Removal of Organic Matter and Nutrients in Sewage Using Sequencing Batch Biofiltration Reactors

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Removal of Organic Matter and Nutrients in Sewage Using Sequencing Batch Biofiltration Reactors

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Removal of Organic Matter and Nutrients in Sewage Using Sequencing Batch Biofiltration Reactors

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Abstract

Conventional biological treatment process has been developed and used as an economical and effective method for treating municipal wastewaters, especially in sewage treatment by using sequencing batch biofiltration systems.

A laboratory-scale process was designed to investigate the effects of changing anoxic/aerobic time and C/N ratios on the performance of Sequencing Batch Biofiltration Reactors (SBBRs) for removal of organic matter and nutrients in sewage. Indices of performance, COD, ammonium nitrogen, total nitrogen, and phosphorus removals were considered for specific analysis.

Two laboratory-scale SBBRs, each with a 12.77 liter the effective liquid volume, and packing with 40% aquacell media, were designed for operating for four 6 hour cycles per day. The daily operating cycle included two sub-cycles of 2.9 hours each, each one alternating anoxic and aerobic conditions. Effluent was withdrawn at the end of the two sub-cycles. Also, five process modes with the anoxic/aerobic time ratios in succession of 0.1, 0.35, 0.5, 0.75 and 1 were designed. Once effective biological nitrogen removal had been acquired, C/N ratios of 3, 5, 7 and

10 were investigated. The total influent nitrogen concentration was maintained constant while the COD concentration was varied to obtain the desired C/N ratio.

The anoxic/aerobic time ratio of the process and the C/N ratio of the influent wastewater were observed to have a substantial effect upon the performance of the SBBR system. From this research, the amount of nitrogen removed by the system increased as the C/N ratio increased. It was also shown that the effluent quality increased when the anoxic/aerobic time ratio was not high.

Finally, it was shown that Mode III operating at an anoxic/aerobic time ratio of 0.5 and a C/N ratio of 10, provided the best performance in the reductions of COD, T-N, and T-P. The treatment was able to achieve 98% removal of COD, up to 97% removal of T-N, 98% removal of NH_4^+-N and 67% removal of T-P. The laboratory-scale treatment study especially demonstrated that SBBRs are capable of achieving high nitrogen removal on wastewater with high C/N and low anoxic/aerobic time ratios. The very small anoxic/aerobic time ratio, however, did not yield greater reductions in COD, T-N, or T-P as compared to higher ratios.

1

Introduction

Conventional biological treatment processes have been developed and is being widely adopted in environmental plants, especially in municipal wastewater treatment, due to its economical and effective manner. However, Randall et al., (1992) have pointed out the problem for the biological treatment process. That is, if one kg of phosphorus is assimilated by algae, 111 kg of new biomass are formed, which is equivalent to 138 kg of COD. One kg of nitrogen could produce 16 kilograms of algae biomass, which would be equivalent to 20 kg of COD. Therefore, small amounts of nutrients can stimulate excessive production of COD in the form of algae growth, leading to eutrophication, loss of oxygen resources, changes in aquatic population and subsequent deterioration of water quality. The benefits for utilizing biological nutrient removal processes have been discussed in great detail by many researchers (McClintock, 1990; Randall et al., 1992; Wable, 1992). Important benefits can be summarized as follows:

- Reduction or elimination of chemicals addition
- Reduction in oxygen requirements during treatment
- Reduction in sludge production
- Recovery of alkalinity by the denitrification process
- Reduction in filamentous growth and improvement in sludge settleability.

Bioreactors consist of two basic physical systems: suspended growth and fixed film. In suspended growth reactors, the microorganisms are suspended as microbial aggregates in the liquid. And in fixed-film reactors, the biomass growth occurs on or within a solid medium that acts as a support.

Fixed-biofilm reactors can be designed as oxic, anoxic, or anaerobic processes. Aerobic systems treating high organic concentrations are frequently limited in their degradation rate because of oxygen transfer. Such systems cause the formation of heavy biomass buildup that leads to a significant oxygen demand. Oxygen transfer can be limited by both a gas phase to liquid phase and a liquid phase to biomass phase. The gas-to-liquid transfer is influenced by bubble size and dissolution pressure. The liquid-to-biomass is transfer influenced by mixing intensity and the biomass structure. Fixed-biofilm treatment systems have a long history of application to wastewater and include the processes using ***fixed-biofilm media*** that were widely employed for municipal wastewater.

At present, the newly developed technology depends on biological film at substratum of supporting medium inside the reactor in order to increase the retention time of microbes, which will lead to a reduction of the waste sludge and an increase in treatment efficiency.

Even though, the biological nutrient removal process has proven to be the most economical means of removing nitrogen and phosphorus from wastewater, the biological nutrient removal process is more complicated and sensitive to operation than conventional activated sludge. In other words, there are many factors that affect the performance, e.g., influent wastewater characteristics, temperature, nitrate, and dissolved oxygen concentrations in the return activated sludge.

Wastewater characteristics has been thoroughly investigated and found to have a strong effect on the performance of biological wastewater treatment processes (Marais et al., 1983). Randall et al., (1992) stated that effluent phosphorus or nitrogen concentrations could be reasonably predicted if influent wastewater had been sufficiently characterized. Janssen (1994) stated that suitable composition of wastewater, i.e., BOD/P and BOD/N ratios, is one of the prerequisites for biological nutrient removal processes.

It is challenge to remove chemical oxygen demand (COD) from many industrial treatment systems with highly concentrated wastewaters prior to discharge into receiving waters or municipal treatment plants. As more information becomes available on the ecological impacts of wastewater discharge, permit limitations are becoming more stringent.

Among the most significant pollutants in municipal or industrial wastewaters are the nutrients nitrogen and phosphorous. Nitrogen can be released into the environment from agricultural areas in three main ways. Nitrogen removal systems have been used in municipal and industrial wastewater treatment systems for a number of years. Most of these systems are large, with several reactors required for the treatment. Randall et al., (1992) introduced the concept of C/N ratio as a factor that influences the performance of biological nutrient removal processes. It should be noted that design of such a biological nitrogen removal system would be specific to particular wastewaters because of the high variability in the parameters that are involved in developing effluent guidelines for the reactor. One of these parameters includes C/N ratio of wastewater. Because the ratio of readily available COD to total nitrogen entering the anoxic zone of the anoxic/aerobic sequencing batch reactor determines whether or not the process is COD limited or nitrogen limited, the effects of this ratio on the system performance needs to be evaluated to determine the significance of the effects. *This information is needed to improve our understanding of wastewater processes using SBBR, and to verify and improve the current design and operating procedures.*

Because of the unavailability of the actual industrial wastewater at the beginning of this study, a synthetic wastewater was prepared after the concentrated wastewater for laboratory use. In composing the synthetic wastewater, substitutions were made for compounds that were unavailable or unknown at the beginning of the research.

There is a little information on treatment methods incorporating COD, phosphorous and nitrogen removal that have been developed specifically for treating wastewater. Therefore, this research effort was undertaken to address the following objectives:

(i) Propose five process modes with the anoxic/aerobic ratio in succession of 0.1, 0.35, 0.5, and 0.75 and determine the suitable anoxic/aerobic ratio for wastewater treatment using Sequencing Batch Biofiltration Reactors (SBBRs) under condition of reduced cycle length to 6 hours

(ii) Use C/N ratios of 3, 5, 7 and 10 for the influent wastewater, and determine an optimum C/N ratio to achieve the most effective conditions for organic matter and nutrients reduction

(iii) Investigate the effects of anoxic/aerobic time and C/N ratios on the performance of SBBRs for the removal of organic matter and nutrients in wastewater. As the indices of the performance, COD, ammonium nitrogen, total nitrogen, and phosphorus removals were considered for specific analysis

(iv) Demonstrate laboratory-scale treatment of the wastewater in SBBRs, as well as to determine the highest efficiency of the proposed process.

2

Literature Review

2.1 Introduction

Characterization of wastewater is an important part of the initial work in the design of a treatment process. Therefore, a short review of sewage characterization studies has been included. In addition, the basics of biological nitrogen removal were discussed and the mechanism of nitrogen removal as well as the strategies used in wastewater treatment was included. Because sequencing batch biofiltration reactor process was used in this research, an in depth review of the SBR was included. Also, a brief synopsis of biofiltration and fixed biofilm reactors was included because of the impact it could have on treatment schemes and removal efficiency of organic matter and nutrients.

2.2 Sewage

It is well known that municipal wastewater constitutes of an important part of the total wastewater from various sources. The amount of pollutants contained in the municipal wastewater has to be treated to meet the standards for direct discharge. This, in fact, may not be the best management strategy for this type of wastewater. Due to dwindling supply and increasing demand for water in the agricultural and industrial sectors, a better alternative to direct discharge of treated municipal wastewater is required to elevate its water quality further to an appropriate level for possible agricultural or industrial reuse. However, relatively little attention has been paid to this aspect.

2.3 Alternative Anoxic-Oxic Process

Alternative anoxic-oxic (ANX) process is a modified method for activated sludge process that provides an anaerobic and aerobic sequence treatment (Fig. 2.1). The process was originally developed for phosphorus removal. However, the ability to provide anaerobic dehalogenation before aerobic mineralization has raised interest among the treatment processes for hazardous chemicals. Adaptation of the ANX process to the treatment of hazardous compounds has resulted in more variations for process operation. The process can be operated as a complete oxic process, as an anoxic process, or as a combined anaerobic-aerobic system. The anoxic process can be operated as a nitrate-reducing, sulfate-reducing, methanogenic metabolism, or sequence combinations. A sequencing operation between oxic and anoxic can be done in the same basin under automatic controls using redox potential and dissolved oxygen levels for operation.

Process configurations that provide for the optimization of multiple metabolic modes are provided in Fig. 2.2. The anaerobic reactor is designed for dehalogenation of more complex aromatic compounds, and the aerobic reactor is designed for mineralization of transformed products. The first reactor also can be used as an anoxic mode for those halogenated compounds that respond to dehalogenation under sulfate-reducing or nitrate-reducing condition.

When all zones are operated as a complete oxic process (Fig. 2.2a), the system will be identical to the typical activated sludge process. Then, the first settling tank is bypassed. This provides a flow regime that is between rapid mix and plug flow. In Fig. 2.2b the first and the second biological contact units are operated in an anoxic mode. Nitrate or sulfate serves as an electron acceptor. The first settling tank can be bypassed or used to recycle the biomass of the first contactor back to the reactor. If the first settling tank is bypassed, then the major microbial species in the biomass will be facultative microorganisms. The strict anaerobes are hindered by the presence of oxygen molecule. The use of settling tank in the flow configuration will depend on the needed of microbial consortium.

In the anaerobic selector mode (Fig. 2.2c), the first bioreactor is operated to support methanogenic metabolism with carbon dioxide as an electron acceptor. Under this mode of operation, settling tank should not be bypassed. It is important to separate the methanogens from the other biomass, since oxygen is toxic to these microorganisms. Exposing this biomass to the aerobic unit will delay the methanogenic response when the sludge is recycled back to the anaerobic unit.

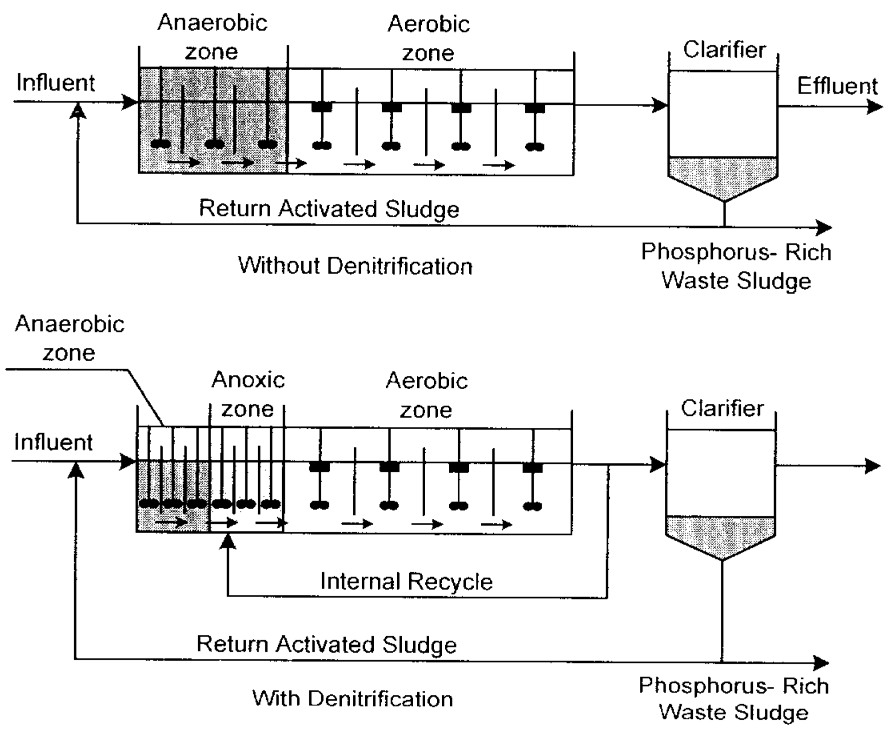
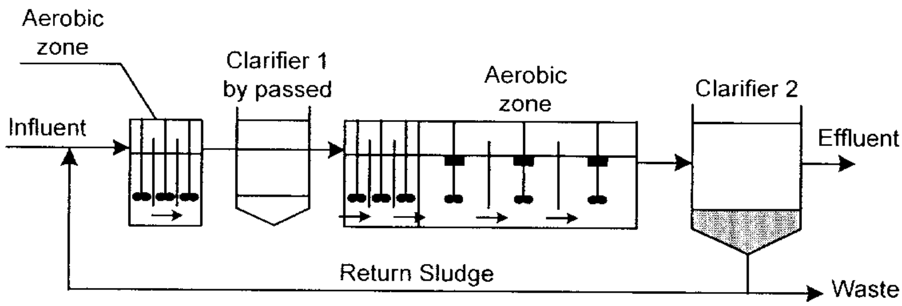
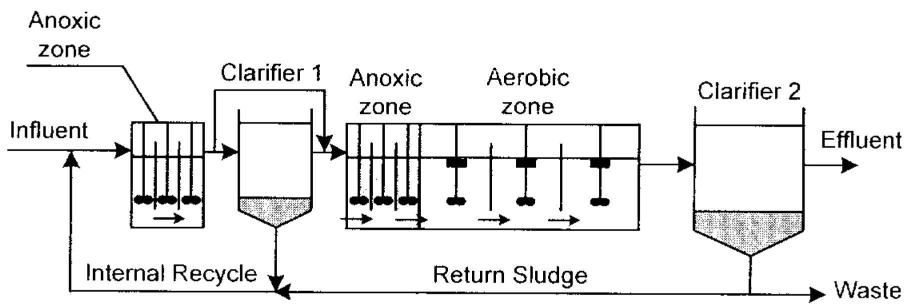


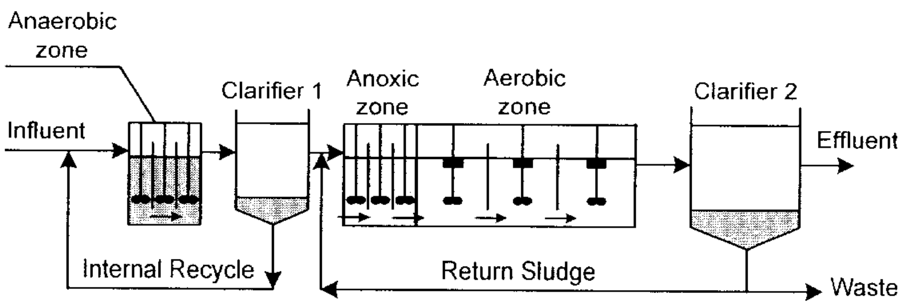
Figure 2-1 Typical arrangements of alternative anoxic-oxic process.



(a) Operated as an Oxic Treatment Facility



(b) Operated as an Anoxic-Oxic Sequence Treatment Facility



(c) Operated as an Anaerobic-Anoxic-Oxic Sequence Treatment Facility

Figure 2-2 Process configurations that provide the flexibility for selecting the most appropriate metabolism modes.

2.4 Sequencing Batch Reactors (SBRs)

The sequencing batch reactor (SBR) is a suspended-growth fill-and-draw system. This process originated in the early 1900s and has recently been applied to as a successful treatment for hazardous compounds. The process was evaluated on landfill leachate that contained hazardous compounds in the early 1980s (Sutton, 1988). Since then, numerous SBR systems have been utilized for the treatment of hazardous waste. The U. S. EPA has recognized this technology under the Innovative and Alternative Technology Assessment (IATA) program.

A sequencing batch reactor consists of a batch reactor that operates under a series of periods that constitutes a cycle. The cycle generally consists of fill, react, settle, decant, and idle periods. The use of these periods allows a single reactor to act as a train of reactors and a clarifier. By manipulating these periods within a single cycle, the SBR can accomplish most of what a continuous flow plant can accomplish with several reactors, each operating under a different condition.

2.4.1 Process Principles

The SBR system consists of a single tank in which timed processes take place sequentially (Fig. 2.3). A complete cycle of unit operations is performed within a single tank by holding the influent waste stream in a batch mode and treating it through a succession of steps.

Irvine and Ketchum (1989) described the SBR and its periods in detail. During the "fill" period, influent wastewater is added to the biomass that was left in the tank from the previous cycle. The length of the "fill" period depends on the number of SBRs, the volume of the SBRs, and the nature of the flow of the wastewater source, which can be intermittent or continuous. The reactor may or may not be mixed during this period. Filling ends when the wastewater reaches the maximum water level or at some fraction of that if multiple fill periods are used during a single cycle.

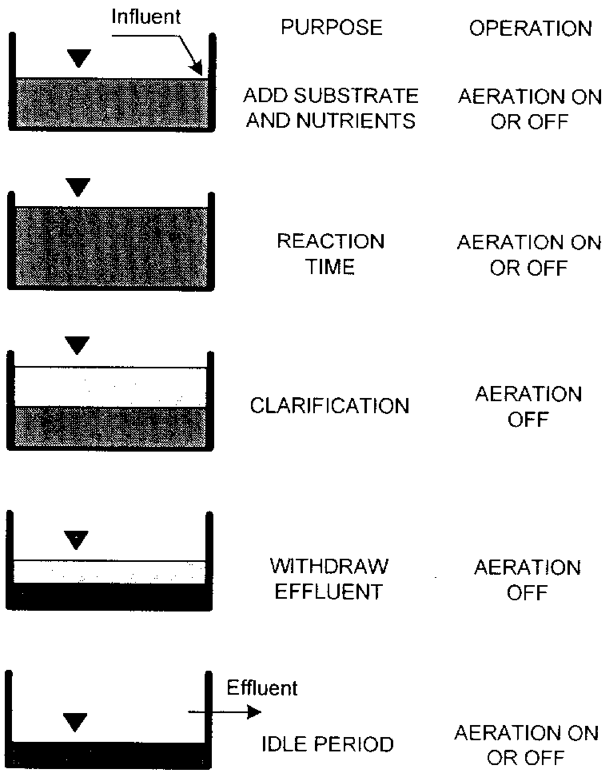


Figure 2-3 Typical sequencing batch reactor operating modes.

The "react" period occurs after the fill period. In most cases the reactor should be mixed during this period. Aeration may or may not be used depending on the reactor's objective and operation. In addition, the react period may be interrupted by fill periods and/or sludge wastage. During the react period, many reactions can take place such as nitrification, denitrification, COD removal, phosphorous removal. After the react period, the "settle" period takes place. During the settle period, the SBR acts as a clarifier. The solids, including biomass and particulate substrate, settle and leave relatively clear effluent on top. The settle period normally lasts between 0.5 and 1.5 hours so that the solids blanket does not float due to the gas buildup. The "decant" occurs at the end of the settle period and is the time when the effluent is drawn off. This may take place with a pipe or a weir.

The decant level should be adjustable so as to make the SBR more adaptable to changes.

Finally, some systems include the "idle period." This period is most necessary when several SBRs are being used with a continuous wastewater source. This allows a small amount of leeway when trying to match the cycles of the SBRs so that one SBR is always on the fill period.

2.4.2 Nitrogen Removal by SBR

2.4.2.1 Biological Transformations of Nitrogen

Three major biological processes directly involved with biological nitrogen removal in wastewater treatment are ammonification, nitrification, and denitrification. Figure 2.4 shows the interaction between the three processes known as Nitrogen Cycle.

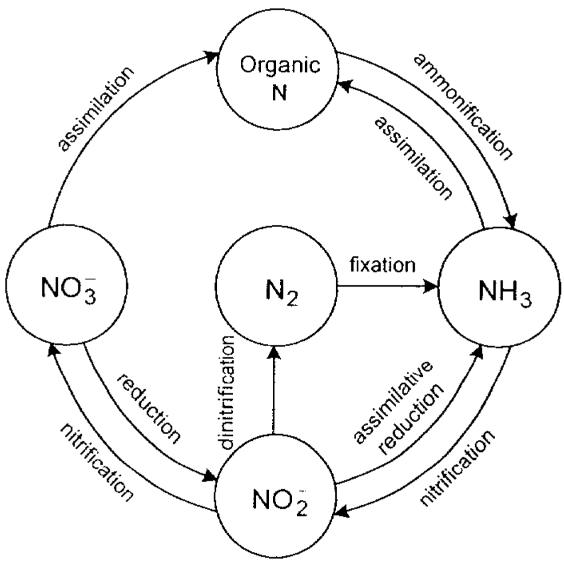


Figure 2-4 Nitrogen cycle in wastewater treatment (Grady, 1999).

Ammonification is defined as the conversion of organic nitrogen to ammonia. Nitrogen as ammonia or nitrate can be assimilated by bacteria to form cellular mass. In most domestic and high strength agricultural wastewaters requiring nitrogen removal, the initial level of nitrogen is high enough that high levels remain even after the bacteria use.

Nitrification is the biological oxidation of ammonium nitrogen and is shown in Figure 2.5. Ammonium nitrogen is oxidized to nitrite by ammonia oxidizing bacteria (AOB) and then to nitrate by nitrite oxidizing bacteria (NOB). Many AOBs and NOBs are autotrophic, although heterotrophic bacteria are known to function as nitrifiers (Painter, 1977). In most situations, very little nitrite exists in a system at any one time because the conversion of ammonium to nitrite by AOBs is generally the rate-limiting step (Antoniou et al., 1990).

$$\underset{\text{ammonium}}{NH_4^+} + O_2 \xrightarrow{AOB} \underset{\text{nitrite}}{NO_2^-} + O_2 \xrightarrow{NOB} \underset{\text{nitrate}}{NO_3^-}$$

Figure 2-5 Nitrification process (adapted from US EPA, 1993).

Denitrification is the key process to remove nitrogen from wastewaters. It occurs when the oxygen concentration in the wastewater becomes low enough that the bacteria begin to utilize nitrate as an electron acceptor under anoxic conditions. Nitrate is reduced by heterotrophic bacteria to the intermediate nitrite and then to nitrogen gas. Nitrogen is then able to leave the wastewater as inert nitrogen gas.

Many types of biological nitrogen removal systems have been developed. Some of the first single sludge treatment processes developed for nitrogen removal were the Modified Ludzack-Ettinger (MLE) (Ludzack and Ettinger, 1962) and the Bardenpho (Barnard, 1975) (Figures 2.6 and 2.7 respectively).

Some of the most recent research on nitrogen removal has involved using a sequencing batch reactor. This type of reactor does not operate continuously as the previously mentioned reactors do, but as a series of operations.

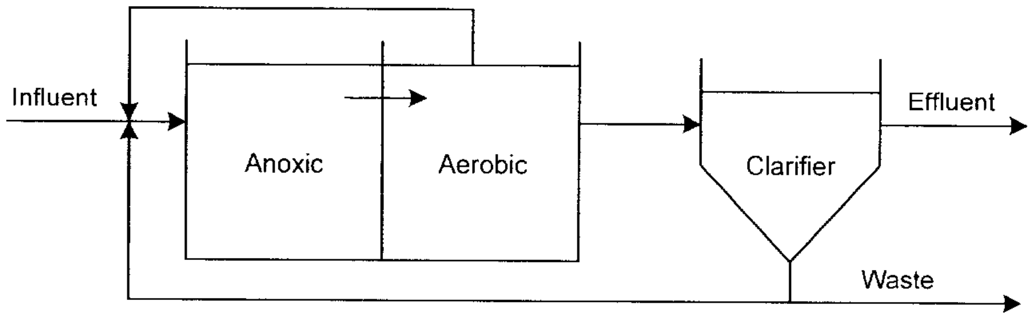


Figure 2-6 Modified Ludzack-Ettinger (MLE) process for biological nitrogen removal.

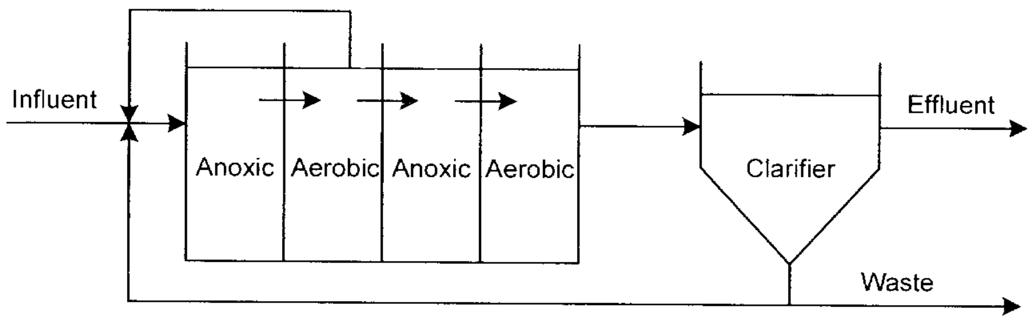


Figure 2-7 Bardenpho process for biological nitrogen removal.

2.4.2.2 Nitrogen Removal by SBR

In general, all SBRs designed with the goal of nitrogen removal have both anoxic and aerobic periods in the cycle. The characteristics that are manipulated in the nitrogen removal SBR are hydraulic retention time (HRT), solids retention time (SRT), anoxic/aerobic ratio, the number of anoxic/aerobic periods, and fill strategy. Much research has been done in the last decade to determine optimum conditions for different wastewaters. Two different scenarios can be used in completing nitrogen removal with SBRs. One scenario is to have two separate nitrification and denitrification phases and the other is to create conditions such that nitrification and denitrification take place under the same macroscopic conditions in the reactor.

Figure 2.8 shows a profile of the major soluble nitrogen species and the soluble chemical oxygen demand (COD) in a reactor, as well as the stages taking place during the cycle. During the fill and first anoxic period, any nitrate/nitrite that is left in the reactor from the previous cycle is denitrified. Once the denitrification is complete, aeration begins. During the aerated react stage, carbon is oxidized and nitrification takes place. After aeration ceases, an anoxic react period begins. During this stage, the oxidized nitrogen species are denitrified by heterotrophs that use endogenous or slowly degradable COD for the carbon and energy source due to the lack of bioavailable soluble COD.

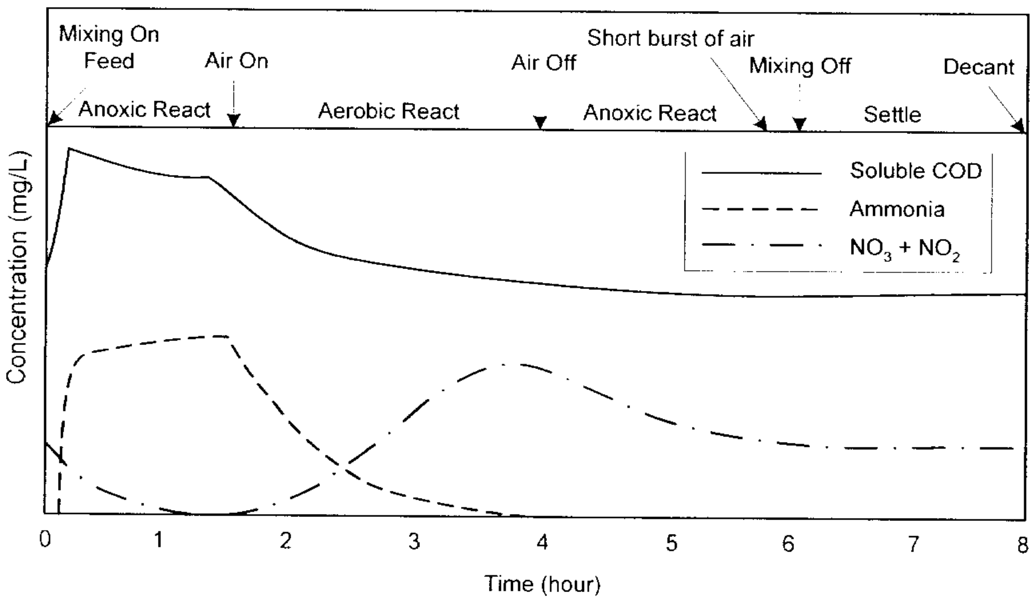


Figure 2-8 Typical SBR cycle incorporating partial nitrogen removal.

2.4.3 Advantages of SBRs

Arora et al., (1985) notes many of the advantages of using the SBR. First, since the SBR is a batch process, the effluent can be held in the reactor until it is treated if there is somewhere for the influent to be stored. This can minimize the deterioration of effluent quality sometimes associated with influent spikes. Also, biomass will not be washed out of a SBR because of flow surges. In addition, simplification of SBRs compared to flow-through activated sludge systems negates the need for return activated sludge to be pumped from the clarifier. Another advantage over conventional systems is that settling occurs when there is no inflow or outflow. Therefore, short-circuiting of the "clarifier" cannot occur.

In addition, the nature of the SBR leaves a lot of room for changes to the system based solely on operation and do not require construction. Edgerton et al. (2000) studied piggery wastewater treatment with SBRs and pointed out that cycle timing can be optimized according to changes in livestock feed cycles. As previously mentioned, the nutrient component in the diet of livestock is a main determining factor in the makeup of agricultural wastewaters.

Therefore, as diets change on a farm, the SBR can be optimized for the changing conditions in the wastewater. Another example of the flexibility of SBR treatment relates to the effect of temperature. Fernandes (1994) demonstrated how temperature affected SBRs treating screened swine manure. The data showed that the temperature of the wastewater could play a major role in the efficiency of treatment. Low temperatures decreased the efficiency of the process. By adjusting cycle times, seasonal temperature effects could be compensated without losing efficiency. Creating a reactor that can nitrify, denitrify, oxidize substrate, and clarify in one vessel saves space and cost and may make wastewater treatment feasible for smaller farms that would have difficulty dealing with a multi-unit treatment train (Edgerton et al., 2000).

2.5 Biofiltration, Fixed Biofilm Reactors and Sequencing Batch Biofilm Reactors (SBBRs)

Bioreactors consist of two basic physical systems: suspended growth and fixed film. With suspended growth reactors, the microorganisms are suspended as microbial aggregates in the liquid. And with fixed-film reactors, the biomass growth occurs on or within a solid medium that acts as a support.

Process design for immobilized biomass is significantly different from that for suspended reactors. The rate of removal is limited by mass transfer and diffusion within the biomass (Fig. 2.9). For fixed-film reactors, removal of contaminants is the result of sorption into the biofilm. The absorbed organic compounds are then degraded. The fixed-biofilm process can be considered as an adsorption system that has a continuous growing and self-generating adsorbent of microbial cells. The organic substrate removal rate is controlled by diffusion through a liquid-film attachment, followed by diffusion into the cellular biomass. Transport through the biomass can be limited as a result of metabolism or diffusion.

The biofilm is developed by recycling a suspension of seed microorganisms from a culture reactor with nutrients and primary substrate at concentrations 2 to 10 times greater than that typically used during actual treatment. Once the biomass is developed to a sufficient level for achieving the desired mean cell residence time, 2 to 5 weeks, then the flow from the culture reactor will stop, and the treatment cycle will be implemented. Even with fixed-biofilm reactors it is wise to maintain growing biomass in the culture reactor to reestablish the biomass during any unexpected biomass loss or population shifts.

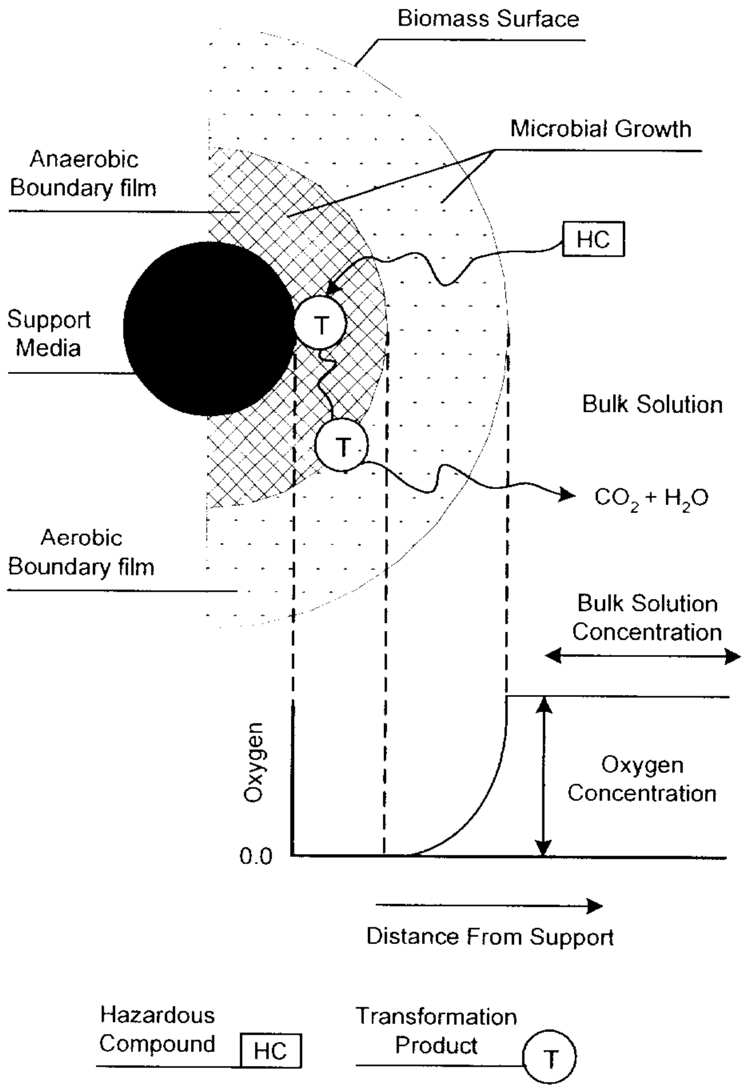


Figure 2-9 Schematic of transfer within the biomass film and oxygen concentration for aerobic systems.

Biofiltration can be designed as oxic, anoxic, or anaerobic processes. Aerobic systems treating high organic concentrations are frequently limited in their degradation rate because of oxygen transfer. Such systems support the formation of heavy biomass buildup that creates a significant oxygen demand. Oxygen transfer can be limited by both the gas phase to liquid phase and the liquid phase to biomass phase. On the other hand, the gas-to-liquid transfer is influenced by bubble size and dissolution pressure. The liquid-to-biomass is influenced by mixing intensity and the biomass structure.

The most common biofilm reactor is the *packed bed*. This is the medium to which the microorganisms are attached is stationary. Historically, large rocks have been used as support media, but today it is more common to use *plastic media* or *pea-sized stones*. Both are lighter and offer greater surface area and pore volume per unit of reactor volume than do large rocks.

Commonly, packed-bed reactors are used for aerobic treatment of wastewaters and are known as biological towers. Here, the wastewater is distributed uniformly over the surface of the bed and allowed to trickle over the surface of the rock or plastic media, giving the packed-bed reactors some plug-flow character. The void space remains open to the passage of air so that oxygen can be transferred to the microorganisms throughout the reactor.

Fixed-biofilm treatment systems have a long history of application to wastewater and include the processes using ***fixed-biofilm media*** that were widely employed for municipal wastewater.

Owing to the numerous advantages of immobilized microorganisms, the fixed-film reactors should find increasing applications and improvement in design configurations. A number of design configurations have been developed for application in the biochemical engineering of industrial products. Fixed-bed bioreactors can be skid-mounted for transportation ease to several treatment locations, or they can be constructed as a more permanent facility. Skid-mounted units have been seeded with microorganisms in the laboratory and then transported to the site for start-up.

At present, the newly developed technology depends on biological film at substratum of supporting medium inside the reactor in order to increase the retention time of microbes, which leads to a reduction of the waste sludge and an increase in treatment efficiency.

Finally, as can be seen from the literature presented, there are many ways to set up a cycle in a biological system for organic matter and nutrients removal. In this thesis, the solution for removal of organic matter and nutrients in sewage using the Sequencing Batch Biofiltration Reactors (SBBRs), packed with aquacell media, is presented. Furthermore, determining the effects of anoxic/aerobic time and C/N ratios on the performance of the SBBRs for removal of organic matter and nutrients in sewage to achieve good quality of the effluent is also desired. As such these were the aims of this study.

3

Materials and Methods

In this chapter, details are given to the experimental design and operational methods used for this research, which involved operation of two laboratory-scale SBBRs.

3.1 Reactors

Two laboratory-scale SBBRs, each with the effective liquid volume of 12.77L, packed with 40% ***aquacell media***, were designed for operating with four 6 hours cycles per day. Each reactor was made with acryl, designing with 140 mm of diameter and 830 mm of height, and operated with air flow rate of 10 mL/min.

Figure 3.1 shows the schematics of laboratory-scale SBBR.

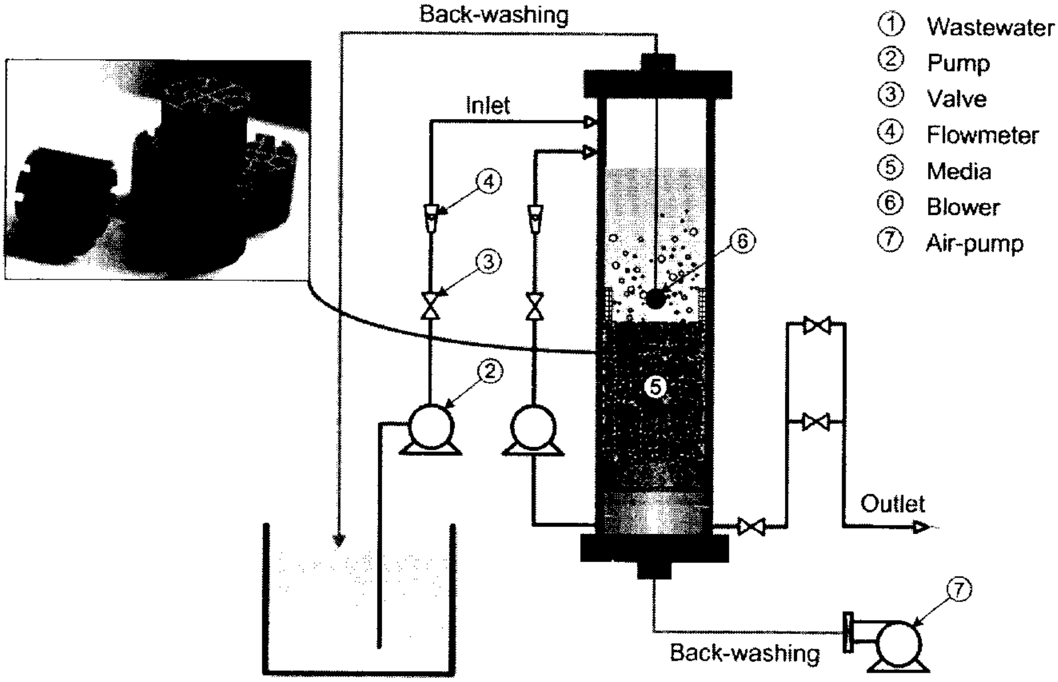


Figure 3-1 Schematics of laboratory-scale SBBR.

3.2 Materials

The SBBRs were seeded with activated sludge (about 2500-3000 mg/L of MLSS) obtained from S-Wastewater Treatment Plant in Pusan, Korea.

3.2.1 Characteristics of Synthetic Wastewater

All laboratory-feed solutions were freshly prepared using room temperature tap water and the following mineral and organic matters:

Table 3-1 Chemical composition of synthetic wastewater

Components	Formula
Glucose	$C_{12}H_{22}O_{11}$
Sodium Bicarbonate	$NaHCO_3$
Ammonium Chloride	NH_4Cl
Sodium Chloride	$NaCl$
Potassium Chloride	KCl
Calcium Chloride	$CaCl_2$
Potassium Dihydrogenphosphate	KH_2PO_4
Magnesium Sulfate	$MgSO_4$

The synthetic wastewater with several C/N ratios was made consisting of the composition of chemicals shown in Table 3.2 and Table 3.3

Table 3-2 Standard content of synthetic wastewater

Chemical Substances	Concentration [mg/L]			
	C/N =3	C/N =5	C/N =7	C/N =10
Glucose	0.1350	0.2250	0.3150	0.4500
NH_4Cl	0.1188	0.1188	0.1188	0.1188
$NaHCO_3$	0.3360	0.3360	0.3360	0.3360
KCl	0.0175	0.0175	0.0175	0.0175
$CaCl_2$	0.0175	0.0175	0.0175	0.0175
$NaCl$	0.0375	0.0375	0.0375	0.0375
KH_2PO_4	0.0351	0.0351	0.0351	0.0351
$MgSO_4$	0.0125	0.0125	0.0125	0.0125

Table 3-3 Characteristics of synthetic wastewater

Item	Unit	Concentration			
		C/N = 3	C/N = 5	C/N = 7	C/N = 10
COD	mg/L	120	200	280	400
NH_4^+-N	mg/L	35-40			
NO_2-N	mg/L	0			
NO_3-N	mg/L	0			
T-P	mg/L	6 – 8			
pH	-	7.0 – 8.0			

3.2.2 Aquacell Media

The aquacell media used as a packing material in SBBRs has the diameter of 2.54 cm, the height of 2.3 cm, the surface area of 74.93cm^2, and the weight of 1.20g/cm^3. The specific surface area and void fraction of the aquacell media in the reactor were 5.0 cm^2/cm^3 and 0.7-0.82, respectively. The compressive strength of the media was fairly high with 250 kgf/cm^2.

Table 3.4 shows the characteristics of Aquacell media used in the research.

Table 3-4 Physical characteristics of Aquacell media

Size (cm)	Surface area (cm^2/EA)	Specific gravity (g/cm^3)	Specific area (cm^2/cm^3)	Bed voidage (-)	Compression intensity (kgf/cm^2)
Ø2.54 x H2.3	74.93	1.20	5.00	0.80	250

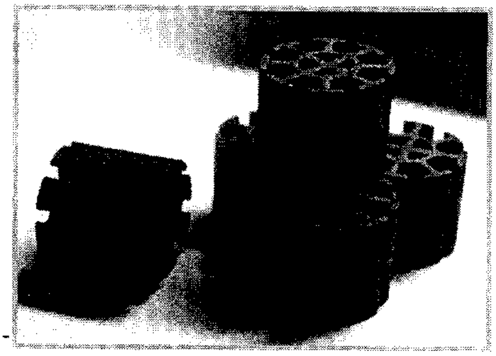


Figure 3-2 Photograph of Aquacell media.

3.3 Operation Methods

The system was fed with the synthetic wastewater and operated at 20°C. The total influent nitrogen concentration was maintained constant while the COD concentration was varied to obtain the desired C/N ratio.

The SBBRs were operated with four 6 hours cycles per day. The daily operating cycle (the main cycle) included two sub-cycles of 2.9 hours each, each one having alternating anoxic and aerobic conditions, and the effluent was withdrawn at the end of the two sub-cycles (Figure 3.3). The remainder of each main cycle time is intended for drawing (10 min) and filling (2 min). During the filling phase (which began with 10 min left in the drawing phase and finished before starting mixing to begin the anoxic phase), 7.7L of synthetic wastewater was introduced into each reactor. The synthetic wastewater was added into 2 tanks for each SBBR once per day.

Moreover, five process modes with the anoxic/aerobic ratio in succession of 0.1, 0.35, 0.5, 0.75 and 1 were designed (Figure 3.4). Once effective biological nitrogen removal has been acquired, the C/N ratios of 3, 5, 7 and 10 were investigated.

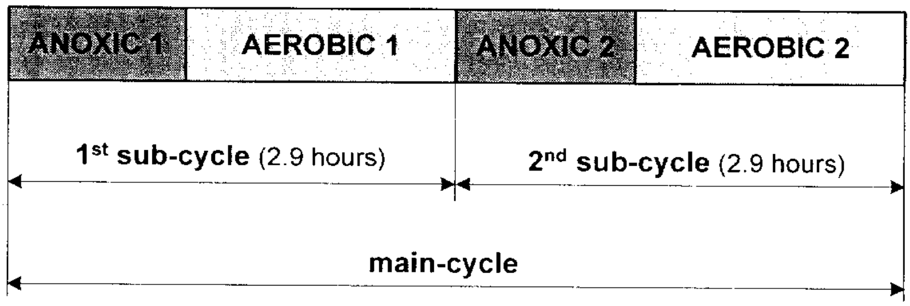


Figure 3-3 Main-cycle of process.

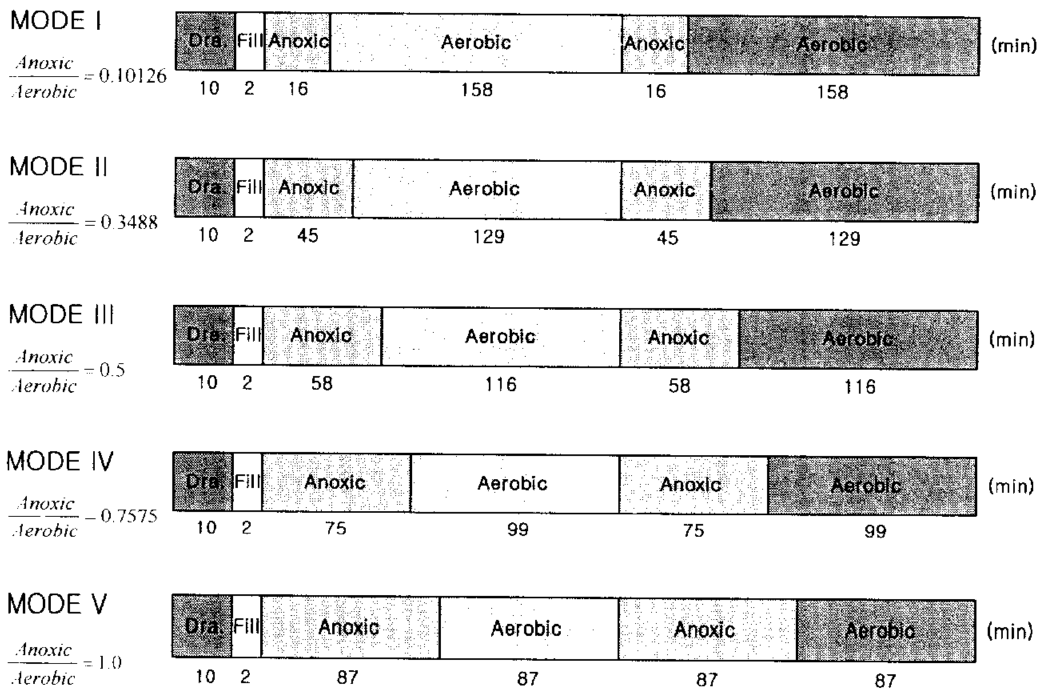


Figure 3-4 Process modes with different anoxic/aerobic time ratios.

3.4 Analytical Methods

Analytical methods were followed by Standard Methods for the Examination of Water and Wastewater (1998) in most cases.

Table 3.5 shows the analytical methods for samples.

Table 3-5 Analytical Methods

No	Item	Unit	Analytical Methods
1	pH	[-]	perpHecT logR meter Model 370 ATIorion
2	Temperature (T)	°C	perpHecT logR meter Model 370 ATIorion
3	Dissolved Oxygen (DO)	mg/L	Istek Model 235D DO meter
4	Chemical Oxygen Demand (COD)	mg/L	Potassium Dichromate Reflux Method
5	Ammonium Nitrogen (NH_4^+ - N)	mg/L	Indophenol Method
6	NO_2^--NO_3^-	mg/L	Ultraviolet Spectrophotometric Screening Method
7	Total Nitrogen (T-N)	mg/L	UV Absorption Method
8	Total Phosphorus (T-P)	mg/L	Titration Method
9	Mixed Liquor Suspended Solids (MLSS)	mg/L	Method 22540 D (AEWW), Total Suspended Solids dried at 103-105°C.

4

Results and Discussion

In this chapter, the results obtained from the experiments are presented. All experiments were conducted at 20°C. The experiments were designed to investigate the effects of anoxic/aerobic time and C/N ratios on the performance of the system which are the main objective of the study.

4.1 Results

4.1.1 Effects of Anoxic/Aerobic Time Ratio on the Performance of the SBBRs

The experimental results for the effects of anoxic/aerobic time ratio on the performance of the SBBRs are shown from Table 4.1-4.10. As the indices of the performance, COD, ammonium nitrogen, total nitrogen, and phosphorus removals were considered for specific analysis.

Table 4-1 Experimental results for the operating condition of Mode I

Item	Removal Efficiency, %		
	Min. Value	Max. Value	Average Value
COD	94.36	98.09	96.22
NH_4^+-N	35.74	80.15	57.95
T-N	28.54	61.69	45.11
T-P	27.37	46.93	37.15

Table 4-2 Experimental results for the operating condition of Mode II

Item	Removal Efficiency, %		
	Min. Value	Max. Value	Average Value
COD	90.66	98.52	94.95
NH_4^+-N	92.31	99.8	96.06
T-N	25.64	83.59	54.62
T-P	29.96	60.39	45.17

Table 4-3 Experimental results for the operating condition of Mode III

Item	Removal Efficiency, %		
	Min. Value	Max. Value	Average Value
COD	94.29	98.11	96.20
NH_4^+-N	98.13	99.40	98.76
T-N	78.07	97.71	87.89
T-P	14.15	66.34	40.25

Table 4-4 Experimental results for the operating condition of Mode IV

Item	Removal Efficiency, %		
	Min. Value	Max. Value	Average Value
COD	90.57	96.72	93.65
NH_4^+-N	65.16	92.35	78.75
T-N	22.67	49.12	35.90
T-P	17.83	59.72	38.77

Table 4-5 Experimental results for the operating condition of Mode V

Item	Removal Efficiency, %		
	Min. Value	Max. Value	Average Value
COD	86.0	96.7	91.35
NH_4^+-N	57.0	76.0	66.5
T-N	28.5	79.8	54.15
T-P	19.15	58.3	38.73

Table 4-6 Variation of concentration and removal efficiency throughout four sub-cycles on operation of Mode I

[mg/L]

Item	Inlet Conc.	Process								Outlet	
		Anoxic 1		Aerobic 1		Anoxic 2		Aerobic 2			
		Conc.	Eff.	Conc.	Eff.	Conc.	Eff.	Conc.	Eff.	Conc.	Eff.
pH	7.8		-		-		-		-		-
COD	400	94	76.5	61.2	84.7	30.8	92.3	8	98.0	8.0	98.09
NH_4^+-N	40	16.8	58.0	16	60.0	14.9	62.7	13.1	67.3	13.6	67.35
T-N	40	25.8	35.5	20.1	49.8	17.6	56.0	15.3	61.69	15.3	61.69
T-P	8	15.76	-97.1	6.5	18.2	6.1	23.7	4.3	45.2	4.38	45.25

Table 4-7 Variation of concentration and removal efficiency throughout four sub-cycles on operation of Mode II

[mg/L]

Item	Inlet Conc.	Process								Outlet	
		Anoxic 1		Aerobic 1		Anoxic 2		Aerobic 2			
		Conc.	Eff.	Conc.	Eff.	Conc.	Eff.	Conc.	Eff.	Conc.	Eff.
pH	7.8	7.5	-	7.8	-	8.0	-	8.1	-	8.1	-
COD	400	76	81	49.6	87.6	26	93.5	6	98.5	5.92	98.52
NH_4^+-N	40	13.7	65.7	5.12	87.2	3.32	91.7	1.8	95.5	1.8	95.50
T-N	40	22.7	43.2	12.5	68.7	10.4	74.0	6.56	83.59	6.56	83.59
T-P	8	14	-75.4	6.3	21.3	6.48	19.0	3.2	60.3	3.2	60.39

Table 4-8 Variation of concentration and removal efficiency throughout four sub-cycles on operation of Mode III

[mg/L]

Item	Inlet Conc.	Process								Outlet	
		Anoxic 1		Aerobic 1		Anoxic 2		Aerobic 2			
		Conc.	Eff.	Conc.	Eff.	Conc.	Eff.	Conc.	Eff.	Conc.	Eff.
pH	7.8	7.6	-	7.73	-	8.0	-	8.1	-	8.1	-
COD	400	43.2	89.2	31.6	92.1	16	96	7.6	98.1	8.0	98.12
NH_4^+-N	40	9.4	76.5	2.92	92.7	1.68	95.8	0.92	97.7	0.92	97.71
T-N	40	18.56	53.6	8.44	78.9	7.2	82.0	0.76	98.1	0.75	98.13
T-P	8	5.97	25.4	4.2	47.5	6.1	23.6	2.61	67.3	2.61	67.34

Table 4-9 Variation of concentration and removal efficiency throughout four sub-cycles on operation of Mode IV

[mg/L]

Item	Inlet Conc.	Process								Outlet	
		Anoxic 1		Aerobic 1		Anoxic 2		Aerobic 2			
		Conc.	Eff.	Conc.	Eff.	Conc.	Eff.	Conc.	Eff.	Conc.	Eff.
pH	7.8	7.5	-	7.72	-	8.16	-	8.23	-	8.20	-
COD	400	49.2	87.7	31.6	92.1	16.4	95.9	13.2	96.7	13.2	96.72
NH_4^+-N	35	16.0	54.2	8.72	75.1	5.7	83.7	2.69	92.3	2.61	92.35
T-N	35	27.5	21.4	23.1	34.0	18.6	46.8	17.9	49.0	17.9	49.12
T-P	8	12.1	-51.5	4.06	49.3	5.8	27.8	3.28	59.0	3.22	59.72

Table 4-10 Variation of concentration and removal efficiency throughout four sub-cycles on operation of Mode V

[mg/L]

Item	Inlet Conc.	Process								Outlet	
		Anoxic 1		Aerobic 1		Anoxic 2		Aerobic 2			
		Conc.	Eff.	Conc.	Eff.	Conc.	Eff.	Conc.	Eff.	Conc.	Eff.
pH	7.8	7.6	-	7.75	-	8.03	-	8.19	-	8.1	-
COD	400	59.2	85.2	40	90	22.8	94.3	13.2	96.7	13.2	96.70
NH_4^+-N	38	21.8	42.7	19.7	48.0	17.2	54.6	15.5	59.2	15.5	59.25
T-N	40	24.7	38.2	24.2	39.6	20.1	47.6	18.8	52.9	18.6	53.42
T-P	8	10.7	-34.4	5.08	36.5	5.6	30.2	3.36	58.0	3.34	58.25

4.1.2 Effects of C/N ratio on the performance of the SBBRs

The experimental results for the effects of C/N ratio on the performance of the SBBRs are shown in Table 4.11.

Table 4-11 Average removal efficiencies of COD, T-P, T-N, and NH_4^+-N with different C/N ratios

No	anoxic/aerobic time ratio	C/N	Average Removal Efficiency (%)			
			COD	T-P	T-N	NH_4^+-N
Mode I	0.10	3	95.69	46.93	35.50	35.74
		5	94.37	27.37	28.54	92.15
		7	95.81	36.63	46.86	44.69
		10	98.09	45.25	61.69	67.35
Mode II	0.35	3	90.67	29.96	25.64	92.31
		5	97.25	39.54	49.40	99.99
		7	97.98	34.54	65.91	96.52
		10	98.52	60.39	83.59	95.50
Mode III	**0.5**	3	94.29	14.14	78.07	99.12
		5	96.51	34.04	80.61	99.03
		7	97.71	46.61	81.32	99.40
		10	**98.12**	**67.34**	**97.71**	**98.13**
Mode IV	0.75	3	90.58	17.83	22.67	82.17
		5	94.15	29.46	29.35	87.62
		7	95.70	36.58	42.56	65.17
		10	96.72	59.72	49.12	92.35
Mode V	1.0	3	86.00	19.15	28.50	76.00
		5	93.80	29.60	63.43	85.50
		7	96.95	33.50	82.00	57.00
		10	96.70	58.25	53.42	59.25

4.2 Discussion

4.2.1 Variation of DO and pH under Various C/N Ratio

Under the condition of process Mode III (the anoxic/aerobic time ratio of 0.5), variation of DO and pH with the changing of C/N ratio is shown in Figure 4.1 and Figure 4.2. In the case of the C/N ratio of 10, experimental results indicate that DO concentration in the aerobic phase is lower than other conditions of C/N ratio (about 5.5 mg/L as compared with 6.8-7.0 mg/L for DO concentration for the C/N ratio of 7). This can explain that the nitrifying bacteria tend to accumulate closer to the attachment surface, while heterotrophs dominate near the outer surface of the biofilm. This "layering" is a natural consequence of the heterotrophs with much faster specific growth rate, which cause them to exist stably in the region of the biofilm experiencing the greatest detachment and predation rates. On the other hand, the nitrifiers accumulate most successfully deeper inside the biofilm, where they are at least partly protected from detachment and predation.

While having the nitrifiers deep inside the biofilm helps stabilize the slow-growing nitrifiers against washout, it also increases their susceptibility to oxygen limitation, since dissolved oxygen must diffuse through the heterotroph layer before it reaches the nitrifiers. The added mass-transport resistance and oxygen consumption by the heterotrophs lower the biofilm DO concentrations available to the nitrifiers. The low DO concentrations inside the biofilm can negate the advantages of protection.

In addition, the oxygen concentration controls whether or not the facultative aerobes respire nitrogen. Oxygen can control denitrification in two ways. The first is repression of the several nitrogen-reductase genes. Research with *Pseudomonas stutzeri* indicates that these genes are repressed by DO concentrations greater than 2.5 to 5 mg O_2/L (Korner and Zumft, 1989). The second control mechanism is inhibition of the activity of the reductase by DO concentrations greater than a few tenths of an mg O_2/L (Tiedje, 1988; Rittmann and Langeland, 1985). The fact that the DO concentrations that suppress the reductase genes are much higher than the concentrations that inhibit their activity means that denitrification can occur when D.O. concentrations are well above zero (Rittmann and Langeland, 1985; deSilva, 1997). This situation is enhanced Electron Donor when the denitrifying bacteria are located inside flocs or biofilms, where the oxygen concentration is lower than in the bulk liquid (Rittmann and Langeland, 1985).

In the case of the C/N ratio of 7, too high DO concentration can lead to the accumulation of the denitrification intermediates, such as NO_2^-, NO_2, and N_2O. Therefore, a high O_2 concentration tends to suppress the nitrite and nitrous oxide reductases before the nitrate reductase is repressed.

Although denitrifiers are not very pH sensitive, pH values outside the optimal range of 7 to 8 can lead to accumulation of intermediates (Figure 4.2). In this case, because of having low alkalinity waters, it is necessary to control pH, because denitrification produces strong base. Base production is illustrated by the balanced reactions in which acetate and H_2 are electron donors for the heterotrophic and autotrophic denitrifiers, respectively.

On the other hand, the kinetic characteristics of heterotrophic denitrifiers and autotrophic nitrifiers are very different. Because the nitrifiers have lower values, are much slower growers, they require substantially longer solids retention times. Furthermore, maximum nitrification rates require a high DO concentration (the C/N ratio of 7), while high DO concentration slows or stops denitrification. Because nitrification is often necessary to provide the NO_3^- or NO_2^- for denitrification, process design and operation must reconcile these conflicting physiological characteristics. It is the reconciliation of the needs of the nitrifiers and heterotrophic denitrifiers that distinguishes the different approaches to denitrification in environmental biotechnology.

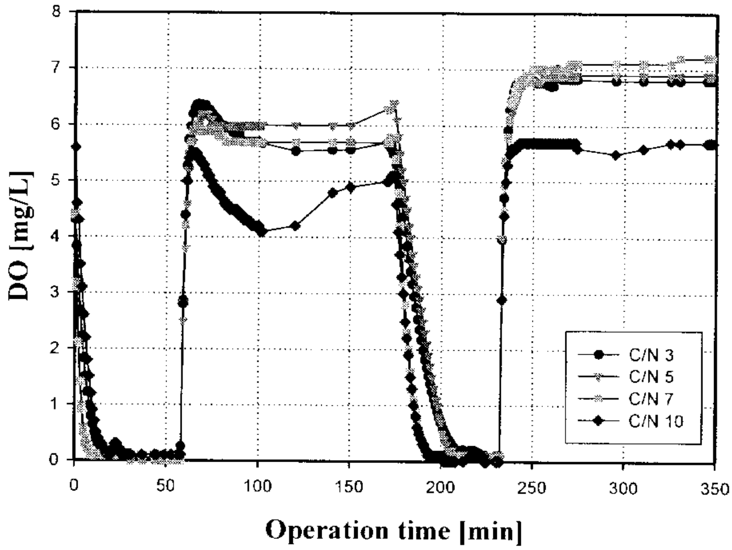


Figure 4-1 Variation of DO with the operating time under each C/N ratio condition of Mode III
[Mode III: anoxic/aerobic/anoxic/aerobic - 58/116/58/116 min].

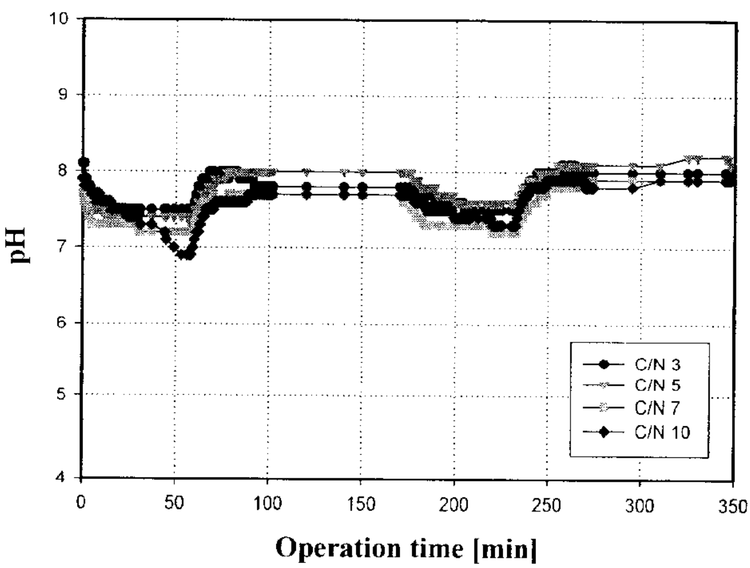


Figure 4-2 Variation of pH with the operating time under each C/N ratio condition of Mode III
[Mode III: anoxic/aerobic/anoxic/aerobic - 58/116/58/116 min].

4.2.2 Effect of Anoxic/Aerobic Time Ratio

Because the investigation about effects of changing anoxic/aerobic time and C/N ratios on the performance of SBBRs for removal of organic matter and nutrients in sewage wastewater effect of anoxic/aerobic time ratio is the second objective of this thesis, some brief discussion was made in this part.

Most COD compound was reduced in the anoxic phase of the first sub-cycle (Figure 4.3 and Table 4.6-4.10). And this was also the same case for NH_4^+-N compound during the operation of five proposed modes (Figure 4.4).

As the anoxic/aerobic time ratio was higher (as 0.75 and 1.0), the pH values was maintained in a more stable state on microbes because the consumption rate of alkalinity was decreased. The organic removal efficiency was increased with decreasing the ratio of anoxic/aerobic time.

The T-N removal efficiency was increased from 61.7% to 97.7% as the ratio of anoxic/aerobic time was higher. Especially, the T-N removal efficiency was the highest when the repeat cycle of anoxic/aerobic was 0.59 (Figure 4.5).

The T-P removal efficiency showed a great difference in each operating condition (Figure 4.6), because it was influenced greatly by the ràtio of anoxic/aerobic time and the repeat cycle of anoxic/aerobic is two time as the operating conditions in this study, and it was the highest in the Mode III which the denitrification rate was the highest.

In addition, nutrients treatment by SBBR was performed with the obtaining so higher removal efficiencies during the definite experimental condition. Nitrogen and phosphorus removal efficiencies were evaluated under different anoxic/aerobic ratio. Both nitrogen and phosphorus removal patterns were investigated through the condition of anoxic/aerobic time ratio ranging 0.10 to 1.0. As anoxic/aerobic time ratio was changed, maximum nitrogen removal efficiency was 97.71% at anoxic/aerobic time ratio of 0.5 (Table 4.11). And, phosphorus removal efficiency was more than 60%.

From the results, the experiment was performed to evaluate the effects of anoxic/aerobic time ratio, for removal of organic matter and nutrients in sewage wastewater treatment by using SBBRs systems.

Finally, in the short conclusion, the result shows that:

(i) The organics removal efficiency was increased in proportion to the increasing anoxic/aerobic time ratio

(ii) Although the ammonium nitrogen removal efficiency was increased at the anoxic/aerobic time ratio was quite higher (0.35 and 0.5) and decreased left than the value of 0.5 of its ratio, the T-N removal efficiency was increased as that was lower.

(iii) As anoxic/aerobic time ratio was lower, the T-P removal efficiency was increased.

(iv) It was also shown that the effluent quality increased when the anoxic/aerobic time ratio was not high, and

(v) The very small anoxic/aerobic time ratio did not yield greater reductions in COD, T-N, or T-P as compared to higher ratios of its.

In the conclusion, the anoxic/aerobic SBBR provided the best overall treatabilitty of the synthetic wastewater based on effluent quality.

Furthermore, the results suggested that quite low anoxic/aerobic time ratio treatment schemes are a viable treatment alternative for sewage wastewaters containing high C/N ratio in its concentrations.

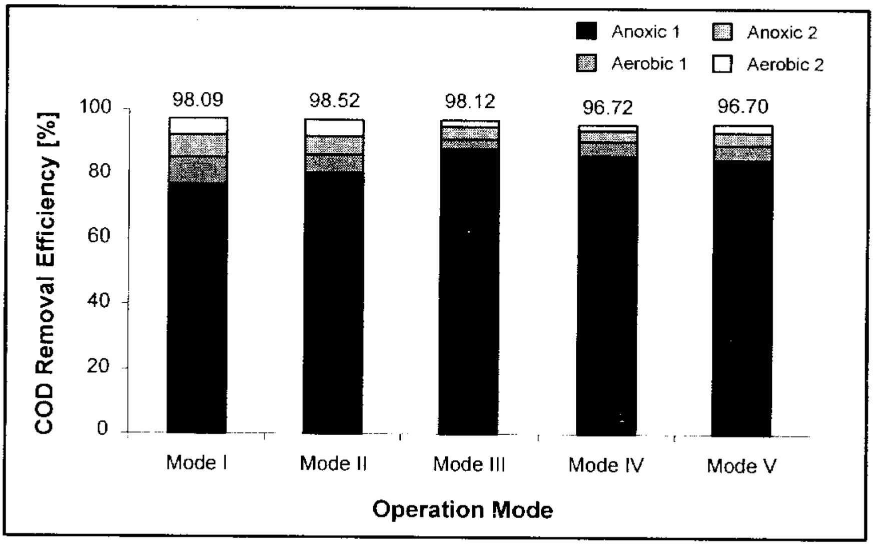


Figure 4-3 Effect of the anoxic/aerobic time ratios on COD removal efficiency *[anoxic/aerobic time ratio: 0.10126 (Mode I), 0.3488 (Mode II), 0.5 (Mode III), 0.75 (Mode IV), 1.0 (Mode 5)]*

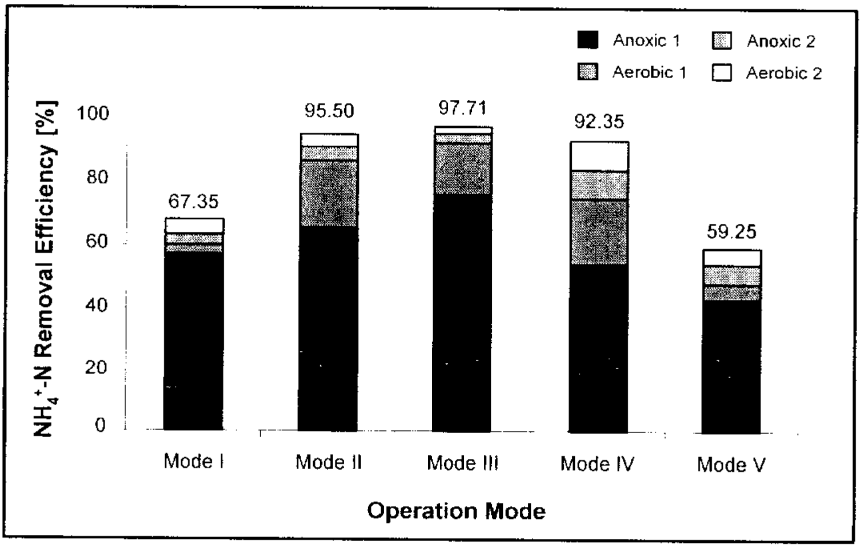


Figure 4-4 Effect of the anoxic/aerobic time ratios on NH_4^+-N removal efficiency *[anoxic/aerobic time ratio: 0.10126 (Mode I), 0.3488 (Mode II), 0.5 (Mode III), 0.75 (Mode IV), 1.0 (Mode 5)]*

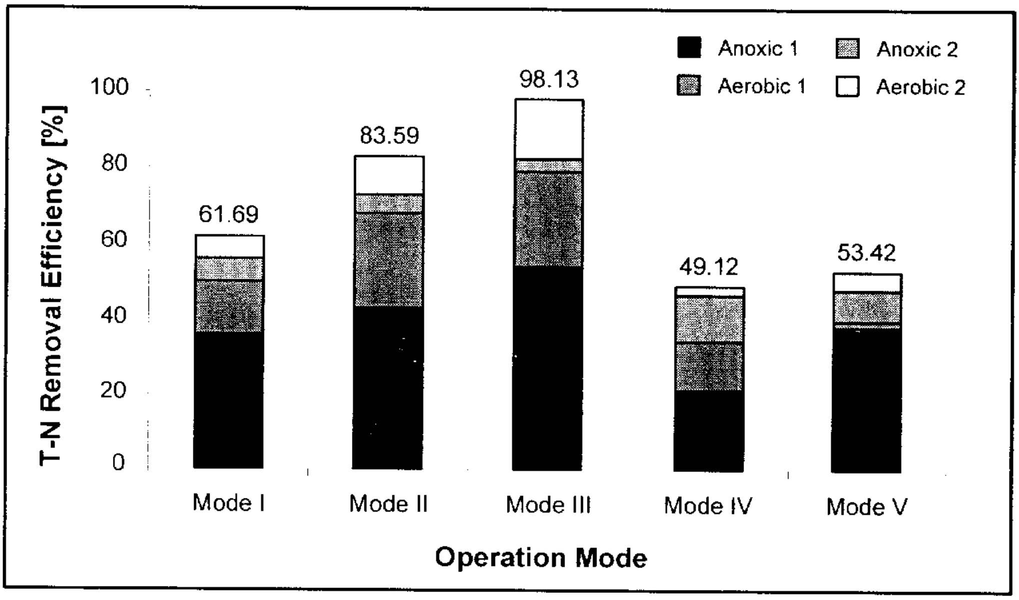


Figure 4-5 Effect of the anoxic/aerobic time ratios on T-N removal efficiency *[anoxic/aerobic time ratio: 0.10126 (Mode I), 0.3488 (Mode II), 0.5 (Mode III), 0.75 (Mode IV), 1.0 (Mode 5)]*

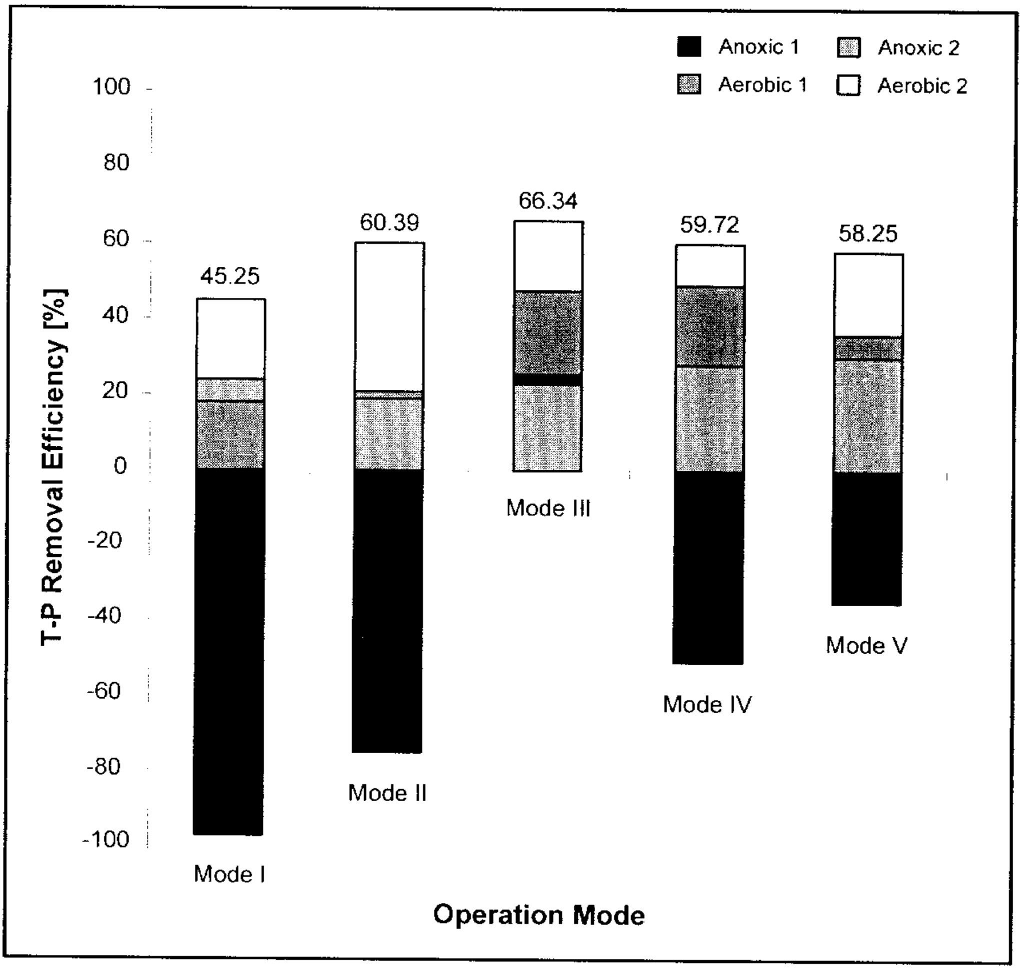


Figure 4-6 Effect of the anoxic/aerobic time ratios on T-P removal efficiency *[anoxic/aerobic time ratio: 0.10126 (Mode I), 0.3488 (Mode II), 0.5 (Mode III), 0.75 (Mode IV), 1.0 (Mode 5)]*

4.2.3 Effect of C/N Ratio on COD Removal

Steady state COD profiles for the five modes of different anoxic/aerobic time ratio are shown in Figure 4.7. The overall COD removal for all modes remained approximately the same. COD removals of around 96 percent or higher was obtained for all experiments. Effect of C/N ratios on COD removal efficiency is shown in Figure 4.7. Figure 4.8 shows that the removal efficiency of COD was increased with increasing in C/N ratio.

At the best process, mode III, COD removal efficiencies above 80 percent were maintained in the first anoxic sub-cycle throughout the experiment with C/N ratios of 3, 5, 7 (Figure 4.7 – 4.8, while 2 to 17 percent COD removal was obtained in the first sub-aerobic zone. Very little COD was available in solution from the second anoxic to the second aerobic zones. Aerobic COD removals ranged from zero to 9.2 percent of the total COD removal However, at the best case, the C/N ratio of 10, the familiar efficiency of COD removal in the first anoxic zone just obtained about 70% (Figure 4.6).

Figure 4.8 shows that essentially complete removal of biodegradable organic was achieved because no soluble COD removal occurred in the last two aerobic phases. To determine the actual changes of COD in each sub-cycle, mass balances across each reactor are presented in Figure 4.8. The anoxic nitrogen release is a function of anoxic COD uptake. As can be seen, higher anoxic COD uptake/anoxic nitrogen release ratios were observed at the higher C/N ratios. In other words, less anoxic nitrogen release per unit of anoxic COD removed was obtained under nitrogen limiting condition. Likewise, less aerobic nitrogen uptake per unit anoxic COD removed was obtained under nitrogen limiting conditions.

As discussed earlier, the uptake of the readily biodegradable organic matter occurred rapidly in the anoxic zone, while subsequent uptake occurred at a much-reduced rate as more slowly biodegradable organic matter was fermented in the non-oxic zones. Comparing the two extreme C/N ratios, more anoxic COD removal was observed under T-N limiting conditions. It appears that because more phosphorus was available under these conditions, it was possible for the bacteria to remove and store COD more efficiently.

It was stated that COD removal in SBBR systems is a function of the nature of organic matter (readily biodegradable versus slowly biodegradable) and the actual system sludge age. The lowest ratio obtained at the C/N ratio of 3, approaches the stoichiometric amount of COD that can be stored by the release of the energy in a phosphorus bond. At the highest C/N ratio (10), more than enough organic matter was available to remove the small amount of nitrogen present in the influent wastewater under T-N limiting conditions, thus resulting in a lower effluent concentration (2 mg/L versus 6 mg/L for C/N of 3).

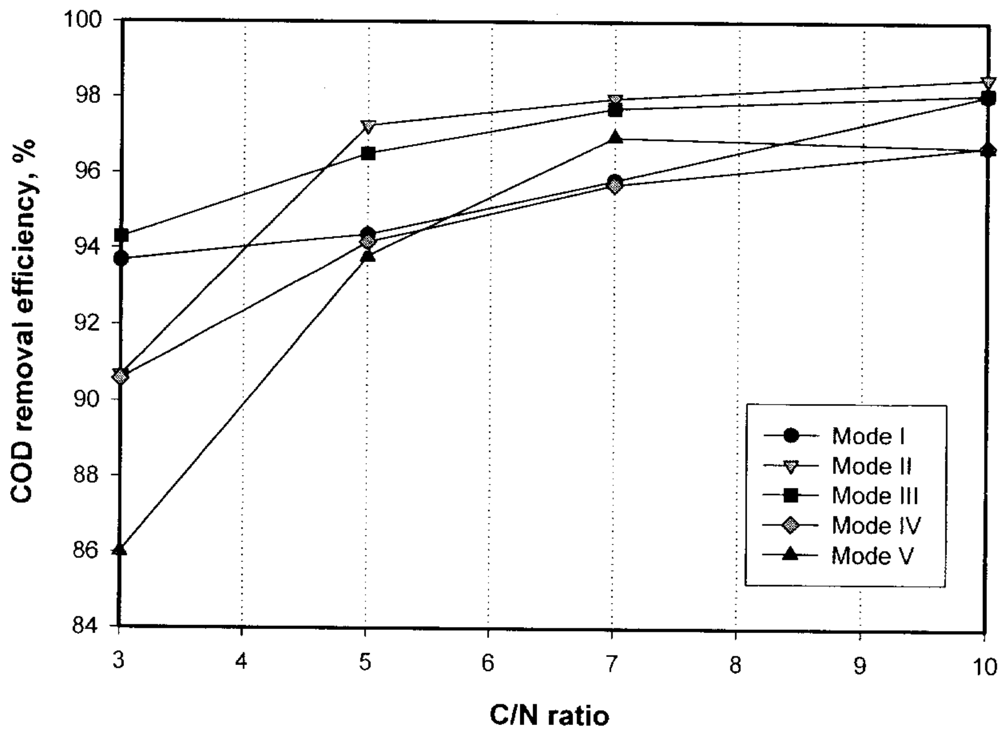


Figure 4-7 Effect of the C/N ratios on COD removal efficiency *[C/N ratio: 3, 5, 7 and 10].*

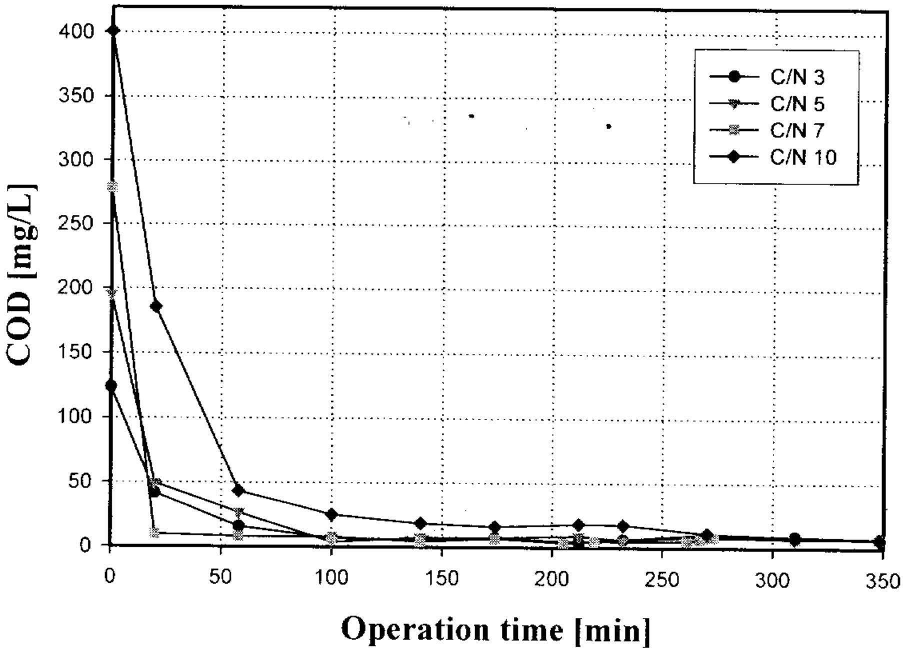


Figure 4-8 Variation of COD on operation of Mode III with C/N ratios of 3, 5, 7, and 10

[Mode III: anoxic/aerobic/anoxic/aerobic - 58/116/58/116 min].

4.2.4 Effect of C/N Ratio on Ammonium Nitrogen Removal

Ammonia and nitrate/nitrite nitrogen removal on research experiment during the first steady sub-cycle was 93% corresponding to an effluent containing no ammonia or nitrite and only 0.15 mg/L NO_x-N.

Based on the ammonia removal efficiency during five mode designing for the research, the best case was chosen following the highest removal efficiency of total nitrogen, Mode III at the C/N ratio of 10, be expressed in Figure 4.9 and 4.10. It is shown that increase in C/N ratio results in an increase in the removal efficiency of ammonia.

The variation of COD, TN, TP and ammonia versus DO, and pH throughout the four sub-cycles on operation of Mode III with C/N ratio in succession of 3, 0.35, 5, 7 and 10, were illustrated (Figure 4.9-4.10). It seems reasonable to assume that all the ammonia in the influent would be nitrified at steady state. Assuming a worse case scenario, which all the ammonia in the third sub-cycle is nitrified and none of the oxidized nitrogen is denitrified during the last anoxic period. The worst ammonia and nitrate/nitrite nitrogen removal efficiency happened in the case of lowest anoxic/aerobic time ratio (Mode I) in every C/N ratio case (Table 4.11). The results showed that using higher anoxic/aerobic time ratio than value of 0.1 is necessary. Thus, the laboratory-scale treatment study is capable of achieving high ammonia removal on wastewater with high C/N and quite low anoxic/aerobic time ratios. It should be noted that the removal would be better than this because some of the ammonia nitrogen goes to producing cell mass rather than being oxidized.

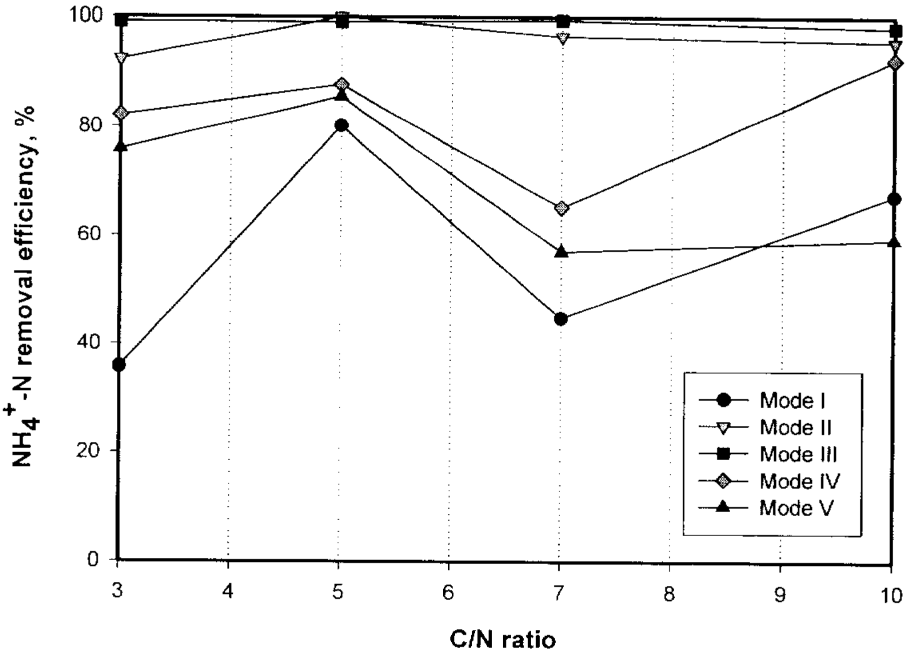


Figure 4-9 Effect of the C/N ratios on NH_4^+-N removal efficiency *[C/N ratio: 3, 5, 7 and 10].*

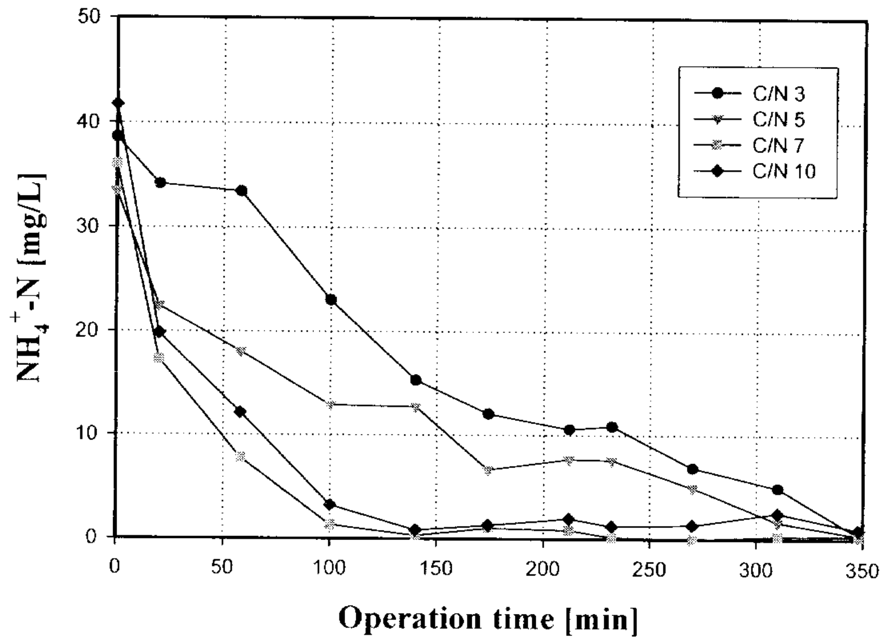


Figure 4-10 Variation of NH4+-N on operation of Mode III with C/N ratios of 3, 5, 7, and 10

[Mode III: anoxic/aerobic/anoxic/aerobic - 58/116/58/116 min].

4.2.5 Effect of C/N Ratio on Nitrogen Removal

Typical system influent T-N concentrations ranged from 35 to 40 mg/L. Effluent TN concentrations of less than 8.2 mg/L were obtained during all experiments. Total nitrogen removals of approximately 40 to 50 percent were typical, and the highest total nitrogen removal efficiency achieved is 98.13% for the operation of Mode III with the C/N ratio of 10 (Table 4.11 and Figure 4.11). The total nitrogen in the system effluent typically consisted of nitrate, with concentrations ranging from 0.76 to 29.64 mg/L. Nitrification was essentially complete, with effluent ammonia concentrations less than 0.5 mg/L throughout the study. As shown in the Figure 4.12 for Mode III with the best total nitrogen removal efficiency, as C/N ratio increases, the extent of total nitrogen removal efficiency increases.

Denitrification in the anoxic zone was generally complete during all experimental phases. Nitrate and nitrite concentrations in the anoxic zone were very low. The average denitrification rate increased as C/N ratio increased. However, denitrification rates were not consistent for each C/N ratio data set due to the fluctuations of the influent wastewater TN concentrations. The average net total nitrogen removal for the C/N ratio of 7 was greater than that for the ratios of 5, 10 and 3, respectively. Aerobic denitrification rates were calculated assuming 12% nitrogen was required for bacterial growth, i.e., and production of MLVSS. Because only negative rates were obtained, it can be assumed that either no denitrification occurred in the aerobic zone or the nitrogen required for bacterial growth was less than 12 % on a MLVSS basis.

In the conclusion, very good total nitrogen removal efficiencies were obtained throughout all experiments (45%-98%). Complete denitrification in the anoxic zone was also consistently observed. The average denitrification rate in the first anoxic zone increased as C/N ratio increased. The ratios of anoxic T-N removed to the amount of nitrate denitrified were also shown in Figure 4.11-4.12 for the case of highest removal efficiency total nitrogen with the C/N ratio in succession of 3, 5, 7, and 10. The average ratio of anoxic COD removed to nitrate denitrified from the C/N ratio of 3 was lower than that of the C/N ratios of 5, 7 and 10, respectively, i.e., more COD being available to the denitrifiers in the anoxic zone at the ratios of 7 and 10. These observations could support the higher degree of the average denitrification rate observed under TN limiting conditions. Aerobic denitrification rates were also calculated to categorize the total nitrification which occurred in the system. Aerobic denitrification rates were calculated assuming 12 % nitrogen was required for bacterial growth. It can be assumed that no denitrification occurred in the aerobic zone because only negative rates were obtained. It is also possible that the nitrogen required for bacterial growth was less than 12% on an MLVSS basis as it was never measured directly.

The C/N ratios are helpful in determining the ease with which biological nitrogen removal can take place. Excellent removal efficiency can be expected for a C/N ratio greater than 9 (Grady et al., 1999). Figure 4.12 shows C/N ratio based on the biodegradable fractions of nitrogen and COD for the second batch of wastewater before and after each stage of pretreatment. Not enough data was available to analyze the first batch of wastewater in this method. Since a large fraction of the total nitrogen was soluble, settling and anoxic treatment reduced the C/N ratio greatly by removing more particulate COD than total nitrogen. The ratio dropped to between 7 and 10 after the last stage of pretreatment. The pretreated wastewater is, therefore, close to being carbon limited for nitrogen removal, but the carbon content will most likely be sufficient from removing most of the nitrogen as long as the reactor setup uses the carbon efficiently. Other researchers working with slaughterhouse waste (Subramaniam et al., 1994) have also noted that too much pretreatment can cause carbon limitations for nitrogen removal.

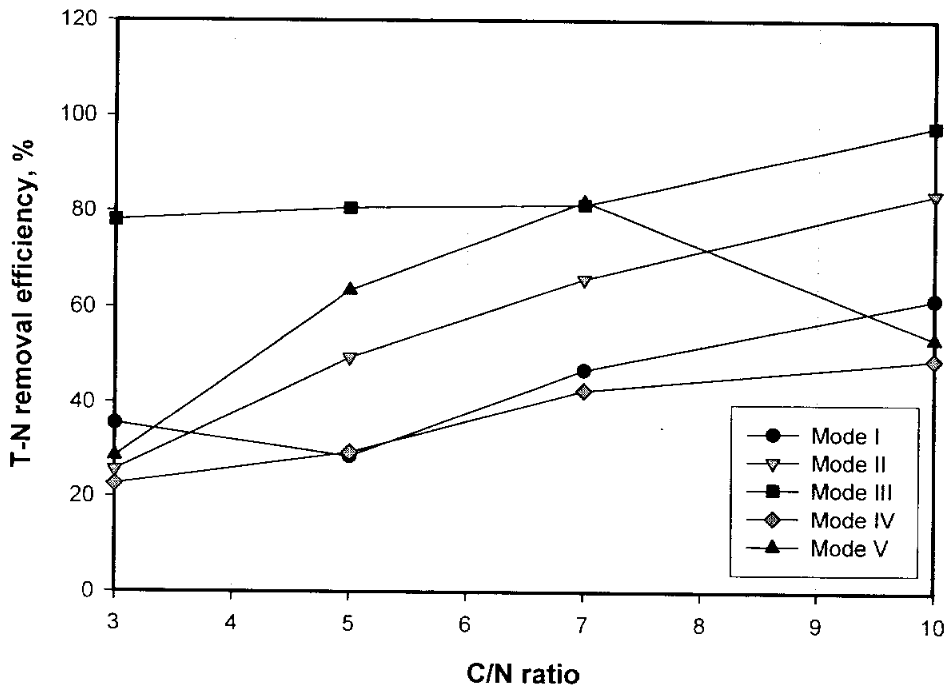


Figure 4-11 Effect of the C/N ratios on T-N removal efficiency

[C/N ratio: 3, 5, 7 and 10].

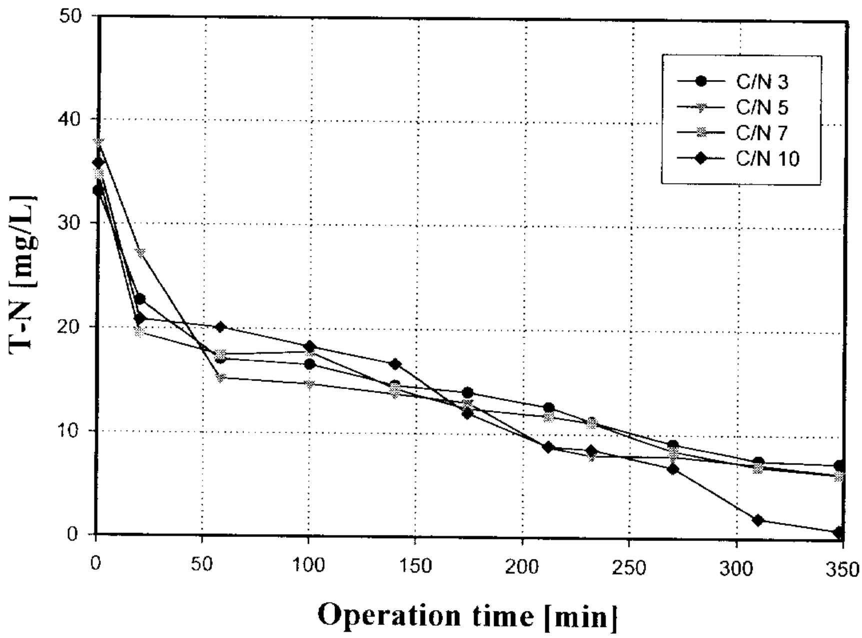


Figure 4-12 Variation of T-N on operation of Mode III with C/N ratios of 3, 5, 7, and 10 *[Mode III: anoxic/aerobic/anoxic/aerobic - 58/116/58/116 min].*

4.2.6 Effect of C/N Ratio on Phosphorus Removal

In the best process, mode III, effluent total phosphorus concentrations were considerably higher than 2 mg/L, as shown in Figure 4.13 – 4.14. System phosphorus removal efficiencies were 14.14 percent, 34.04 percent, 46.61 percent and 66.34 percent for C/N ratios of 3, 5, 7 and 10, respectively. As shown in Figure 4.13, with the lowest C/N ratios, very little phosphorus was removed from influent throughout the main 6 hours cycle, from the first anoxic to the second aerobic sub-cycles. The removal efficiency of phosphorus in this case was only 14.14 percent.

The Figure 4.14 show that almost phosphorus compound was removed in the first anoxic and the first aerobic zones. And at the C/N ratios of 7 and 10, phosphorus concentration in wastewater was steadily reduced during the main cycle length (Figure 4.13 – 4.14). The highest phosphorus removal efficiency, as 66.34 percent, was achieved at C/N ratio of 10.

In general, the majority of the phosphorus release occurred in the anoxic zone, followed by subsequent uptake of phosphorus in the aerobic zone in all experiments. The C/N ratio of 10 produced the greatest anoxic phosphorus release, and the C/N ratio of 3 produced the least phosphorus release. Likewise, the C/N ratio of 10 caused the largest aerobic phosphorus uptake while the ratio of 3 caused the least aerobic phosphorus uptake. There was phosphorus release in the second anoxic sub-cycle for all four sub-cycles.

The percent phosphorus removal from wastewater was increased with a increase in C/N ratio (Figure 4.14). The system, however, is capable of achieving high phosphorus removal on wastewater with quite low anoxic/aerobic time ratio (Mode III - anoxic/aerobic time ratio of 0.5).

In the Figure 4.13, as C/N ratio increases, the extent of T-P removal efficiency increases. In other words, this tendency excludes the case both of so quite low

C/N ratio (3 of C/N) and anoxic/aerobic time ratio (0.1 of anoxic/aerobic).

The lowest effluent concentration of phosphorus was obtained under phosphorus limiting conditions. The highest net phosphorus removal (mg/d) occurred under COD limiting conditions in all experiments.

Phosphorus release and uptake patterns were used to describe the EBPR phenomena which occurred in the SBBR lab-scale system. It was observed that phosphorus release for the C/N ratio of 3 was greater than that for the ratios of 5, 7 and 10, respectively. The system phosphorus removals were 0.95 mg/L for the ratio of 3, 2.31 mg/L for the ratio of 5, 3.22 mg/L for the ratio of 7, and 4.62 mg/L for the ratio of 10. As predicted, effluent phosphorus concentrations decreased as the C/N ratio increased, i.e., as phosphorus became limiting. Effluent phosphorus concentrations were 5.332 mg/L, 4.292 mg/L, 3.538 mg/L, and 2.318 mg/L for the C/N ratios of 3, 5, 7 and 10, respectively. It appears that phosphorus is harder to remove from wastewater when small amounts of phosphorus are present in the influent wastewater. In this study, the amount of COD required to remove a unit of phosphorus under COD limiting conditions had the lowest value. In other words, the minimum COD required to remove a unit of phosphorus present in wastewater occurred under COD limiting conditions. Phosphorus uptake in the aerobic zone was observed to be directly proportional to anoxic phosphorus release. Similar observations have been reported from the studies by Siebritz et al. (1983), Wentzel et al. (1985) and Abu-ghararah and Randall (1991). Aerobic phosphorus uptake increases as anoxic phosphorus release increases if secondary release is insignificant. The phosphorus uptake to phosphorus release ratio of 1 is the theoretical ratio expected for EBPR.

Phosphorus release in the anoxic zone was observed in all experiment, however, anoxic phosphorus release was a relatively small fraction of the overall system phosphorus release. Anoxic phosphorus release could occur as a result of the fermentation of slowly biodegradable organic matter in this zone. Gerber (1987) stated that phosphorus release is not limited to anoxic conditions. Phosphorus release can occur under anoxic or aerobic conditions as long as some short chain fatty acid is present under those conditions. Iwema and Meunier (1985) also reported from batch studies that anoxic phosphorus release was observed when acetic acid was present even when a high concentration of nitrate (40 mg/L) was measured.

Carbon to nitrogen ratios is also used to determine the efficiency at which biological phosphorus removal might occur in a wastewater. The results showed that low efficiency phosphorus removal processes need a C/N ratio of lower than 3. Figure 4.14 shows that the C/N ratios for the wastewaters meet this requirement if all the COD is allowed to be used for phosphorus removal. A second set of C/N ratios were calculated to show what the ratio would be if carbon was preferentially used on a 5/1 (C/N) basis to remove nitrogen, and only the remaining carbon would be available strictly for phosphorus removal. The wastewater will, therefore, become carbon limited if the goal is phosphorus removal as well as nitrogen removal. A slight change in the particulate COD or phosphorus removal efficiencies during pretreatment could change this ratio. In addition, it should be noted that these ratios depend on the values that were determined for the inert particulate and colloidal COD, which were lower than the actual values because of the methods used. This causes the calculated nutrient ratios to be higher than the true ratios are.

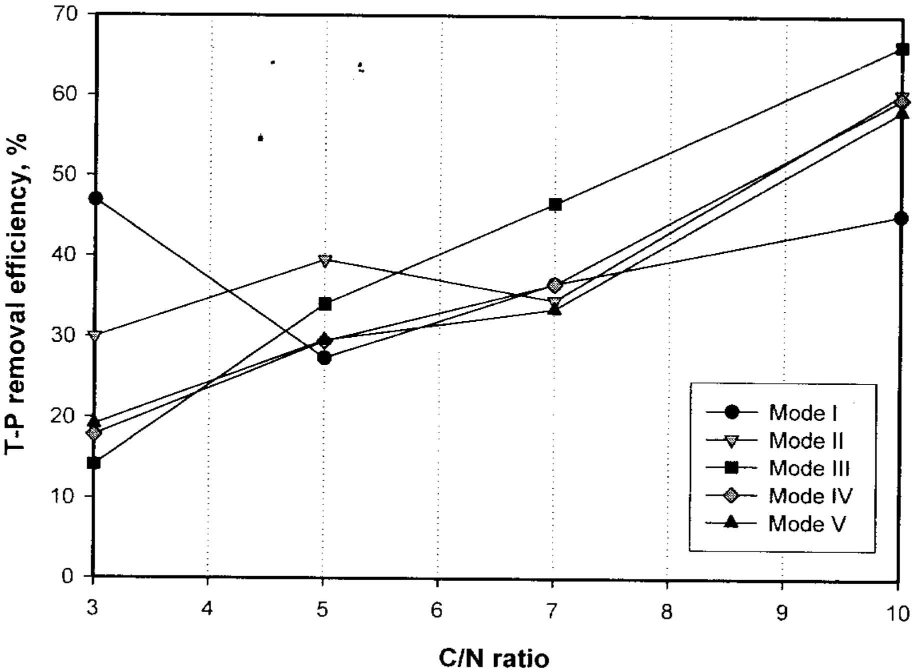


Figure 4-13 Effect of the C/N ratios on T-P removal efficiency

[C/N ratio: 3, 5, 7 and 10].

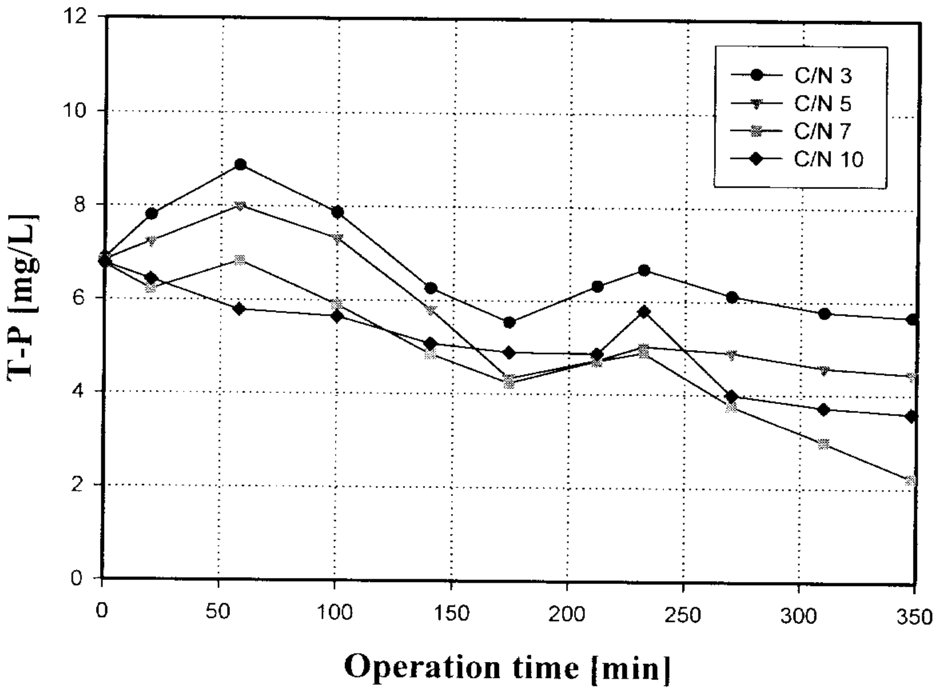


Figure 4-14 Variation of T-P on operation of Mode III with C/N ratios of 3, 5, 7, and 10

[Mode III: anoxic/aerobic/anoxic/aerobic - 58/116/58/116 min].

❖ Startup Problems and Upsets: Nitrification Inhibition

One major issue that developed while running the two SBBRs was inhibition of nitrification during the aerobic zone of the main cycle. As was discussed previously, many factors can contribute to nitrification inhibition, but inhibition by free ammonia levels is considered to be most significant. As pH increases, the fraction of ammonia in the unionized state increases. A significant increase in pH with a high concentration of ammonia in the reactor can lead to level of free ammonia well above inhibitory levels.

It was noted that the influent for the two SBBRs was the anoxic effluent that had been diluted 50% with tap water. Anoxic treatment leads to a production of carbon dioxide. It is believed that the covered anoxic reactor used in this study led to high levels of dissolved carbon dioxide in the wastewater. Two conditions in this reactor most likely led to this. One is that the different anoxic/aerobic time ratio was operated. Therefore, there was little outside disturbance to encourage dissipation of the carbon dioxide. Second, the cover over the reactor likely led to a higher than normal partial pressure of carbon dioxide in the headspace of the reactor. This would have slowed the mass transfer of the carbon dioxide from the liquid phase as well. The carbon dioxide is not in itself a problem to nitrification. In fact, nitrifying autotrophs use carbon dioxide as a carbon source. It was believed that carbon dioxide stripping through aeration and mixing in the SBR led to problems.

Experiments supported the notions that carbon dioxide stripping caused an increase in the pH of the reactor. This was apparently a chemical effect rather than an interaction with the biomass. Effluent was taken from the anoxic reactor and aerated in a batch reactor without any addition of biomass. The pH rose rapidly over the period of 45 minutes to an hour from approximately 7.2 to near 8.2. Again, this pH would not be harmful to nitrification on its own because nitrification actually approaches a maximum near pH 8 (US EPA, 1993). However, when this high pH is associated with high values of ammonia, the free ammonia levels can rise to inhibitory levels.

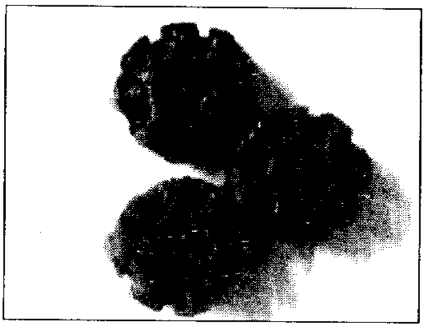
4.2.7 Effect of Back-Washing on the Performance of the SBBRs

Several phenomena were also observed during this research, especially in the back-washing. Firstly, the performance of the SBBRs system for removal of organic matter and nutrients in sewage wastewater was greatly enhanced by good operating on the back-washing.

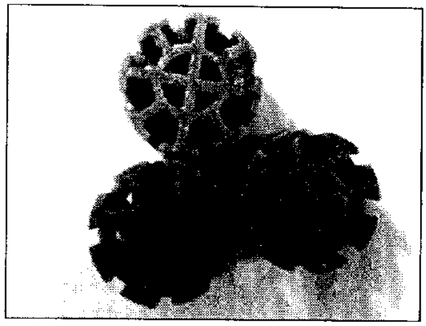
Secondly, the correlation between total nitrogen inside the SBBRs and total nitrogen of the input wastewater for researching, the back-wasing was taking into consideration. This indicated that the homogeneous of the surface of aquacell media was one of some important factor affecting the result of this study.

Finally, no numerical data was observed in the back-washing from this study, based on the conjecture and assumption and that the condition of aquacell media as well as the sludge is homogeneousness during all cases, and it is ensured that the input data was satisfied the requirements for the research.

There are some images of aquacell media before and after back-washing showing in Figures 4.15-4.18 for references.

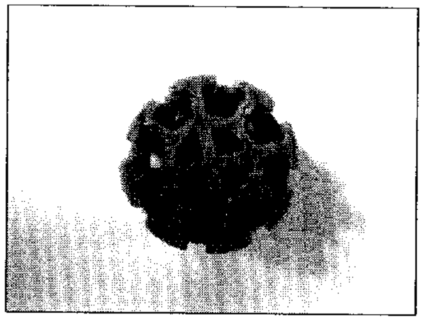


Before back-washing



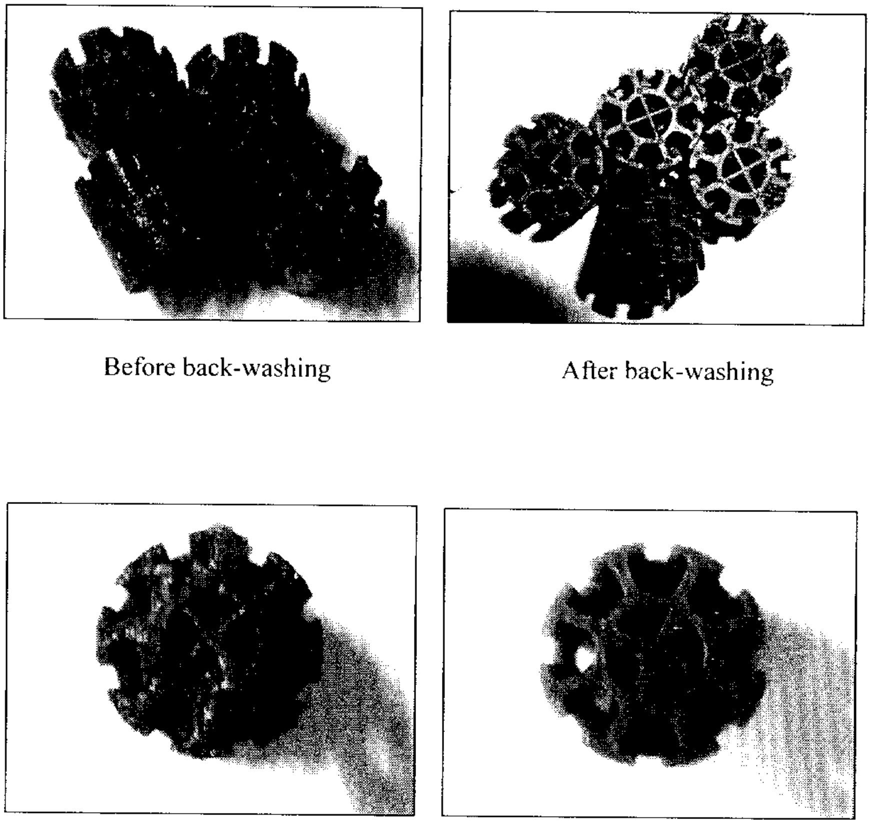
After back-washing

Before back-washing



After back-washing

Figure 4.15 Image of Aquacell media before and after back-washing in the case of Mode III and C/N ratio of 3



Before back-washing | After back-washing

Before back-washing | After back-washing

Figure 4.16 Image of Aquacell media before and after back-washing in the case of Mode III and C/N ratio of 5

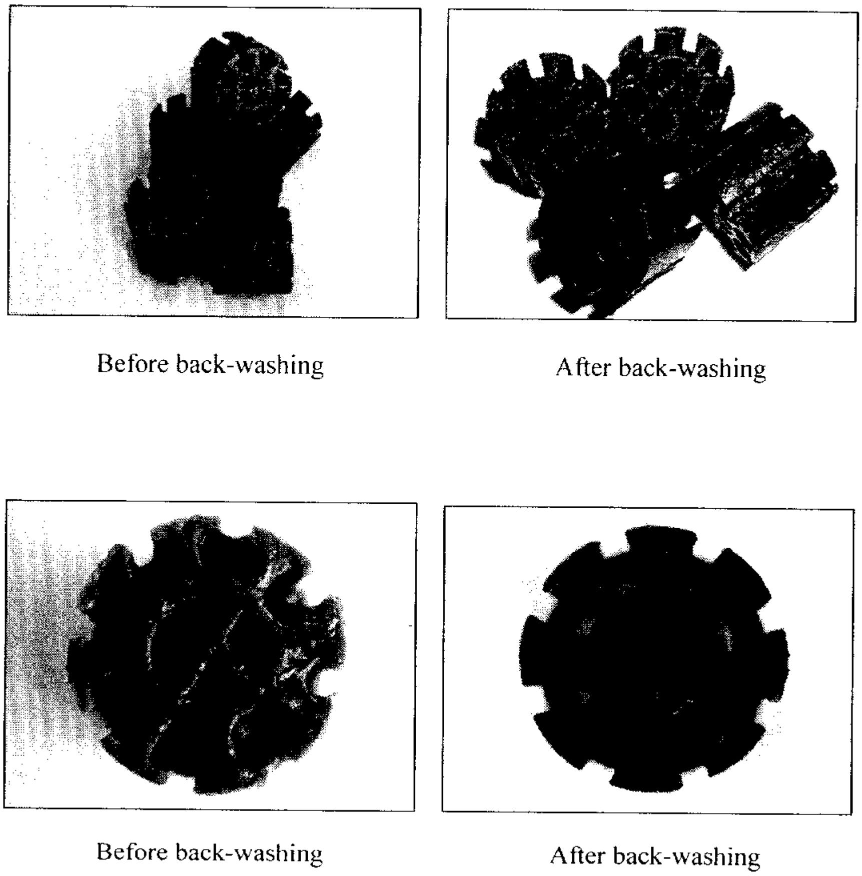
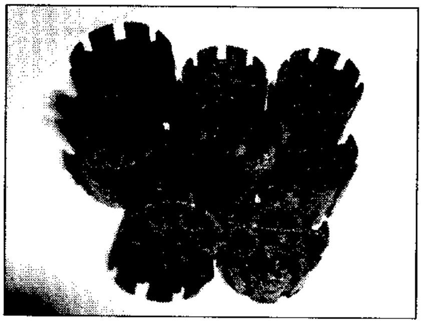
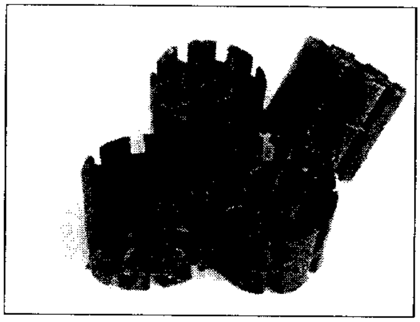


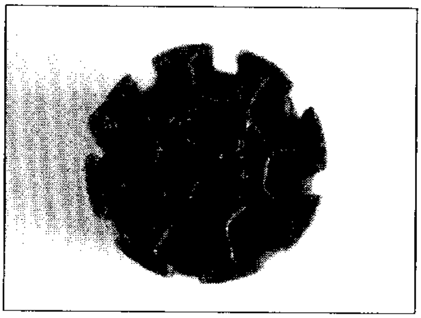
Figure 4.17 Image of Aquacell media before and after back-washing in the case of Mode III and C/N ratio of 7



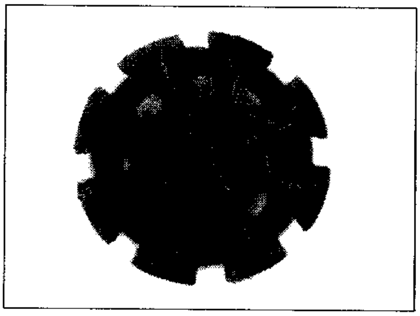
Before back-washing



After back-washing



Before back-washing



After back-washing

Figure 4.18 Image of Aquacell media before and after back-washing in the case of Mode III and C/N ratio of 10

5

Conclusions

This thesis addresses effects of the anoxic/aerobic time and C/N ratios on the performance of the Sequencing Batch Biofiltration Reactors (SBBRs). To design the operating condition, five process modes with the anoxic/aerobic ratio in succession of 0.1, 0.35, 0.5, and 0.75 were proposed, and laboratory scale reactors were operated with C/N ratio in succession of 3, 5, 7 and 10. The system was fed with the synthetic wastewater and operated at 20°C. The total influent nitrogen concentration was maintained constant while the COD concentration was varied to obtain the desired C/N ratio. The daily operating cycle of the SBBR reactor included two sub-cycles of 2.9 hours each, each one alternating anoxic and aerobic conditions, and the effluent was withdrawn at the end of the two sub-cycles. The length of the main cycle is 6 hours.

In this study, to achieve the objectives and goals in the demands at first, the above proposal to the anoxic/aerobic time and C/N ratios were considered in the best mode of operation. The anoxic/aerobic time ratio of process and C/N ratio of the influent wastewater were observed to have a substantial effect upon the performance of the SBBRs system. From this research, the amount of nitrogen removed by the system increased as C/N ratio increased. It was also shown that the effluent quality increased when the anoxic/aerobic time ratio was not high.

The following conclusions can be drawn from the wastewater characterization, parameters measured, lab-scale treatment, and designed process modes:

1. Mode III with the anoxic/aerobic ratio of 0.5 and the C/N ratio of 10 was defined the best condition for the SBBR performance on the above characteristics.
2. The result showed that treatment using the SBBRs was able to achieve 98% removal of COD as well as up to 97% removal of T-N, 98% removal of ammonia and 67% removal of T-P. These highest efficiencies were obtained at the C/N ratio of 10 with mode III operating condition having the anoxic/aerobic time ratio of 0.5.
3. The laboratory-scale treatment study especially demonstrated that the SBBRs are capable of attaining high nitrogen removal on wastewater with high C/N and quite low anoxic/aerobic time ratios.

The performance differences between the anoxic/aerobic time and C/N ratios stress the importance of treatability studies on C/N ratio of wastewater.

The demonstration from laboratory-scale experimental results as well as the achievement of high efficiencies was foundation to reference or correlation.

Finally, using the wastewater characterization and parameters measured, the sensitivity of the SBBRs to select suitable operating condition, as well as investigating effects of anoxic/aerobic time and C/N ratios on the rates of COD, phosphorus, and nitrogen removals was determined.

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