

PROCESS DEVELOPMENT STUDIES ON THE BIOLOGICAL UTILIZATION OF NITROGEN
IN A DOMESTIC WASTEWATER TREATMENT SYSTEM

A Thesis
Presented to
the Faculty of the Department of Civil Engineering
Cullen College of Engineering
The University of Houston

In Partial Fulfillment
of the Requirements for the Degree
Master of Science

by
B. W. Ryan
May, 1974

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ABSTRACT

This research document presents the results of a study of a full-scale advanced wastewater treatment facility which included a suspended growth nitrification reactor and attached growth denitrification reactors.

A literature review is made of nitrification and denitrification phenomena and of the design criteria developed for utilizing these phenomena in laboratory and pilot plant studies of nitrogen removal from wastewater.

Conclusions reached as a result of this study included the following:

1. The addition of the suspended growth nitrification reactor resulted in almost complete conversion of ammonia nitrogen to nitrates.
2. Attached growth denitrification reactors utilizing either granular or plastic media provide a feasible means of removing nitrate nitrogen from domestic wastewater.
3. Plastic media with higher porosity required less operator attention and gave better denitrification results.

Evaluation of the full-scale facility led to recommendations that methanol dose rate should be more carefully controlled and that additional studies be conducted to optimize denitrification, to delineate the role of the filtration phenomenon in removal of suspended solids by the granular media denitrification reactor, and to determine the influence of temperature and influent concentration gradients on each unit process.

TABLE OF CONTENTS

<u>Chapter</u>		<u>Page</u>
I	INTRODUCTION	1
II	LITERATURE REVIEW	5
	Development of Wastewater Treatment Technology	5
	Bacterial Utilization of Nitrogen Species in Natural Waters	10
	Biochemistry of Microbial Utilization of Nitrogen	14
	Biological Nitrogen Control Processes in Wastewater Treatment	19
III	EXPERIMENTAL PROGRAM	30
	Introduction	30
	Analytical Program	40
	Plant Operation	40
IV	PRESENTATION AND DISCUSSION OF RESULTS	46
	Nitrification Reactor Evaluation	46
	Granular Media Denitrification Reactor Evaluation	49
	Plastic Media Denitrification Reactor Evaluation	53
	Comparison of Granular Media and Plastic Media Denitrification Reactors	57
V	CONCLUSIONS	58
VI	RECOMMENDATIONS.....	60
	BIBLIOGRAPHY	61
	APPENDIX A - LABORATORY PROCEDURES	66
	APPENDIX B - TEST DATA	68

LIST OF TABLES

<u>Table</u>		<u>Page</u>
1	Calculated Loadings for the El Lago Facility	32
2	Typical Values for El Lago Raw Waste-Water for the Period May, 1970 through August 1973	33
3	Values for Primary and Final Effluent for the Period May 1970 through June 1972	35
4	Specifications for Nitrification Reactor	36
5	Design Specifications for Granular Media Denitrification Towers	38
6	Design Specifications for Plastic Media Denitrification Towers	39
7	Nitrification Reactor Performance	47
8	Conditions for Initial Evaluation of Granular Media Denitrification Towers	50
9	Initial Evaluation of Granular Media Denitrification Towers	51
10	Conditions for Initial Evaluation of Plastic Media Denitrification Towers	54
11	Initial Evaluation of Plastic Media Denitrification Towers	55

LIST OF FIGURES

<u>Figure</u>		<u>Page</u>
1	The Nitrogen Cycle	3
2	Nitrate and Nitrite Concentration in Reactor (1-hr Residence Time)	27
3	Nitrate and Nitrite Concentration in Reactor (2-hr Residence Time)	27
4	Nitrate and Nitrite Concentration in Reactor (4-hr Residence Time)	28
5	Cell Concentration in Reactor	28
6	Original El Lago Wastewater Treatment Facility	31
7	El Lago Advanced Waste Treatment Facility	41

CHAPTER I

INTRODUCTION

The American people have become more aware than ever before of the importance of an adequate supply of clean water. During the same time that per capita consumption of water for domestic, commercial and industrial purpose was rising rapidly, the nation's population was increasing dramatically and critical water shortages developed as ground water tables dropped and surface waters became more polluted.

The concern of the American People was made manifest in a far reaching way when the ninety-second Congress passed P.L. 92-500, known as the Federal Water Pollution Control Act Amendments of 1972. This act, which became law on October 18, 1972, establishes the most comprehensive program ever enacted for cleaning up America's water. In addition to establishing the goal of no discharges of pollution into the nation's waters by 1985, it established some more immediate requirements which are aimed at achieving that goal. These include the requirement that industries use the "best practicable" water technology by July 1, 1983. In addition, all publicly owned waste treatment plants must be using "best practicable" treatment technology by July 1, 1983.

Engineers and scientists in the water pollution control field are being challenged to develop new treatment processes and to make nationwide use of the best existing processes, some of which have been used only in the laboratory or in the field under unusual circumstances.

The treatment of wastewater to meet the goals of P.L. 92-500 and to permit more extensive water reuse will include more effective removal of dissolved organics and suspended solids and the control of inorganic compounds that might degrade the receiving waters or make the effluent unsuitable for immediate reuse. The latter category includes the nitrogen species of ammonia, nitrite and nitrate.

These species of nitrogen are common in nature and are essential to life processes. Their role in the nitrogen cycle is shown in Figure 1. Some of the undesirable effects of selected nitrogen species are as follows:

- (1) Ammonia in effluents can exert a nitrogenous oxygen demand (NOD) and cause a depletion in dissolved oxygen in receiving waters (1,2).
- (2) Ammonia is corrosive to copper plumbing (2).
- (3) Ammonia requires large quantities of chlorine for break-point chlorination and increases the required contact time for disinfection (2,3).
- (4) Ammonia can be toxic to fish (2).
- (5) Nitrates in drinking water can be a health hazard to humans (causes methemoglobinemia in infants) and may cause losses in livestock due to goiter and bloating (2,4,5).
- (6) Nitrogenous compounds are nutrients for aquatic plants and may cause eutrophication of lakes, streams and estuaries (2,6,7).

The undesirable effects of ammonia may be eliminated by the

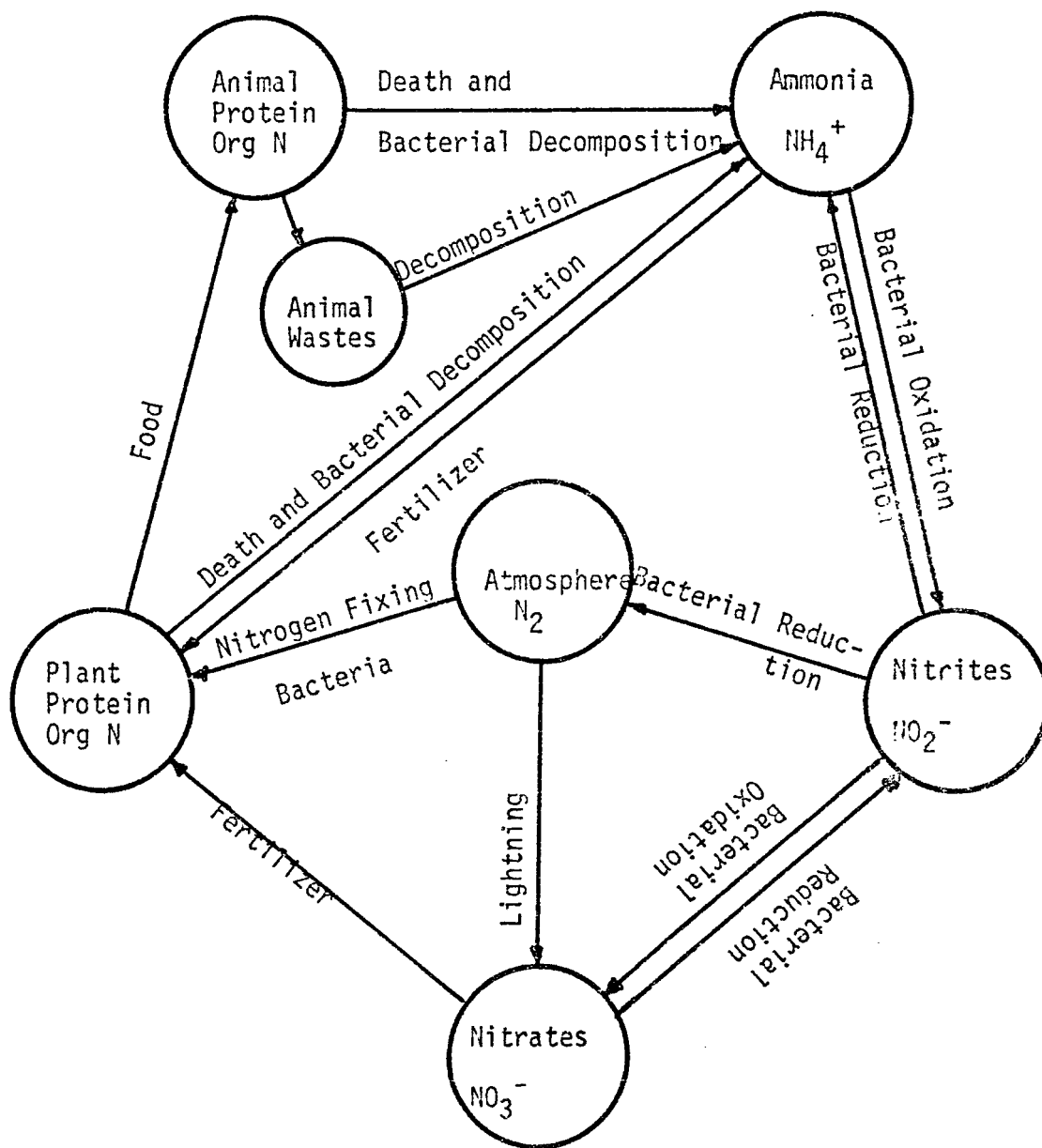


FIGURE 1 THE NITROGEN CYCLE

oxidation of this reduced form of nitrogen to nitrate by biological nitrification before it is released from the treatment plant. The last two harmful effects of nitrogen are significant only if the treated wastewater is to be reused for drinking by humans or livestock or if it is to be released to receiving water where nitrogen is the limiting nutrient for undesirable aquatic plants. Only then would actual removal of nitrogen from wastewater be justified. Removal of nitrogen from wastewater is currently required in only a limited number of places in the United States. These are the San Joaquin basin drainage into San Francisco Bay, Tampa Bay in Florida, the municipal Blue Plains plant in Washington, D.C., the Chicahoming River and Occoquan Creek watersheds in Virginia, and all Class AA,A and B waters in Hawaii (2).

The purposes of this research program were to review the development of wastewater treatment technology with special emphasis on nitrogen control, to summarize the current knowledge in the field of microbial utilization of nitrogen, and to investigate and report on a full-scale application of a biological nitrification-denitrification system. The investigation was primarily concerned with reunification and establishment of design criteria, development of process controls and operations procedures, and the evaluation of two types of fixed growth denitrification reactors.

CHAPTER II

LITERATURE REVIEW

For centuries man has had to contend with the waste products resulting from his preparation of plants and animals for food and other uses and from natural body functions. Man, the nomad, could simply move away when the accumulation of waste became too noxious; but when he began to till the soil and gather together in villages and towns, more ingenious methods of ridding himself of his waste had to be developed since he was no longer free to move away and leave it. A part of the literature deriving from man's efforts to solve this problem is reviewed and reported in this chapter in five sections. The first section reviews the development of wastewater treatment technology and the second reviews bacterial utilization of nitrogen species in natural waters. Then the biochemistry of microbial utilization of nitrogen is presented in the third, followed by a survey of biological nitrogen control processes in the fourth section. The last section summarizes the literature review and defines the current state of the art in nitrogen control technology.

Development of Wastewater Treatment Technology

Down through the Middle Ages sewage was disposed of even in the most advanced countries of Western Europe simply by dumping it into public streets and alleys. Even though elaborate storm-drainage systems existed in many of the larger cities, their use for the disposal of any wastes other than kitchen slops was prohibited by law until about 1815 in London and 1880 in Paris. As a result of these restrictions great amounts of decomposing

organic matter accumulated in the streets and alleys and finally led to the practice of dumping human wastes into the storm drains. During the 1840's the water-carriage system of waste removal was introduced and true sanitary sewers came into existence. These transferred the responsibility for handling of wastes from the individual to the community. Initially, the unsanitary conditions were moved from the centers of cities to the outskirts. This resulted in the concentration of these wastes into a few main streams which could be treated as the need for it became apparent (8,9).

In this country and in Britain, the first commonly used form of biological treatment of wastewater was the sewage farm. This practice involved pouring the sewage from urban drainage systems onto fields for irrigation and fertilization of farm crops. As the demand increased for land in proximity to urban areas, it became uneconomical to devote large areas to sewage farming and other methods of treatment were developed. The natural sand deposits of New England led to the practice of intermittent sand filtration which was made more efficient by pretreatment of wastewaters in settling tanks and later in biological units as well. Since sands in England were too tight for intermittent sand filtration, contact beds were developed. These first consisted of shallow tanks with layers of slate supported on bricks. Later tanks were simply filled with slag or stone. Large populations of microbes colonized the beds of slate, slag or stone and received their sustenance from the non-settling and dissolved solids of the influent wastewater. The beds were operated on a cycle of fill, draw and rest until it was discovered that

performance could be improved by continuously spraying the wastewater onto the media. The tanks were no longer kept filled with wastewater, but were fitted with underdrains which permitted circulation of air through the beds. This aeration along with the oxygen dissolved into the wastewater as it was sprayed onto the contact surfaces kept the system aerobic throughout its depth. In England these were first referred to as bacteria beds and later as percolating filters. In the United States this technology became known as trickling filters (3). The first large scale application of percolating or trickling filters was in Lancashire, England in 1893 (8). The next significant development in wastewater treatment was the activated sludge unit in which wastewater and flocs of microorganisms were kept aerobic and in suspension by air blown into the bottom of the tank (3). This occurred just prior to World War I and was based on studies on oxidation of sewage conducted by Avdern and Lockett (9).

Later, the idea of the submerged solid media of the contact beds combined with aeration gave rise to the submerged-contact aerators on aerated contact beds (3). Asbestos cement or plastic sheets were suspended vertically in a tank and were aerated by air forced up between them. The microbial colonies attached to the sheets were kept aerobic in this manner. This is sometimes referred to as contact aeration and is well adapted to use in small installations.

In rural areas and in smaller communities, septic tanks utilizing subsurface irrigation to handle the effluent came into common use after being introduced by Cameron in Exeter, England in 1905 (3,9). The septic tank required only a small initial investment and worked quite

well where the soil was not too tight for adequate subsurface dispersion of the effluent.

Often because of tight soils which made septic tanks impractical, small communities needed public wastewater treatment facilities and the demand for a very economical system led to the development of oxidation on biological stabilization ponds. These were first referred to as sewage lagoons and were used simply as holding ponds until it became apparent that stabilization of organics was occurring as dense growths of green algae developed. Through the process of photosynthesis, the algae utilized sun light for energy and combined carbon dioxide from the wastewater to form biomass through growth. During active growth periods oxygen was given off and dissolved into the surrounding water (this oxygen was used by the bacteria in the system to consume the organic pollutants in the wastewater). Oxidation ponds found wide acceptance for treating wastes from small towns in arid or semi-arid regions where land was not too expensive (3,10).

These methods of treating wastewater have been refined and modified over the years and in some instances new names have been given to the resulting processes. However, the basic processes described here and newer variations of them all have two common purposes. These are, first, to remove the readily biodegradable organic matter from wastewater and convert it to an innocuous form so that pollution of our natural waters is prevented or controlled and, second, to prevent the spread of water-borne diseases (9).

These processes do much to achieve the purposes, particularly

when this treatment is followed by disinfection by some means such as chlorination to destroy residual potential pathogenic organisms; but some of the compounds resulting from the biological breakdown of unstable organic materials are capable of exerting a detrimental effect on receiving waters or of lowering effluent acceptability for reuse. Two groups of such compounds are the refractory organic pollutants which are highly resistant to further biological treatment and the inorganic compounds that may be toxic to desirable plant or animal life or serve as growth stimulants to unwanted plants. Compounds containing nitrogen in the form of ammonia, nitrite or nitrate are prominent in the latter group; and many studies have been conducted to find practical methods for the control or removal of these forms of nitrogen.

Bacterial Utilization of Nitrogen Species in Natural Waters

The study of bacterial utilization of different nitrogen compounds present in natural waters indicates that the transformations of nitrogen in aquatic ecosystems are quite similar qualitatively to those occurring in the soil ecosystems which have been studied more extensively. While the kinds of transformations occurring in natural waters are well understood, the rates and mechanisms controlling these reactions are not yet adequately documented (11).

Bacterial utilization of nitrogen species in natural waters is of interest to environmental scientists and engineers for two primary reasons. First, those processes occurring in nature may be amenable to optimization for wastewater treatment plant applications; and, second, knowledge of bacterial utilization of the various forms of nitrogen provides some insight into which forms and in what quantities nitrogen may be discharged into natural waters without degrading them beyond acceptable limits.

An increase in the input of nitrogen and phosphorous compounds into fresh-water ecosystems can accelerate the complex biological, chemical and physical interactions which govern the growth of plant, animal and microbial populations. Such an increase in the concentration of available nutrients in an aquatic system is known as eutrophication; and the gross changes most commonly observed include increased growth of littoral vegetation, development of algae blooms and deoxygenation of the deeper or hypolimnic waters (12). While some scientists maintain that phosphorous is almost always the limiting nutrient factor in algal growth, Wuhrman (13) does not share their view. He points out that in algae the

proportion or ratio of nitrogen to phosphorous atoms is between 15 and 20 to one so that a "physiologically equilibrated" growth involves the uptake of approximately twenty times as many nitrate ions as phosphorous ions. This consideration tends to indicate that nitrogen is relatively less available than phosphorous in the concentrations normally found in the upper layers of lakes where algae find enough sunlight to grow; and, therefore, nitrogen may be the true limiting factor in at least as many instances as phosphorous. Lund (14) states that the low nitrate ($\text{NO}_3\text{-N}$) levels observed in summer in surface waters of eutrophic lakes might indicate N limitation at that time. Gerloff and Skoog (15) suggest that nitrogen is the important limiting nutrient for algal growth in Lake Mendota, Wisconsin; and Serruya and Berman (16) came to a similar conclusion regarding Lake Kinneret, Israel. Skapiro, in 1970, (17) and Ryther and Dunston, in 1971, (18) reported evidence to indicate that nitrogen rather than phosphorous is the critical factor in algal growth and eutrophication in coastal waters. Nitrogen is thus seen as exerting an important influence, and in some cases a critical influence, upon the natural receiving waters.

It has long been known that pollutants released to natural waters are subjected to "self-purification" processes in which microorganisms are involved. One of the most important of these processes is that in which aerobic bacteria oxidize certain pollutants. This process usually proceeds at a slow rate due to the small population densities of bacteria involved. This process of oxidation is a beneficial one in that it converts certain chemically active pollutants to stable or relatively inert forms, but it may also cause problems as the organisms remove dis-

solved oxygen from the water to meet their respiratory needs. This reduction in dissolved oxygen may reach a point at which fish and other aquatic life cannot survive. In extreme cases the water becomes completely deoxygenated and anaerobic bacteria may act on sulfates and other sulfur compounds to release poisonous and foul smelling hydrogen sulfide (H_2S) to the atmosphere (19). The effect of this reduction of dissolved oxygen is observed downstream from points at which biodegradable pollutants are dumped into flowing streams. This drop in dissolved oxygen level is commonly known as "oxygen sag" and is frequently the result of aerobic bacteria oxidizing organic or carbonaceous material present in domestic wastewater. However, it has been shown (20) that reduced forms of nitrogen such as ammonia and ammonia compounds, known as ammonia nitrogen (NH_4-N), also exert an oxygen demand on receiving waters as bacteria of the genus Nitrosomonas oxidize NH_3-N to nitrites (NO_2-N) and then bacteria belonging to the genus Nitrobacter oxidize NO_2-N to nitrates (NO_3-N).

This noncarbonaceous biochemical oxygen demand does not occur as rapidly as that for most dissolved carbonaceous material but may be most evident during the second week after the ammonia nitrogen is discharged into the receiving water (21). In some cases this process may be responsible for reducing the dissolved oxygen level below the critical point necessary for the support of desirable forms of aquatic life.

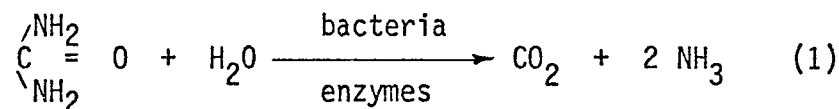
If the water containing dissolved nitrates does become deoxygenated, one of the resulting biological processes will usually involve saprophytic bacteria metabolizing any available organic energy source while utilizing the nitrates as electron acceptors (12,22). The nitrate nitrogen is thereby reduced to nitrogen gas and is no longer

chemically active or available to most aquatic organisms.

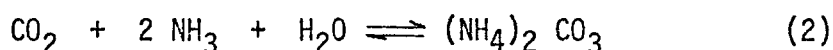
Bacterial utilization of nitrogen in natural waters is thus recognized as being a contributor to the "oxygen sag" problem in some instances as well as providing clues for nitrogen control by adaptation of natural bacteriological processes for conversion of ammonia nitrogen to nitrates to molecular nitrogen in advanced wastewater treatment plants. A more detailed review of the processes involved is presented in the following section.

Biochemistry of Microbial Utilization of Nitrogen

Nitrogen in fresh, raw domestic wastewater is mainly in the form of urea and proteins. Urea in solution is hydrolyzed to carbon dioxide and ammonia by bacteria and these compounds then form ammonium carbonate (23).

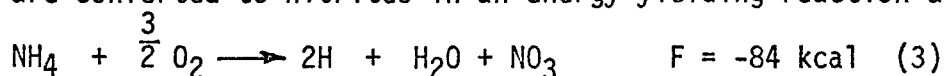


and



If decomposition of protein occurs under aerobic conditions it is referred to as decay. A variety of bacteria including Proteus vulgaris, Pseudomonas aeruginosa, Bacillus mycoides and some Actinomyces as well as some molds are capable of carrying this complex decomposition process from proteins to amino acids to ammonia under aerobic conditions. Anaerobic decomposition of protein is known as putrefaction and results in offensive odors. Clostridium putrificum is one of a number of micro-organisms involved in this process (24).

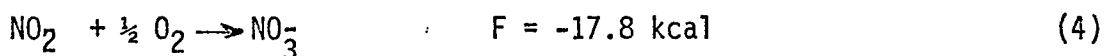
Ammonia is converted to nitrates mainly by two highly specialized groups of autotrophic, obligate aerobic bacteria in a process known as nitrification. This conversion was shown to be a biological process as early as 1877, and Winogradsky had isolated nitrifying organisms by 1890 (25). Nitrification occurs in two steps. In the first, ammonium ions are converted to nitrites in an energy yielding reaction as follows:



The genera Nitrosomonas and Nitrosococcus are the most common nitrifiers, but there are also three other genera of autotrophic nitrifiers. These are Nitrosaspira, Nitrocystis and Nitrosogloea (26). There are three

heterotrophic bacteria known to be capable of oxidizing ammonia to nitrite. These micro-organisms belong to the genera Pseudomonas, Streptomyces and Nocardia (27).

The second step in nitrification is the oxidation of nitrite to nitrate by bacteria of the genus Nitrobacter in the following energy yielding reaction (6,28,29):

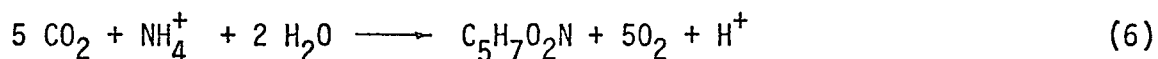


While oxidation of nitrite to nitrate is thought to be accomplished only by Nitrobacter, it is known that some heterotrophic bacteria can oxidize ammonia all the way to nitrate; and a fungus, Aspergillus flavus, is capable of oxidizing amino and ammonia nitrogen to the nitrate form (18,30,31,32). However, nitrification by the heterotrophic bacteria does not yield nearly as much energy as when accomplished by the autotrophic micro-organisms and is less prevalent in nature (1).

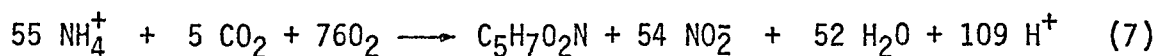
A comparison of energy yields shown in Equations 3 and 4 shows that the Nitrosomonas oxidation of ammonium ion to nitrate yields much more energy than the conversion of nitrite to nitrate by Nitrobacter. McKinney (33) states that Nitrobacter must process approximately three times as much substrate to obtain the same energy. This accounts for the fact that large build-ups of nitrite seldom occur in nitrification and the conclusion that the rate of conversion of ammonia to nitrite controls the overall nitrification reaction (28).

The overall energy reaction for conversion of ammonia to nitrate is $\text{NH}_4 + 2\text{O}_2 \longrightarrow \text{NO}_3^- + 2\text{H}^+ + \text{H}_2\text{O}$ (5)
This reaction furnishes energy for growth of the nitrifying bacteria; and during this growth process, some of the ammonium ion is assimilated into

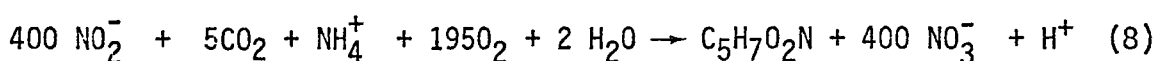
bacterial protoplasm with CO_2 being the source of cell carbon. The assimilation reaction, showing the resulting cell as $\text{C}_5\text{H}_7\text{O}_2\text{N}$, may be written as follows (1,29):



The overall reactions of nitrification and assimilation or cell synthesis may be written as follows (34): by Nitrosomonas spp.



by Nitrobacter spp.



On the basis of these equations, nitrification of 20 mg/l ammonium nitrogen ($\text{NH}_3\text{-N}$) would produce less than 3 mg/l of Nitrosomonas and about $\frac{1}{2}$ mg/l of Nitrobacter. Approximately 85 mg/l of dissolved O_2 would be consumed and about 2 moles of H^+ would be produced for each mole of $\text{NH}_3\text{-N}$ oxidized. Only about 2% of the original $\text{NH}_3\text{-N}$ would be utilized for cell growth (1).

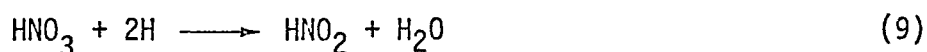
Denitrification is a biological process in which bacteria reduce nitrates and nitrites to molecular nitrogen (35). The phenomenon of denitrification was recognized as early as 1860 when it was observed that when fermentations took place in the presence of nitrates, it was usually accompanied by the production of nitrite, nitrous oxide and nitrogen gas (36). There is a lack of agreement on the exact enzymatic pathways involved and the inhibiting effect of dissolved oxygen, but it has been well established that a wide variety of common facultative bacteria have the ability to reduce the nitrate ion when dissolved oxygen is absent or very limited, and when a carbon energy source is available. Most of the denitrifiers are facultative anaerobic heterotrophs of the genera Pseudomonas,

Micrococcus, Spirillum and Achromobacter (28). According to Frey (37), the most common denitrifiers are Pseudomonas fluorescens, Pseudomonas stutzeri, Micrococcus denitrificans and various spore forming bacillus. Autotrophic bacteria known to be capable of denitrification include Thiobacillus denitrificans and Thiobacillus thioparus (38).

Most of the denitrifying bacteria are commonly found in waste-water treatment plants under aerobic conditions since these same organisms are also active in proteolysis, ammonification and other reactions; therefore, their presence in large numbers does not necessarily indicate that conditions are suitable for denitrification (39,40).

Denitrification is a respiratory mechanism in which the nitrate ion replaces molecular oxygen. This is quite different from assimilation in which NO_3^- is reduced to NH_4^+ and used in the synthesis of cell protein (1). Schroeder and Busch (41) emphasized this distinction in 1968 and referred to the respiratory mechanism as dissimilatory nitrate reduction. These researchers defined this phenomenon in a broad sense as the reduction of nitrate where the nitrate ion serves as the exogenous hydrogen acceptor for the oxidation of a substrate. Even though denitrification occurs in the absence of free oxygen, the nitrate ion serving as a hydrogen acceptor permits the bacterial cell to maintain aerobic metabolism (39). The carbon energy source serves as the hydrogen donor and can be any one of a variety of organic compounds.

The metabolic pathways of denitrification have been presented by Bremner and Shaw (42) in a simplified sequence of reactions as follows.





Possible end products of denitrification appear to be the gases nitric oxide (NO), nitrous oxide (N₂O) and nitrogen gas (N₂). The presence and relative amounts of these gases depends upon the conditions provided for growth (28).

Biological Nitrogen Control Processes in Wastewater Treatment

Biological processes currently appear to be destined to play a leading role in the removal of nitrogen species in wastewater since there is no generally accepted physical-chemical process for nitrogen control. Existing physical-chemical processes, including air stripping of ammonia, ion exchange, reverse osmosis, electrodialysis, distillation, breakpoint chlorination, and chemical denitrification using ferrous sulfate, involve nitrogen removal. To date, these operations have not been demonstrated to be economically feasible when only nitrification is required (43). Biological processes for nitrogen control offer a number of alternatives for meeting changing effluent standards. Therefore, the addition of the appropriate unit processes is normally all that would be required to upgrade an existing conventional treatment plant to meet ammonia or total nitrogen discharge limitations. The advantages of such alternatives have been illustrated in the Clear Creek Basin, south of Houston, Texas, where a total phosphorus limit and more stringent BOD₅ and total suspended solids limits were established in 1970 for municipal wastewater treatment plants in an effort to upgrade the quality of water in Clear Lake (44). After three years of further study, the Texas Water Quality Board decided that the desired level of dissolved oxygen could not be maintained in Clear Lake unless 1969 standards were revised (45). The proposed revision included a further reduction in allowable BOD₅ and a limitation on ammonia nitrogen. Adoption of the ammonia nitrogen limitation would make it necessary for each of approximately twenty municipal wastewater treatment plants to make changes in operations or facilities to meet the new standards. Some plants could probably meet the ammonia nitrogen limitation with minor changes while

others would require addition of separate nitrification facilities.

The variety of possible configurations for biological nitrogen control should provide the range of alternatives necessary for optimizing the upgrading of each plant with its own unique combination of unit processes, design capacities, loadings, efficiencies, and space available for new construction.

Within the last ten years in Europe and the United States substantial research has been conducted at the laboratory and field scale level to develop fundamental data on the application of biological processes for removal of nitrogen to varying degrees. Biological reactors for this purpose have been divided into two major types: suspended growth reactors and attached growth reactors. Both types have been utilized for nitrification and denitrification (1, 2, 29, 40).

Suspended growth reactors have been applied to activated sludge nitrification and denitrification processes (1,28,40,46,47). Suspended growth nitrification systems utilized either diffused air or mechanical stirring to keep the micro-organisms in suspension, to foster contact between the substrate and the micro-organisms, and to supply the required oxygen (1,28). Suspended growth denitrification reactors utilized mechanical mixing for maintaining the micro-organisms in suspension and for improving contact between the substrate and the denitrifying bacteria (46). Diffused air was not used for these two purposes since denitrification proceeds more efficiently in the absence of dissolved oxygen. For the same reason, rigorous mechanical mixing was avoided to prevent surface agitation which would have resulted in increased aeration of the mixed liquor (48).

Attached growth or fixed film reactors included trickling

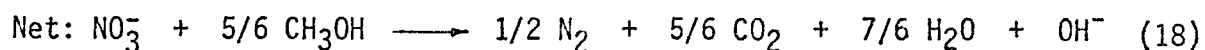
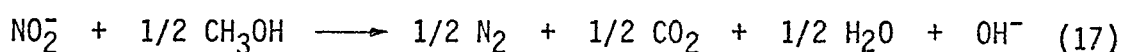
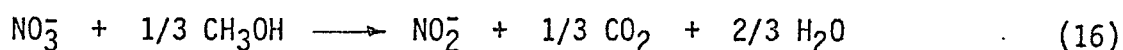
filters, submerged filters and packed towers. These were either upflow or downflow, were gravity or pressure fed and utilized a wide variety of media for physical support and retention of the microbiological population (6,49,50,51). Oxygen was supplied to the nitrification reactor by the injection of air or pure oxygen in the case of submerged filters and packed towers and by natural or forced circulation of air in trickling filters.

A carbon energy source was required for denitrification in both suspended growth and attached growth reactors. Investigators in Europe traditionally sought to use raw wastewater (20-40% of total plant flow) or the solids from aerobic biological treatment for this purpose (51). The primary reason for this was economic since these carbon energy sources were available at no cost. Disadvantages of using raw wastewater included the introduction of significant quantities of ammonia and organic nitrogen which could not be removed by biological denitrification. Nitrogen removal was limited to a maximum of 60-80% when raw wastewater was used for the carbon energy source. Biological solids from the aerobic nitrification process were used in an alternating cycle in which the solids were exposed first to aerobic and then to anaerobic conditions followed by recirculation to aerobic conditions. Harremoës (51) reported that several European scientists had tested this approach, but Karl Wuhrmann was the only one to achieve a high degree of nitrogen removal. Harremoës attributed the general lack of success with this approach to inadequate nitrification. American researchers, Mechals, et al. (52), conducted investigations of this approach with a more sophisticated arrangement of two mirror image attached growth reactors. The flow alternated in direction periodically as each reactor changed functions, with nitrifi-

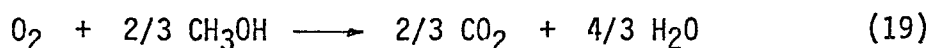
cation occurring in the leading reactor and denitrification taking place in the other. Nitrogen removal of about 95% from wastewater containing 49.4 mg/l using a small scale system to process 4 to 8 gal/day. During these studies it was determined that nitrification proceeded 5 to 8 times faster than denitrification which would require a balanced operational system to have a series of reactors in which 1/5 to 1/8 of the reactor volume would be periodically alternated between aerobic and anaerobic modes.

Organic materials considered suitable for supplying the energy required for denitrification included sugar, acetic acid, ethanol, acetone and methanol (1,22,28,54). While the use of any of these eliminated the inefficiency of nitrogen removal that resulted from addition of raw wastewater with its $\text{NH}_3\text{-N}$, Barth, et. al. (55), selected methanol because it was found that the lower alcohols were primarily oxidized instead of being synthesized to cellular material and because glucose caused bulking sludge in suspended growth reactors. Because of these advantages as well as the fact that it was relatively inexpensive and readily available, most investigators in the United States used methanol as the carbon energy source (1,4,22,54,55,56,57,58,59,60).

McCarty, Beck and St. Amant (54) reported on denitrification of agricultural wastewater using upflow, attached growth reactors. Methanol (CH_3OH) requirements were shown in the denitrification reactions as follows:



It can be seen that one mole of nitrate required at least 5/6 moles of methanol for denitrification. This was also expressed as 1.9 mg/l of methanol for each mg/l of nitrate nitrogen to be converted to nitrogen gas. In addition to the methanol requirement shown above, any dissolved oxygen had to be removed before denitrification could take place and methanol was provided for this as shown in the following equation:



Thus, 0.67 mg/l of methanol had to be provided for each mg/l of dissolved oxygen. An amount equal to about 30% of the stoichiometric amounts discussed above then had to be supplied for bacterial growth. This total requirement led to definition of the consumptive ratio as "the ratio of the total quantity of an organic chemical consumed during denitrification to the stoichiometric requirement for denitrification and deoxygenation alone". McCarty, et al. (54) reported this consumptive ratio for methanol as 1.3 and this has been confirmed as a suitable ratio by Stensel, et al. (59); Smith, et al. (6); and Dholakia, et al. (61). McCarty, et al. (54) developed the following formulas:

Methanol Requirement:

$$C_m = 2.47 N_0 + 1.53 N_1 + 0.87 D_0 \quad (20)$$

Biomass Production:

$$C_b = 0.53 N_0 + 0.32 N_1 + 0.19 D_0 \quad (21)$$

Where C_m = required methanol concentration, mg/l

C_b = biomass production, mg/l

N_0 = initial nitrite concentration, mg/l

N_1 = initial nitrate concentration, mg/l

D_0 = initial dissolved oxygen concentration, mg/l

Mulbarger (46) reported on modifications of the activated sludge process for nitrification and denitrification in a 0.2 mgd pilot plant. The study evaluated three process alternatives: single sludge, dual sludge and three sludge systems. The three sludge system was the only one in which organic carbon removal, nitrification, and denitrification were accomplished in separate reactors, each followed by a clarifier from which biological solids could be returned as necessary to the reactor in which the micro-organisms had developed. Mulbarger concluded that the three sludge system provided the "most dependable, consistent operation". This system achieved its superior performance in carbon removal, nitrification and denitrification by utilizing three isolated, optimized cultures of micro-organisms. Mulbarger's investigation of the three sludge system indicated that incomplete organic nitrogen hydrolysis to ammonia occurred in the first reactor, complete nitrogen oxidation occurred after two hours in the nitrifying reactor, and the addition of methanol for denitrification would allow complete removal of all nitrogen except for an apparently nonbiodegradable residual. The nitrification reactor was operated at a cell residence time of 10 to 15 days and an aeration detention time of 2.5 to 3.0 hours in constant flow studies. For variable flows, Mulbarger recommended detention times of 3.5 to 4.0 hours at nominal flow and 2.5 hours at peak flow. Solids growth was reported to range from a net loss to production of about 0.10 pounds of non acid-soluble volatile solids per pound of chemical oxygen demand (COD) removed. Sludge settling rates indicated that a surface overflow rate of 1320 ± 300 gpd/ft² could be utilized for clarifier design. Sludge volume indices (SVI) ranged from 50 to 80 during most of the operation. Mulbarger pointed out that since a pH

of 8 was optimum for nitrification, external sources of alkalinity would be required in an inadequately buffered wastewater. Alkalinity depletion was reported to average 6.1 pounds of CaCO_3 /pound $\text{NO}_3\text{-N}$ as compared to the theoretical value of 7.1 pounds of CaCO_3 /pound $\text{NO}_3\text{-N}$. Mulbarger reported that the methanol added for denitrification was preferentially oxidized rather than synthesized and volatile solids production generally ranged between 0.04 and 0.10 pounds of non acid-soluble volatile solids per pound of COD removed. Methanol was added at a rate of 3.0 to 3.5 parts per part of nitrate nitrogen and oxidized nitrogen was decreased from 17.9 mg/l to less than 1 mg/l.

Carry, et al. (50), investigated columnar denitrification of wastewater using two pressure sand filters operated in series. Each vessel contained 3.3 ft³ of No. 16 sand (0.97 mm) and provided a combined total empty bed detention time of nine minutes and a hydraulic application rate of 2.5 gpm/ft² when operated at a flow rate of 5.5 gpm. Both reactors were backwashed daily at 15 gpm/ft² using about 700 gallons, or 9% of flow. The investigators found that even with an excess of methanol, complete removal of nitrate did not occur consistently. While almost 100% removal was achieved at times, average removal was 83% (11 mg/l to 2 mg/l) at empty bed contact time of 9 minutes. Dissolved oxygen in the influent was about 1.1 mg/l and in the effluent it was zero.

Requa and Schroeder (60), reported on studies of kinetics of denitrification in a packed-bed, up-flow reactor. The reactor column was 4 inches in diameter, 66 inches tall and packed with Plexiglas Raschig rings, 1.1 inches long and 0.625 and 0.5 inches in outer and inner diameters, respectively. Total surface area of the packing was

4,740 square inches (specific surface area of about $80 \text{ ft}^2/\text{ft}^3$). Almost complete removal of nitrate occurred after steady state was approached at flow rates giving 1-hour, 2-hour and 4-hour residence times with 20 mg/l $\text{NO}_3\text{-N}$ in the influent. The $\text{NO}_2\text{-N}$ in the effluent ranged from almost zero with 4-hour residence time to 0.1 mg/l with 1-hour residence time (Figures 2,3, and 4). The researchers found that nitrate removal rate depended on nitrate concentration and bacterial cell concentration and that effluent quality changed very little as residence time was reduced because production of cells increased as the mass loading rate (mass of nitrate/time) increased. The curve of effluent quality versus residence time was found to be very flat. Cell concentration in the reactor was highest near the bottom of the up-flow reactor and lowest near the top. As shown in Figure 5, cell concentration ranged from about $12,000 \text{ mg/l}$ at the bottom to $2,000 \text{ mg/l}$ at the top for 1-hour residence time. The range was about $3,000 \text{ mg/l}$ to 300 mg/l for 4-hour residence time. Plug-ging and hydraulic washout of cells were the factors which limited mass loading rate.

The literature indicates that from a process point of view, biological removal of nitrogen from wastewater by nitrification and denitrification is feasible. However, to date no data has been published to indicate that a full-scale system utilizing a second stage suspended growth reactor for nitrification followed by an attached growth reactor for nitrification has been investigated to establish design criteria and develop operating procedures. This study at the El Lago Advanced Waste Treatment Facility was initiated for the purpose of obtaining this information by operating such a system to treat domestic wastewater under normal conditions involving diurnal variations in

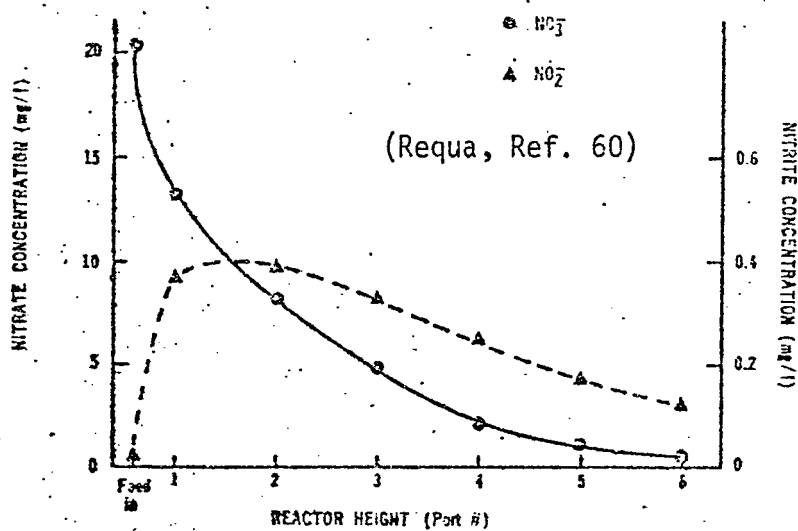


Figure 2.

Nitrate And Nitrite Concentration In
Reactor (1-Hr Residence Time)

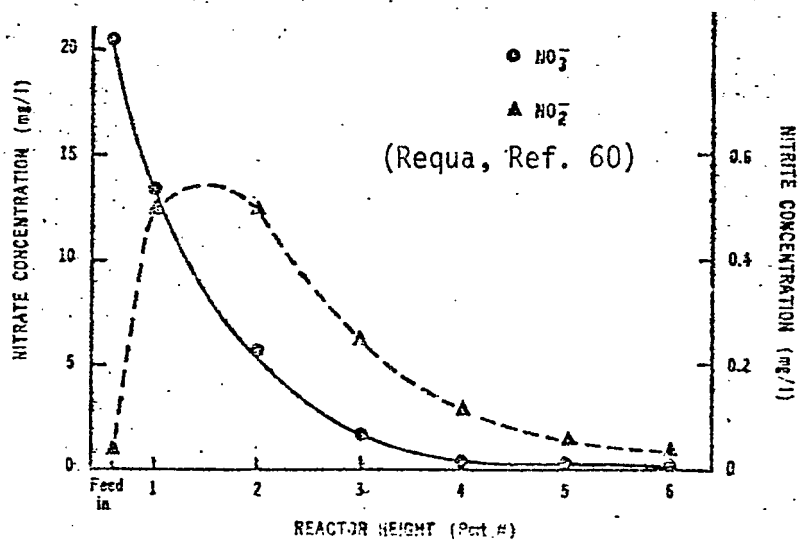


Figure 3

Nitrate And Nitrite Concentration In
Reactor (2-Hr Residence Time)

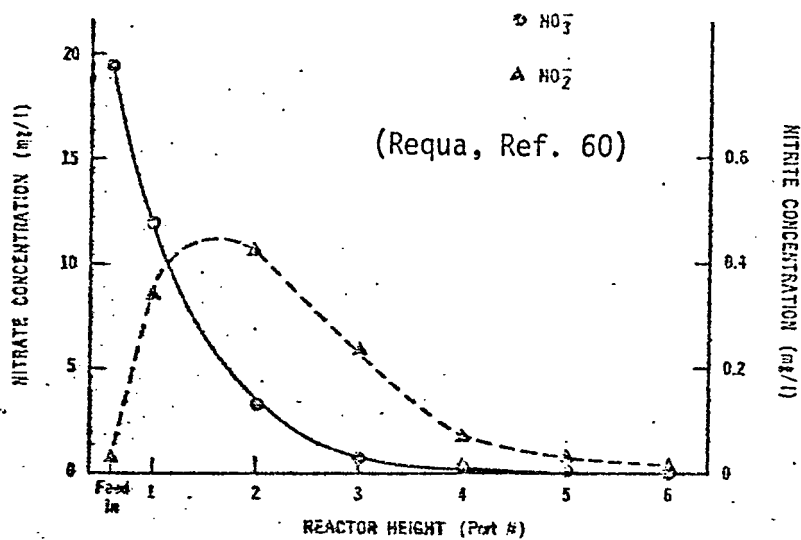


Figure 4

Nitrate And Nitrite Concentration In
Reactor (4-Hr Residence Time)

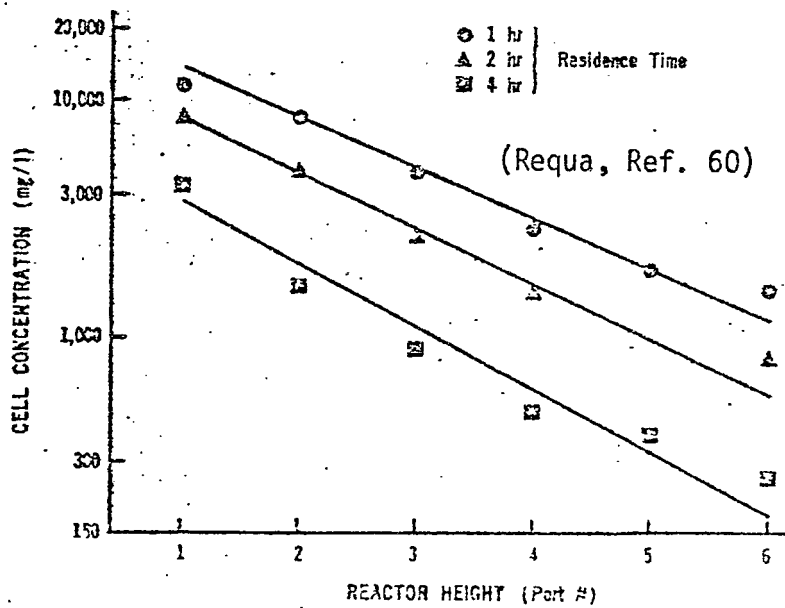


Figure 5

Cell Concentration In Reactor

plant loading and hydraulic stressing due to stormwater inflow. An additional objective was to evaluate the effectiveness of two different types of media to support the denitrifying micro-organisms.

CHAPTER III

EXPERIMENTAL PROGRAM

Introduction

The objectives of the experimental program were to utilize a full-scale nitrification-denitrification facility to evaluate existing design criteria, to develop additional criteria as required, to develop process controls and operations procedures in a normal operating environment, and to evaluate two types of media in fixed growth denitrification reactors.

Investigations for this study were conducted at the El Lago Advanced Waste Treatment Facility using a full-scale wastewater treatment plant. An existing plant consisting of primary sedimentation, dissolved organic pollutant removal via aerobic biological trickling filtration, sedimentation of biomass, disinfection with the use of chlorine in a contact basin and volume reduction and stabilization of suspended solids by anaerobic digesters was modified to permit investigations of nitrification and denitrification as well as phosphorus and colloidal suspended solids removal.

A schematic flow diagram of the El Lago Wastewater Treatment Facility prior to modification is shown in Figure 6. The facility consisted of two side-by-side plants receiving influent from a common wet well. Plant No. 1 was constructed in 1962 and had a design capacity of 200,000 gal/day while Plant No. 2 was completed in 1969 with a design capacity of 300,000 gal/day. Calculated loadings for the facility are shown in Table 1 and typical values for El Lago raw wastewater for the period May 1970 through August 1973 are shown in Table 2. The effective-

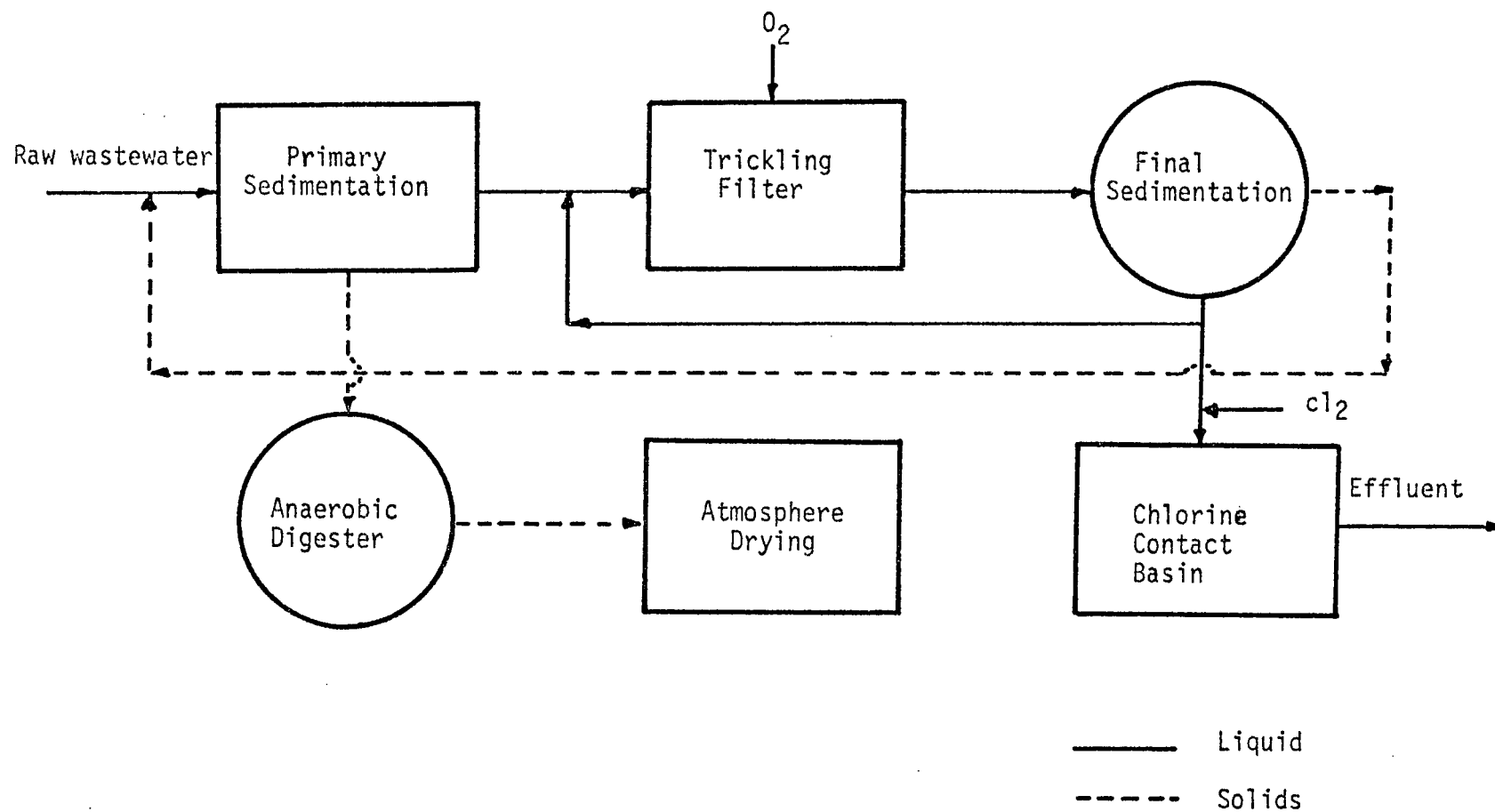


FIGURE 6 ORIGINAL EL LAGO WASTEWATER TREATMENT FACILITY

CALCULATED LOADINGS FOR THE EL LAGO FACILITY

Unit	Parameter	Intensity or Description
District	Population	3,000
Influent Flow	Avg. Dry Weather (ADW)	0.3 mgd
	Wet Weather (WW)	1.0 mgd
Primary Clarifiers	Detention Time	
	ADW	1.6 hr
	WW	0.5 hr
	Surface Overflow Rate	
	ADW	440 gpd/ft ²
	WW	1,480 gpd/ft ²
Trickling Filters	Media	Natural rock
		Nominal 4 inch
	Depth	6.5 feet
	Organic Load	12 lbs/day/1,000 ft ³
	Hydraulic Load	
	ADW	92 gpd/ft ²
	WW	310 gpd/ft ²
	Recirculation	Constant 0.3 mgd
Final Clarifiers	Detention Time	
	ADW	5.4 hr
	WW	1.5 hr
	Surface Overflow Rate	
	ADW	320 gpd/ft ²
	WW	1,060 gpd/ft ²
Chlorine Contact	Detention Time	
	ADW	1 hr
	WW	18 minutes
Anaerobic Digesters	Volume	8,830 ft ²
	Volume/Capita	2.9 ft ²
Sand Drying Bed	Area	6,300 ft ²
	Area/Capita	2.1 ft ²

TABLE 2

TYPICAL VALUES FOR EL LAGO
RAW WASTEWATER FOR THE PERIOD
MAY 1970 THROUGH AUGUST 1973

Item	mg/l	
	Average	Range
BOD ₅	161	93 - 223
COD	287	89 - 654
SS	195	18 - 256
TP	13.6	3.7 - 27
Ammonia Nitrogen	24	2.4 - 49
Organic Nitrogen	13.5	2.4 - 25
Oxidized Nitrogen	0	-
pH Units (median)	7.6	7.0 - 8.1
Alkalinity as Calcium Carbonate	345	-
Total Oxygen Demand	455	-

ness of the biological treatment prior to plant modification is shown in Table 3 which presents the tabulated values for primary clarifier effluent and plant final effluent.

Evaluation of data led to the conclusion that while total oxygen demand was being greatly reduced and partial nitrification was taking place in the trickling filters, effective denitrification would be dependent upon a more completely nitrified wastewater. Provisions for this was made by the addition of a suspended growth nitrification reactor immediately following the trickling filters. This reactor was divided into two chambers so that reactor volume, and thus detention time, could be varied by using one or both of the chambers. Specifications for the suspended growth nitrification reactors are given in Table 4.

The effluent or mixed liquor from the nitrification reactor was routed to the existing final clarifiers which then became known as the intermediate clarifiers. Air lift pumps were installed in the intermediate clarifiers to return settled solids to the nitrification reactor. The existing capability for wasting any excess solids to the head of the plant was retained.

A pump sump was constructed to receive the effluent from the clarifiers. Two variable speed, five-stage vertical turbine pumps (Goulds, Model VIT) were provided to deliver nitrified wastewater to the denitrification reactors both for feed and backwash. Each pump had a capacity of 210 gpm at a total pumping head of 115 feet. The pumps were controlled by an Autocon system which varied pump speed and output proportional to depth of water in the sump. During periods of very low flow, only one pump operated at a slow speed, with that pump increasing speed and the other pump coming into operation as flow into the sump

TABLE 3

VALUES FOR PRIMARY AND FINAL EFFLUENT
FOR THE PERIOD MAY 1970 THROUGH JUNE 1972

Item	mg/l			
	Primary Effluent		Final Effluent	
	Average	Range	Average	Range
BOD ₅	121	70-140	12	5-25
COD	229	119-260	67	52-80
SS	56	30-106	12	4-20
TP	13.6	10-20	13.8	8-19
Ammonia Nitrogen	15.7	14-17	5.5	5-10
Organic Nitrogen	3.5	2-5	2	1-2
Oxidized Nitrogen	0	-	11	5-12
pH Units	7.2	6.9-7.8	7.5	7.0-8.0
Total Oxygen Demand	315	-	101	-

TABLE 4

SPECIFICATIONS FOR NITRIFICATION REACTOR

Section	Volume	Detention Time, hr. ADW	Detention Time, hr. WW
Main Bay	56,988 gal	4.5	1.3
Second Bay	18,817 gal	-	-
Both Bays	75,805 gal	6.1	1.8
Nitrified Effluent Sump	18,817 gal	-	-

Diffusers:

Two headers in main bay 3.9 m (12.4 ft) each, with diffusers spaced 0.3 m (1 ft). One header in second bay 2.4 m (8 ft) with diffusers spaced 0.3 m. All headers located on south wall.

Compressors:

Two 12.6 cu m/minute (450 cu ft/min), alternately operated; to supply air for nitrification bays, air lift pumps, and filter scour.

Air Lift Pumps:

Two each, located in each clarifier to return settled mixed liquor to main nitrification bay.

Intermediate Clarifiers:

As shown on Table 1 as final clarifiers.

increased. This provided flow and positive internal hydraulic pressure to the denitrification reactors under a wide range of flow conditions (50-600 gpm).

Denitrification reactors were of the attached growth type and two sets of towers were installed in order to test two different media. Two towers were erected for each media in order to get the desired bed depth without having the towers so high as to be esthetically unacceptable in the residential area where the plant is located. The two granular media towers contained 3-4 mm round sand particles and were operated downflow in series. These towers were a proprietary design of Dravo Corporation. Design specifications for the granular media towers are given in Table 5. The two plastic media towers were packed with 5/8" cylindrical polyethylene Flexirings (Koch Engineering, Inc.) and were operated upflow in series. These towers were designed specifically for this study and the steel vessels were shop-fabricated and field erected by local contractors. Design specifications for the plastic media towers are shown in Table 6.

Methanol was selected as the carbon energy source to be added to the denitrification feed stream and two variable speed pumps were installed for this purpose. Each pump was capable of delivering up to 25 gal/day against a head of 15 psi. The length of stroke on these diaphragm pumps was manually adjusted and the speed was controlled by means of a signal from the Autocon system so that methanol pump speed was directly proportional to denitrification feed pump speed. Methanol was injected into the denitrification feed stream shortly after it came from the vertical turbine feed pumps.

Installation of a tertiary filter and connection of the chlorine

TABLE 5

DESIGN SPECIFICATIONS FOR GRANULAR
MEDIA DENITRIFICATION TOWERS

Item	Specifications
Vessels	2 - connected in series, downflow
Pressure Test	to 50 psi
Material	steel tanks, sandblasted and shop coated
Diameter	6 feet
Media height	6.5 feet each tower
Media type	3-4 mm rounded sand of glacial origin
Porosity	40 percent. Surface area 250 ft ² /ft ³
Empty bed contact time	15 minutes
Process hydraulic rate:	
at 0.3 mgd	7.4 gpm/ft ²
at 0.5 mgd	12.3 gpm/ft ²
Backwash water source	nitrified effluent
Backwash rate	20 gpm/ft ²
Air scouring rate	8 cfm/ft ²
Freeboard	30 percent bed expansion

TABLE 6

DESIGN SPECIFICATIONS FOR PLASTIC
MEDIA DENITRIFICATION TOWERS

Item	
Vessels	2 - connected in series, upflow
Pressure test	to 50 psi
Material	steel tanks, sandblasted and shop coated
Diameter	10 feet
Media height	10 feet
Vessel interior	Divided into quarters by solid walls, from the bottom head to the top of the media. Freeboard section common to all quarters. Separate influent connections for each quarter. Wire mesh across top of media.
Media type	5/8 inch flexirings, polypropylene
Porosity	92 percent. Surface area 105 ft ² /ft ³
Empty bed contact time	60 minutes
Process hydraulic rate:	
at 0.3 mgd	2.5 gpm/ft ²
at 0.5 mgd	4.1 gpm/ft ²
Backwash water source	nitrified effluent
Backwash rate	20 gpm/ft ²
Freeboard	Not needed. Media retained by grating.

contact chambers in series completed the significant parts of the plant modification.

A schematic flow diagram of the modified plant, known as the El Lago Advanced Waste Treatment Facility, is shown in Figure 7.

Analytical Program

The major sampling points were numbered 1 through 6 on Figure 7. Most samples were 24-hour composites consisting of 4 equal elements collected at 10:00 a.m., 1:30 p.m., 4:00 p.m. and 7:30 a.m. Grab samples were taken for determination of total suspended solids, volatile suspended solids and settleability of the nitrification mixed liquor suspended solids. Grab samples were also taken of the raw wastewater at 4-hour intervals over a 24-hour interval in order to determine diurnal variations in organic loading.

Analyses of samples were in accordance with Standard Methods for the Examination of Water and Wastewater, 13th Edition (62), and the Environmental Protection Agency, Analytical Quality Control Laboratory manual, Methods for Chemical Analysis of Water and Wastes (63). All samples were refrigerated immediately after being collected; and, where appropriate, analyses were performed on the same day the sample was collected.

Dissolved oxygen levels were monitored in nitrification and denitrification influent and effluent with a Weston and Stack Model 300 portable oxygen analyzer (64).

Plant Operation

In January 1973, the nitrification reactor was put into operation by diverting the underdrain from the trickling filters into the

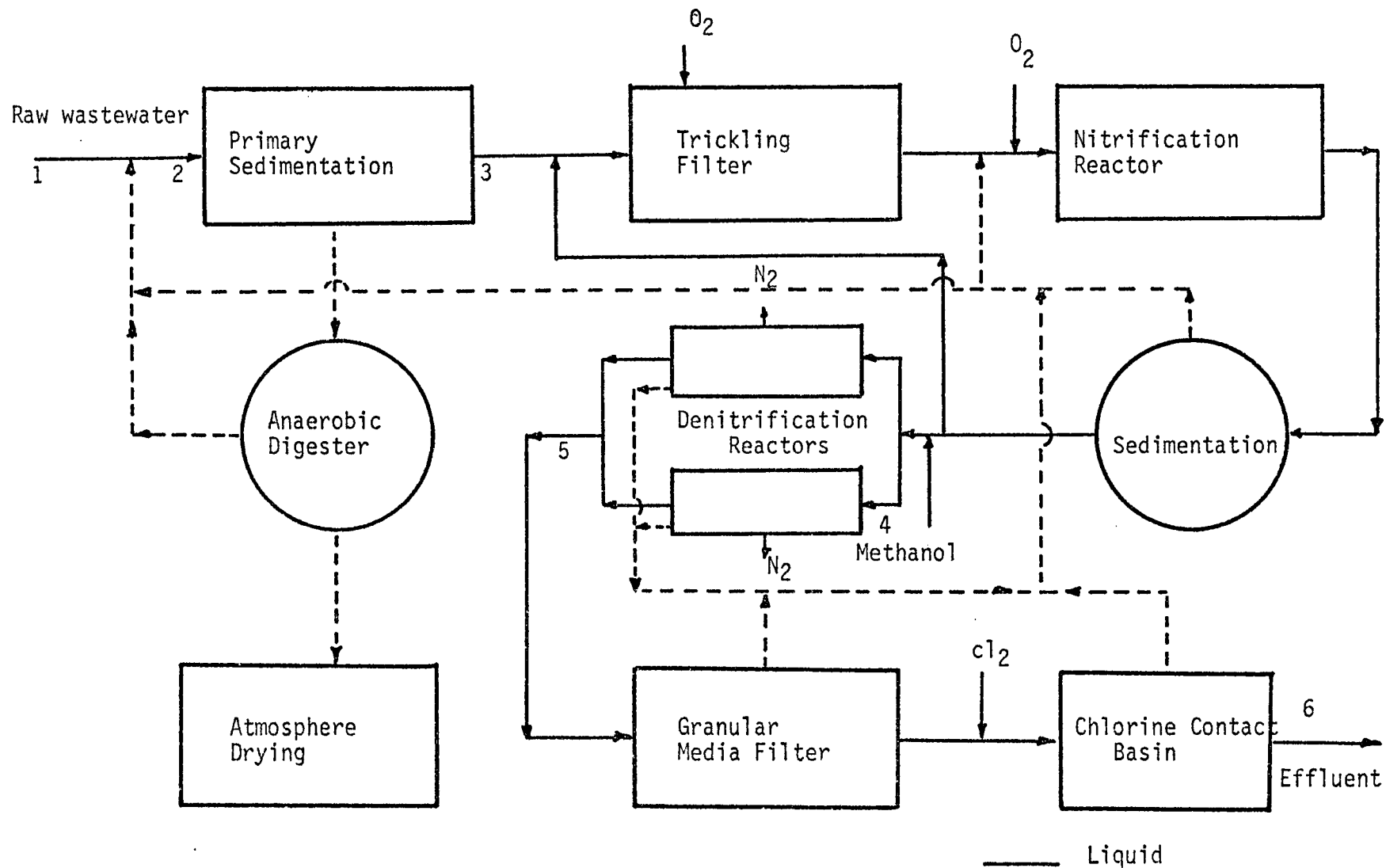


FIGURE 7 EL LAGO ADVANCED WASTE TREATMENT FACILITY

reactor and providing air for mixing and aeration. The effluent from the reactor was routed to the intermediate clarifiers from which approximately 200 gpm of the clarified effluent was recirculated to the trickling filters. The biomass that settled out of the mixed liquor was returned to the nitrification reactor by the airlift pumps. Difficulty was encountered in maintaining suspended solids in the system due to the high flow during periods of heavy rain which tended to wash out the solids and the low concentration of BOD_5 in the trickling filter underdrain which was insufficient to support flocculant growth in the nitrification reactor. Wash out was minimized by isolating the nitrification reactor during the short periods of peak flow due to inflow of surface water into the collection system. Better flocculating bacterial growth was achieved by diverting a 100 gpm stream of primary effluent into the nitrification reactor. This was done from March 12, 1973 to March 26, 1973; and during this two week period, mixed liquor solids increased from 40 mg/l to 1,000 mg/l. Addition of the primary effluent was discontinued on March 26, 1973, but the practice of isolating the reactor during heavy rainfall periods was continued. By June, 1973 the mixed liquor contained about 2,500 mg/l total suspended solids and 1,000 mg/l volatile suspended solids. During the ensuing four months, no solids were withdrawn.

With air being supplied to the nitrification reactor at 375 cu. ft/min., dissolved oxygen levels were found to be approximately 5.0 mg/l at the point where the trickling filter underdrain entered the reactor and 6.8 mg/l in both nitrification effluent and the intermediate clarifier effluent. Throttling the air to decrease the dissolved oxygen level resulted in inadequate mixing in the reactor and thus proved

impractical.

Results of testing to determine nitrification reactor performance appear in Chapter IV.

The granular media, down flow, denitrification towers were operated from early May to early July 1973. During the first 10 days of operation, the ratio of methanol to nitrate nitrogen was gradually increased from 1:1 to 3:1. Bacterial cell growth in the sand media which has a porosity of forty percent was such that backwashing of the first of two towers in series was required at approximately 24-hour intervals. It was assumed that backwashing was needed when pressure drop across a tower reached 20 psi at a flow rate of 250 gpm. The second tower normally required backwashing at 48-hour intervals. Backwash required approximately 7,000 gallons of nitrified wastewater per day which was slightly more than two percent of the normal throughput volume. Loading of the granular media towers by bacterial cells with the resulting resistance to flow made it impossible to maintain a constant ratio of methanol to wastewater with the Autocon flow control system. Speed of the denitrification feed pumps was dependent upon depth of nitrified wastewater in the sump; and as plugging of the media occurred, the flow rate decreased and the sump level rose. The flow controller which signalled the wastewater pumps to increase speed to maintain the flow rate also sent a signal to the methanol pump controls to increase speed and methanol delivery. This resulted in a high ratio of methanol to wastewater. The problem was minimized by adjusting the methanol pumps to somewhat underdose under clean tower conditions which resulted in only a slight overdose as the tower reached the state of solids loading at which backwash was required.

Evaluation of the granular media reactor performance was based on data taken over a 33 day period during which time a relatively steady state condition had been attained and operational problems were at a minimum. A summary of the operational conditions and the data is presented in Chapter IV, Tables 8 and 9.

The plastic media denitrification towers were filled with nitrified wastewater and pressure tested for leaks in January, 1973. After testing, the pressure was reduced to ambient and the water left in the idle vessels for almost six months. Within three days after start-up, denitrification appeared to have reached steady state, probably as a result of micro-organisms becoming established on the plastic media during the preceding months. Upflow operation continued for six weeks before it became necessary to deliberately wash solids from the towers. This was not dictated by pressure drop through the towers, but rather by an increase in effluent solids. The plastic media towers were not back-washed in the true sense of the term since direction of flow was not reversed. Solids were purged by valving off the flow to three of the quarters and surging the total flow through the fourth quarter. This procedure was repeated until all quarters of each vessel were purged. The effluent from this operation was returned to the primary clarifiers where the solids were settled out and routed to the anaerobic digesters. During resumption of operations following a period of two days when the towers were out of service because of pump problems, it was discovered that a large quantity of solids had floated to the top of the towers. The effluent contained such a high concentration of solids that the final polishing filters were almost immediately loaded to such an extent that flow practically ceased. As a result of these experiences,

routine surging of the plastic media towers was commenced at 4-week intervals and towers were surged before being taken out of service and again upon resumption of operations. Maintaining a uniform methanol feed rate was relatively easy during operation of the plastic media towers since the resistance to flow due to biological growth on the media was not as variable as it had been with the granular media towers.

Data for performance evaluation of the plastic media reactors was collected over a period of 55 days, commencing two days after start-up. The operating conditions and performance data are presented in Chapter IV, Tables 10 and 11.

CHAPTER IV

PRESENTATION AND DISCUSSION OF RESULTS

This investigation consisted of full-scale plant studies of suspended growth nitrification and attached growth denitrification under normal operating conditions including maximum hydraulic loading during periods of high rainfall.

Nitrification Reactor Evaluation

The evaluation of the nitrification capability of the modified plant was based primarily upon data taken during the months of July and August 1973 and summarized in Table 7. Significant variables measured during this period were ammonia nitrogen, organic nitrogen and nitrate nitrogen. Dissolved oxygen levels and pH were monitored throughout the evaluation period. A comparison was made between this data and the limited data obtained on plant operations during the summer of 1970 in order to determine overall improvement in nitrification.

Because of the high recirculation rate from the intermediate clarifier to the trickling filters, it was impossible to isolate the nitrification reactor performance from that of the trickling filters. At average dry weather flow the recirculation ratio was about 1:1, and this diluted the trickling filter underdrain which had a high content of ammonia nitrogen and organic nitrogen with the highly nitrified clarifier effluent. The trickling filters were thus made to appear to have a high nitrification capability. From the results as summarized in Table 7, it was determined that total nitrogen content of the diluted trickling filter underdrain consisted of 11% ammonia nitrogen, 15% organic nitrogen

TABLE 7

NITRIFICATION REACTOR PERFORMANCE

33 Sample Periods in July and August 1973								
mg/l								
Raw Wastewater		Primary Effluent		Rock Filter Underdrain		Nitrified Effluent		
Constituent	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range
NH ₃ -N	15.4	24.6-3.9	15.1	24.0-4.0	2.3	3.4-0.8	1.0	2.3-0.3
Org-N	14.6	15.4-10.0	10.6	11.5-8.0	3.1	5.4-1.4	1.6	5.3-0.5
NO ₃ -N	-	-	-	-	15.6	22.9-7.3	13.6	23.8-5.9

nitrogen and 74% nitrate nitrogen. It was also determined that after passing through the suspended growth nitrification reactor, the nitrogen species were 6% ammonia nitrogen, 10% organic nitrogen and 84% nitrate nitrogen. The effectiveness of the suspended growth reactor was also apparent when the results of the evaluation were compared with data taken in the summer of 1970 to evaluate the nitrifying capability of the trickling filters with 1:1 recirculation from the clarifiers. Data from 1970 showed that the trickling filter influent contained 19.5 mg/l of total Kjeldohl nitrogen (TKN) and no measurable nitrates and the effluent contained 9.0 mg/l TKN and 7.2 mg/l nitrate nitrogen. Unoxidized species were decreased 54% by the trickling filters with recirculation. This observation compared to a 90% decrease in the combined trickling filter - suspended growth reactor system. A loss of about 10 mg/l of total nitrogen occurred in the trickling filter, suspended growth reactor and intermediate clarifier combination. This loss was attributed to a combination of air stripping of ammonia in the trickling filter and to coincidental denitrification. Conditions favorable to denitrification occurred while the nitrified effluent was contacted with the primary effluent in approximately 40 feet of closed channel enroute to the trickling filter media and as the filter underdrain passed through another 60 to 80 feet of closed channel to the nitrification reactor.

Throughout this period, dissolved oxygen in the trickling filter underdrain remained at about 5.0 mg/l and that of the nitrification reactor effluent was on the order of 6.8 mg/l. The pH of trickling filter underdrain and nitrification reactor effluent was about 7.8 and 7.9 respectively. Air required to provide adequate mixing in the nitrification reactor supplied an abundance of dissolved oxygen for biological

activity and the alkalinity of the wastewater made it unnecessary to add chemicals for pH control.

Granular Media Denitrification Reactor Evaluation

The performance of the granular media denitrification reactor was evaluated on the basis of data taken during a 33-day period extending from June 4 to July 6, 1973. Pertinent variables measured to assess reactor performance included suspended solids (SS), ammonia nitrogen ($\text{NH}_3\text{-N}$), total Kjeldahl nitrogen (TKN), nitrate nitrogen ($\text{NO}_3\text{-N}$), five-day biochemical oxygen demand (BOD_5), chemical oxygen demand (COD) and methanol dose rate. These results, as well as data on total and soluble phosphorus (TP and SP), rainfall, and flow rates, were summarized for presentation in Tables 8 and 9.

During the evaluation period, inflow of storm water into the sanitary sewer system caused total plant flow to reach a high of 1.00 mgd on a day when rainfall measured 5.7 inches. Limited denitrification capacity necessitated diversion of that flow in excess of 0.420 mgd directly from the intermediate clarifiers to the chlorine contact chambers. Average denitrification flow for the period was 0.254 mgd. Empty bed contact time averaged slightly less than 18 minutes.

Nitrate nitrogen averaged 15.2 mg/l in the influent and 2.6 mg/l in the effluent for removal of 83%. Overall plant removal of total nitrogen during this period was 87%. The average value for total nitrogen in the raw wastewater was 42.6 mg/l and the final effluent contained an average of 5.6 mg/l.

The average dose rate for methanol during the evaluation period was 47 mg/l. Since the dissolved oxygen level (D_0) in the nitrified clarifier effluent was consistently on the order of 6.8 mg/l,

TABLE 8

CONDITIONS FOR INITIAL EVALUATION OF
GRANULAR MEDIA DENITRIFICATION TOWERS

Length of study period: 33 days

Rain during study period:

12.2 inches
5.7 inches peak day

Total flow to plant:

0.307 mgd average
0.160 mgd low day
1.000 mgd high day

Flow to fine media denitrification towers:

0.254 mgd average
0.160 mgd low day
0.420 mgd high day

Wastewater temperature: 78°F

Total number of analytical measurements: 433

Phosphorus removal: Due to high flow on some days of study period metal salt and polymer were not dosed continuously.

Nitrogen removal: Nitrite nitrogen is not given in Table 9 because all values were less than 0.2 mg/l.

TABLE 9
INITIAL EVALUATION OF GRANULAR MEDIA DENITRIFICATION TOWERS

June 4 to July 6, 1973												
All Values, mg/l												
() = Number of Samples												
	Raw Wastewater		Primary Influent		Primary Effluent		Nitrified Effluent		Denitrified Effluent		Final Effluent	
Item	Avg	Range	Avg	Range	Avg	Range	Avg	Range	Avg	Range	Avg	Range
TP	12.8	3.7-21.8 (4)	15.4	4.8-22.8 (10)	8.4	5.1-15.6 (10)	7.3	4.0-16.1 (5)	6.6	1.5-11.5 (12)	4.8	4.1-5.4 (2)
SP	10.3	2.4-17.0 (4)	4.7	0.9-12 (10)	4.1	1.6-6.9 (10)	3.4	2.1-3.9 (4)	5.5	1.0-11.0 (12)	3.6	2.1-5.0 (2)
SS	113	21-200 (4)	289	98-754 (10)	72	37-114 (10)	37	8-57 (22)	17	2-56 (19)	3	1-6 (14)
NH ₃ -N	18.7	2.4-35.2 (4)	21.7	16.2-26.2 (7)	21.5	16.2-23.9 (6)	0.9	0.4-2.2 (22)	0.8	0.5-1.8 (19)	0.6	0.4-0.7 (3)
TKN	42.6	7.7-64.7 (4)	38.6	30.8-49.3 (7)	30.2	29.3-31.6 (5)	3.7	0.8-10.8 (22)	2.4	0.9-6.2 (19)	3.3	1.5-6.2 (3)
NO ₃ -N	-	-	-	-	-	-	15.2	5.4-24.8 (22)	2.6	0-9.7 (19)	2.3	0-5.4 (3)
BOD ₅	175	(1)	222	220-223 (2)	-	-	65*	58-72 (2)	9	6-12 (2)	9	5-18 (4)
COD	297	89-391 (4)	488	244-720 (10)	181	101-240 (10)	121*	51-224 (19)	72	16-176 (18)	51	36-90 (11)
Methanol Dose	-	-	-	-	-	-	47	20-81 (23)	-	-	-	-

*Includes demand due to added methanol.

nitrate nitrogen (N_O) averaged 15.2 mg/l and nitrite nitrogen was near zero, the theoretical methanol requirement (C_m) was calculated using Equation 20 as follows:

$$C_m = 2.47 (15.2) + 0.87 (6.8) = 43 \text{ mg/l}$$

The average methanol feed exceeded denitrification requirements by approximately 4 mg/l. Since each mg/l of methanol requires about 1.5 mg/l of dissolved oxygen for its biological oxidation, this excess methanol appeared in the effluent as 6 mg/l of BOD_5 .

Average temperature of the denitrification influent was 78°F (25.6°C) and pH ranged from 7.80 to 8.40. Effluent pH ranged from 7.75 to 8.30.

Design parameters for the granular media reactors used in this study differed somewhat from those used by Carry, et. al (50). The El Lago reactors made use of larger media (3-4 mm sand vs 0.97 mm sand), had greater total bed depth (13.0 ft vs 6.6 ft), and provided somewhat longer empty bed contact time (15 min. vs 9 min.). A lower percentage of throughput volume was used for backwash purposes (2% vs 9%), probably due to the larger media and less solids removal from actual filtration. Both systems required daily backwash operations.

Methanol dose rates were very nearly the same and nitrate removal (83%) was the same. The greater bed depth and longer empty bed detention times in the El Lago reactors were probably responsible for achieving similar nitrogen removals while being subjected to ten-fold variations in flow rates (60 to 600 gpm) instead of the uniform flow rate imposed by Carry.

Design parameters for the El Lago granular media nitrification reactors appeared to be reasonable, but improved performance probably

would have been achieved with a flow control system which would have fed methanol into the reactors directly proportional to wastewater flow. An even greater improvement would have been possible with a system incorporating an automatic analyzer for continuous monitoring of nitrate and control of methanol dose rate.

Plastic Media Denitrification Reactor Evaluation

Data for evaluation of the plastic media denitrification reactor were taken over a 55-day period from July 8 through August 31, 1973. A summary of the data is presented in Tables 10 and 11.

The wastewater treatment plant as a whole was subjected to a lower hydraulic loading during evaluation of the plastic media reactors than during the studies on the granular media reactors; but due to the lower head loss through the plastic media, higher denitrification flows were attainable. Flow through the plastic media towers averaged 0.315 mgd as compared to 0.254 mgd for the granular media. During the periods of peak plant flow due to inflow of storm water into the sanitary sewer system, it was necessary to divert some flow directly from the intermediate clarifier to the chlorine contact chambers. The capacity of the two pumps which fed the nitrification reactors and the final polishing filters was inadequate to handle peak flow due primarily to head loss across the final polishing filters. The average daily flow through the reactors (0.315 mgd) was 5% above the design capacity (0.300 mgd) and the peak daily flow (0.632 mgd) was more than 200% of design capacity. Empty bed contact time averaged about 55 minutes.

Nitrate nitrogen averaged 13.6 mg/l in the influent and 0.9 mg/l in the effluent for removal of 93%. Overall plant removal of

TABLE 10

CONDITIONS FOR INITIAL EVALUATION OF PLASTIC
MEDIA DENITRIFICATION TOWERS

Length of study period: 55 days

Rain during study period:

9.91 inches
2.46 inches peak day

Total flow to plant:

0.320 mgd average
0.171 mgd low day
0.900 mgd high day

Flow to plastic media denitrification towers:

0.315 mgd average
0.171 mgd low day
0.632 mgd high day

Wastewater temperature: 81°F

Total number of analytical measurements: 1,254

Phosphorus removal: At different periods of time ferric chloride and aluminum sulfate were used to insolubilize phosphorus.

Nitrogen removal: During one 24-hour period during the study the methyl alcohol pumps were inoperative. Nitrite nitrogen is not given in Table 11 because all values were less than 0.2 mg/l.

TABLE 11
INITIAL EVALUATION OF PLASTIC MEDIA DENITRIFICATION TOWERS

July 8 through Aug. 31, 1973													
All Values, mg/l													
() = Number of Samples													
	Raw Wastewater		Primary Influent		Primary Effluent		Nitrified Effluent		Denitrified Effluent		Final Effluent		
Item	Avg	Range	Avg	Range	Avg	Range	Avg	Range	Avg	Range	Avg	Range	
TP	12.3		13.1		6.7		-		-		2.8		
		6.2-18.5		6.5-22.1		1.0-17.4		-		-		1.0-8.2	
		(25)		(36)		(40)						(31)	
SP	10.3		3.1		2.4		-		-		2.3		
		3.3-15.9		0.5-9.6		0-6.5		-		-		0.5-6.2	
		(25)		(36)		(36)						(31)	
SS	102		231		63		43		19		4.5		
		43-219		104-456		17-136		2.0-90		2-71		0.4-24	
		(26)		(36)		(36)		(37)		(38)		(33)	
NH ₃ -N	15.6		15.1		14.0		0.8		1.0		0.8		
		3.9-24.6		3.1-29.3		3.9-20		0.0-2.3		0.0-3.0		0.0-1.8	
		(26)		(32)		(36)		(38)		(38)		(32)	
TKN	29.7		31.8		26.7		2.6		2.5		1.7		
		13.9-40.0		19.3-46.2		16.2-35.4		0.8-7.6		0.5-6.1		0.9-3.5	
		(26)		(32)		(36)		(37)		(38)		(32)	
NO ₃ -N	-		-		-		13.6		0.9		0.6		
		-		-		-		5.9-23.8		0-3.0		0-3.5	
BOD ₅	143		156		87		43*		15		8		
		60-260		86-243		47-124		11-66		3-38		0.8-20	
		(11)		(12)		(13)		(14)		(14)		(14)	
COD	248		336		167		107*		52		38		
		136-380		111-590		97-329		50-207		23-182		20-63	
		(26)		(32)		(36)		(38)		(38)		(33)	
Methanol Dose	-		-		-		34		-		-		
		-		-		-		16-69		-		-	
								(49)					

*Includes demand due to added methanol.

total nitrogen during the period was 92%. Total nitrogen in the raw wastewater averaged 29.7 mg/l and the final effluent contained an average of 2.3 mg/l.

The average dose rate for methanol was 34 mg/l. The theoretical methanol requirement (C_m) was calculated using Equation 20 with the average influent nitrate value of 13.6 mg/l, disregarding the average nitrite value of less than 0.2 mg/l and assuming dissolved oxygen to be 6.8 mg/l.

$$C_m = 2.47 (13.6) + 0.87 (6.8) = 40 \text{ mg/l}$$

The average methanol feed was below the calculated average requirement by approximately 6 mg/l.

The temperature of the denitrification influent averaged 81°F (27.2°C) and pH ranged from 7.48 to 8.25. Effluent pH ranged from 7.60 to 8.00.

The average actual contact time in the plastic media reactors was about 51 minutes (55 minutes empty bed contact time X 0.92 porosity). This was very near the one hour residence time used in one of the runs of the study by Requa, et al (60). That study used 4-inch diameter columns and the denitrification influent contained 20 mg/l of nitrate vs an average nitrate content of 13.6 mg/l in the El Lago influent. The effluent in the Requa study contained 0.1 mg/l of nitrate for about 98% removal. The El Lago study produced effluent containing 0.9 mg/l of nitrate to achieve 93% removal. The full scale El Lago reactor compared very favorably to the laboratory column, especially when note was taken of the 10% overdose of methanol by Requa and the 17% underdose at El Lago.

Design parameters were considered to very satisfactory for a plant subject to significant variations in flow rates. As in the case

of the granular media reactor, improved performance would have been expected if a methanol feed system controlled by an automatic nitrate analyzer could have been installed.

Comparison of Granular Media and Plastic Media Denitrification Reactors

The plastic media reactor achieved 93% removal of nitrate with a substantial underdose of methanol as compared to only 83% removal of nitrate with a slight overdose of methanol in the granular media reactor. The plastic media reactor required a negligible amount of water for surging to remove solids and the granular media reactor used 2% of throughput volume for backwash. In addition to these apparent advantages, the plastic media reactor required less operator time since the daily backwash cycle was not required.

CHAPTER V

CONCLUSIONS

The operation of a conventional domestic wastewater treatment plant modified to incorporate a suspended growth nitrification reactor and the first full-scale attached growth denitrification reactors ever put into service provided a unique opportunity to study biological nitrification and denitrification phenomena under actual treatment plant conditions. Nitrification data from 33 sample periods in July and August 1973, as well as data taken on granular media and plastic media denitrification reactors over 33 and 55-day periods, respectively, provided the basis for a number of conclusions.

The El Lago scheme for removing dissolved organic pollutants and nitrifying nitrogenous wastes by a combination of aerobic trickling filters, suspended growth nitrification reactor and intermediate clarifiers with high recirculation was capable of providing wastewater treatment to meet expected effluent standards of 5 mg/l, BOD_5 and 5 mg/l ammonia nitrogen.

Operational procedures were developed to provide satisfactory performance of nitrification and denitrification reactors over a broad spectrum of hydraulic and nitrogen loadings. Detailed operating procedures were highly dependent upon characteristics of applied wastewater, variations in flow, type of media and sophistication of flow control equipment.

A suspended growth nitrification reactor provided a practical means of achieving almost total nitrification of ammonia present in trickling filter underdrain.

Attached growth denitrification reactors utilizing either

granular media or plastic media provided a feasible means of removing nitrate nitrogen from domestic wastewater.

Denitrification reactors packed with plastic media having a high specific surface area and porosity provided better denitrification in a simple system with limited methanol dose rate control than reactors utilizing fine granular media. Plastic media reactors require less operator attention and time due to longer runs between surging or backwashing to remove solids.

The performance of the full-scale nitrification and denitrification facilities verified previous laboratory studies and the design criteria used in this facility.

CHAPTER VI

RECOMMENDATIONS

As a result of operational experience and evaluation of performance data on a suspended growth nitrification reactor and on two types of attached growth denitrification reactors, a number of recommendations were formulated.

Controls capable of providing a constant, manually selected methanol dose rate are essential to efficient denitrification and should be installed. Additional studies should then be conducted to optimize denitrification, conserve methanol and minimize COD and BOD₅ in the plant effluent.

Additional studies should be conducted to delineate the role of the filtration phenomenon in removal of suspended solids by the granular media denitrification reactor.

Long term studies should be conducted to establish the influence of temperature and influent concentration gradients on each unit process.

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APPENDIX A

LABORATORY PROCEDURES

LABORATORY PROCEDURES
ANALYTICAL METHODS

All references are to Standard Methods for the Examination of Water and Wastewater, 13th Edition, unless otherwise indicated.

pH Value - Glass Electrode Method, Sec. 144A.

Phosphorus, Total (TP) - Perchloric Acid Digestion and Vanadomolybdophosphoric Acid Colorimetric Method, Sec. 223.

Phosphorus, Soluble (SP) - Filtration through 0.45- μ membrane filter and Vanadomolybdophosphoric Acid Colorimetric Method, Sec. 223.

Suspended Solids (SS) - Total Suspended Matter (Nonfiltrable Residue), Sec. 135.

Ammonia Nitrogen ($\text{NH}_3\text{-N}$) - Nesslerization Method, Sec. 132 and 212; and Titrimetric Method, Sec. 135.

Total Kjeldahl Nitrogen (TKN) - Nitrogen (Organic) Determination, Sec. 135 and 216.

Nitrate Nitrogen ($\text{NO}_3\text{-N}$) - Baucine Method, Sec. 213, and EPA Methods for Chemical Analysis of Water and Wastes, 1971, pp. 170-174.

Nitrite Nitrogen ($\text{NO}_2\text{-N}$) - Diazotization Method, Sec. 134.

Biochemical Oxygen Demand, 5-Day (BOD_5) - Oxygen Demand (Biochemical) with dissolved determination by Modified Winkler Method, Sec. 218 and 219.

Chemical Oxygen Demand (COD) - Oxygen Demand (Chemical), Sec. 220.

APPENDIX B

TEST DATA

DATA
FROM
GRANULAR MEDIA DENITRIFICATION REACTOR
EXPERIMENT

All values are in mg/l except as otherwise indicated.

RAW WASTEWATER

Date	Total Flow (mdg)	Total P	Sol. P	Sus. Solids	NH ₃ -N	TKN	BOD ₅	COD	pH (units)
June 4	.160	8.2	7.7	21	35.2	60.1		338	8.13
5	.265	17.5	17.0	135	21.6	37.7		391	7.85
6	.364								
7	.246								
8	.243								
9	.240								
10	.247								
11	.240								
12	1.000								
13	.780	3.7	2.4	94	2.4	7.7		89	7.75
14	.560								
15	.424								
16	.310								
17	.240								
18	.220								
19	.284								
20	.256								
21	.248								
22	.238								
23	.242								
24	.240								
25	.202								
26	.251								
27	.266								
28	.268								
29	.234	21.8	14.0	200	15.4	64.7	175	372	7.88
30	.257								
July 1	.250								
2	.270								
3	.280								
4	.250								
5	.263								
6	.294								

PRIMARY INFLUENT

Date	Total P	Sol. P	Sus. Solids	NH ₃ -N	TKN	BOD ₅	COD ₅	pH (units)
June 4								
5								
6								
7								
8	4.8	.9	251				478	7.70
9								
10								
11	8.5	1.9	98				244	7.45
12								
13								
14								
15								
16								
17								
18								
19	22.8	1.6	754				713	7.35
20	16.0	3.4	256	16.2	40.0	223	396	7.70
21	13.0	3.5	344	24.6	39.3		436	7.15
22	20.3	12.4	170	16.9	29.3		720	7.23
23								
24								
25								
26								
27								
28								
29								
30								
July 1								
2	13.5	2.7	295	26.2	44.7		520	7.75
3	20.9	8.2	179	21.6	30.8		389	7.38
4								
5	18.9	5.9	404	20.8	49.3	220	714	7.50
6	15.0	6.0	136	15.4	37.0		276	7.52

PRIMARY EFFLUENT

Date	Total P	Sol. P	Sus. Solids	NH ₃ -N	TKN	COD	pH (units)
June 4							
5							
6							
7							
8	5.8	1.9	61			173	7.13
9							
10							
11	6.9	3.2	53			141	7.45
12							
13							
14							
15							
16							
17							
18							
19	5.1	3.4	37			101	7.30
20	7.5	5.4	63			159	7.60
21	7.1	3.9	71	21.6	30.0	178	7.50
22	12.4	6.9	114	23.9	30.8	225	7.28
23							
24							
25							
26							
27							
28							
29							
30							
July 1							
2	6.0	1.6	41	23.9	29.3	152	7.40
3	15.6	5.3	81	16.2		219	7.50
4							
5	7.2	3.2	88	23.1	31.6	219	7.45
6	10.4	5.9	113	20.0	29.3	240	7.45

NITRIFIED EFFLUENT										
Date	Total P	Sol. P	Sus. Solids	NH ₃ -N	TKN	NO ₃ -N	BOD ₅	COD	pH (units)	Methanol Dose
June 4	6.7	3.9	8	.4	5.0	15.8		96	8.28	56
5	5.4	3.9	45	.6	2.3	19.9		51	8.28	56
6	4.3	2.1	39	1.2	5.4	11.0		77	8.12	76
7			44	.6	10.8	15.4		77	8.18	66
8			23	.8	5.4	11.6			8.40	46
9										
10										64
11			47	.6	2.9	12.3		163	8.23	66
12			34	.4	5.6	12.0				
13										
14										
15			30	.7	5.6	10.0			8.10	33
16										
17										
18			26	1.2	2.8	6.6	58	117	7.90	27
19			31	1.8	2.6	5.4		124	8.00	20
20			36	1.4	2.0	12.3		89	8.00	30
21			35	.7	3.0	15.0		115	7.85	48
22			37	.8	2.0	22.0		66	7.88	35
23										
24										
25	16.1		38	.9	4.5	14.1		151	7.85	56
26			39	.8	3.0	18.5		111	8.20	81
27			51	.5	4.6	19.4		116	7.85	34
28			57	.5	3.6	20.2		204	7.80	34
29			51	.9	1.9	20.2	72	148	7.88	36
30										40
July 1										41
2			41	.9	2.0	21.1		224	7.95	41
3			23	2.2	.8	15.8		123	7.85	58
4										
5			48	1.2	1.9	24.8		104	7.90	51
6	4.0	3.6	30	.9	2.6	11.4		156	7.90	53

DENITRIFIED EFFLUENT

Date	Total P	Sol. P	Sus. Solids	NH ₃ -N	TKN	NO ₃ -N	BOD ₅	COD	pH (units)	Flow (mgd)
June 4										.160
5										.265
6	4.1	2.1	45	.7	6.2	3.5		142	7.85	.364
7			19	1.8	2.3	.0		176	8.15	.221
8			10	.9	1.5	.1			8.10	.242
9										.238
10										.245
11			5	.6	1.8	.1		85	8.30	.170
12										.420
13										
14										
15			26	.9	2.7	.1		78	7.75	.420
16										.308
17										.238
18			3	.6	2.0	.0	6	39	8.05	.217
19			7	1.1	1.8	1.5		16	8.05	.283
20			3	.8	2.0	3.9		35	8.10	.255
21			21	.5	2.8	9.7		75	7.95	.246
22			2	.5	1.4	9.7		23	7.90	.236
23										.240
24										.238
25			7	.9	3.0	.9		43	8.10	.200
26			9	.5	2.0	5.3		43	8.25	.250
27			25	.5	3.5	.9		104	7.95	.265
28			56	.7	2.3	.9		60	8.10	.267
29			14	.7	2.0	.9	12	48	8.20	.233
30										.257
July 1										.247
2			14	.9	2.8	.9		64	8.07	.265
3			11	1.4	0.9	4.4		100	8.03	.272
4										.245
5			19	.8	1.4	1.8		46	8.03	.256
6	5.4	5.0	35	.9	2.6	4.4		132	7.80	.288

FINAL EFFLUENT

Date	Total P	Sol. P	Sus. Solids	NH ₃ -N	TKN	NO ₃ -N	BOD ₅	COD	pH (units)
June 4			3.2						7.77
5			1.2						8.03
6			5.6						8.00
7	4.2	4.0	1.2						8.18
8	8.7	8.5	2.4					90	8.10
9									
10									
11	9.5	9.0	1.6						8.13
12				.4	2.2	1.5		54	8.20
13	5.2	3.1		.7	6.2	5.4	5	70	7.75
14									
15	1.5	1.0	3.8					47	7.95
16									
17									
18									
19									
20	5.2	4.9	2.0				5	36	7.60
21									
22									
23									
24									
25	5.9	5.5	1.6					43	7.55
26									
27									
28									
29	11.5	11.0	6.2	.7	1.5	.0	10	36	8.10
30									
July 1									
2	7.3	6.8	1.2					40	7.95
3	4.1	3.9	1.2					46	7.75
4									
5	10.8	4.2	4.4				18	38	7.60
6	4.7	3.6	4.4					60	7.68

DATA
FROM
PLASTIC MEDIA DENITRIFICATION REACTOR
EXPERIMENT

All values are in mg/l except as otherwise indicated.

RAW WASTEWATER

Date	Total Flow (mgd)	Total P	Sol. P	Sus. Solids	NH ₃ -N	TKN	BOD ₅	COD	pH (units)
July 8	.490								
9	.292								
10	.308								
11	.272								
12	.262								
13	.365								
14	.260								
15	.241								
16	.276								
17	.171								
18	.261								
19	.280								
20	.273	16.5	15.9	102	24.6	37.0	178	293	7.67
21	.317								
22	.254								
23	.266	9.5	9.6	116	19.3	30.8	109	166	7.85
24	.310			196	21.8	30.8	119	229	7.82
25	.311	14.5	11.0	91	15.4	23.9	142	307	7.68
26	.276	11.2	8.8	126	16.9	33.9		303	7.82
27	.363	11.3	10.5	45	15.4	27.0		252	7.65
28	.279								
29	.250								
30	.243	12.2	7.0	159	17.7	33.1	116	241	7.83
31	.270	18.5	15.9	114	18.5	27.7	146	318	7.70
Aug. 1	.256	11.3	9.0	116	18.5	30.0	130	248	7.62
2	.900	6.2	3.3	96	4.6	13.9		136	7.42
3	.347	7.4	6.6	82	8.5	19.3		188	7.80
4	.278								
5	.272								
6	.269								
7	.278	14.4	13.2	219	15.4	33.9		380	7.80
8	.381	16.0	10.0	89	13.9	26.2	60	265	7.66
9	.367								
10	.539								
11	.336								
12	.300								
13	.305	7.5	4.3	180	17.7	33.9		182	7.60
14	.308	15.5	8.0						
15	.277								
16	.259			64	16.2	30.8	88	178	7.95
17	.266	13.0	12.3	43	3.9	27.7		190	8.05
18	.325								
19	.326								
20	.333	6.3	5.8	72	21.6	32.3		185	7.70
21	.320	11.4	10.0	85	16.9	34.7		261	7.96
22	.302	11.5	10.5	87	20.0	25.4	260	296	7.43
23	.282	14.0	12.5	62	3.9	23.9		217	7.81
24	.296	11.3	9.6	59	4.6	28.5		183	7.76
25	.288								
26	.345								
27	.235	13.5	13.2	94	16.9	34.7		307	7.82
28	.311	16.2	14.6	86	22.3	40.0		261	7.83
29	.451	7.6	6.7	103	13.7	27.7		269	7.95
30	.310	11.5	11.0	97	15.4	31.6	230	269	8.05
31	.280	18.1	17.4	68	20.8	32.3		321	7.72

PRIMARY INFLUENT

Date	Total P	Soluble P	Sus. Solids	NH ₃ -N	TKN	BOD ₅	COD	pH (units)
July 8								
9	9.0	.6	104	10.7	21.6		111	7.28
10	10.3	1.0	234	8.5	33.9		250	7.55
11	12.0	.5	262	15.0	24.6	111	337	7.10
12	16.2	5.8	234	8.5	30.8		403	7.35
13	14.9	.5	263	9.2	26.2		422	7.10
14								
15								
16	14.5	2.5	220	29.3	30.8		349	7.30
17	18.5	7.5	230	11.6	27.7		396	7.23
18	14.0	3.5	188	16.9	35.4	134	384	7.65
19	18.5	.9	338	13.1	38.5	188	384	6.95
20	14.9	.5	128	16.2	30.8		428	7.28
21								
22								
23	6.5	3.6	456	17.7	26.2	89	214	
24	13.3	2.4	192	27.0	42.4	177	314	7.35
25	18.0	1.1	202	15.2	23.9	169	284	7.40
26	11.0	3.3	129	17.7	31.6		296	7.40
27	15.5	1.9	253	13.9	37.7		140	7.22
28								
29								
30	9.0	2.0	120	13.3	33.9	86	186	7.50
31	22.1	8.4	227	17.7	33.2	206	229	7.50
Aug. 1	12.8	1.0	139	21.8	35.4	163	388	7.02
2	6.6	.8	411	7.7	19.3		407	6.73
3	6.8	.8	247	11.6	23.9		236	7.56
4								
5								
6	10.0	9.6	143	24.6	40.8		281	7.65
7	11.1	1.5	239	15.4	30.0		307	7.40
8	11.5	1.0	277	16.2	30.8	164	341	6.92
9								
10								
11								
12								
13	16.0	1.5	262	18.5	35.4		407	7.30
14	19.0	4.5						
15								
16			426	18.5	38.5	139	352	7.68
17	9.0	4.5	168	3.1	27.7		253	7.40
18								
19								
20	10.5	2.5	334	20.0	37.7		332	7.43
21	18.5	7.6	182	16.9	33.8		376	7.38
22	14.5	5.0	214	13.7	31.6	243	365	7.23
23	15.0	3.7	232	6.9	28.5		334	7.55
24	8.8	4.5	109	8.5	29.2		186	7.41
25								
26								
27	13.5	3.5	118	17.7	46.2		396	7.45
28	14.5	5.5	198					7.88
29	11.5	1.3	362					7.52
30	12.5	3.0	278					7.35
31	12.8	5.1	182					7.20

Date	PRIMARY EFFLUENT							pH (units)
	Total P	Soluble P	Sus. Solids	NH ₃ -N	TKN	BOD ₅	COD	
July 8								
9	2.5	.7	36	12.3	19.3		111	7.30
10	1.0	.5	17	7.7	21.6		99	6.85
11	2.0	.5	22	16.9	30.0	47	107	6.95
12	9.2	5.8	75	12.1	30.0		157	7.40
13	5.9	.6	50	15.4	29.3		119	7.05
14								
15								
16	8.0	1.8	104	16.9	33.9		221	7.00
17	13.0	4.0	99	19.3	32.2		239	7.05
18	5.0	1.5	42	16.9	26.2	86	161	7.30
19	10.4	1.8	36	11.6	27.0	124	181	7.10
20	5.6	1.0	70	13.9	26.2	82	178	7.50
21								
22								
23	6.9	.0	114	16.9	26.2	70	151	7.08
24	11.4	4.6	106	20.0	23.9	79	186	7.60
25	6.0	2.0	56	13.9	23.1	86	160	7.48
26	5.4	2.0	46	16.9	27.7		154	7.40
27	7.0	3.0	69	14.6	26.2		140	7.80
28								
29								
30	5.6	2.0	59	16.9	26.2	84	186	7.60
31	17.4	5.1	107	18.5	32.3	110	229	7.60
Aug. 1	4.6	1.7	100	19.6	28.5	101	186	6.97
2	3.1	.7	62	10.8	16.2		97	7.10
3	3.3	1.2	47	3.9	19.3		108	7.35
4								
5								
6	7.0	6.5	71	19.3	33.2		243	7.60
7	6.6	2.8	77	16.9	27.0		190	7.10
8	2.7	.5	38	16.9	27.0	55	126	7.03
9								
10								
11								
12								
13	6.0	2.8	64	7.8	28.5		116	7.25
14	6.5	3.3						
15								
16			45	16.9	35.4	86	158	8.00
17	10.0	2.8	47	6.2	28.5		329	7.42
18								
19								
20	3.5	1.0	37	14.6	23.9		150	7.48
21	6.7	4.3	63	15.4	26.2		157	7.42
22	5.8	3.6	44	13.1	25.4	122	188	7.60
23	5.2	2.6	48	10.8	26.2		171	7.28
24	7.2	3.6	51	10.8	25.4		132	7.52
25								
26								
27	5.7	4.0	53	14.6	31.6		211	7.50
28	7.0	3.5	61	11.6	23.9		166	7.30
29	4.6	1.3	64	13.7	24.6		154	7.48
30	4.6	1.6	136	13.7	24.6		151	7.25
31	5.8	3.3	57	7.7	25.4		189	7.28

NITRIFIED EFFLUENT								Methanol Dose
Date	Sus. Solids	NH ₃ -N	TKN	NO ₃ -N	BOD ₅	COD	pH (units)	
July 8								29
9	87	.5	2.3	16.7		107	7.90	28
10	27	.6	0.8	18.5		131	7.75	35
11	50	1.0	2.3	17.6	29	99	7.65	22
12	44	.7	1.8	8.3		97	7.80	26
13	42	.4	1.4	23.8		108	7.65	28
14								47
15								39
16		.5	3.2	21.1		169	7.65	69
17	60	.5	2.0	22.9		81	7.90	44
18	67	.5	1.6	22.9	64	123	7.78	
19	90	.7	3.6	17.6	66	119	7.72	
20	50	.5	1.5	14.2	27	83	7.72	32
21								36
22								23
23	10	.7	1.2	18.5	57	99	7.85	54
24	34	.7	1.4	11.4	52	109	8.00	37
25	33	1.4	2.4	11.4	49	123	7.74	37
26	43	2.3	4.7	10.3		188	8.00	53
27	27	1.2	5.1	10.0		124	7.58	63
28								0
29								0
30	27	.5	1.5	13.2	11	50	7.96	0
31	38	.3	1.9	13.3	47	93	8.10	49
Aug. 1	44	.3	1.2	14.4	42	89	7.81	20
2	33	1.6	3.8	8.6		78	7.68	19
3	28	1.5	1.8	10.6		100	7.82	34
4								16
5								27
6	28	.9	3.2	10.8		95	7.82	38
7	35	.5	2.7	8.6		78	7.88	16
8	39	.4	2.3	5.9	54	107	7.58	34
9	34	.0	1.4	11.2		112	7.78	43
10	35	.0	7.6	10.0		130	7.78	27
11								30
12								42
13	60	.9	4.2	13.2		105	7.48	45
14								41
15								29
16	84	.9	3.6	10.6	27	75	8.25	46
17	46	1.2	3.4	8.9		63	7.70	26
18								33
19								33
20	2	1.1		11.9		207	7.88	21
21	43	1.2	2.8	13.2		134	7.88	24
22	42	1.2	2.3	10.6	57	96	7.90	24
23	7	1.4	2.6	10.6		74	7.72	18
24	56	1.6	2.4	18.0		116	7.75	30
25								41
26								26
27	50	1.6	3.8	12.6		115	7.98	41
28	37	1.1	2.2	14.7		127	7.72	40
29	55	.7	3.0	12.1		95	7.84	36
30	53	.7	2.8	13.3	27	83	7.65	40
31	39	.7	1.5	13.8		87	7.83	34

DENITRIFIED EFFLUENT

Date	Sus. Solids	NH ₃ -N	TKN	NO ₃ -N	BOD ₅	COD	pH (units)	Flow (mgd)
July 8								.490
9	25	.4	1.9	.9		103	7.72	.292
10	41	.5	2.2	.9		59	7.85	.308
11	5	.8	1.6	.0	3	44	8.00	.273
12	2	.8	1.4	2.5		35	7.85	.262
13	8	.1	.7	.0		35	8.00	.365
14								.260
15								.241
16	8	.5	1.4	.0		47	7.90	.276
17	6	.8	1.6	.9		31	7.95	.171
18	11	.8	1.6	1.8	24	54	7.88	.261
19	62	.8	3.6	.9	34	85	7.80	.280
20	60	.8	2.7	1.8	38	48	7.80	.273
21								.317
22								.254
23	22	1.6	6.1	.1	27	36	8.00	.266
24	10	1.1	1.6	.7	5	31	8.14	.310
25	7	1.5	3.1	.1	9	54	7.92	.311
26	10	3.0	4.1	.1		65	8.02	.276
27	27	1.2	3.2	.1		54	7.75	.363
28								.279
29								.250
30	3	2.0	3.1	2.5	4.3	58	7.98	.243
31	7	.4	1.8	3.0	4.5	27	8.00	.270
Aug. 1	13	.5	2.0	1.4	5.6	43	7.92	.256
2	45	1.6	4.3	2.1		62	7.68	.632
3	2	.3	1.6	.6		36	7.80	.347
4								.278
5								.272
6	6	.9	2.4	.3		32	7.90	.269
7	7	1.4	2.2	.1		47	7.85	.278
8	5	.9	1.6	.3	16.4	24	7.80	.381
9	3	.0	.5	.1		39	7.60	.367
10	6	.1	2.0	2.5		40	7.72	.539
11								.336
12								.300
13	71	.8	4.5	.3		124	7.90	.305
14								.308
15								.277
16	54	.8	3.4	2.2	14.1	63	7.68	.259
17	3	1.1	2.3	1.5		32	7.85	.266
18								.325
19								.326
20	22	1.4	3.1	.4		46	7.90	.333
21	8	1.5	2.7	.4		31	7.80	.320
22	8	1.6	2.7	.4	8.6	58	7.80	.302
23	5	2.0	3.2	2.0		23	7.80	.282
24	37	1.9	3.2	.7		50	7.69	.296
25								.288
26								.345
27	35	1.9	4.2	.1		73	7.95	.235
28	25	.9	2.6	.1		182	8.00	.310
29	22	1.2	2.2	.7		47	7.90	.451
30	9	.8	2.0	.2	9.2	40	8.00	.310
31	13	.9	2.3	.4		31	7.80	.280

FINAL EFFLUENT									
Date	Total P	Sol. P	Sus. Solids	NH ₃ -N	TKN	NO ₃ -N	BOD ₅	COD	pH (units)
July 8									
9			10.8					44	7.68
10									
11	2.0	1.9	.8	.4	1.2	3.5	2.5	32	7.40
12									
13									
14									
15									
16									
17									
18	2.7	2.0	4.5	.8	1.2	.9	16.9	46	7.70
19	2.1	1.8	6.0	.7	1.4	.0	16.7	61	7.70
20	1.5	1.3	24.0	.5	1.1	.0	19.6	36	7.65
21									
22									
23	1.5	.5	4.0	.8	.9	.0	4.5	24	7.58
24	3.4	2.9	8.0	.5	1.5	.0	3.4	31	7.89
25	4.6	3.4	4.0	1.8	1.9	.0	6.9	38	7.95
26	4.1	3.4	13.0	1.5	3.5	.0		58	8.00
27	8.2	6.2	5.5	1.2	2.7	.0		39	7.57
28									
29									
30	3.2	3.0	.4	1.8	2.8	2.5	8.5	31	7.98
31	3.5	3.4	2.5	.4	1.6	2.5	2.4	23	7.90
Aug. 1	3.9	3.7	4.5	.5	1.8	1.3	4.5	31	7.98
2	2.2	2.1	3.0	1.8	3.0	1.8		35	7.60
3	2.6	1.1	.8	.4	1.6	.3		32	7.90
4									
5									
6	5.6	5.2	.8	1.1	2.3	.2		32	8.00
7	5.1	4.8	.4	.9	1.9	.1		43	7.90
8	3.5	2.3	1.5	.8	1.5	.0	.8	20	7.88
9			1.6	.5	1.1	.0		34	7.90
10	1.4	1.1	.8	.0	.9	1.9		36	6.92
11									
12									
13	1.9	1.8	3.6	.7	1.2	.1		35	7.52
14	1.6	.8							
15									
16			6.0	.9	1.5	.1	13.8	55	7.80
17	1.0	1.0	2.5	.9	2.0	1.2		24	7.68
18									
19									
20	3.1	3.0	2.0	.7	1.5	.0		31	7.60
21	2.8	2.7	2.0	.8	1.9	.1		38	7.75
22	2.3	2.3	3.0	.9	1.5	.1	7.1	42	7.68
23	1.0	.8	7.5	1.2	2.2	1.3		27	7.65
24	1.7	1.5	1.2	1.6	2.3	1.0		47	7.55
25									
26									
27	2.6	2.5	3.5	.9	2.4	.0		38	7.78
28	2.5	2.1	6.0	.4	1.1	.0		40	7.85
29	1.2	1.2	4.0	.4	1.2	.1		63	8.00
30	1.6	1.3	2.5	.3	1.1	.0	1.6	36	7.62
31	1.7	1.3	8.0	.4	1.5	.1		39	7.50