@ Chemostat operation

Organics Removal by Granular Activated Carbon

The effectiveness of granular activated carbon (GAC) in reducing the COD from the effluent of one of the systems was briefly investigated. A GAC jar test was conducted on day 444. Five hundred milliliters of Control System effluent was placed in each of six jar test flasks. GAC (Calgon 400) was placed in five of the flasks in 0.125, 0.25, 0.5, 0.75 and 1.00 gram amounts. The jars were then mixed to keep the carbon suspended for approximately 40 hours. The COD of the solution in each jar after 40 hours was then determined. The results are presented in Table 17. The GAC reduced the COD from a high of 76.1 mg/l in the jar without any carbon to a low of 12.0 mg/l in the jar with 1.00 gram of carbon. It was also noted that during the test, the GAC removed color. The Control System effluent had a faint yellowish tint but was virtually clear after carbon treatment.

Table 17. COD Removal by GAC

Flask	Volume (ml)	Carbon (M) (mg)	Final COD (mg/l)	COD Removed (X) (mg)	X/M (mg/mg)
1	500	0.00	76.1	----	----
2	500	125	54.6	10.8	0.086
3	500	250	40.6	17.8	0.071
4	500	500	22.3	26.9	0.054
5	500	750	14.1	31.0	0.041
6	500	1000	12.0	32.1	0.032

Effectiveness of Powdered Activated Carbon

The second activated carbon test involved adding powdered activated carbon (PAC) directly to an activated sludge system treating the leachate. PAC was added directly to the Control System while it was operating in the MLE mode. Dosing with PAC began on day 572 and continued until day 585. Table 18 presents the amount of carbon added and the Control System's influent and effluent COD concentrations each day during the test.

No attempt was made to determine the optimum carbon addition needed to minimize the effluent COD. The goal of this test was only to determine if the PAC could be effective in reducing the COD of the treatment system effluent. Clearly, from the results presented in Table 18, the PAC was effective in reducing the COD from a high of 91.0 mg/l the day the test was begun down to a minimum of 19.0 mg/l. Throughout the study, the addition of PAC was able to continually reduce the effluent COD concentration to about one half the influent COD. During this time period, the Experimental System was acting as a control as PAC was not being added to this system. The effluent COD concentration of the Experimental System was monitored several times and found to average 85.5 mg/l. As was discussed earlier, the nitrification capabilities of both systems were lost while

Table 18. Powdered Activated Carbon Test; Control System

Day	PAC Added (grams)	Influent COD (mg/l)	Effluent COD (mg/l)
572	3.00	82.4	91.0
573	0.36	82.4	19.0
574	0.36	82.4	----
575	0.75	82.4	41.7
576	0.00	82.4	34.4
577	1.00	82.4	50.5
578	1.00	82.4	----
579	1.00	82.4	29.6
580	1.00	82.4	----
581	1.00	82.4	----
582	1.00	82.4	47.7
583	1.00	92.9	----
584	0.00	92.9	40.6

this test was being conducted. However, it is believed that the addition of PAC was not the cause for this loss of treatment capability.

METALS REMOVAL

Table B-11 in Appendix B presents total influent and soluble effluent iron and manganese concentrations for both systems at various times during the study. Iron and manganese are the only metals presented because the other metals in the leachate were present in very low concentration (see Table 1). As can be seen, both systems consistently reduced the iron concentrations to below 1.00 mg/l. Manganese reductions were not as great as iron, but the influent manganese concentrations were normally below 1.00 mg/l to begin with.

ADDITIONAL DATA

Mixed Liquor Suspended Solids

Table B-6 in Appendix B presents the mixed liquor suspended solids (MLSS) and mixed liquor volatile suspended solids (MLVSS) concentrations during the study. These data are summarized in Tables 19 and 20 for each operating condi-

Table 19. Solids Concentrations in Reactors; Days 59 to 220

Days	MCRT*	Exp. System**		Control System	
		Suspended	Volatile	Suspended	Volatile
59-73	15	1480	1194	665	473
84-90	8	1136	973	325	218
99-116	3	351	299	92	63
121-124	2@	203	171	73	54
156-177	2#	297	248	65	29
184-192	2\$	214	165	51	31
203-220	1\$	108	89	31	21
* Aerobic Sludge Age ** Concentrations in mg/l @ Operation with 1.0 liter clarifier # Operation with 0.5 liter clarifier \$ Chemostat Operation					

Table 20. Solids Concentration in Reactors; Days 234 to 428

Days	MCRT*	Exp. System**		Control System	
		Suspended	Volatile	Suspended	Volatile
234-238	3#	----	----	273	238
248-321	3@	281	252	283	260
331-349	2.5	258	233	262	240
354-394	2	241	216	235	206
405-428	1.5	165	133	197	163

* Total System Sludge Age

** Concentrations in mg/l

Control System Operation Before Solids Transfer From Exp.

@ Operation After Solids Transfer

tion and MCRT up until day 428 (end of 1.5 day MCRT operation). Table 19 presents the solids concentrations during the "aerobic MCRT" period when the Control System consisted of the 3 liter completely aerobic reactor. Table 20 presents the concentrations during the "total system MCRT" period when the Control System consisted of the two stage 4.5 liter completely aerobic system.

Soluble Effluent Orthophosphorus

Table B-12 in Appendix B presents the effluent soluble orthophosphorus concentrations for both systems. This analysis was performed occasionally throughout the study to ensure that sufficient phosphorus was being added to prevent phosphorus limitations. Investigators at the University of British Columbia had recommended that the soluble orthophosphorus concentration in the reactors of a leachate treatment system not drop below 0.5 mg/l (54). As can be seen from Table B-12, the soluble orthophosphorus concentrations were well above this minimum level throughout the study.

Leachate and System pH Values

Table B-13 in the Appendix presents the pH values of the influent leachate, the mixed liquor from the systems' 1.5 liter reactors and of the system effluents. The pH of the

leachate varied from 6.48 to 7.02 whereas the pH's of the systems ranged from about 7.5 to about 8.5 with no unusually large variations.

Chapter V. DISCUSSION

COMPARISON OF NITRIFICATION RATES

The utilization of the MLE process which places an anoxic zone in front of an aerobic reactor to promote denitrification caused no detrimental effects to nitrification and resulted in several positive effects. As was shown in the results provided in Chapter 4, the average effluent ammonia-N concentrations from the Experimental System using the MLE process were the same as or lower than the concentrations from the Control System during operations at three and two day aerobic MCRT's (see Tables 5 and 6). In addition during operation at the low MCRT's (three days and below) when nitrification was not complete, the Experimental System consistently nitrified as much as or more nitrogen per liter of aerobic volume than did the Control System. This can be seen by comparing the nitrification amounts and rates of both

systems for two periods of operation, days 59 to 177 and days 221 to 428, which are presented below.

Day 59 to Day 177

Tables 21 and 22 present nitrification and denitrification rates for this period of the study. The Experimental System was operating as a MLE process and the Control System was a 3.0 liter completely aerobic system. Methanol was being fed only to the Experimental System. The MCRT's presented are aerobic MCRT's and not total system MCRT's. The amount of nitrogen nitrified and denitrified per day (in mg/day) was determined by appropriate nitrogen mass balances around the systems. An example calculation is provided in Appendix A. The nitrification values were then divided by 3.0 liters (the aerated volume) to obtain the amount of nitrogen nitrified each day per unit volume (in mg/l-day). The percent nitrified was determined by dividing the amount nitrified by the amount of nitrogen available for nitrogen. For the Experimental System, the amount denitrified was divided by 1.5 liters (the volume of the anoxic tank) to obtain the amount of nitrogen denitrified each day per unit volume (in mg/l-day). For the Control System the amount denitrified was divided by the total 3.0 liter system volume

since it did not contain an anoxic volume. The percent denitrified was determined by dividing the amount of nitrogen denitrified by the amount of nitrate available. The nitrification and denitrification rates (in l/day) were determined by dividing the amount of nitrogen nitrified or denitrified per day (in mg/day) by the mixed liquor volatile suspended solids (in mg/l) and then dividing by the appropriate volume (in liters).

As can be seen by comparing the two tables, the nitrification rates of the Control System are many times higher than those of the Experimental System at all MCRT's even though the amount of nitrogen nitrified by both systems was about the same. This was due to the fact that methanol was added to the Experimental System but not the Control System which caused the MLVSS of the Experimental System to be much higher. As this methanol addition and corresponding higher MLVSS concentration were not related to nitrification, the utilization of the traditional nitrification rates for determining nitrification capacity or capability is not useful in this situation. However, the amount of nitrification per day per unit volume is a better indicator of nitrification ability.

As can be seen when comparing the two tables, the Control System nitrified more nitrogen than did the Experimental System at the 15 and 8 day aerobic MCRT's. However, this was

Table 21. Nitrification/Denitrification Rates; Experimental System; Days 59 to 177

	Aerobic MCRT (days)				
	15	8	3	2@	2#
MLVSS (mg/l)	1194	973	299	174	248
N-Nitrified (mg/day)	296	278	283	297	174
(mg/l-day)	98.7	92.7	94.3	99.0	58.0
% Nitrified	100	100	97.7	99.0	58.5
Nitrification Rate (1/day)	0.083	0.095	0.316	0.579	0.233
N-Denitrified (mg/day)	253	231	211	194	147
(mg/l-day)	169	154	141	129	98.0
% Denitrified	81.5	82.4	74.0	64.9	84.0
Denitrification Rate (1/day)	0.141	0.158	0.471	0.76	0.395
@ Operation with 1.0 liter clarifier					
# Operation with 0.5 liter clarifier					

Table 22. Nitrification/Denitrification Rates; Control System; Days 59 to 177

	Aerobic MCRT (days)				
	15	8	3	2@	2#
MLVSS (mg/l)	473	218	63	54	29
N-Nitrified (mg/day)	326	320	281	326	158
(mg/l-day)	109	107	93.7	109	52.7
% Nitrified	100	100	85.5	97.4	48.2
Nitrification Rate (1/day)	0.23	0.49	1.49	2.03	1.79
N-Denitrified (mg/day)	34	64	55	51	28
(mg/l-day)	11	21	18	17	9.3
% Denitrified	10.0	19.8	19.4	15.6	17.7
Denitrification Rate (1/day)	0.024	0.099	0.291	0.314	0.316
@ Operation with 1.0 liter clarifier					
# Operation with 0.5 liter clarifier					

because there was more nitrogen available for nitrification. Due to methanol addition, the Experimental System's MLVSS concentration was much higher than the Control System's MLVSS concentration. Therefore, more solids were wasted from the Experimental System each day than from the Control System; and with the solids, more nitrogen. In addition there was more organic nitrogen in the Experimental System effluent than in the Control System effluent as evidenced by higher levels of effluent TKN concentrations. This left less nitrogen in the Experimental System to be nitrified. The fact that the effluent ammonia-N concentrations from both systems were zero at 15 and 8 day MCRT's shows that all the available nitrogen was nitrified and that the nitrification capacities of both systems were not exceeded.

As the aerobic MCRT of both systems were decreased down to three days, ammonia-N started to appear in the effluents of both systems indicating that the nitrification capacity of both systems had been exceeded. Judging from the average effluent ammonia-N concentrations from both systems (1.10 mg/l from the Experimental System and 7.94 mg/l from the Control System) it would appear that the Experimental System was nitrifying at a higher rate than the Control System. However, when reviewing the amount nitrified in Tables 21 and 22, both systems nitrified approximately the same amount of

nitrogen (94.3 mg/l-day for the Experimental System and 93.7 mg/l-day for the Control System).

During two day MCRT operation with 1.0 liter clarifiers installed, the amount of nitrogen nitrified increased in both systems (99.0 mg/l-day for the Experimental System and 109 mg/l-day for the Control System). As with the 15 and 8 day MCRT operating periods, the lower Experimental System nitrification amount was due to the fact that there was essentially no nitrogen left to nitrify as signified by a percent nitrified value of 99%. This indicates that the additional retention time in the clarifiers allowed for continuing nitrification to occur as was discussed earlier.

During two day MCRT operation with the original 0.5 liter clarifiers, the nitrification capacity of both systems was clearly exceeded. Figure 5 in Chapter 4 shows that significant amounts of ammonia-N were found in the effluents of both systems during this period of operation (days 156 to 177) everytime a sample was taken. The average amount of nitrogen nitrified in the Experimental System was higher than that nitrified by the Control System (58.0 mg/l-day versus 52.7 mg/l-day). However, as was discussed in Chapter 4, the t-test for this period indicated that the effluent ammonia-N concentrations for the two systems were not different which would also indicate that the amount nitrified by the two

systems was statistically the same. The minimum that can be concluded is that, during this period, the Experimental System nitrified at least as much as the Control System.

In summary, during this period the MLE system nitrified as least as much nitrogen as the completely aerobic Control System during low MCRT operating periods when incomplete nitrification occurred. This demonstrates that the placement of an anoxic zone in front of the aerobic zone causes no detrimental effects to nitrification. In addition, the need to add methanol to the Experimental System to fuel denitrification and the resulting higher MLVSS concentrations caused the average effluent ammonia-N concentrations from the Experimental System to be equal to or lower than those from the Control System during low MCRT operation.

Day 221 to Day 428

Tables 23 and 24 present the nitrification and denitrification amounts and rates for the Experimental and Control Systems respectively during this period of the study. during this time period, both systems were operated as conventional systems with clarifiers and sludge recycle lines installed. The Control System was made identical in physical configura-

Table 23. Nitrification/Denitrification Rates; Experimental System; Days 221 to 428

	MCRT (days)			
	3	2.5	2	1.5
MLVSS (mg/l)	252	233	216	133
N-Nitrified (mg/day)	303	333	300	255
(mg/l-day)	101	111	100	85
% Nitrified	95	95.7	92.6	84.3
Nitrification Rate (1/day)	0.401	0.476	0.463	0.639
N-Denitrified (mg/day)	220	249	223	165
(mg/l-day)	147	166	149	110
% Denitrified	71.7	73.7	73.0	64.4
Denitrification Rate (1/day)	0.58	0.69	0.69	0.82

Table 24. Nitrification/Denitrification Rates; Control System; Days 221 to 428

	MCRT (days)				
	3*	3	2.5	2	1.5
MLVSS	238	260	240	206	163
N-Nitrified (mg/day)	154	325	338	329	288
(mg/l-day)	34	72	75	73	64
% Nitrified	51.4	98.1	93.4	98.9	92.9
Nitrification Rate (1/day)	0.144	0.278	0.313	0.355	0.393
N-Denitrified (mg/day)	56	65	98	41	30
(mg/l-day)	12	14	22	9.1	6.7
% Denitrified	35.5	19.8	28.6	12.3	10.4
Denitrification Rate (1/day)	0.0511	0.0556	0.0778	0.0444	0.18
* Operation prior to solids transfer					

tion to the Experimental System with the exception that it was completely aerated. Methanol was also fed to the Control System at the same rate as the Experimental System. The goal for this period of the study was to compare the nitrification capabilities of the two systems treating the exact same waste (leachate plus methanol). As was shown in Figure 7 and can be seen from Table 12, the Control System nitrification was poor during days 234 to 238. From Table 24, only an average of 154 mg/day of nitrogen was nitrified. This is compared to 325 mg/day during days 248 to 320; after mixed liquor was transferred from the Experimental System to the Control System. It is not clear what caused the nitrification inhibition during days 234 to 238. The System had been nitrifying well during chemostat operation just prior to this time period. The MLVSS concentration was 238 mg/l during this period which compared well with the Experimental System solids concentration of 252 mg/l during three day MCRT operation. Conversion from chemostat operation back to a conventional system with solids recycle should not have caused the inhibition because nitrification in the Experimental System which underwent the same conversion was virtually complete during this period as the effluent ammonia-N concentrations for this system were less than 5 mg/l. The additional 1.5 liters of aerated volume added to the Control System should

not have caused the problem. The only major change to the Control System was the initiation of methanol addition. Hooper and Terry (56) found that methanol at a concentration of 5×10^{-3} M completely inhibited the conversion of ammonia to nitrate by *Nitrosomonas europaea*. An addition rate of approximately 167 mg of methanol per liter of leachate to the Control System would have caused the influent methanol concentration to be 5.22×10^{-3} M. In the experiments, Hooper and Terry utilized a *Nitrosomonas* concentration of 200 mg/l for their studies. However, this concentration was based on the wet weight of the microorganisms. If the dry weight was only 10% of this wet weight, the microorganism concentration would have been 20 mg/l. The volatile suspended solids concentration in the Control System during one day MCRT chemostat operation prior to the addition of methanol was 21 mg/l. Therefore, it appears that the methanol to nitrifier ratio in the Control System at the initiation of methanol addition could have been equivalent to the ratio at which Hooper and Terry found the methanol to be inhibiting. Based on these considerations, there is a possibility that the nitrifiers in the Control system could not acclimate to the methanol.

Another possibility for the poor nitrification is that the nitrifier growth rate was reduced because the hetero-

trophs growing rapidly on the newly added methanol out competed the nitrifiers for nutrients such as dissolved oxygen.

Nitrification improved dramatically after mixed liquor was transferred from the Experimental System to the Control System. A possible explanation is that the mixed liquor introduced into the Control System a strain(s) of *Nitrosomonas* from the Experimental System that was already acclimated to the methanol. Another possibility is that this transfer could have increased the nitrifier concentrations high enough to overcome competition from heterotrophs.

Once nitrification was established in the Control System, an examination of Tables 23 and 24 shows that the Control System nitrified more nitrogen than did the Experimental System at each MCRT. Also, as was discussed earlier in Chapter 4, the Control System MCRT had to be reduced to 1.5 days before ammonia was routinely observed in the effluent as compared to a two day MCRT for the Experimental System. This higher total nitrification capability of the Control System would be expected because of its 4.5 liter aerated volume as compared to only 3 aerated liters for the Experimental System.

However, of more interest is the comparison of the total amount of nitrogen nitrified and the amount nitrified per

liter of aerated volume (in mg/l-day). During the 1.5 day MCRT period of operation, when the nitrification capacity of both systems was clearly exceeded, the Experimental System nitrified 255 mg/day whereas the Control System nitrified 288 mg/day. A t-test at a .05 significance level was performed on the amounts nitrified during this time period. The results showed that the amount nitrified by the Control System was statistically higher than that nitrified by the Experimental System ($t\text{-obs} = 2.89$; $t\text{-crit} = 2.09$). However, when divided by the aerobic volumes (3 liters for the Experimental System; 4.5 liters for the Control), the amount nitrified per liter of aerated volume in mg/l-day by the Experimental System was higher (85 mg/l-day) than that nitrified by the Control System (64 mg/l-day). A t-test performed on these results showed that the Experimental System did nitrify more nitrogen per unit volume ($t\text{-obs} = 6.43$; $t\text{-crit} = 2.09$). The Control System, therefore, nitrified more nitrogen but needed 50% more aerobic volume to do so and nitrified less nitrogen per unit volume.

Denitrification in the anoxic zone of the Experimental System was not complete at any MCRT during this period of the study as indicated by the fact that substantial amounts of nitrate and nitrite were found in the anoxic zone during this period (see Table 13 in Chapter 4). As can be seen in Table

23, the percent of nitrates denitrified during the 3, 2.5, and 2 day MCRT periods of operation remained steady at about 73%. This is about what would be expected based on the recycle rate. With a total recycle rate of 3:1 (mixed liquor rate 2:1; sludge recycle 1:1) about 75% of the nitrates formed in nitrification would be recycled back to the anoxic zone for denitrification. However, the percent denitrified dropped to 65% during the 1.5 MCRT period. This substantial drop in denitrification capacity occurred even though the loading of nitrates to the anoxic zone was low because of limited nitrification (only 255 mg/l-day nitrogen nitrified).

In summary, the results of this operating period have shown that the Experimental system operating as an MLE process nitrified more nitrogen per day per unit volume than a completely aerobic system treating the same influent. The results of the 1.5 day MCRT period, indicate that the MLE process would need less aerobic volume to nitrify the same amount of nitrogen. However, further investigation is needed to conclude this. The additional studies would need to be designed such that incomplete nitrification occurs in both systems at a variety of MCRT's higher than one or two days. These studies, therefore, would most likely need to be conducted at temperatures below 20C.

Although not complete at the low MCRT's, denitrification in the Experimental system was not substantially degraded until the MCRT was reduced below two days.

EFFECT OF DOUBLING MIXED LIQUOR RECYCLE FLOWRATE

Day 436 to Day 553

During this time period, the Control System was modified to utilize the MLE process. On day 494, the nitrate recycle flowrate was doubled from 2Q to 4Q. As was presented earlier, this increase in flowrate decreased the effluent nitrate-N concentration from 13.7 mg/l to 7.18 mg/l, thereby increasing the percent of nitrogen removed from 71.1% to 83.7%; the highest percent removed during the study. Table 25 presents the nitrification and denitrification amounts and rates for the Control System during this period. These results show that during the 4Q period (days 516 to 553) the system denitrified approximately 40 mg of nitrate per day more than was denitrified by the system during the 2Q period (days 474 to 483). The percent denitrified increased from 74% to 88%. No nitrates were detected in the anoxic reactor, which indicated that the denitrification capacity of

Table 25. Nitrification/Denitrification Rates; Control System; Effect of Increasing Mixed Liquor Recycle Flowrate

	Recycle Flowrate (l/day)	
	12	24
MLVSS	486	836
N-Nitrified		
(mg/day)	306	291
(mg/l-day)	102	97
Nitrification		
Rate (l/day)	0.21	0.116
N-Denitrified		
(mg/day)	224	262
(mg/l-day)	149	175
% Denitrified	74.2	87.8
Denitrification		
Rate (l/day)	0.307	0.209

the reactor was not exceeded under these conditions.

RESIDUAL EFFLUENT TKN

A review of the TKN concentrations presented in Table 5 for the 15 and 8 day aerobic MCRT operating periods shows that the average Experimental System effluent TKN concentrations were 2.52 and 2.73 mg/l respectively. The effluent TKN concentrations from the Control System during the same operating period were 0.10 and 0.43 mg/l. This soluble residual TKN must have been organic nitrogen as there was no ammonia in the effluents of either system. This residual TKN was not nonbiodegradable organic nitrogen from the influent leachate that did not break down while passing through the systems. This can be proved by reviewing the Control System results for the 15 and 8 day MCRT operating periods. During this time period, the influent organic nitrogen concentration was approximately 5 mg/l (influent TKN minus the influent ammonia-N concentrations) whereas the Control system effluent organic nitrogen concentration (TKN) for the same period averaged only about 0.26 mg/l. This indicates that the organic nitrogen in the leachate is essentially completely deaminated and nitrified while in the treatment systems. The

residual TKN therefore, must be produced during the treatment process. An explanation could be that the residual TKN was from the production of extracellular material by heterotrophic bacteria metabolizing methanol. As the Control System was not being fed methanol and had a much lower MLVSS concentration during this time period, the Control System's residual TKN was much lower.

Further review of Tables 5, 8, 11, 12 and 14 indicates that residual TKN continued to be present throughout the study. This residual organic nitrogen was originally low in the Control System effluent but increased when methanol addition was started on day 227.

NITRIFICATION INHIBITION

A definitive explanation for the two periods (days 441 to 475 and days 573 to 584) during which nitrification failed can not be found. It is most probable that some component intermittently entering and leaving the leachate, either a toxic organic compound or a metal, caused the nitrification inhibition for the following reasons:

1. Nitrification inhibition occurred in both systems at

the same time.

2. Until the inhibition periods occurred, both systems had operated for well over a year without any similar problems occurring.

3. The temperature during these periods of the study was constant at 20C and the pH of the systems was between 8 and 8.5; both parameters being in the range for good nitrification.

4. The soluble orthophosphorus concentrations during the inhibition periods were found to be well above the recommended minimum 0.5 mg/l recommended (Table B-12). No indication that either system was nutrient limiting ever occurred throughout the study.

5. The leachate that was being treated when the first inhibition period began (collected on day 436) was just about the most concentrated batch of leachate treated during the entire study. The COD concentration was 106.8 mg/l (the highest throughout the entire study). The TKN concentration was 64.9 mg/l which, although not the highest during the study, was well above the average TKN of 58.3 mg/l. The iron concentration of the leachate was 22.5 mg/l which was above the average of 18.6 mg/l. The next quantity of leachate treated (collected on day 454) was also relatively strong with a COD concentration of 99.0 mg/l, a TKN concentration of

64.4 mg/l and an iron concentration of 24.2 mg/l. Although none of the concentrations listed above are unusually large, they do indicate that the leachate being treated during the inhibition period was of comparatively high strength. In addition, the results from daphnia toxicity testing (Table 15) indicate that the two leachates discussed above were more toxic to daphnia than other batches of leachate that were tested.

6. The first period of inhibition in the Control System ended abruptly on day 475. A new quantity of leachate had been obtained and was fed to the systems beginning with day 472. The strength of this leachate was considerably weaker than that of the previous two quantities discussed in 5 above. This leachate had a COD concentration of 86.4 mg/l, and a TKN concentration of 57.8 mg/l (metals were not analyzed for this leachate). In addition, this leachate did not cause any daphnia toxicity.

7. The second period of inhibition began while treating leachate collected on day 561. Unlike the leachate during the first period, this leachate possessed approximately only average strength (TKN concentration of 61.4 mg/l, COD concentration of 92.9 mg/l) with the exception of the iron concentration which was high at 28.3 mg/l. However, as occurred during the first inhibition period, the leachates

collected on day 561 and on day 581 and used during the inhibition period were relatively toxic to daphnia.

COD UTILIZED DURING DENITRIFICATION

As was discussed earlier, 167 mg/l of methanol was added to the Experimental System and, after day 227, to the Control System. The results from the Experimental System operation at 15 day MCRT (days 59 to 73) can be used to estimate the amount of COD utilized per mg of nitrate denitrified.

At a leachate flowrate of 6 liters per day, and with a methanol feed concentration of 167 mg/l, a total of 1 gram of methanol was fed to the anoxic tank each day. This would provide a COD of 1500 mg each day to the anoxic tank. This COD loading rate was essentially the optimum. Table B-8 in the Appendix presents the Experimental System influent raw leachate COD concentrations and the Experimental System effluent COD concentrations during the 15 day MCRT operation. The average influent leachate (without the methanol) COD during this period was 74.5 mg/l and the system effluent concentration was 71.8 mg/l. The COD concentration in the effluent of the anoxic reactor was also monitored during this period of the study and was found to average 74.2 mg/l.

Therefore, since the anoxic zone COD concentration was almost equal to the effluent COD, essentially all the methanol added was utilized in the anoxic zone during denitrification. In addition, the concentrations of nitrates and nitrites in the anoxic reactor were near zero on every day during the 15 day MCRT period of operation with the exception of day 63 during which the nitrate-N concentration was 3.20 mg/l (Table B-5 in the Appendix). This indicated that virtually all the nitrate loaded into the anoxic reactor was denitrified and that denitrification was not limited by a lack of nutrients or by a lack of COD.

A COD balance around the anoxic reactor determined that 1460 mg/day of COD was utilized in the reactor. A nitrogen balance determined that 253 mg/day of nitrate-N was denitrified. This results in 5.77 mg of COD utilized for every mg of nitrate-N denitrified.

CARBON ADSORPTION ISOTHERM

Table 16 in Chapter 4 shows that little COD removal occurred in either biological treatment system. The BOD₅ concentration of the effluent solutions was occasionally monitored and was generally well under 10 mg/l (see Table B-9

in the Appendix). TOC concentrations of the effluents were also monitored occasionally (Table B-10 in the Appendix) and were generally found to be approximately 2 mg/l less than the influent leachate TOC concentrations. The fact that the effluent COD concentrations are almost equal to the influent leachate COD concentrations at all operating conditions and that there was little BOD in the influent leachate and the effluents, indicates that the organics in the leachate were not biodegraded in the systems and were the major contributors to the effluent COD concentrations.

As discussed in Chapter 2, the literature indicates that the organics in leachates from old landfills are normally nonbiodegradable. The results in Chapter 4 clearly support the conclusions of the previous studies. The literature also indicates that activated carbon has been shown to be effective in removing organics from old landfill leachates. The results presented in Chapter 4 also support these conclusions as both granular activated carbon (GAC) and powdered activated carbon (PAC) were found to be effective in reducing the COD in the effluent of the Control System.

The results presented in Table 17 for COD removal by GAC fit the Freundlich adsorption isotherm. Plotting the amount of COD removed per mass of carbon (X/M) versus the final equilibrium COD concentration in mg/l results in a straight

line on a log-log plot as is shown in Figure 12.

Using the slope and intercept, the following equation can be developed for the GAC:

$$X/M = 0.00739C^{0.60}$$

Where: X = COD adsorbed in grams

M = Mass of carbon in grams

C = Equilibrium COD concentration in
mg/l

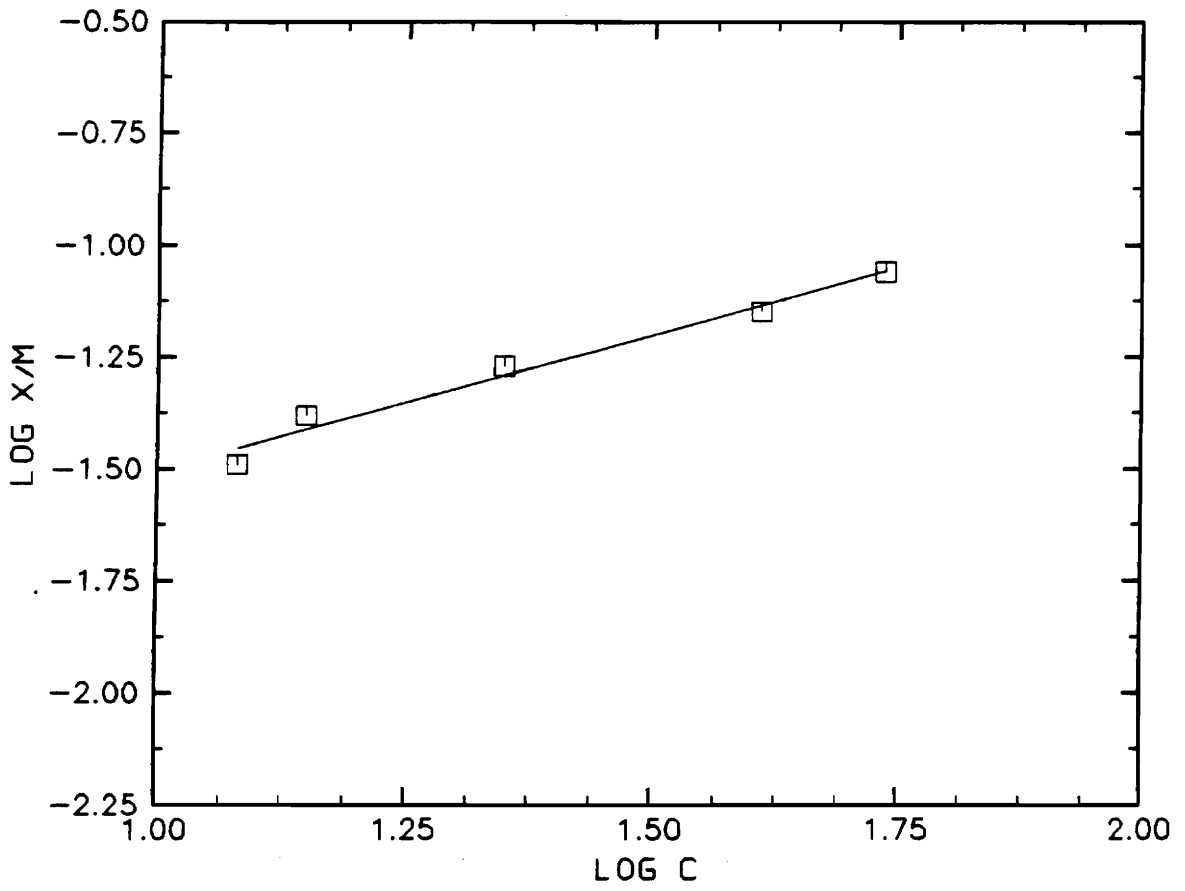


Figure 12. Carbon Adsorption Isotherm

Chapter VI. CONCLUSIONS

Based on the results obtained during this study, the following conclusions were reached:

1. The Modified Ludzack Ettinger (MLE) process provides a very effective method for removing nitrogen from an old leachate such as the leachate from the Dixie Caverns Land-fill. As much as 84% of the influent nitrogen was removed by the MLE process compared with a maximum of only 30% removed by a completely aerobic activated sludge system.

2. The amount of nitrogen removed by the MLE process was a function of the operating mean cell residence time (MCRT) and of the mixed liquor recycle flowrate. Higher MCRT's and higher recycle rates improved nitrogen removal.

3. Transient toxicity could be a significant problem for biological treatment of old leachates. Toxicity was

observed during two separate periods late in the study. This toxicity severely inhibited nitrification and increased the death rate of daphnia neonates.

4. The addition of an anoxic zone in front of an aerobic reactor to promote denitrification caused no detrimental effects to nitrification in the aerobic reactor. The 4.5 liter MLE system (with a 3 liter aerobic reactor) nitrified as well as the three liter aerobic control system at low aerobic MCRT's.

5. An MLE system with methanol addition for denitrification will produce equal or lower effluent ammonia concentrations than a completely aerobic nitrification system treating the same wastewater when operated at low MCRT's. This is due to heterotrophic biomass growth reducing the amount of nitrogen available for nitrification. The average effluent ammonia-N concentrations from the MLE system at three and two day aerobic MCRT's were 1.10 mg/l and 20.6 mg/l respectively. The three liter completely aerobic nitrification system produced effluent ammonia-N concentrations of 7.94 mg/l and 28.3 mg/l when operated at the same aerobic MCRT's.

6. The MLE system nitrified more total nitrogen per day per liter of aerated volume than the two stage completely aerobic control system when the wastewater and all operating conditions were the same during these experiments. This phenomenon needs further investigation.

7. Acclimation to methanol is needed in a nitrification/denitrification activated sludge system. Nitrification in the completely aerobic system, which had been operating for six months without methanol addition, was adversely effected when the addition of methanol was begun.

8. The addition of methanol to a nitrifying activated sludge system causes an increase in the effluent organic nitrogen concentration. During this study, this increase was two to three mg/l.

9. Operation at low MCRT's was detrimental to denitrification, indicating partial wash-out of denitrifiers. Denitrification was virtually complete at 15 and 8 day aerobic MCRT's with average anoxic reactor nitrate concentrations of 0.46 and 0.34 mg/l, respectively. Incomplete denitrification occurred at MCRT's of three days and below as evidenced by anoxic nitrate and nitrite concentrations each consistent-

ly well above 1.0 mg/l. Denitrification was substantially degraded when the MCRT was reduced to below two days.

10. It was determined that 5.77 mg of methanol as COD was used for every mg of nitrate-N denitrified during the 15 day aerobic MCRT period of operation.

11. The organics in the leachate generally resisted biodegradation. This was expected for a leachate from an "old" landfill with little biodegradable organics left. Neither the MLE system nor the completely aerobic Control system without methanol feed were effective in reducing the COD of the leachate, removing only 4.8% and 2.8% of the influent COD respectively.

12. Granular activated carbon was found to be an effective method for reducing the COD of the completely aerobic system effluent. During a 40 hour jar test the aerobic system effluent COD was reduced from 76 mg/l down to a low of 12 mg/l with a carbon concentration of 2 grams/liter. The results from the jar test could be described by the Freundlich Isotherm.

13. Powdered activated carbon added directly to the

activated sludge reactors of the Control system was found to be effective in reducing the effluent COD. The minimum COD attained was 19 mg/l at a carbon dose of 3 grams. A daily dose of 1 gram reduced the effluent COD to about 40 mg/l.

14. Both systems successfully oxidized and removed iron, consistently reducing the soluble concentrations to below 1 mg/l.

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APPENDIX A
SAMPLE CALCULATIONS FOR
NITRIFICATION AND DENITRIFICATION RATES
AND T-TEST

A1 Determination of Nitrification Rate

A1-1 Average Results From Days 59 to 73 (15 day aerobic MCRT); Experimental System Operated as an MLE Process

1. Influent:
 TKN = 55.9 mg/l
 Flowrate (Q) = 6 l/day
 Influent $\text{NO}_x\text{-N}$ = 2.40 mg/l
2. Effluent:
 TKN = 2.52 mg/l
 Ammonia-N = 0.01 mg/l
 Effluent VSS = 2.62 mg/l
 $\text{NO}_x\text{-N}$ = 9.67 mg/l
3. Reactors:
 Mixed Liquor VSS = 1194 mg/l
 VSS Wasted Daily = 224 mg/day
 Volume Wasted = 186 ml/day

A1-2 Determination of Nitrification Rate

1. Total Influent TKN = $55.9 \text{ mg/l} \times 6 \text{ l/day} = 335.4 \text{ mg/day}$
2. Total Effluent TKN = $2.52 \text{ mg/l} \times 6 \text{ l/day} = 15.1 \text{ mg/day}$
3. Total Nitrogen Removed with Solids =
 $\{2.6 \text{ mg/l} \times (6 - .186 \text{ l/day}) + 224 \text{ mg/day}\} \times 0.1 = 24 \text{ mg/day}$
4. Total N-Nitrified = $335.4 - 15.1 - 24 = 296 \text{ mg/day}$
5. Total N-Nitrified per unit of aerobic volume =
 $296 \text{ mg/day} \times 1/3 \text{ liters} = 98.7 \text{ mg/l-day}$
6. Nitrification Rate =
 $296 \text{ mg/day} \times 1 \text{ liter}/1194 \text{ mg} \times 1/3 \text{ liters} = 0.083 \text{ l/day}$

A1-3 Determination of Percent Nitrified

1. Total Nitrogen Available for Nitrification =
 Total Influent TKN - (Total Effluent TKN - Total

Effluent Ammonia-N) - Nitrogen Removed with Solids)

$$= 335.4 - \{15.1 - (0.01) \cdot 6\} - 24 = 296 \text{ mg/day}$$

2. Total N-Nitrified = 296 mg/day

3. Percent Nitrified =

$$\frac{\text{Total N-Nitrified} \cdot 100\%}{\text{Total Nitrogen Available for Nitrification}} \\ = 296 \cdot 100\% / 296 = 100\%$$

A2 Determination of Denitrification Rate

A2-1 Determination of Denitrification Rate

1. Total N-Denitrified = N-Nitrified + $\text{NO}_x\text{-N}_{\text{in}} \cdot Q - \text{NO}_x\text{-N}_{\text{out}} \cdot Q$

$$= 296 + (2.40 - 9.67) \cdot 6 = 253 \text{ mg/day}$$

2. Total N-Denitrified per unit of Anoxic Volume =

$$253 \text{ mg/day} \cdot 1 / 1.5 \text{ liters} = 169 \text{ mg/-day}$$

3. Denitrification Rate =

$$253 \text{ mg/day} \cdot 1 \text{ liter} / 1194 \text{ mg} \cdot 1 / 1.5 \text{ liters} = 0.141 \text{ 1/day}$$

A2-2 Determination of Percent Denitrified

1. Total Nitrogen Available for Denitrification = N-Nitrified + $\text{NO}_x\text{-N}_{\text{in}} \cdot Q$

2. % Denitrified = $253 \cdot 100\% / (296 + 2.4 \cdot 6) = 81.5\%$

A3 Statistical T-Test Example

A3-1 Statistics of Effluent Ammonia-N Concentrations From Days 99 to 116 (3 day aerobic MCRT)

1. Experimental System:
Average effluent ammonia-N = 1.10 mg/l
Standard deviation = 1.81; variance = 3.27
2. Control System:
Average effluent ammonia-N = 7.94 mg/l
Standard deviation = 6.31; variance = 39.9
3. Number of observations = 9

A3-2 Null Hypothesis = The average effluent ammonia-N concentrations from the two systems are equal

A3-3 Alternative Hypothesis = The ammonia-N concentrations from the two systems are not equal

A3-4 Calculation of Test Statistic

1. $t\text{-obs} = (y_1 - y_2) / S_p \sqrt{(1/n + 1/n)}$
2. $S_p = (3.27 + 39.9) / 2 = 4.65$
3. $t\text{-obs} = (7.94 - 1.10) / 4.65 \sqrt{(1/9 + 1/9)}$
 $= 3.12$

A3-5 Determination of t-crit

1. Significance level = .05
2. Two sided test
3. Degrees of freedom = $n + n - 2 = 16$
4. $t\text{-crit} = 2.12$ (Table 4 in An Introduction to Statistical Methods (62))

A3-6 Conclusion

1. $t\text{-obs} > t\text{-crit}$; reject null hypothesis
2. The Control System's effluent ammonia-N is higher than

the effluent ammonia-N from the Experimental system

APPENDIX B
TREATMENT OF DIXIE CAVERNS LANDFILL LEACHATE
OPERATING RESULTS

TABLE B - 1. COMPARISON OF TKN REMOVALS

Date	Day #	MCRT	TKN CONCENTRATION, MG/L		
			Influent Leachate	Effluent	
				Exp	Cont
5/28/88	59	15	55.9	2.03	0.00
5/29/88	60	15	55.9	2.35	0.00
5/30/88	61	15	55.9	2.14	0.00
6/1/88	63	15	55.9	2.56	0.00
6/2/88	64	15	55.9	2.78	0.64
6/4/88	66	15	55.9	2.99	0.00
6/5/88	67	15	55.9	2.56	0.00
6/6/88	68	15	55.9	2.46	0.32
6/7/88	69	15	55.9	2.35	0.00
6/11/88	73	15	56.0	2.99	0.00
6/27/88	89	8	55.3	2.99	0.85
6/28/88	90	8	55.3	2.46	0.00
7/7/88	99	3	55.3	2.99	0.00
7/8/88	100	3	55.3	5.98	0.32
7/9/88	101	3	55.3	9.40	2.99
7/10/88	102	3	55.3	4.06	5.02
7/13/88	105	3	58.4	1.28	7.48
7/14/88	106	3	58.4	4.06	10.8
7/20/88	112	3	58.4	3.63	25.6
7/21/88	113	3	58.4	3.84	13.4
7/24/88	116	3	58.4	7.05	13.8
7/29/88	121	2	58.4	4.06	1.71
7/30/88	122	2	58.4	3.63	1.82
7/31/88	123	2	58.4	5.55	4.49
8/1/88	124	2	58.4	4.49	2.45
9/2/88	156	2	60.0	13.9	42.3
9/4/88	158	2	60.0	24.6	39.9
9/11/88	165	2	60.0	36.1	30.0
9/23/88	177	2	62.4	26.9	22.2
9/30/88	184	2*	62.4	2.79	0.00
10/7/88	192	2*	62.4	3.91	1.34
10/18/88	203	1*	62.4	7.69	2.14
10/20/88	205	1*	62.4	3.12	0.00
11/4/88	220	1*	60.2	5.13	2.23
11/18/88	234	3	60.2	5.13	32.2
11/20/88	236	3	60.2	8.12	28.2
11/21/88	237	3	60.2	6.41	28.8
11/22/88	238	3	60.2	6.62	24.6
12/2/88	248	3	54.5	4.27	3.85

*Chemostat Operation

1/14/89	291	3	65.7	11.1	2.56
1/15/89	292	3	65.7	10.0	1.92
1/17/89	294	3	65.7	8.83	2.29
1/22/89	299	3	65.7	3.71	11.6
1/29/89	306	3	64.4	4.14	2.18
1/30/89	307	3	64.4	4.36	2.51
2/1/89	309	3	64.4	3.92	2.40
2/4/89	312	3	64.4	7.63	3.29
2/5/89	313	3	64.4	7.19	3.71
2/9/89	317	3	64.4	5.02	2.62
2/12/89	320	3	64.4	2.94	0.87
2/25/89	333	2.5	70.6	3.27	2.18
2/28/89	336	2.5	70.6	5.01	4.58
3/3/89	339	2.5	65.7	4.58	2.83
3/6/89	342	2.5	65.7	8.18	8.94
3/9/89	345	2.5	65.7	15.8	24.5
3/12/89	348	2.5	65.7	3.92	2.62
3/13/89	349	2.5	65.7	3.49	2.40
3/19/89	355	2	65.7	5.67	2.83
3/20/89	356	2	65.7	5.23	2.18
3/31/89	367	2	62.5	5.45	2.73
4/5/89	372	2	62.5	9.60	3.49
4/14/89	381	2	66.3	9.59	2.40
4/27/89	394	2	67.8	8.18	0.33
5/9/89	406	1.5	67.8	23.8	13.0
5/10/89	407	1.5	67.8	11.1	7.41
5/11/89	408	1.5	67.8	11.9	9.70
5/15/89	412	1.5	53.4	3.82	1.85
5/17/89	414	1.5	53.4	8.13	4.92
5/18/89	415	1.5	53.4	4.28	1.07
5/19/89	416	1.5	53.4	7.60	3.10
5/25/89	422	1.5	63.7	16.1	2.14
5/26/89	423	1.5	63.7	16.5	6.42
5/30/89	427	1.5	63.7	14.8	3.10
5/31/89	428	1.5	63.7	11.0	1.42
6/13/89	441	4	64.9	4.10	2.50
6/21/89	449	4	64.9	3.21	19.5
6/25/89	453	4	64.9	13.6	20.5
7/16/89	474	8	57.9	32.3	5.14
7/19/89	477	8	57.9	36.5	1.39
7/25/89	483	8	57.9	41.3	2.57
8/5/89	494	8	59.0	43.2	2.43
8/13/89	502	8	59.0	2.98	3.40
8/20/89	509	8	59.0	9.72	1.40
9/7/89	527	8	59.1	8.38	2.87
9/9/89	529	8	59.1	14.3	2.20
9/14/89	534	8	59.1	6.50	2.92

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9/18/89	538	8	59.1	11.2	3.19
9/26/89	546	8	58.0	29.6	1.76
10/11/8	561	8	61.4		
10/31/8	581	8	61.7		

TABLE B - 2. COMPARISON OF NH-3 REMOVALS

Date	Day #	MCRT	NH-3-N CONCENTRATION, MG/L		
			Influent Leachate	Effluent	
				EXP	Cont
5/28/88	59	15	51.0	0.00	0.00
5/29/88	60	15	51.0	0.00	0.00
5/30/88	61	15	51.0	0.00	0.00
6/1/88	63	15	51.0	0.11	0.00
6/2/88	64	15	51.0	0.00	0.00
6/4/88	66	15	51.0	0.00	0.00
6/5/88	67	15	51.0	0.00	0.00
6/6/88	68	15	51.0	0.00	0.00
6/7/88	69	15	51.0	0.00	0.00
6/11/88	73	15	47.0	0.00	0.00
6/27/88	89	8	50.2	0.00	0.00
6/28/88	90	8	50.2	0.00	0.00
7/7/88	99	3	50.2	0.43	0.69
7/8/88	100	3	50.2	2.99	2.46
7/9/88	101	3	50.2	5.23	3.74
7/10/88	102	3	50.2	0.11	4.22
7/13/88	105	3	52.1	0.00	7.38
7/14/88	106	3	52.1	0.32	9.83
7/20/88	112	3	52.1	0.75	21.5
7/21/88	113	3	52.1	0.00	11.4
7/24/88	116	3	52.1	0.11	10.2
7/29/88	121	2	52.1	0.11	0.11
7/30/88	122	2	52.1	0.43	1.17
7/31/88	123	2	52.1	1.92	2.72
8/1/88	124	2	52.1	0.91	1.66
9/2/88	156	2	55.9	9.29	34.8
9/4/88	158	2	55.9	18.1	34.9
9/11/88	165	2	55.9	33.5	25.7
9/23/88	177	2	56.7	21.4	17.6
9/30/88	184	2*	56.7	0.00	0.00
10/7/88	192	2*	56.7	0.00	0.00
10/18/88	203	1*	56.7	4.69	0.89
10/20/88	205	1*	56.7	0.93	0.33
11/4/88	220	1*	54.2	0.91	0.80
11/18/88	234	3	54.2	1.92	27.7
11/20/88	236	3	54.2	4.59	23.9
11/21/88	237	3	54.2	3.26	23.8
11/22/88	238	3	54.2	4.54	21.9
12/2/88	248	3	50.5	0.64	0.00

* Chemostat Operation

1/14/89	291	3	61.0	8.38	0.85
1/15/89	292	3	61.0	6.41	0.21
1/17/89	294	3	61.0	5.18	0.49
1/22/89	299	3	61.0	0.22	8.83
1/29/89	306	3	59.6	0.00	0.00
1/30/89	307	3	59.6	1.14	0.81
2/1/89	309	3	59.6	0.00	0.00
2/4/89	312	3	59.6	3.71	0.33
2/5/89	313	3	59.6	2.89	1.04
2/9/89	317	3	59.6	0.00	0.00
2/12/89	320	3	59.6	0.00	0.00
2/25/89	333	2.5	64.6	0.00	0.00
2/28/89	336	2.5	64.6	0.76	2.07
3/3/89	339	2.5	61.4	1.53	0.27
3/6/89	342	2.5	61.4	2.78	4.63
3/9/89	345	2.5	61.4	12.2	20.9
3/12/89	348	2.5	61.4	0.00	0.00
3/13/89	349	2.5	61.4	0.00	0.55
3/19/89	355	2	61.4	2.18	1.36
3/20/89	356	2	61.4	1.20	1.20
3/31/89	367	2	59.3	3.82	0.55
4/5/89	372	2	59.3	6.10	0.00
4/14/89	381	2	62.5	7.09	0.55
4/27/89	394	2	64.3	3.71	0.00
5/9/89	406	1.5	64.3	14.7	11.4
5/10/89	407	1.5	64.3	7.36	4.90
5/11/89	408	1.5	64.3	8.94	7.96
5/15/89	412	1.5	49.1	0.00	0.00
5/17/89	414	1.5	49.1	4.60	3.20
5/18/89	415	1.5	49.1	1.74	1.23
5/19/89	416	1.5	49.1	4.23	1.09
5/25/89	422	1.5	58.3	13.0	1.28
5/26/89	423	1.5	58.3	13.6	5.20
5/30/89	427	1.5	58.3	11.0	3.30
5/31/89	428	1.5	58.3	7.83	0.87
6/13/89	441	4	59.8	0.00	0.00
6/21/89	449	4	59.8	0.00	15.3
6/22/89	450	4	59.8	----	21.1
6/23/89	451	4	59.8	----	19.8
6/24/89	452	4	59.8	8.56	16.5
6/25/89	453	4	59.8	10.9	16.9
6/29/89	457	4	60.3	7.49	15.0
7/1/89	459	4	60.3	9.63	16.2
7/2/89	460	4	60.3	8.35	19.8
7/3/89	461	8	60.3	0.00	16.6
7/4/89	462	8	60.3	2.78	10.8
7/5/89	463	8	60.3	1.63	8.02

7/6/89	464	8	60.3	0.00	1.74
7/7/89	465	8	60.3	0.00	0.00
7/10/89	468	8	60.3	5.27	3.40
7/11/89	469	8	60.3	14.6	11.3
7/12/89	470	8	60.3	19.1	13.3
7/13/89	471	8	60.3	27.8	17.4
7/16/89	474	8	50.5	24.9	0.97
7/17/89	475	8	50.5	28.1	0.00
7/18/89	476	8	50.5	30.8	0.00
7/19/89	477	8	50.5	29.8	0.00
7/25/89	483	8	50.5	35.5	0.00
7/28/89	486	8	50.5	34.1	0.00
7/29/89	487	8	50.5	33.7	1.04
8/1/89	490	8	53.4	28.7	0.00
8/3/89	492	8	53.4	33.7	0.00
8/5/89	494	8	53.4	38.1	0.00
8/13/89	502	8	53.4	0.00	0.00
8/14/89	503	8	53.4	0.00	0.00
8/20/89	509	8	57.5	0.00	0.00
8/24/89	513	8	57.5	0.00	0.00
8/27/89	516	8	57.5	0.00	0.00
8/30/89	519	8	57.5	7.98	0.00
8/31/89	520	8	57.5	12.3	0.00
9/7/89	527	8	56.6	5.13	0.00
9/9/89	529	8	56.6	7.83	0.00
9/10/89	530	8	56.6	1.70	0.00
9/11/89	531	8	56.6	1.10	0.00
9/14/89	534	8	56.6	3.04	0.00
9/18/89	538	8	56.6	6.24	0.00
9/26/89	546	8	54.1	23.3	0.00
10/2/89	552	8	54.1	23.0	0.00
10/3/89	553	8	54.1	24.6	0.00
10/9/89	559	8	54.1	9.94	0.00
10/19/89	569	8	57.4	22.0	0.00
10/23/89	573	8	57.4	0.00	0.00
10/25/89	575	8	57.4	1.66	1.20
10/26/89	576	8	57.4	4.58	8.11
10/27/89	577	8	57.4	9.49	15.1
10/29/89	579	8	57.4	18.2	13.8
11/03/89	584	8	57.4	32.6	32.6

TABLE B - 3. INFLUENT LEACHATE NITRATE CONCENTRATION

Date	Influent Leachate Nitrate Concentration, Mg/l
5/25/88	2.40
6/9/88	0.24
6/24/88	0.41
7/11/88	0.34
8/26/88	0.20
9/16/88	0.11
10/28/88	0.57
11/22/88	0.70
12/9/88	0.59
1/4/89	0.24
1/27/89	0.70
2/14/89	0.13
3/2/89	0.88
3/21/89	0.22
4/6/89	0.30
4/25/89	0.26
5/12/89	0.00
5/23/89	0.20
6/8/89	0.00
6/26/89	0.00
7/12/89	0.00
7/31/89	0.62
8/19/89	0.38
9/5/89	2.58
9/21/89	0.68
10/11/89	0.07

TABLE B - 4. COMPARISON OF EFFLUENT NO₃-N(NO₂-N) CONCENTRATIONS

Date	Day #	MCRT	NO ₃ -N(NO ₂ -N) CONCENTRATIONS* MG/L	
			Exp	Control
5/28/88	59	15	10.6	38.6
5/29/88	60	15	9.97	56.0
5/30/88	61	15	10.7	46.4
5/31/88	62	15	10.8	48.6
6/1/88	63	15	9.29	56.0
6/2/88	64	15	9.87	55.4
6/4/88	66	15	8.15	53.9
6/5/88	67	15	10.1	50.3
6/6/88	68	15	9.20	52.2
6/7/88	69	15	9.62	56.0
6/11/88	73	15	8.07	49.0
6/27/88	89	8	9.43	49.9
6/28/88	90	8	6.96	36.2
7/7/88	99	3	9.14	50.2 (0.44)
7/8/88	100	3	9.33	38.4 (2.04)
7/9/88	101	3	7.60	44.6 (1.57)
7/10/88	102	3	11.0	40.7 (1.52)
7/13/88	105	3	11.6	32.6 (1.61)
7/14/88	106	3	13.4	32.5 (2.22)
7/20/88	112	3	12.5	10.7 (8.99)
7/21/88	113	3	23.4	12.6 (23.4)
7/24/88	116	3	14.0	15.2 (23.8)
7/29/88	121	2	18.1 (4.50)	28.8 (10.6)
7/30/88	122	2	14.3 (3.70)	31.5 (20.0)
7/31/88	123	2	10.6 (3.50)	26.3 (17.9)
8/1/88	124	2	12.6 (3.22)	27.5 (22.7)
9/2/88	156	2	6.42	4.89 (6.77)
9/4/88	158	2	3.14	3.69 (8.03)
9/11/88	165	2	0.49	9.83 (9.57)
9/23/88	177	2	1.30 (6.87)	37.9 (6.42)
9/30/88	184	2**	5.70 (11.1)	57.7
10/7/88	192	2**	2.90 (2.72)	43.7 (4.00)
10/18/88	203	1**	10.9 (8.76)	53.5 (1.41)
10/20/88	205	1**	13.0 (11.7)	56.2 (0.47)
11/4/88	220	1**	17.3 (0.71)	51.8 (0.30)

* Nitrite concentrations (NO₂-N) are in parenthesis

** Chemostat operation

11/18/88	234	3	17.7	5.66(3.64)
11/20/88	236	3	14.7(4.90)	11.2(5.43)
11/21/88	237	3	11.0(5.04)	9.16(9.50)
11/22/88	238	3	9.70(5.02)	13.2(10.7)
12/2/89	248	3	10.1(3.04)	39.1(3.74)
1/14/89	291	3	9.06	46.7(7.56)
1/15/89	292	3	7.29(4.30)	50.2(6.57)
1/17/89	294	3	10.7	39.7(4.61)
1/22/89	299	3	16.8(2.62)	18.4(1.40)
1/29/89	306	3	10.6(3.52)	27.7(1.77)
1/30/89	307	3	8.81(4.01)	46.8(1.08)
2/1/89	309	3	7.38(2.73)	43.7(2.62)
2/4/89	312	3	5.81(3.99)	39.7(0.73)
2/5/89	313	3	7.53(4.98)	47.6(3.07)
2/9/89	317	3	8.37(1.14)	39.4(0.65)
2/12/89	320	3	25.1(2.76)	52.3(0.60)
2/25/89	333	2.5	10.1(2.37)	46.9(0.27)
2/28/89	336	2.5	15.8(4.20)	36.7(1.62)
3/3/89	339	2.5	10.7(2.98)	41.5(0.80)
3/6/89	342	2.5	12.5(3.65)	39.9(2.11)
3/9/89	345	2.5	9.18(2.37)	18.9(0.16)
3/12/89	348	2.5	12.0(2.70)	36.5(3.95)
3/13/89	349	2.5	13.0(1.86)	33.6(1.02)
3/19/89	355	2.0	14.1(2.88)	35.6(1.41)
3/20/89	356	2.0	14.2(2.57)	48.9(1.03)
3/31/89	367	2.0	11.3(2.41)	44.9(6.61)
4/5/89	372	2.0	10.4(5.21)	45.0(2.57)
4/14/89	381	2.0	5.02(1.86)	47.6(2.62)
4/27/89	394	2.0	8.81(2.72)	55.5(1.29)
5/9/89	406	1.5	13.7(3.29)	43.3(5.11)
5/10/89	407	1.5	15.3(4.00)	39.6(4.69)
5/11/89	408	1.5	14.9(4.40)	42.6(4.35)
5/15/89	412	1.5	18.3(4.00)	43.3(1.75)
5/17/89	414	1.5	13.0(2.08)	36.5(1.63)
5/18/89	415	1.5	11.6(2.38)	37.8(2.07)
5/19/89	416	1.5	12.4(2.59)	39.0(1.27)
5/25/89	422	1.5	6.90(2.44)	40.6(2.58)
5/26/89	423	1.5	7.04(2.65)	36.6(3.04)
5/30/89	427	1.5	9.34(3.33)	40.7(2.47)
5/31/89	428	1.5	11.1(3.70)	42.2(1.84)
6/13/89	441	4	14.2	14.7
6/21/89	449	4	12.6(0.69)	7.30
6/24/89	452	4	9.00(1.00)	8.00(1.00)
6/25/89	453	4	8.70	6.58
7/3/89	461	8	16.2(1.74)	9.39(1.40)
7/10/89	468	8	18.6(1.18)	9.04(1.06)
7/11/89	469	8	24.4(2.86)	8.07(0.93)

7/12/89	470	8	19.7 (5.91)	7.59 (0.88)
7/13/89	471	8	12.6 (8.85)	6.73
7/16/89	474	8	4.20 (9.40)	13.1 (1.90)
7/17/89	475	8	1.76 (5.87)	12.9 (1.34)
7/18/89	476	8	0.90 (3.54)	12.6 (1.26)
7/19/89	477	8	0.44 (1.95)	12.7 (1.24)
7/25/89	483	8	0.64	11.7
8/5/89	494	8	0.50 (1.23)	10.4 (1.78)
8/13/89	502	8	3.06 (7.90)	16.2
8/14/89	503	8	2.72 (6.00)	8.80
8/15/89	504	8	2.38 (6.85)	11.4 (0.89)
8/19/89	508	8	1.39 (5.12)	6.30
8/20/89	509	8	0.50 (5.00)	7.00
8/21/89	510	8	2.25 (5.00)	
8/22/89	511	8	4.00 (4.00)	
8/23/89	512	8	7.00 (1.50)	8.00
8/24/89	513	8	8.00	9.00
8/25/89	514	8	7.19	8.27
8/27/89	516	8	6.74	7.92
8/30/89	519	8	3.38 (1.73)	8.20
8/31/89	520	8	6.26 (2.10)	8.05
9/7/89	527	8	2.81 (4.32)	6.55
9/9/89	529	8	4.50 (7.40)	9.67 (1.04)
9/10/89	530	8	4.20 (6.07)	5.60
9/11/89	531	8	3.69 (7.84)	6.60
9/14/89	534	8	2.90 (13.1)	8.28
9/18/89	538	8	1.95 (18.9)	5.76
9/26/89	546	8	0.48 (7.50)	6.60
10/2/89	552	8	0.88 (6.43)	6.08
10/3/89	553	8	0.63 (5.94)	5.71
10/9/89	559	8	9.81 (2.94)	4.57
10/12/89	562	8	8.10 (2.78)	6.25
10/23/89	573	8	5.92 (2.85)	9.36 (0.18)
10/25/89	575	8	4.12 (1.50)	7.07

TABLE B - 5. COMPARISON OF ANOXIC ZONE
NO3-N(NO2-N) CONCENTRATIONS

Date	Day #	MCRT	ANOXIC ZONE NO3-N(NO2-N) * CONCENTRATION, MG/L	
			MLE	Control
5/28/88	59	15	0.07	
5/29/88	60	15	0.08	
5/30/88	61	15	0.09	
5/31/88	62	15	0.59	
6/1/88	63	15	3.20	
6/2/88	64	15	0.03	
6/4/88	66	15	0.02	
6/5/88	67	15	0.80	
6/6/88	68	15	0.03	
6/7/88	69	15	0.15	
6/11/88	73	15	0.02	
6/27/88	89	8	0.34	
7/7/88	99	3	0.34	
7/8/88	100	3	1.08	
7/9/88	101	3	0.72	
7/10/88	102	3	0.58	
7/13/88	105	3	1.58	
7/14/88	106	3	3.30	
7/20/88	112	3	1.63	
7/21/88	113	3	6.73	
7/24/88	116	3	4.17	
7/29/88	121	2	10.2	
7/30/88	122	2	5.58 (1.00)	
9/2/88	156	2	0.11	
9/4/88	158	2	0.49	
9/11/88	165	2	0.00	
9/23/88	177	2	0.12 (1.67)	
9/30/88	184	2**	1.85 (7.12)	
10/7/88	192	2**	0.39	
10/18/88	203	1**	6.70 (6.18)	
10/20/88	205	1**	4.95 (9.64)	
11/4/88	220	1**	8.65 (1.63)	
11/18/88	234	3	10.1 (2.06)	
11/20/88	236	3	3.57 (1.80)	
11/21/88	237	3	2.16 (1.43)	

* Nitrite concentrations (NO2-N) in parentheses

** Chemostat operation

11/22/88	238	3	1.36 (1.29)	
12/2/88	248	3	0.82	
1/14/89	291	3	0.93	
1/15/89	292	3	1.08 (1.25)	
1/17/89	294	3	0.68 (1.20)	
1/29/89	306	3	0.20	
1/30/89	307	3	0.38	
2/1/89	309	3	0.68 (1.14)	35.0 (1.67)
2/4/89	312	3	0.63 (1.47)	29.9 (1.59)
2/5/89	313	3	0.64 (1.31)	29.1 (1.55)
2/9/89	317	3	1.85 (1.58)	34.8 (1.78)
2/12/89	320	3	9.12 (2.32)	37.5 (2.02)
2/25/89	333	2.5	0.28	37.7 (1.61)
2/28/89	336	2.5	2.48 (1.14)	33.4 (2.93)
3/3/89	339	2.5	0.75 (1.33)	30.7 (2.33)
3/6/89	342	2.5	2.90 (2.11)	33.1 (3.00)
3/9/89	345	2.5	0.50 (0.88)	18.0 (0.62)
3/12/89	348	2.5	1.66 (1.49)	28.4 (2.97)
3/13/89	349	2.5	2.52 (0.79)	30.3 (2.18)
3/19/89	355	2.0	3.17 (1.52)	32.2 (2.40)
3/20/89	356	2.0	3.07 (1.43)	36.7 (3.48)
3/31/89	367	2.0	0.96 (1.00)	29.6 (6.27)
4/5/89	372	2.0	0.90 (1.61)	37.3 (4.32)
4/14/89	381	2.0	0.76 (0.95)	37.1 (3.95)
4/27/89	394	2.0	1.10 (1.23)	44.4 (2.99)
5/9/89	406	1.5	3.43	30.4 (3.17)
5/10/89	407	1.5	4.32	31.0 (3.69)
5/11/89	408	1.5	4.00 (2.62)	30.0 (3.11)
5/15/89	412	1.5	3.93 (3.80)	31.0 (2.21)
5/17/89	414	1.5	1.55 (1.37)	26.4 (1.47)
5/18/89	415	1.5	1.02 (0.53)	26.8 (2.01)
5/19/89	416	1.5	2.11 (1.65)	23.0 (1.85)
5/25/89	422	1.5	0.53 (1.15)	24.9 (1.70)
5/26/89	423	1.5	0.71 (1.22)	20.8 (2.03)
5/30/89	427	1.5	0.81 (0.97)	33.0 (2.09)
5/31/89	428	1.5		29.5 (1.70)
6/13/89	441	4	1.10	5.06
6/15/89	443	4	0.78	5.08 (1.07)
6/21/89	449	4	0.95	0.75 (0.86)
6/25/89	453	4	0.00	0.00
7/11/89	469	8		
7/12/89	470	8	13.0 (4.09)	0.40
7/13/89	471	8		
7/16/89	474	8	2.35 (6.09)	0.24
7/19/89	477	8	0.00 (1.27)	0.00
8/5/89	494	8	0.00	0.00
8/15/89	504	8	0.88	1.75

8/13/89	502	8	0.00 (1.55)	3.86
8/14/89	503	8	0.00	0.00
8/19/89	508	8	0.00	0.00
8/24/89	513	8	0.00	0.00
8/27/89	516	8	0.00	0.00
8/30/89	519	8	0.00	0.00
8/31/89	520	8	0.00	0.00
9/7/89	527	8	0.00	0.00
9/9/89	529	8	1.14 (3.85)	0.00
9/10/89	530	8	0.82 (2.84)	0.00
9/11/89	531	8	0.00 (5.19)	0.00
9/14/89	534	8	0.00 (9.96)	0.00
9/18/89	538	8	0.00 (12.9)	0.00
9/26/89	546	8	0.00 (6.54)	0.00
10/2/89	552	8	0.00 (5.21)	0.00
10/3/89	553	8	0.00 (4.75)	0.00
10/9/89	559	8	6.55 (1.00)	0.00
10/12/89	562	8	2.11	0.00
10/23/89	573	8		0.71
10/25/89	575	8	0.18	0.53

TABLE B - 6. COMPARISON OF MLSS AND MLVSS CONCENTRATIONS

Date	Day #	MLSS, MG/L		MLVSS, MG/L	
		System		System	
		Exp.	Control	Exp.	Control
5/28/88	59	1330	1110	1037	810
5/29/88	60	1480	1140	1193	820
5/30/88	61	1547	1080	1217	790
5/31/88	62	1620	890	1314	680
6/1/88	63	1443	1230	1093	860
6/2/88	64	1453	755	1170	540
6/4/88	66	1510	660	1245	490
6/5/88	67	1523	730	1262	505
6/6/88	68	1488	655	1222	455
6/7/88	69	1395	665	1138	455
6/11/88	73	1495	525	1248	395
6/22/88	84	1232	320	1060	225
6/23/88	85	1115	360	953	245
6/25/88	87	1158	365	993	225
6/27/88	89	1112	290	940	200
6/28/88	90	1061	290	917	195
7/7/88	99	312	63	270	55
7/8/88	100	353	93	293	58
7/9/88	101	399	142	327	80
7/10/88	102	356	85	337	65
7/13/88	105	374	115	305	70
7/14/88	106	348	75	297	65
7/20/88	112	364	93	310	58
7/21/88	113	350	85	300	55
7/24/88	116	300	78	256	60
7/29/88	121	231	74	192	50
7/30/88	122	224	76	193	56
7/31/88	123	176	66	150	56
8/1/88	124	180	74	150	52
9/2/88	156	378	63	330	29
9/4/88	158	351	67	310	30
9/9/88	163	333	103	281	40
9/11/88	165	275	90	227	34
9/21/88	175	223	44	169	26
9/23/88	177	211	24	172	17
9/30/88	184	233	68	171	37
10/7/88	192	195	33	158	24
10/18/88	203	127	28	104	19
10/20/88	205	81	33	59	21
11/4/88	220	115	33	103	24

11/18/88	234	230	260	195	220
11/20/88	236	255	327	202	280
11/21/88	237	225	293	201	250
11/22/88	238	165	213	157	200
12/2/88	248	332	296	285	257
1/11/89	288	247	287	210	267
1/12/89	289	228	293	198	273
1/14/89	291	265	300	237	267
1/15/89	292	263	290	235	277
1/17/89	294	330	280	305	258
1/22/89	299	218	240	207	235
1/29/89	306	458	277	422	253
1/30/89	307	397	278	365	255
1/31/89	308	432	290	400	275
2/1/89	309	297	263	267	245
2/3/89	311	332	303	302	278
2/4/89	312	257	297	228	282
2/5/89	313	310	300	285	273
2/8/89	316	240	265	208	228
2/9/89	317	228	247	218	238
2/10/89	318	233	255	208	242
2/12/89	320	257	323	235	283
2/13/89	321	268	300	227	263
2/23/89	331	259	250	236	243
2/24/89	332	273	267	241	237
2/28/89	336	227	219	206	194
3/1/89	337	238	225	198	195
3/3/89	339	286	278	268	267
3/5/89	341	227	283	216	265
3/6/89	342	175	216	154	198
3/8/89	344	254	209	241	192
3/9/89	345	296	269	261	232
3/11/89	347	310	279	290	275
3/12/89	348	304	339	268	304
3/13/89	349	245	307	214	283
3/18/89	354	194	258	168	222
3/19/89	355	210	202	200	187
3/20/89	356	179	206	159	178
3/29/89	365	252	247	227	220
3/30/89	366	308	272	281	251
3/31/89	367	246	243	223	210
4/4/89	371	279	318	258	284
4/5/89	372	304	231	262	206
4/13/89	380	222	226	183	185
4/14/89	381	171	204	161	187
4/18/89	385	358	268	325	239
4/21/89	388	247	195	222	158

4/26/89	393	205	242	177	213
4/27/89	394	205	172	178	149
5/8/89	405	115	134	104	123
5/9/89	406	145	148	132	133
5/10/89	407	114	162	106	144
5/11/89	408	111	165	108	148
5/15/89	412	152	197	115	157
5/17/89	414	183	167	138	144
5/18/89	415	146	223	127	170
5/19/89	416	163	222	134	186
5/25/89	422	204	223	165	189
5/26/89	423	242	248	176	188
5/29/89	426	178	254	129	199
5/30/89	427	198	221	152	172
5/31/89	428	190	194	146	164
6/10/89	438	650	753	503	600
6/11/89	439	577	670	477	567
6/12/89	440	623	663	517	553
6/13/89	441	513	643	467	553
6/15/89	443	523	414	427	327
6/19/89	447	368	417	313	295
6/20/89	448	518	448	415	323
6/21/89	449	503	488	415	380
6/25/89	453	565	538	468	440
7/2/89	460	802	533	640	430
7/3/89	461	723	492	622	453
7/4/89	462	840	613	658	500
7/5/89	463	783	593	600	452
7/7/89	465	768	608	623	517
7/9/89	467	610	652	552	557
7/10/89	468	758	653	631	538
7/12/89	470	646	670	428	488
7/15/89	473	632	622	455	487
7/16/89	474	623	597	475	510
7/19/89	477	620	575	413	462
7/25/89	483	550	1200	320	982
7/26/89	484	440	1057	193	830
7/28/89	486	594	993	413	793
8/1/89	490	490	823	347	733
8/5/89	494	617	976	320	737
8/13/89	502	837	857	600	653
8/20/89	509	977	820	867	777
8/24/89	513	1030	846	920	723
8/25/89	514	963	853	813	750
8/27/89	516	956	750	760	640
8/29/89	518	1097	903	930	830
8/31/89	520	1080	890	900	830

9/6/89	526	1143	1017	950	847
9/8/89	528	1163	1007	883	853
9/10/89	530	1213	1013	963	823
9/11/89	531	1197	920	1000	840
9/14/89	534	1150	1007	923	927
9/18/89	538	1453	957	1093	773
9/26/89	546	1053	1010	807	900
10/2/89	552	1047	960	733	803
10/3/89	553	1020	1100	660	967
10/9/89	559	540	1010	420	870
10/12/89	562	580	763	487	670
10/13/89	563	393	813	340	680
10/25/89	575	736	1413	667	1260
10/26/89	576	785	1577	713	1443
10/30/89	580	747	1770	637	1640

TABLE B - 7. EFFLUENT TSS AND VSS CONCENTRATIONS

Date	Day #	EFFLUENT TSS MG/L		EFFLUENT VSS MG/L	
		Exp.	Control	Exp.	Control
5/28/88	59	0.98	3.64	0.25	1.82
5/29/88	60	2.80	7.60	1.40	3.80
5/30/88	61	2.80	8.40	2.80	8.40
5/31/88	62	2.26	7.84	2.66	6.37
6/1/88	63	2.00	6.80	2.0	4.40
6/2/88	64	3.20	8.00	3.20	4.00
6/4/88	66	2.40	11.1	2.40	8.33
6/5/88	67	7.98	14.3	5.46	10.6
6/6/88	68	4.20	11.7	4.20	9.05
6/7/88	69	4.03	10.3	0.81	6.44
6/11/88	73	5.20	10.6	3.60	4.38
6/22/88	84	2.60	2.00	2.60	2.00
6/23/88	85	3.20	3.20	2.40	2.40
6/25/88	87	5.20	4.00	5.20	4.00
6/27/88	89	6.60	6.80	4.20	4.80
6/28/88	90	4.30	8.10	4.30	6.60
7/7/88	99	2.20	3.80	2.20	3.80
7/8/88	100	2.80	4.00	2.80	2.40
7/9/88	101	2.50	6.60	2.50	5.30
7/10/88	102	2.90	4.70	2.90	4.70
7/13/88	105	7.50	12.6	4.40	6.01
7/14/88	106	9.60	18.5	9.60	15.3
7/20/88	112	2.80	8.80	2.80	4.80
7/21/88	113	4.30	2.90	3.60	2.20
7/24/88	116	4.80	3.60	4.80	3.20
7/29/88	121	5.90	5.70	5.50	4.10
7/30/88	122	2.40	3.20	2.40	3.20
7/31/88	123	7.40	8.60	6.70	5.90
8/1/88	124	3.80	5.60	3.80	4.00
9/2/88	156	4.30	2.80	4.30	1.80
9/4/88	158	5.40	4.50	5.40	2.60
9/9/88	163	8.20	2.30	7.46	1.41
9/11/88	165	5.94	5.10	4.40	2.90
9/21/88	175	4.10	3.42	2.90	1.30
9/23/88	177	21.1	2.40	16.8	2.00
11/18/88	234	3.40	19.1	3.40	16.4
11/20/88	236	12.4	22.0	10.4	20.0
11/21/88	237	10.8	24.0	9.60	19.0
11/22/88	238	11.2	34.4	10.4	23.4
12/2/88	248	10.3	22.0	10.3	22.0

1/11/89	288	6.52	20.0	5.40	17.5
1/12/89	289	3.50	11.3	3.50	10.0
1/14/89	291	6.50	17.0	6.50	16.0
1/15/89	292	25.7	16.5	24.3	15.5
1/17/89	294	27.3	4.30	26.4	4.30
1/22/89	299	9.00	3.50	9.00	3.50
1/29/89	306	11.5	6.00	10.0	5.00
1/30/89	307	13.5	4.00	12.0	4.00
1/31/89	308	4.50	5.00	4.50	5.00
2/1/89	309	13.0	7.50	12.5	7.50
2/3/89	311	5.00	9.00	4.50	5.50
2/4/89	312	2.80	2.94	2.80	2.90
2/5/89	313	12.8	4.50	11.0	4.50
2/8/89	316	17.5	5.00	15.8	5.00
2/9/89	317	38.0	6.00	34.1	6.00
2/10/89	318	23.5	6.67	21.0	6.67
2/12/89	320	3.40	10.6	3.40	10.6
2/13/89	321	4.33	9.33	3.00	8.00
2/23/89	331	13.2	8.20	12.6	8.20
2/24/89	332	12.1	6.10	11.3	6.10
2/28/89	336	7.44	7.30	7.40	7.30
3/1/89	337	18.0	6.50	16.0	5.00
3/3/89	339	5.80	3.72	5.80	3.20
3/5/89	341	13.8	5.31	13.8	5.31
3/6/89	342	22.0	6.00	20.5	5.70
3/8/89	344	48.2	5.30	46.0	5.30
3/9/89	345	81.1	29.8	67.8	25.8
3/11/89	347	54.0	74.0	53.0	74.0
3/12/89	348	31.0	6.20	26.0	5.90
3/13/89	349	30.0	9.70	28.7	9.70
3/18/89	354	44.7	6.00	40.7	6.00
3/19/89	355	48.0	6.40	45.6	6.40
3/20/89	356	36.1	3.64	30.6	3.60
3/29/89	365	11.5	6.00	11.5	6.00
3/30/89	366	11.3	3.30	11.3	3.30
3/31/89	367	5.70	5.70	5.70	5.70
4/4/89	371	6.20	4.80	6.20	4.40
4/5/89	372	17.4	3.70	14.2	3.20
4/13/89	380	11.0	4.00	9.00	2.00
4/14/89	381	58.1	9.00	50.0	9.00
4/18/89	385	7.80	3.70	7.80	3.70
4/21/89	388	7.00	6.90	5.70	6.90
4/26/89	393	22.9	3.50	21.4	3.50
4/27/89	394	46.0	8.10	40.0	6.60
5/8/89	405	26.8	5.50	25.6	5.50
5/9/89	406	33.0	8.00	29.7	8.00
5/10/89	407	27.9	8.00	27.4	8.00

5/11/89	408	24.5	6.40	24.5	6.40
5/15/89	412	35.3	31.6	25.3	25.8
5/17/89	414	32.0	4.80	24.0	4.80
5/18/89	415	20.0	3.90	13.9	3.90
5/19/89	416	24.6	3.50	20.4	3.50
5/25/89	422	37.5	2.40	33.3	2.40
5/26/89	423	35.7	4.60	28.6	4.20
5/29/89	426	28.8	2.00	24.4	1.60
5/30/89	427	27.4	1.90	24.0	1.90
5/31/89	428	30.6	2.70	25.0	2.70
6/10/89	438	6.84	2.40	6.84	2.40
6/11/89	439	8.81	4.20	7.50	4.28
6/12/89	440	10.8	2.40	9.00	2.40
6/13/89	441	6.80	----	6.80	1.60
6/15/89	443	11.7	2.00	10.8	2.00
6/19/89	447	4.00	2.40	4.00	2.40
6/20/89	448	5.90	4.10	5.40	4.10
6/21/89	449	5.30	4.30	4.40	4.30
6/25/89	453	6.52	10.4	4.80	8.30
7/2/89	460	8.80	8.40	6.00	5.20
7/3/89	461	15.0	10.2	15.0	10.2
7/4/89	462	7.20	8.40	5.60	6.80
7/5/89	463	7.89	9.10	4.70	8.10
7/7/89	465	7.80	4.30	7.80	4.30
7/9/89	467	13.3	19.1	12.8	19.1
7/10/89	468	32.9	9.01	27.7	8.60
7/12/89	470	64.6	7.14	39.4	2.10
7/15/89	473	100	12.4	70.0	8.40
7/16/89	474	88.0	8.60	74.0	8.60
7/19/89	477	82.5	24.3	82.5	22.4
7/25/89	483	55.0	153	32.5	120
7/26/89	484	28.5	96.3	18.0	86.2
7/28/89	486	41.2	14.9	35.9	13.5
8/1/89	490	40.7	43.7	36.0	42.1
8/5/89	494	55.0	46.7	40.0	33.3
8/13/89	502	52.5	19.2	43.3	13.9
8/20/89	509	92.8	14.8	85.7	16.1
8/24/89	513	60.9	14.5	57.0	14.5
8/25/89	514	57.5	9.50	48.8	9.00
8/27/89	516	70.0	6.20	58.0	5.10
8/29/89	518	67.0	12.8	61.0	12.8
8/31/89	520	69.0	18.4	62.0	14.6
9/6/89	526	117	33.6	95.7	28.0
9/8/89	528	131	34.4	89.8	23.4
9/10/89	530	121	30.1	83.0	22.8
9/11/89	531	80.9	75.0	35.2	30.4
9/14/89	534	81.4	37.9	74.3	36.1

9/18/89	538	42.0	17.3	29.3	12.3
9/26/89	546	53.8	13.6	39.7	12.7
10/2/89	552	170	23.5	123	17.5
10/3/89	553	137	19.0	100	17.0
10/9/89	559	140	56.6	106	43.3
10/12/89	562	208	111	168	98.6
10/13/89	563	53.3	9.03	42.2	8.30
10/25/89	575	41.2	15.7	36.8	11.0
10/26/89	576	34.5	15.8	25.7	12.5
10/30/89	580	40.4	22.9	38.6	21.6

TABLE B - 8. COMPARISON OF COD REMOVALS

Date	Day #	MCRT	COD CONCENTRATION (MG/L)		
			Influent Leachate	Effluent	
				Exp.	Cont
5/28/88	59	15	74.5	63.1	68.2
5/29/88	60	15	74.5	64.7	64.7
5/30/88	61	15	74.5	65.8	66.9
5/31/88	62	15	74.5	75.3	79.0
6/1/88	63	15	74.5	70.3	67.0
6/2/88	64	15	74.5	67.4	72.0
6/4/88	66	15	74.5	77.2	54.8
6/5/88	67	15	74.5	64.6	60.0
6/6/88	68	15	74.5	74.6	86.5
6/7/88	69	15	74.5	95.0	83.5
6/27/88	89	8	85.4	78.4	74.7
6/28/88	90	8	85.4	97.8	95.2
7/7/88	99	3	85.4	75.7	75.5
7/8/88	100	3	85.4	81.6	72.8
7/9/88	101	3	85.4	83.7	79.8
7/10/88	102	3	85.4	85.4	78.5
7/13/88	105	3	88.7	80.2	80.6
7/14/88	106	3	88.7	78.6	86.3
7/20/88	112	3	88.7	82.1	91.5
7/21/88	113	3	88.7	77.0	98.7
7/24/88	116	3	88.7	81.4	116.1
7/29/88	121	2	88.7	84.4	99.1
7/30/88	122	2	88.7	84.9	100.9
7/31/88	123	2	88.7	83.0	100.7
8/1/88	123	2	88.7	81.8	107.4
9/2/88	156	2	82.6	81.5	81.8
9/4/88	158	2	82.6	96.3	97.3
9/11/88	165	2	82.6	98.5	106.2
9/23/88	177	2	93.9	88.7	77.6
9/30/88	184	2*	93.9	85.7	77.8
10/7/88	192	2*	93.9	94.4	80.1
10/18/88	203	1*	93.9	86.0	88.4
10/20/88	205	1*	93.9	84.2	87.5
11/4/88	220	1*	91.6	77.4	88.0
11/18/88	234	3	91.6	92.7	116.3
11/20/88	236	3	91.6	77.2	97.3
11/21/88	237	3	91.6	87.6	99.7
11/22/88	238	3	91.6	88.1	95.4

* Chemostat operation

12/2/88	248	3	92.2	88.2	92.2
1/14/89	291	3	91.4	87.4	95.4
1/15/89	292	3	91.4	87.0	88.6
1/17/89	294	3	91.4	97.1	101.2
1/22/89	299	3	91.4	100.0	97.5
1/29/89	306	3	91.4	103.8	105.0
1/30/89	307	3	91.4	97.3	98.9
2/1/89	309	3	91.4	98.2	98.2
2/4/89	312	3	91.4	114.8	114.8
2/5/89	313	3	91.4	111.5	114.1
2/9/89	317	3	100.2	99.4	94.5
2/12/89	320	3	100.2	109.8	99.3
2/25/89	333	2.5	100.2	109.0	97.3
2/28/89	336	2.5	100.2	122.9	117.9
3/3/89	339	2.5	95.0	107.1	105.5
3/6/89	342	2.5	95.0	101.4	101.4
3/9/89	345	2.5	95.0	109.1	106.7
3/12/89	348	2.5	95.0	103.4	86.5
3/13/89	349	2.5	95.0	104.0	89.3
3/19/89	355	2.0	95.0	108.5	100.4
3/20/89	356	2.0	95.0	100.9	95.2
3/31/89	367	2.0	71.4	109.0	78.6
4/5/89	372	2.0	71.4	81.0	79.3
4/14/89	381	2.0	79.5	88.6	93.5
4/27/89	394	2.0	92.4	107.2	107.2
5/9/89	406	1.5	92.4	99.8	93.2
5/10/89	407	1.5	92.4	90.7	98.1
5/11/89	408	1.5	92.4	89.9	94.5
5/15/89	412	1.5	81.6	97.9	80.0
5/17/89	414	1.5	81.6	77.3	74.0
5/18/89	415	1.5	81.6	79.7	74.8
5/19/89	416	1.5	81.6	75.6	71.5
5/25/89	422	1.5	81.6	154.0	120.0
5/26/89	423	1.5	81.6	121.2	110.6
5/30/89	427	1.5	100.2	94.9	81.9
5/31/89	428	1.5	100.2	89.0	93.8
6/21/89	449	4	106.8	82.2	80.5
6/25/89	453	4	106.8	90.7	94.2
7/3/89	461	8	99.0	76.1	81.8
7/12/89	470	8	99.0	86.4	67.1
7/16/89	474	8	86.4	87.2	56.8
7/17/89	475	8	86.4	101.9	75.8
7/19/89	477	8	86.4	106.4	79.2
7/25/89	483	8	86.4	430.0	69.8
7/29/89	487	8	86.4	124.4	80.8
8/1/89	490	8	99.4	135.0	85.0
8/5/89	494	8	99.4	130.0	67.4

8/13/89	502	8	99.4	92.8	72.8
8/30/89	519	8	84.8	91.7	76.0
9/7/89	527	8	94.0	87.3	68.7
9/9/89	529	8	94.0	102.1	76.2
9/14/89	534	8	94.0	112.7	74.1
9/18/89	538	8	94.0	102.0	70.1
9/26/89	546	8	98.4	122.0	65.6
10/22/89	572	8	82.4	----	91.0
10/23/89	573	8	82.4	----	19.0
10/25/89	575	8	82.4	----	41.7
10/26/89	576	8	82.4	----	34.3
10/27/89	578	8	82.4	----	50.5
10/29/89	580	8	82.4	----	29.6
11/01/89	583	8	82.4	95.3	47.7
11/03/89	585	8	92.9	83.6	40.6

TABLE B - 9. COMPARISON OF BOD-5 CONCENTRATIONS

Date	Day #	BOD-5 CONCENTRATION, MG/L		
		Influent Leachate	Effluent Exp. Control	
4/24/88	25	9.41	----	----
5/9/88	40	6.17	----	----
5/19/88	50	6.17	8.60	1.90
5/28/88	59	3.92	0.72	0.54
5/31/88	62	3.92	0.84	1.38
6/6/88	68	3.92	3.72	2.16
6/9/88	71	6.54	----	----
6/24/88	86	4.88	----	----
7/24/88	116	7.72	2.22	17.0
8/26/88	163	5.47	----	----
10/18/88	203	2.00	2.64	4.08
10/28/88	213	1.26	----	----
11/22/88	238	3.81	----	----
12/9/88	255	0.86	----	----
1/5/89	282	6.46	----	----
6/26/89	454	4.22	----	----

TABLE B - 10. COMPARISON OF TOC CONCENTRATIONS

Date	Day #	TOC CONCENTRATION, MG/L		
		Influent Leachate	Effluent Exp. Control	
4/23/88	24	29.3	----	----
5/9/88	40	27.6	----	----
5/28/88	59	26.5	24.6	23.6
5/29/88	60	26.5	24.1	24.0
5/30/88	61	26.5	24.7	23.5
5/31/88	62	26.5	23.5	24.4
6/1/88	63	26.5	23.9	24.5
6/2/88	64	26.5	23.7	24.0
6/4/88	66	26.5	23.8	22.9
6/5/88	67	26.5	23.7	24.2
6/6/88	68	26.5	18.1	23.4
6/7/88	69	26.5	26.2	21.0
6/11/88	73	27.6	24.4	24.3

TABLE B - 11. INFLUENT AND EFFLUENT METALS CONCENTRATIONS

Date	Iron*			Manganese		
	Influent	Exp.	Control	Influent	Exp.	Control
5/24/88	8.99	0.85	0.24	1.20	0.22	0.06
5/29/88	8.99	0.97	0.58	1.20	1.74	0.24
5/31/88	8.99	0.26	0.18	1.20	0.59	0.13
7/9/88	----	0.07	0.09	----	0.28	0.42
9/11/88	9.40	0.07	0.27	0.63	0.31	0.33
6/22/89	22.5	----	0.66	0.84	----	0.56
6/25/89	22.5	----	0.47	0.84	----	0.62
8/29/89	----	----	0.03	----	----	0.34
9/18/89	14.5	0.49	----	0.64	0.42	----
9/26/89	12.5	0.97	----	0.66	0.39	----

* All concentrations in mg/l

TABLE B - 12. EFFLUENT SOLUBLE ORTHO-PHOSPHATE

Date	EFFLUENT PO ₄ -P CONCENTRATIONS (MG/L)	
	Exp.	Control
5/26/88	6.60	3.46
5/31/88	7.25	3.62
6/6/88	6.25	5.44
7/9/88	7.81	6.34
7/21/88	6.17	8.44
9/4/88	5.50	4.44
11/4/88	3.83	5.14
11/21/88	3.09	5.02
1/17/89	5.22	4.19
2/12/89	3.16	5.68
3/3/89	2.82	----
3/13/89	5.45	3.34
5/15/89	2.53	2.43
7/10/89	6.78	5.17
7/11/89	7.14	7.70
7/16/89	6.32	7.38
7/25/89	9.03	11.6
9/26/89	9.13	8.74

TABLE B - 13. COMPARISON OF pH VALUES

Date	Influent Leachate	Experimental		Control	
		Anoxic	Eff.	Anoxic	Eff.
5/28/88	6.65	7.98	8.20		8.00
5/29/88	6.65	8.00	8.14		7.95
5/30/88	6.65	8.05	8.12		7.90
5/31/88	6.65	8.00	8.08		7.91
6/1/88	6.65	7.94	8.26		7.99
6/2/88	6.65	7.94	8.22		8.04
6/4/88	6.65	7.89	8.27		8.10
6/5/88	6.65	7.77	8.21		8.05
6/6/88	6.65	8.12	8.23		8.03
6/7/88	6.65	7.85	8.27		8.06
6/11/88	6.74	7.92	8.25		8.08
6/27/88	7.02	7.97	8.16		8.02
6/28/88	7.02	7.97	8.27		8.14
7/7/88	7.02	7.93	8.22		7.97
7/8/88	7.02	7.92	8.27		8.02
7/9/88	7.02	8.00	8.34		8.12
7/10/88	7.02	8.04	8.32		8.12
7/13/88	6.83	7.97	8.29		8.07
7/14/88	6.83	7.95	8.27		8.10
7/20/88	6.83	7.85	8.25		8.21
7/21/88	6.83	8.02	8.24		8.16
7/24/88	6.83	7.88	8.25		8.08
7/29/88	6.83	8.10	8.28		8.07
7/30/88	6.83	7.98	8.23		8.02
7/31/88	6.83	7.97	8.29		7.98
8/1/88	6.83	8.13	8.23		8.01
9/2/88	6.58	7.88	8.15		8.31
9/4/88	6.58	7.78	8.19		8.35
9/11/88	6.58	8.07	8.29		8.25
9/23/88	6.77	8.09	8.19		8.11
9/30/88	6.77	8.10	8.35		7.95
10/7/88	6.77	7.91	8.18		8.03
10/18/88	6.77	7.98	8.21		7.84
10/20/88	6.77	7.74	8.13		7.85
11/4/88	6.75	7.82	8.08		7.80
11/18/88	6.75	7.88	7.95		8.06
11/20/88	6.75	7.99	8.08		8.14
11/21/88	6.75	7.84	8.02		8.09
11/22/88	6.75	7.98	8.07		8.14
12/2/88	6.82	7.83	7.94	7.80	7.74
1/14/89	6.87	7.97	8.01	7.82	7.81
1/15/89	6.87	7.84	7.99	7.79	7.84

1/17/89	6.87	7.88	7.88	7.74	7.71
1/22/89	6.87	7.71	7.86	7.66	7.89
1/29/89	6.87	7.91	8.07	7.94	7.94
1/30/89	6.87	7.81	7.96	7.90	7.82
2/1/89	6.87	7.95	8.11	7.92	7.97
2/4/89	6.87	7.84	8.04	7.90	7.90
2/5/89	6.87	7.79	7.92	7.87	7.82
2/9/89	6.87	7.85	7.98	7.80	7.83
2/12/89	6.87	7.71	7.87	7.73	7.75
2/25/89	6.90	7.89	8.04	7.82	7.91
2/28/89	6.90	7.90	7.96	7.80	7.80
3/3/89	6.95	7.67	7.90	7.59	7.74
3/6/89	6.95	7.93	8.09	7.93	7.93
3/9/89	6.95	7.78	8.02	8.02	8.12
3/12/89	6.95	8.42	8.31	8.27	8.24
3/13/89	6.95	7.91	8.21	7.93	7.91
3/19/89	6.95	7.88	7.91	7.87	7.90
3/20/89	6.95	----	8.07	7.87	7.87
3/31/89	6.65	7.85	7.97	7.92	7.94
4/5/89	6.65	7.89	8.04	7.95	7.95
4/14/89	6.48	7.80	7.97	7.92	7.97
4/27/89	6.62	8.03	8.07	8.04	7.94
5/9/89	6.62	7.51	6.91	7.96	7.90
5/10/89	6.62	7.54	8.07	7.93	7.95
5/11/89	6.62	7.50	8.02	7.95	7.91
5/15/89	6.50	7.44	7.89	7.78	7.88
5/17/89	6.50	7.61	8.06	7.93	8.00
5/18/89	6.50	7.31	8.01	7.81	7.94
5/19/89	6.50	7.48	8.08	7.60	8.03
5/25/89	----	7.79	8.00	7.90	7.79
5/26/89	----	7.41	8.00	7.88	7.92
5/30/89	----	7.48	7.97	7.83	7.94
5/31/89	----	7.88	8.10	----	7.99
6/2/89	----	----	8.35	----	8.36
6/13/89	6.50	----	8.34	----	8.30
6/21/89	6.50	8.07	8.27	8.10	8.32
7/10/89	6.59	----	7.93	----	8.07
7/12/89	6.59	7.59	7.98	7.65	7.88
7/16/89	6.57	7.49	7.95	7.60	7.95
7/25/89	6.57	8.22	8.44	8.04	8.23
8/5/89	6.62	8.06	8.40	8.11	8.49
8/13/89	6.62	8.05	8.33	7.96	8.36
8/30/89	6.58	8.09	8.13	8.05	8.32
9/7/89	6.90	8.00	8.25	7.98	8.28
9/9/89	6.90	8.20	8.43	8.21	8.45
10/14/89	6.71	8.06	8.34	8.21	8.40
10/26/89	6.71	8.18	8.36	8.21	8.49

Vita

Donald Marickovich was born on August 20, 1953, in Sewickley, Pennsylvania. In 1976 he was graduated from Drexel University in Philadelphia, Pennsylvania, with a Bachelor of Science in Physics. From 1976 until 1980, he was an officer in the United States Navy and assigned as an instructor at the Naval Nuclear Power School in Orlando, Florida. From 1980 until 1987 he worked as a nuclear engineer at the Naval Sea Systems Command, Nuclear Propulsion Directorate in Arlington, Virginia. Starting in 1986 he began pursuing a Master's degree in Environmental Engineering on a part time basis at a Virginia Polytechnic Institute and State University graduate center in Northern Virginia. In the summer of 1987, he moved to Blacksburg, Virginia to pursue his degree full time.

Donald Marickovich