pH AS A REAL-TIME CONTROL PARAMETER
IN SWINE WASTEWATER TREATMENT

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ABSTRACT

The main objective of this research was to evaluate the effectiveness of pH as a real-time control parameter in swine wastewater treatment. This real-time control process was implemented into a Two Stage Sequencing Batch Reactor (TSSBRs) system. The first SBR is known as the Anoxic/Oxic (A/O) reactor. The real time process was applied here as most of the reactions occurred in this reactor. The second reactor is known as the anoxic reactor and was mainly designed for nitrate removal. The reproducibility of the pH patterns was tested under three different organic loadings. The treatment efficiencies for C, N, and P were examined to determine the capability of this newly developed process. The pH pattern was also compared to the more established ORP pattern for the purpose of cross-referencing. The differences between an attached growth system using Ringlace as a medium and a suspended growth system, both using the same real-time control strategy, were also examined in this research.

It was determined from the results of this study that pH can be a valuable parameter in a real-time control process. The pH patterns were consistent and uniform, despite the fluctuations in wastewater concentrations. The pH pattern clearly demonstrated its ability to monitor all the respiratory states. Several well defined control points on the pH curve such as the nitrate apex during the anoxic phase, and the Nitrogen Break Point (NBP) and Residual Carbon Manipulation Point (RCMP) during the aerobic phase were identified. The clarity of each feature makes real-time control using pH possible. It should be noted that the time allotted for the anoxic phase was four hours,

while the aerobic phase was adjusted by the real time control process with the control point set in between the NBP and RCMP.

The success of this real-time control process was marked by the flexible HRT provided for the system. It was found in this research that the application of the real-time process either ensured a complete removal of carbon and ammonia, or reduced the overall cycle time (maximized energy savings). A constant effluent quality was also produced for each run.

It was demonstrated in this study that the treatment efficiency for nitrate and phosphorus can be greatly affected by the carbon content in the wastewater. It was found that the optimal treatment efficiency for each nutrient (C, N, and P) was achieved during the high organic loading run. The high nutrient loading rate ensured rapid denitrification and phosphorus release during the anoxic phase, while creating a favourable environment for carbon oxidation, nitrification, and phosphorus uptake during the aerobic stage. It was also found that the anoxic reactor was not needed under conditions of high organic loadings. A high carbon content in the influent created simultaneous nitrification/denitrification in the aerobic phase. This reduced the nitrate loading into the anoxic reactor.

From the results of this research, it was found that pH pattern correlated well with the ORP curve. Each distinct feature identified on the pH curve coincided with the ORP pattern. The ORP pattern was affected more by the aeration rate than pH. Nonetheless, it can be concluded that both pH and ORP can be used as a real-time control parameter.

The results obtained from this research suggested that the only difference between the attached growth and suspended growth systems was the hydraulic retention time. The

higher biomass retained in the attached growth system enabled the bacteria to complete

the necessary reactions in a shorter period of time. The effluent quality produced from

both systems were virtually the same for each run.

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ACKNOWLEDEGMENTS

GLOSSARY OF TERMS

A/O .... Anoxic/Oxic

TSSBRs .... Two Stage Sequencing Batch Reactors

Bio-P .... Biological Phosphorus removal

DO .... Dissolved Oxygen

ORP .... Oxidation Reduction Potential

HRT .... Hydraulic Retention Time

SRT .... Solids Retention Time

RCMP .... Residual Carbon Manipulation Point

NBP .... Nitrogen Break Point

AGS .... Attached Growth System

SGS .... Suspended Growth System

C/N .... Carbon/Nitrogen

BNR .... Biological Nutrient Removal

PHA .... Polyhydroxyalkanoates

PHB .... Polyhydroxybutyrate

RBC .... Rotating Biological Contactor

BOD_5 .... 5 days Biochemical Oxygen Demand

COD .... Chemical Oxygen Demand

TOC .... Total Organic Carbon

TS .... Total Solids

TVS .... Total Volatile Solids

CHAPTER 1

INTRODUCTION

Serious environmental problems are associated with pig production. Due to the increasing demand of pigs by the public as a source of food, and hence the rising prices in the market, large volumes of manure are generated which often cannot be accommodated safely by the land in the vicinity of the operation.

Waste handling is one of the major problems resulting from the industrialization of swine production. The main factors affecting systems for the removal, transport, storage and utilization of the wastes are public health considerations and technical feasibility requirements including economic factors.

Swine wastewaters are often used as fertilizers in land irrigation. This leads to serious problems with land and water pollution. The surface run-offs and groundwater are often contaminated by swine wastewaters which contain a high concentration of organic carbon, nitrogen, and phosphorus. Regulations should be made in order to protect any water resource from swine wastewater contamination.

Traditional wastewater treatment systems such as activated sludge are not often used when dealing with swine wastewater. The variability in the concentration of the wastewater poses an enormous problem for this type of treatment system as they are often operated under a fixed time control. The quality of the effluent from the treatment system heavily depends on the concentrations of the influent (Benefield, 1977). The time that it

takes for the reactions to complete, therefore, depends on the influent concentrations. Hence a fixed time control system cannot guarantee a constant effluent quality.

In recent years, real time systems have been developed to control the wastewater treatment processes. The application of Oxidation Reduction Potential (ORP) to control carbon oxidation, nitrification and denitrification as a means of optimizing the system performance has been developed (Charpentier et al. 1987, Charpentier et al. 1989, Jenkins et al., 1989, Koch et al. 1985, Koch et al, 1988, Peddie et al, 1988, Peddie et al 1990, Saune et al, 1996, Yu et al. 1997).

In dealing with real-time control systems, a reproducible pattern from the controlling parameter (such as pH) must be observed before the system can be put into operation. Both the pH and ORP measurements can provide some significant information in an anaerobic/aerobic Biological Nutrient Removal (BNR) system. The Nitrogen Break Point (NBP) in the pH/ORP profile during the aerobic phase helps to determine and identify bacterial activity and transitions. A correlation between pH/ORP and nitrate concentration during the anoxic phase have also been found by other researchers. The distinct features that are showcased in the pH and ORP profiles will make it possible for real-time control to be successful.

The success of any BNR systems heavily depends on bacterial activities. The Solids Retention Time (SRT), therefore, is crucial to system performance. The ability for an attached growth system to retain more bacteria (comparing to a suspended growth system) often leads to a better removal efficiency in an BNR system.

This research involves the use of pH (a hypothesis by Chang-Six Ra) as a controlling parameter for the real time process of a Two Stage Sequencing Batch Reactor

(TSSBR). It will use the specific features provided by the pH curve to achieve real time control and thereby evaluate the effectiveness of pH as a control parameter. The removal efficiency between a suspended growth and an attached growth system will also be documented in this research.

CHAPTER 2

OBJECTIVES

This research was conducted with the objectives listed as follows;

1) To evaluate the effectiveness of pH as a real-time controlling parameter

    1 a) To determine the effect of different concentrations of wastewater on system performance

    1 b) To determine the reproducibility of the pH pattern under three different concentrations of wastewater

2) To compare the pH and ORP patterns under different concentrations of swine wastewater

3) To compare the treatment efficiencies between an attached growth (Ringlace) and a suspended growth treatment systems

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CHAPTER 3

LITERATURE REVIEW

3. 1 Characteristics of Swine Wastewater

The performance of a waste treatment system can be greatly affected by the characteristics of the wastewater. When dealing with swine manure, it is necessary to differentiate between the feces and urine. Many factors influence the quantity and the nutrient value such as BOD_5, ammonia, etc. of the wastewater. It has been found that manure production is proportional to animal age, weight, and feed intake. On the average, in the case of swine, the weight of manure produced is 1.75 kg/day/100kg of swine weight, and the moisture content is about 92% which gives the manure fluid characteristics (Fernandes, 1988). The typical BOD_5 value for swine manure ranges from 25,000-35,000 mg/L, with its COD value being 80,000-90,000 mg/L. The COD/BOD_5 ratio, therefore, is approximately 3, meaning that pig slurries have a high concentration of biodegradable materials. Animal housing, floor washing and the manure collection technology also influence the quality of manure.

Inadequate manure storage facilities, land spreading of manure on frozen soils, or application of land spreading in an excessive rate could seriously damaged the environment through surface runoffs or groundwater contamination. The high concentration of carbon, nitrogen and phosphorus and the high production rate of swine manure can affect the location of the barn, land application of the waste, or use of

traditional municipal waste treatment technologies. Table 3. 1 illustrated the composition of typical swine manure.

Table 3. 1 Manure production and characteristics per 1000kg live pig per day (A. S. A. E., 1992)

Parameter	Units	Production
Total Manure	kg/1000kg.day	84
BOD$_5$	kg/1000kg.day	3.1
COD	kg/1000kg.day	8.4
Total Solids	kg/1000kg.day	11
Volatile Solids	kg/1000kg.day	8.5
Urine	kg/1000kg.day	39
Total Nitrogen	kg/1000kg.day	0.52
Total Phosphorus	kg/1000kg.day	0.18
pH	kg/1000kg.day	7.5
Potassium	kg/1000kg.day	0.29
Calcium	kg/1000kg.day	0.33
Magnesium	kg/1000kg.day	0.070
Sulfur	kg/1000kg.day	0.076
Sodium	kg/1000kg.day	0.067
Chloride	kg/1000kg.day	0.26
Iron	kg/1000kg.day	0.016
Manganese	kg/1000kg.day	0.0019
Boron	kg/1000kg.day	0.0031
Molybdenum	kg/1000kg.day	0.00003
Zinc	kg/1000kg.day	0.0050
Copper	kg/1000kg.day	0.0012
Cadmium	kg/1000kg.day	0.000027
Lead	kg/1000kg.day	0.000084

3. 2 Fundamentals of Biological Wastewater Treatment

3. 2. 1 Removal of Organic Matter

The major objective in most biological treatment processes is the reduction of organic content in the wastewater. In accomplishing this type of treatment, the chemoheterotrophic organisms are of primary importance because of their requirement for organic compounds in addition to both carbon and energy source.

Aerobic processes are used effectively in wastewater treatment. In these processes a heterogeneous microbial community or sludge, composed mostly of bacteria, fungi, protozoa and rotifers, metabolizes organic matter in the presence of oxygen and the resulting end products are carbon dioxide, water and new cells.

Many organisms consume oxygen for survival. When molecular oxygen is used as the electron acceptor in respiratory metabolism, the process is known as aerobic respiration. Much of the products of aerobic respiration are in the form of carbon dioxide and water. The following equation is a simplified version of the reaction that have taken place under aerobic condition:

$$\textit{organic matter} + O_2 \rightarrow CO_2 + H_2O + \textit{energy} + \textit{cell mass}$$
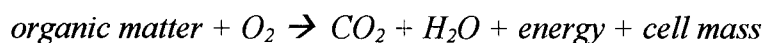
However, in order to form new bacterial cells, other forms of nutrients must be present as the organic matter only serve as the carbon source in the reaction. In a wastewater treatment system, the bacterial culture carries out the conversion in general accordance with the stoichiometry shown in the following two equations:

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Oxidation and Synthesis:

organic matter (COHNS) + O_2 + *nutrients* $\rightarrow CO_2$ + NH_3 + $C_5H_7NO_2$ + *other*

end products

Endogenous Respiration:

$C_5H_7NO_2$ + $5O_2$ $\rightarrow$ $5CO_2$ + $2H_2O$ + NH_3 + *energy*
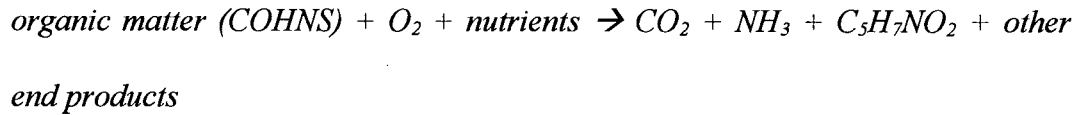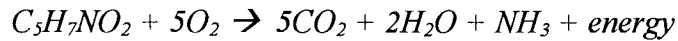
Endogenous respiration occurs when there is a lack of nutrients in the reactor which forces the microorganisms to consume their own cellular mass until they lyse and lose their viability and the ability for reproduction. When lysing predominates the cell nutrients are released into environment and become available to the other active organisms as substrate.

The principal significance of oxygen as the external electron acceptor is that aerobic heterotrophs oxidize the substrate completely and therefore the maximum amount of energy is generated. Whereas anaerobic microorganisms produce reduced compounds but the energy yield is much lower than aerobic process. Because of their fast growing characteristics, aerobic microorganisms and aerobic treatment have been preferred where reaction time is an important operating parameter.

3. 2. 2 Nitrogen Removal

The release of nitrogen into surface and ground waters at an excessive rate can lead to serious environmental problems. One major problem encountered with nitrogen was the direct depletion of DO in receiving waters. In-stream nitrification directly

consumes oxygen, while bio-stimulation of aquatic plant growth lowers oxygen indirectly when the plant growth dies and undergoes bacterial decomposition. Ammonia can also be very toxic to aquatic animals, while nitrate can cause severe public health concern as they are often related to issues such as cancer and blue babies syndrome.

To lower the nitrogen concentration discharged into the environment, a wastewater treatment system usually incorporated some kind of nitrification-denitrification mechanism for the removal of ammonia and nitrate. The first step of this process is to convert ammonia into nitrate under an aerobic environment. However, nitrogen has merely changed forms and not yet been removed. The second step then is to convert nitrate into nitrogen gas, a stable gaseous product, under an anoxic environment.

3. 2. 2. 1 Nitrification

Ammonia is oxidized to nitrate in the environment and in biological wastewater treatment by two groups of chemo-autotrophic bacteria which operate in sequence. Both groups of bacteria are autotrophic organisms as they derive energy for growth from the oxidation of inorganic nitrogen compounds. In contrast, heterotrophic bacteria derive energy from the oxidation of organic matter. Another feature of these organisms is that they use inorganic carbon (carbon dioxide) for synthesis rather than organic carbon. The first group of bacteria in this process of nitrification, represented principally by members of genus *Nitrosomonas*, oxidize ammonia to nitrite which is then further oxidized to nitrate by the second group, usually represented by members of the genus *Nitrobacter*. These oxidations can be represented as follows (Metcalf & Eddy, 1991).

For *Nitrosomonas*, the equation is

$$55NH_4^+ + 76O_2 + 109HCO_3^- \rightarrow C_5H_7O_2N + 54NO_2^- + 57H_2O + 104H_2CO_3$$
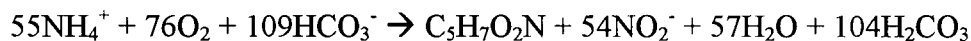
For *Nitrobacter*, the equation is

$$400NO_2^- + NH_4^+ + 4H_2CO_3 + HCO_3^- + 195O_2 \rightarrow C_5H_7O_2N + 3H_2O + 400NO_3^-$$
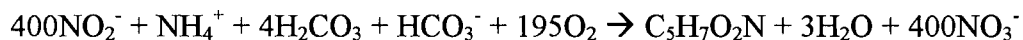
Energy released in these oxidations is used in synthesizing cell material from carbon dioxide. Two important aspects of the process which are evident from the above equations are the requirement for oxygen and the liberation of hydrogen ions. Oxygen demand in nitrification is an important feature in treatment plant aeration system design and may also be important in effluent receiving waters. Acid production in nitrification may result in depression of the pH in poorly buffered wastewaters and lead to loss of process stability.

3. 2. 2. 2 Denitrification

Denitrification is a process that involves microbial reduction of nitrate to nitrite, and subsequently to nitrogen gas. Denitrification takes place when there is an absence of molecular dissolved oxygen. Instead, heterotrophic bacteria uses nitrate as the electron acceptor to carry out their reduction reactions. Energy yield from anoxic respiration is much lower than that from oxygen respiration. An external carbon source is also necessary for the maximum reaction rate, although the process will proceed, at a reduced rate, by 'endogenous respiration'. Carbon sources used include settled wastewater and carbohydrate wastes but methanol has often been preferred for nitrogen removal because

it has been relatively cheap, readily available and permits simpler process control. The stoichiometric equations for denitrification using methanol as the carbon source is illustrated by the following equations (McCarty et al., 1969).

For nitrate removal:

$$NO_3^- + 1.08CH_3OH + H^+ \rightarrow 0.065C_5H_7O_2N + 0.47N_2 + 0.76CO_2 + 2.44H_2O$$

For nitrite removal:

$$NO_2^- + 0.67CH_3OH + H^+ \rightarrow 0.04C_5H_7O_2N + 0.48N_2 + 0.47CO_2 + 1.7H_2O$$
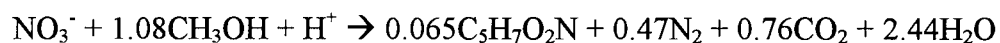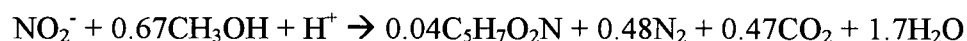
Unlike nitrification, a broad range of bacteria can carry out denitrification. Denitrifiers are readily available in the environment because of their facultative nature. In another words, most denitrifiers can use either oxygen or nitrate as their terminal electron acceptor. Denitrifiers can survive in aerobic systems because of their ability to use oxygen and efficiently oxidize organic matter. The abundance of denitrifiers minimizes the need to create favorable conditions for their survival, as must be done for nitrifiers.

3. 2. 3 Phosphorus Removal

Increased algal and plant growth caused severe damage to the water quality of a number of lakes and rivers. High phosphorus content in these water has been cited as the cause for such changes. Incompetent sewage treatment system often discharges effluent at a very high phosphorus concentration. Many physical and chemical tertiary treatment methods for phosphorus removal are too costly at the present time. Thus biological methods of phosphate removal have been investigated heavily in the past decade.

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Phosphorus is an essential nutrient for cell synthesis by bacteria. Phosphorus usually appears in three different forms: ortho-phosphate (PO_4^{-3}), polyphosphate (P_2O_7), and organically bound phosphorus. Microbes often utilize approximately 10 to 30% of the influent phosphorus for cell synthesis and energy transport. However, uptake beyond their normal levels is required in order to achieve a low effluent concentration levels. *Acinetobacter* are one of the principal organisms responsible for the removal of phosphorus. Other researches found that *Aeromonas* and *Pseudomonas* (Lotter, 1985), *Bacillus cereus* (Hascoet et al., 1984), and *Pseudomonas vesiculcris* (Suresh et al., 1984) are also capable of polyphosphate accumulation.

The generally accepted theory for biological phosphorus removal is to expose the microbes to alternating anaerobic and aerobic conditions. This process is known as "luxury uptake", which expresses the ability of a biological sludge to uptake phosphorus that exceeds the 2.3% phosphorus by weight typical of conventional wastewater treatment system biomass, when submitted to cyclic anaerobic/aerobic conditions (Randall et al., 1997). Fermentation products and sufficient organic carbon must be present during the anaerobic phase in order for rapid phosphorus release to take place. Figure 3. 1 illustrates the mechanism for biological phosphorus removal. The basics of biological phosphorus removal model is as follows. Volatile fatty acids and other products are produced from fermentation in the anaerobic zone. These fermentation products are generally derived from soluble portion of the influent BOD as there is often not sufficient time for the hydrolysis and conversion of the influent particulate BOD. These fermentation products are preferred and readily assimilated and stored by the microorganisms capable of excess biological phosphorus removal (E.P.A, 1987). It was found that the organic matter was

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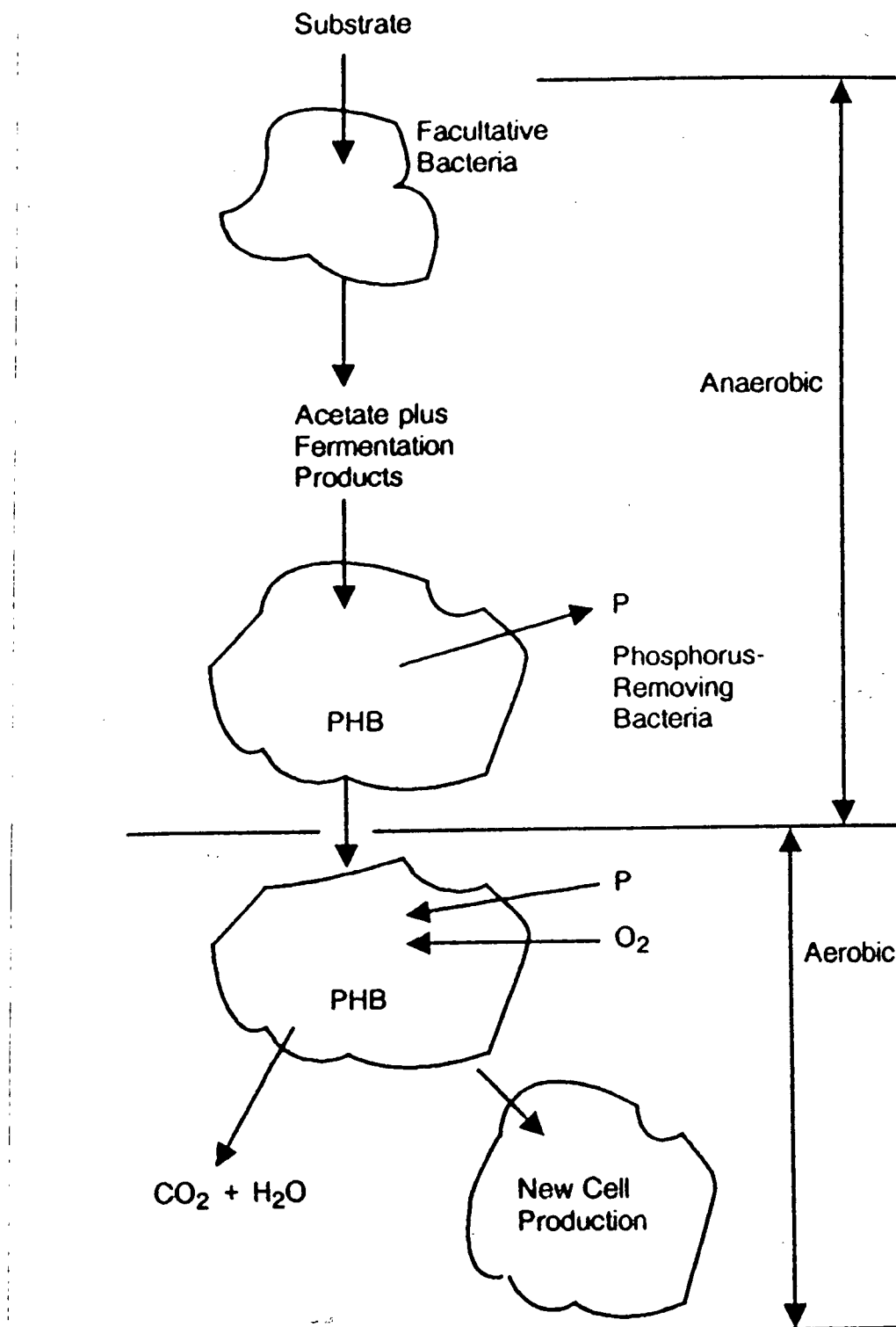
Figure 3. 1 Mechanisms for biological phosphorus removal

stored as polyhydroxyalkanoates (PHA), of which polyhydroxybutyrate (PHB) is the best known member. It is thought that the energy required for the storage of PHA is produced by phosphorus accumulating bacteria by decomposing polyphosphate from an intercellular store. As a result, the phosphorus accumulating bacteria will release phosphate in connection with the storage of organic matter. During the subsequent aerobic phase, the PHAs are metabolized and the energy derived from them is stored as polyphosphate. There are two different models presented by Comeau et al, 1986, and Arun et al., 1989 about the reducing power needed to synthesize PHA from fermentation products. In the model presented by Comeau, the reducing power is supplied by the anaerobic operation of the tricarboxylic acid cycle (TCA cycle). In the model presented by Arun, reducing power is supplied by the utilization of glycogen.

It is accepted widely that the introduction of nitrate to the anaerobic zone is detrimental to the phosphorus removal process because nitrate will be utilized as an electron acceptor for the growth of non-poly-P heterotrophs. This reduces the amount of substrate available for the polyP organisms, which in turn reduces the amount of P removal that can be achieved. However, it has been found by other researches that phosphorus uptake by polyP organisms does occur in the anoxic zones, indicating the polyP organisms are capable of denitrification (Hascoet et al., 1985, Malnou et al., 1984, Comeau et al., 1986). It was found that nitrate can serve as an electron acceptor for the oxidation of stored PHB; however, indications are that not all polyP organisms have this ability. It was also found that nitrate is not as efficient as oxygen for P uptake. Batch tests indicated that more stored carbon (PHA) is utilized for a give amount of P taken up

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when nitrate is the electron acceptor in place of oxygen (Gerber et al., 1987, Osborn and Nicolls, 1978).

3. 3 Sequencing Batch Reactor

The Sequencing Batch Reactor (SBR) is an activated process designed to operate under non-steady state conditions. The process involved in the SBR and conventional activated sludge systems are similar. They both carried out aeration and sedimentation/clarification. The major difference between the two systems is that the SBR operates in a batch mode with aeration and sludge settlement both occurring in the same tank (in time sequence), whereas the conventional systems rely on different tanks to carry out different functions in a continuous manner. In addition, the SBR system can be designed with the ability to treat a wide range of influent volumes whereas the continuous system is based upon a fixed influent flowrate. Thus, there is a degree of flexibility associated with working in a time rather than in a space sequence.

The SBR systems are usually operated in five stages (see figure 3. 1). It is usually carried out in sequence as follows: fill, react (aeration), settle, draw, and react.

Fill - The reactor is gradually filled with the influent over a set time or volume. Mixing and/or aeration can be supplied during this period.

React - The reaction that begins during the fill period is completed during the react period. The SBR can be operated to achieve any combination of carbon oxidation, nitrogen reduction, and phosphorus removal. Depending on the

Influent

Fill   Well Mixed
Air On/Off

React   Well Mixed
Air On

Air Off

Settle

Sludge

Air Off

Draw

Effluent

Sludge

Air Off

Idle

Wasted
Sludge

Sludge
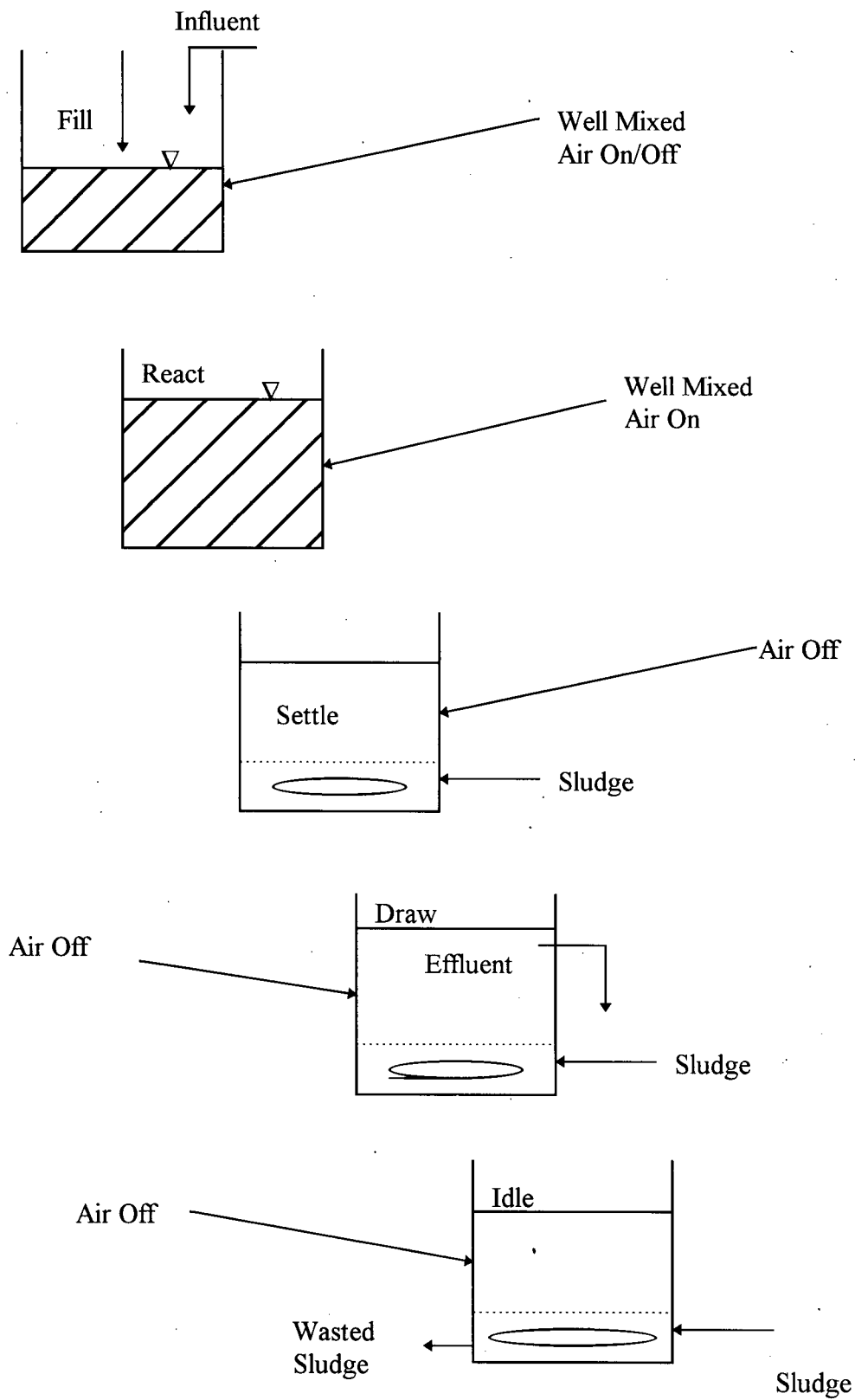
Figure 3. 2 Typical Operating Sequence for a SBR

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objective, aerobic/anoxic environments can be applied. Aeration is not provided if the objective is to achieve denitrification and phosphorus release. The presence of oxygen is often found for nitrification, BOD removal, and phosphorus uptake. After the substrate is consumed famine stage starts. During this stage some microorganisms will die because of the lack of food and will help reduce the volume of settling sludge. The range of cycle times for this period is approximately 35-80% of the total cycle time. Towards the end of this period, sludge wasting can be done to maintain the sludge age. The sludge age in days would be equal to the reciprocal of the fraction of the maximum volume wasted each day.

Settle - In this period, separation of liquid and solids is carried out under quiescent conditions (aeration terminated). No liquids should enter or leave the tank to avoid turbulence in the supernatant. The time for settling is usually between 0.5 to 1 hour.

Draw - The clarified supernatant is discharged during this stage. Decanting must be done without disturbing the settled sludge. The percent of the cycle time can range from 5 to more than 30%.

Idle - The idle period is used whenever the influent wastewater flow is irregular. It can also function as a pause in a multi-tank system, by providing time for one reactor to complete its full cycle before switching to another unit. The length of time in idle will be determined by the wastewater flow rate pattern. Provision for aeration mixing and sludge wasting are optional in this stage. In this period, the

microbial population is maintained by endogenous respiration and can be readily activated by the incoming wastewater during the fill period.

Sludge wasting is an important step in the SBR operation that greatly affects performance. Wasting is not included as one of the five basic process steps because there is no set time period within the cycle dedicated to wasting. The amount and frequency of sludge wasting is determined by performance requirements, as with a conventional continuous-flow system. Since both aeration and settling occur in the same chamber, no sludge is lost in the react step, and none has to be returned from the clarifier to maintain the sludge content in the aeration chamber.

SBR systems have been applied to the treatment of wastewater from various sources with good results. The advantages that SBR have over the conventional treatment systems are: 1) reliable effluent under shock loading and fluctuating influent concentrations, 2) high oxygen transfer efficiency, 3) nitrogen removal via nitrification/denitrification, 4) phosphorus removal without addition of chemicals, and 5) less reactor volume.

There are certain disadvantages of using SBR as a treatment system. The system is not as reliable as the unit gets larger because of the difficulty to install timing devices and sensors for control purposes. Also clogging of aeration devices can become a major problem during the settling, drawing, or idling stages.

3. 3. 1 Application of SBRs into swine wastewater treatment

The SBR system has been successfully used in the treatment of swine wastewater. The results from these researches suggested that SBR can reduce swine wastewater pollution into the environment. They also show excellent carbon and nitrogen removal efficiency. Anaerobic SBR can also be applied for reduction of COD and VS. The following are a few examples of SBR used in swine wastewater treatment.

The effects of nitrogen removal under aerobic and anoxic conditions were investigated (Fernandes et al., 1991). Laboratory studies were conducted on a SBR for the treatment of screened liquid swine manure. The SBR was operated on the basis of a 24 hour cycle at 6 and 9 days hydraulic retention time and 20 days solids biological retention time. Nitrification and denitrification processes were carried out in the same reactor by alternating aerobic/anoxic conditions, and the tested time ratios were 19/3, 16/6, 14/8, and 10/12 (hours). For reactors operating at 6/20 and 9/20 (HRT/SRT) with an anoxic fill plus react of 3 to 8 and 3 to 12 hours the removal of TSS, COD, and inorganic nitrogen ranges from 95 to 97%, 94 to 98%, and 80 to 93%, respectively. It was found that the anoxic operation did not adversely affect the activity of the nitrifiers. Based on the overall performance it is not recommended to operate the SBR for more than 8 hours anoxic, between fill and react sequences, because the sludge settling process deteriorates and odour can be a problem. It was also found from this study that air supply could be reduced by about 42%, without affecting process performance.

Fernandes and McKyes, 1991, tested different strength influents, measured in terms of COD, of swine manure (9500, 19000, 28500, 38000 mg/L). A mathematical

model was developed to describe the changes in concentration with time of COD, ammonia, nitrate, and nitrite. Total cycle time was 24 hours: fill (3h), react(19h), settle (1h), draw(0.5h), and idle(0.5h). The results showed that the single reactor SBR process is capable of reducing the potential polluting carbon and nitrogen components of a concentrated wastewater to a high degree when it is operated with 7 to 9 days of HRT and 20 days of SRT. The prediction of COD removal in the fill periods showed good agreement with observed values but this study failed to reproduce the prediction using their data and model and proposed coefficients. Overall efficiency of COD removal was 96%.

A study was done to examine the effect of temperature (5, 10, 21°C) on the performance of a SBR treating liquid swine manure (Fernandes, 1993). The HRT and SRT were set at 9 and 20 days respectively. The SBR performance was seriously affected at an operating temperature of 5°C, but at 10°C a significant improvement in the system performance was achieved. There wasn't much difference between the 10 and 21°C results. The COD and NH₄-N removals were 96.7% and 97% at 10°C and 97.3% and 99.8% at 21°C, respectively.

A full scale system consisting of 3 SBRs to treat diluted swine wastewater was investigated (Lo et al., 1991). The reactor design was based on the results of a bench-scale study of swine wastewater. The cycle consisted of the following: 6 minutes of aerated fill, 200 minutes of reaction time, 30 minutes of settling, and 6 minutes for decanting. The entire cycle was 4 hours long. The volume of wastewater treated and discharged each cycle amounted to 1,400L. The SRT was maintained at 14 days after the

15th week. The results indicated that very high BOD_5 removal of 91% was achieved. However, high nitrogen removal efficiency was not observed.

Anaerobic sequencing batch reactor (ASBR) has also been studied by several researchers. (Schmit et al., 1993, Degue et al., 1992). The researches investigated the effect of temperature on swine wastewater treatment. The difference between this and a regular SBR is that no aeration was provided throughout the entire process. It was found that ASBR is capable of sustaining volatile solids destruction over VS loads of 0.9 to 5 g/L/d between 40% to 60% at a temperature of 20°C. This ASBR reactor is capable of adapting to lower temperatures by holding solids longer at lower temperatures. The adaptive ability is less from 25°C to 20°C than from 35°C to 25°C. The ASBR demonstrated a great ability in COD and VS reduction. They concluded that the ASBR is capable of achieving much higher degrees of swine waste conversion to biogas than is possible with conventional completely mixed reactors.

3. 4 Application of Real Time Control System

It is necessary to develop a system to achieve a complete removal of carbon, nitrogen, and phosphorus from wastewater. The fluctuations in the concentrations of wastewater often lead a poor system performance using a conventional steady state (fixed time) system. Therefore, a more dynamic system is needed to secure and optimize the aeration periods so that their length is adjusted to the influent load variations. The incapacity of Dissolved Oxygen (DO) monitoring to give information during anoxic and anaerobic periods created a new interest for the ORP and pH measurements. Real time

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control system have been applied to many wastewater system. DO and ORP monitoring system had been investigated intensively by other researchers and are more established, while the use of pH as a real-time control parameter is still in the investigating stage.

3. 4. 1 Oxidation-Reduction Potential

Since DO is pretty self explanatory and very well established, and pH as a real-time control parameter is still in an investigating stage, only ORP will be discussed in details in this section, as numerous researches have been done to investigate ORP as a real-time control parameter. The purpose of this section is to introduce the readers to some background information so to familiarize them with the ORP parameter.

The utilization of chemical energy in living organisms involves oxidation-reduction reactions. Chemically, an oxidation is defined as the removal of an electron or electrons from a substance. A reduction is defined as the addition of an electron or electrons to a substances. Substances vary in their tendencies to give up electrons and become oxidized or to accept electrons and become reduced. This tendency is expressed as the oxidation reduction potential. This potential can be expressed by the Nernst equation:

$$E_h = E^o + \frac{RT}{nF} \ln \frac{(Ox)}{(\text{Re}d)}$$

where E_h = the potential referred to the normal hydrogen electrode

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E°   = the standard potential of the system at 25°C when the activities of all

reactants are unity

R   = universal gas constant (8.315 volt-joules coulombs)

T   = temperature (degree Kelvin)

n   = number of moles of electrons transferred

F   = Faraday constant (23,061 cal/mole-volt)

Ox   = Oxidized species

Red   = Reduced species

In most complex biological systems, such as encountered in wastewater treatment applications, the many chemical and biological oxidation-reduction reactions taking place are not in equilibrium and the observed ORP cannot be interpreted thermodynamically. The observed ORP represents the net electron activity of all the redox reactions taking place and thus indicated the general oxidative status of the system. ORP is a highly sensitive and instantaneous on-line instrumentation in the aeration tank. It reflects the amount of materials such as DO, organic substrate, activity of organisms and some toxic compounds. It can also indicate some operation conditions such as overloading, under-loading, over-aerating, and under-aerating. Researchers have found that ORP is much more sensitive at very low oxygen levels as the ORP varied linearly with the log of oxygen concentration, suggesting the ORP is much more sensitive when the DO level is low (Peddie et al., 1990). Koch & Oldham, 1985, were also able to determine that a linear relationship exists between ORP and nitrate concentration in the anoxic zone, and a similar

relationship between ORP and concentration of phosphate also appears in the anaerobic and anoxic zones.

ORP has been used for monitoring and control purposes recently. Due to the inability of DO measurements to monitor any anaerobic process, ORP has been used extensively to monitor and control anaerobic fermentation processes such as beer making and methane production (Ishizaki et al., 1974, Koch & Oldham, 1985). Methane production was found to be optimized at ORP levels between -500 and -520 mV (Koch & Oldham, 1985). Also many aerobic fermentation processes take place at a DO concentration that can not be accurately measured by commercial oxygen probes (Kjaergaard, 1977). Other research involved the finding that optimal production of specific amino acids occurred at different ORP levels between -265 and -220 mV (Radjai et al., 1984).

The application of oxidation and reduction potential in wastewater treatment system for control has not been adopted until recently (past two decades). The difficulty in getting reliable ORP measurements and the impossibility of unambiguous interpretations are the reasons for such a late development in this field of research (Koch and Oldham, 1985). Many factors would affect the ORP readings. The absolute values of ORP in the system can be affected by parameters such as chemical species, biological activities, pH, temperature, etc. All of these factors may also vary within a given time frame. Therefore, it is impossible to have a control strategy based on the absolute values of ORP. Instead, the ORP profile is used to determine the control process for the wastewater treatment system.

3. 4. 2 Preferences in Choosing a Real-Time Control Parameter

Using different control parameters in a biological process will yield different results. Among real-time parameters such as DO, ORP and pH, DO, in particular when used with oxygen uptake rate data, has been the most meaningful and widely used parameter in biological wastewater treatment. As mentioned before, however, the shortcoming of DO in a aerobic-anaerobic process is evident: DO cannot indicate any respiratory states beyond aerobic conditions. Therefore, there is a new interest in developing a parameter that can reflect on all the respiratory states (aerobic, anoxic, and anaerobic).

Numerous researches have been done to investigate ORP as a real-time control parameter in alternating aerobic-anaerobic process. In fact, the features on the ORP profile have been used to control the length of the aerobic and anoxic cycles in a waste or wastewater treatment process (Wareham et al., 1993, Sasaki et al., 1993). In contrast, pH has seldom been solely used as a control parameter for biological processes. There are certain advantages of using pH over ORP as a control parameter. It has been noted that ORP values depended on the initial treatment of platinum probes (Heduit et al., 1992). Therefore, the values were found to be dependent on the surface characteristics of the metal. Also, the sensitivity of the ORP probes can create all sorts of problems for the treatment process. An entire treatment system consists of many components such as electrical mixers, aerators, etc. Such equipment can generate electrical fields that can disturb the readings from the ORP probes. This will lead to a fluctuation in ORP readings, which makes the control process impossible. Another disadvantage of using ORP as a

monitoring parameter is that the ORP values observed are very site specific. pH does not suffer from such shortcoming as a given pH value has the same meaning regardless of the type of probe or system measured. This allows the use of specific pH values as a control parameter. In addition, ORP is only an indication of the overall oxidative-reductive state of the system; the absolute ORP value per se does not have any process significance. Data obtained from the pH pattern, on the other hand, particularly when used with alkalinity, may be used to better correlate the process, offer more meaningful information for process evaluation, and enhance current understanding of the process mechanisms. In addition, the changes in pH are well defined by denitrification and nitrification reactions, while the change on ORP curve cannot be directly related to the process changes. Also, ORP values are subject to change due to other factors and therefore may not be consistent. It can be seen from Al-Ghusain et al., 1995, that the absolute ORP values rapidly changed within a period of 40 hours. Another advantage of using pH over ORP as a control parameter is that the ammonia valley (or nitrogen break point) can easily be observed on the pH curve despite imbalances between aeration and oxygen demand. The ORP curve cannot identify this specific point under conditions of excessive-aeration, which can result in control failure.

An advantage of using ORP is that it can reduce the operating costs of a system. When the nitrogen break point is not observed on the ORP curve, it indicates that the system is under an excessive aeration condition. If this condition is seen by the operator, the aeration rate can be decreased which optimizes the aeration costs.

3. 4. 3 Application of Real Time Control in Environmental Research

The behavior of nitrogen and phosphorus during batch aerobic digestion of waste activated sludge was compared between continuous aeration and intermittent aeration by control of DO (Matsuda et al., 1988). Waste activated sludge samples from a municipal wastewater treatment plant were digested under 30 days batch operation. If the cycle time of intermittent aeration was decided so that the DO in a digestion tank did not become zero, the reduction rate of sludge solids and the behavior of nitrogen and phosphorus in the intermittent aeration were similar to those in the continuous aeration. This illustrated that the control of aeration by DO monitoring reduced the power cost for aeration but produced the same results (without control) at the same time.

The investigations of using ORP as a real-time control parameter were done by several researchers (Peddie et al., 1990, Plisson-Saune et al., 1996). The characteristics of the ORP curve were also studied using aerobic/anoxic sludge digesters (Peddie) and a low-load activated sludge wastewater treatment plant using intermittent aeration (Saune). Features on the ORP curve such as the nitrate knee in the anoxic phase and the nitrogen break point in the aerobic phase were discovered. These distinctive features were directly related to the changes in the system chemistry and biological activity. The profiles discovered by both researchers were reproducible and can easily be implemented in a real-time control system as each feature on the graph has a obvious slope change. These researches concluded that ORP is a better monitoring tool than DO because of its ability to monitor both the aerobic and anaerobic respiration. Peddie concluded that ORP is directly related to low DO concentrations and nitrate. The reproducibility of the ORP

profiles and sensitivity of the measured potential to changes in biological or chemical activity appears to make it ideal parameter for automated monitoring and process control purposes.

Some authors have described control methods based on the use of ORP absolute values (Charpentier et al., 1987, Charpentier et al., 1989, Lo et al., 1994). When a fixed low level is reached (denitrification) aeration is switched on, an stopped when a fixed high level is obtained (nitrification). In Charpentier's first study, the variations in ORP were correlated with the effluent ammonia and nitrate concentration in a low load activated sludge process. Subsequently, two ORP setpoints (-80 to +120 mV) were targeted and the aeration cycle was controlled by these ORP setpoints. In this study, constant quality of effluent was obtained, determining the electricity consumption for aeration. The second Charpentier study proposed that the targeting of upper and lower ORP setpoints in the aeration cycle simultaneously optimized the effluent quality and electricity consumption. In Lo's research there was discussion about the effect of different ORP setpoints on the performance of lab-scale extended aeration treatment system. The system exhibited excellent carbon, nitrogen, and phosphorus removal. It was concluded that the optimum ORP setpoint for total nitrogen removal is 110 mV.

More dynamic control has been developed by several researchers (Wareham et al., 1993, Sasaki et al., 1993). Sludge digestion using ORP regulated aerobic-anoxic cycles were studied. The real-time control strategy involved in that process relied on a computer to detect the disappearance of nitrate in the system by recognizing the nitrate knee on the ORP curve. This strategy was compared with a fixed time procedure where the anoxic period was set for 3 hours. Comparison between the two systems revealed that the ORP

regulated reactor removed slightly more nitrogen and solids than the reactor with the fixed cycle length. In addition the ORP regulated reactor was better accommodated to several disturbances which influenced the time of nitrate disappearance. Sasaki's approach was a little different than the traditional dynamic ORP control method. His research involved an anaerobic-aerobic activated sludge process. He used the nitrate break point as the control index to adjust the aerobic, anoxic, and anaerobic periods. After aeration was performed for a predetermined period, the time when nitrate break point appeared on ORP curve was calculated and used a predetermined value of next cycle. His results showed a high nutrient removal efficiency.

On-line monitoring of pH for control purposes is still in the investigative stage. Some research has indicated that pH can be a very effective real-time control parameter (Yu et al., 1997, Ra et al., 1998, Al-Ghusain et al., 1995). In Yu's research, he suggested that the ORP and pH profile that he obtained from a fixed-time control study could accurately represent the dynamic characteristics of the system. The control process was applied to a continuos-flow activated sludge batch reactor. The results showed that the measurement of ORP and pH can accurately detect the ends of nitrification and denitrification. The ORP and pH profiles were used to establish a real-time control strategy to determine the transfer of operation stages. The breakpoints on the ORP and pH curve were used as a cross-reference utility to verify the control points. He concluded that the real-time system showed a better nitrogen removal efficiency than the fixed-time control. Also, the overall cycle time was reduced by approximately 35% and 42% of the energy for aeration was saved using the real-time control system.

In Ra's research, a detailed real-time control strategy using either ORP or pH was mentioned in the discussion. A real-time control process was implemented into his newly developed two stage sequencing batch reactor process to treat swine wastewater. He suggested that although the two curves are slightly different, the control points identified (Nitrogen Break Point (NBP) and Residual Carbon Manipulation Point (RCMP)) are equivalent with respect to identification of the chemical and biological state of wastewater, and therefore equivalent in terms of their use in the control strategy. In his research loading rate control was also achieved by using an ORP set-point. The rationale behind the loading rate control was to achieve a flexible loading volume of influent, depending on the influent quality and the oxidation state of the reactor, to optimize the system's performance. He found that the decrease in ORP was proportional to the rate of denitrification. By applying the loading rate control, sufficient soluble carbon can be assured for denitrification and phosphorus release. The removal efficiency of over 95% in carbon, nitrogen and phosphorus demonstrated the effectiveness of a real-time control system.

In Al-Ghusain's research on sludge digestion, he preferred using pH as a real-time control parameter over ORP because of the ease of identifying the ammonia valley (same as NBP in Ra's research) using pH. He indicated that the ammonia valley and the nitrate apex (same as the nitrate knee) can be used as a control point. The results of this alternating aerobic-anoxic sludge digestion experiment using upper and lower pH limits of 8 and 6 yielded excellent performance in terms of VSS destruction, energy saving, overall system pH stability, and lower nutrient concentrations in the supernatant.

3. 5 Attached Growth Systems

In general, wastewater treatment systems often uses suspended biomass to carry out the necessary reactions for effluent discharge. However, the coexistence of different microorganisms presents a competitive relationship for oxygen between the phosphate accumulating bacteria and the nitrifying bacteria. Researchers have found that only partial nitrification was obtained when an excellent phosphate removal occurred and vice versa in a SBR system with suspended growth (Hang-Sik et al., 1993). It was reported that efficient phosphate removal and complete nitrification using a biofilm SBR was achieved. An attached growth system offers the following advantages: 1) higher solids retention time; 2) elimination of long sludge settling periods; 3) coexistence of aerobic and anoxic metabolic activity in the same ecosystems.

An attached growth system basically involves providing a solid support medium on which biomass develops in thin layers. The wastewater will come into contact with the support media. The outermost biomass within the medium will receive much of the nutrients and oxygen available in the wastewater. Organic loading into the system is really important to the diversity of bacteria population because heterotrophic bacteria have a faster growth rate than autotrophic bacteria and can successfully out-compete autotrophs for oxygen under conditions where soluble carbon is in abundance. A high organic loading will increase the heterotrophic bacterial population attached onto the medium. The inner layer biomass heavily depends on diffusion in order to obtain nutrients and electron acceptors for survival. The bacteria in the inner layers are more capable of making use of the by-products from the outer layer bacterial metabolism. The inner layer

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bacteria are mostly anaerobes or depend on nitrate as a electron acceptor for cell synthesis. The entire biomass in all the layers is a diverse array of species and not easily characterized.

Attached growth systems are usually mentioned along with aerobic processes. There are mainly three types of attached growth systems; the trickling filter, rotating biological contactor (RBC), and submerged media carrier. The operation of a trickling filter involves wastewater percolating through a stationary medium to which microorganisms are attached onto. The RBC consists of a series of closely spaced circular disks that are submerged in wastewater and rotated slowly through it. In operation, bacteria are attached onto the surface of the disks. The rotation of the disks forced the biomass to uptake the nutrients while in the wastewater, and then the oxygen in the atmosphere when the disks rotated out of the mixed liquor. The submerged media carrier is a stationary process where the media is completely submerged in the process mixed liquor. The bacteria over time will attach themselves onto the medium. Nutrients and dissolved oxygen for cell synthesis are then obtained from the surrounding mixed liquor.

3. 5. 1 Application of Attached Growth Systems

Various research has been carried to investigate biological nutrient removal using an attached growth system. The possibility of joining biological phosphorus and nitrogen removal in a biofilm SBR was studied using an operation strategy with four reaction phase: Anaerobic/Aerobic/Anoxic/Aerobic (Garzon-Zuniga et al., 1996). A 1,000L pilot scale reactor, filled with Pall-Rings as biofilm support was fed with municipal wastewater.

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The system worked successfully obtaining removal of COD, phosphates and ammonia nitrogen of 89%, 75% and 87%, respectively. He concluded that the high removal efficiencies of N and P were attributed to the balance of population between nitrifying bacteria and phosphate accumulating bacteria.

Another biofilm pilot SBR was studied to determine the effects of different operation strategies on the capacity of the biofilm (Pall-Rings) to remove C, N, and P (Munoz-Colunga). The wastewater was enriched with a molasses and phosphate solution when needed to vary the nutrients concentration. The treatment cycles were adjusted with four stages: filling, anaerobic phase, aerobic phase and draw of treated wastewater. Cycle duration of 8 and 12 hours were tested with different anaerobic/aerobic time ratios. The highest COD and PO_4-P removal rates were obtained with 12 hours cycles and phase duration of 37/63 percent anaerobic/aerobic. When the organic loading rate was higher than 5 $gCOD/m^2d$ the activity of the phosphate accumulating bacteria and nitrification could not be observed. The best results regarding phosphate removal and nitrification were obtained when the mean organic load was 3 $gCOD/m^2d$.

The use of biofilm was also studied in a continuous system. A lab scale experiment using a combined biofilm and activated sludge process to enhance biological nitrogen and phosphorus removal was examined (Liu et al., 1996). In the system, fibrous carriers were packed in the anoxic tank for the attached growth of denitrifying bacteria and the sludge of the clarifier was returned to the anaerobic tank to release phosphate. An average removal efficiency of 75% total nitrogen compounds, 92% total phosphorus and 88% COD from domestic wastewater was achieved. It was concluded that biofilm denitrification was very effective and its operation was relatively simple. This

demonstrated that the ability of an attached growth system to survive under a continuous anoxic phase.

A full scale test on Ringlace was done to study the effects of media positioning within the aerobic section of a plug flow nitrogen removal process (Sen et al., 1993). A similar test using the same process but with suspended biomass was also done to compare the differeneces between the ammonia levels at succeeding points along the aeration section before and after the Ringlace. These tests showed, under optimum concentrations of ammonia, nitrification in the process with Ringlace increased by 60% over the suspended growth control system.

CHAPTER 4

MATERIALS AND METHODS

A system known as a Two Stage Sequencing Batch Reactor (TSSBRs) was designed for this research in order to remove the high concentrations of carbon, nitrogen, and phosphorus in swine wastewater. A real time system was implemented as part of the control strategy for the entire process. Two different systems were set-up for this specific project. An attached growth system that utilized Ringlace as its growth medium was compared with a suspended growth system. Various investigations were done to determine the effectiveness of pH as a control parameter in this real-time automated systems. The efficiency of each system was also evaluated and compared. The operating conditions and strategies used for this system are described in the following in detail.

4. 1 Experimental Design and Setup

The components of the batch reactor used for this study are shown in figure 4. 1. Both the main (A/O) reactor and post (Anoxic) reactors were made out of Plexiglas. The A/O reactor for both the attached growth and suspended growth systems was approximately 2 ft in height and 8.5 inches in diameter. For the suspended growth system, the A/O reactor had a working volume of 12L. The attached growth system had a working volume of only 10.5L as the Ringlace displaced the liquid up to the 12L mark. The anoxic reactor for both systems was approximately 1 ft tall and 7.5 inches in diameter.

feed

A/O
reactor

∇

probes

amplifier

Computer

aerator

effluent

Relay Box
(controls
pumps, mixers,
&) aerator

Anoxic
reactor

sludge
wasting

sludge
wasting

final
effluent
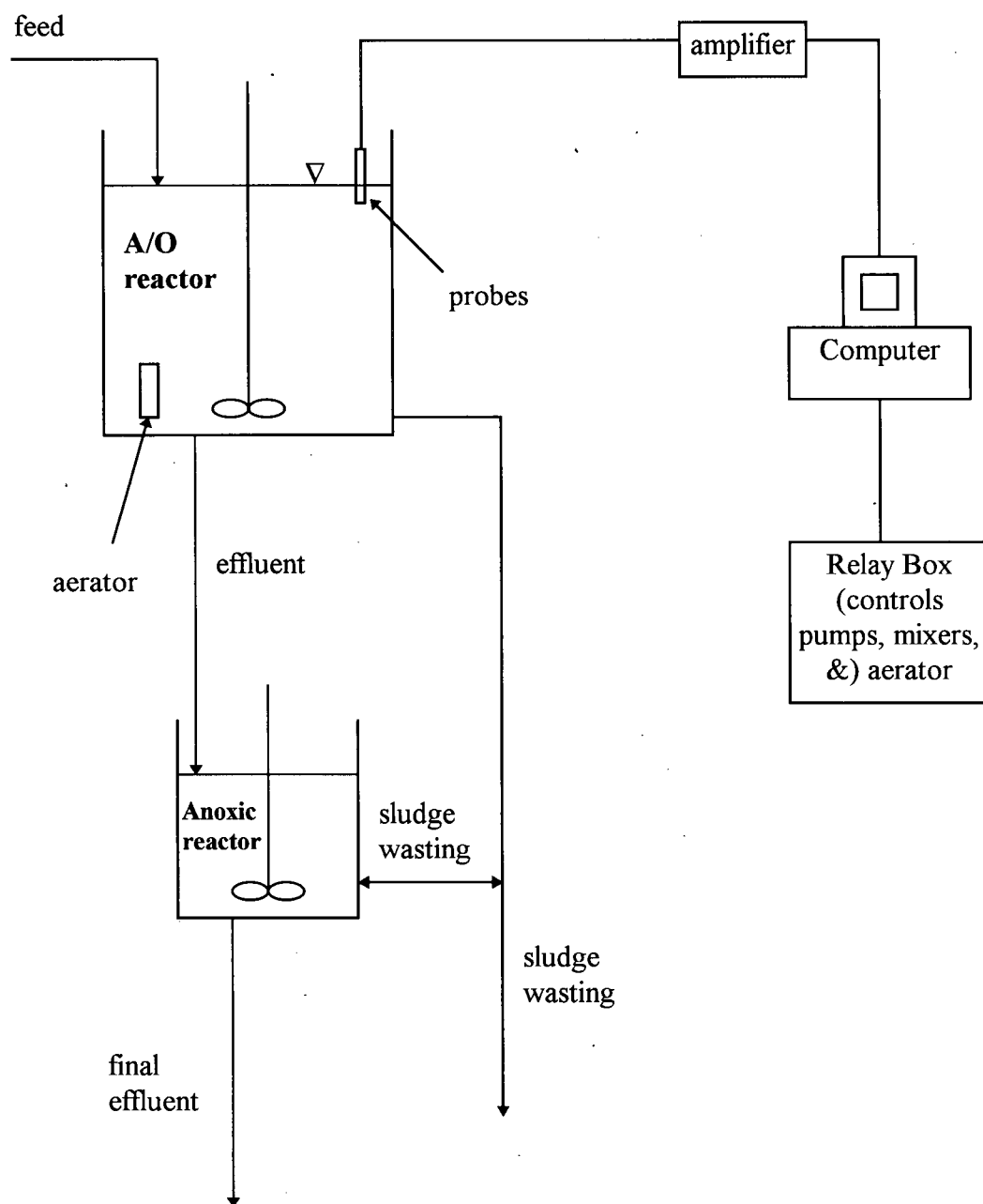
Figure 4. 1 Schematic of the TSSBRs with Real Time Control System

It had a working volume of 7L. Tubing was placed down to the 10L mark in the A/O reactor and the 5L mark in the anoxic reactor for the purpose of decanting. A valve was installed at the bottom of each reactor for wasting sludge.

Mixers were installed in both reactors to ensure complete mixing for the system. Due to the sensitivities of the pH and ORP probes, a shaft made out of Plexiglas with an appropriate blade was used to avoid any electrical interference.

Air for the A/O reactor was provided by an aerator through an air-stone placed at the bottom of the reactor. The air flow rate was adjusted accordingly by the dial on the aerator.

Three peristaltic pumps were used for the entire system (in batch-mode). The first pump was used to discharge the effluent from the anoxic reactor (after the sludge has been settled). The second pump was used to transfer the supernatant from the A/O reactor to the Anoxic reactor. The third pump was used to feed the influent into the A/O reactor. The pumping rates were controlled by a speed controller. A mixer was installed in the influent tank to ensure a well mixed feed for the system. The entire system is shown in figure 4. 2 and 4. 3.

4. 2 Set-Up of Electronic Devices

Four pH probes and two ORP probes were placed into the A/O reactors. These probes were placed at different positions in the reactor to monitor the changes of each parameter in the reactor. The average of these values were taken so that accurate

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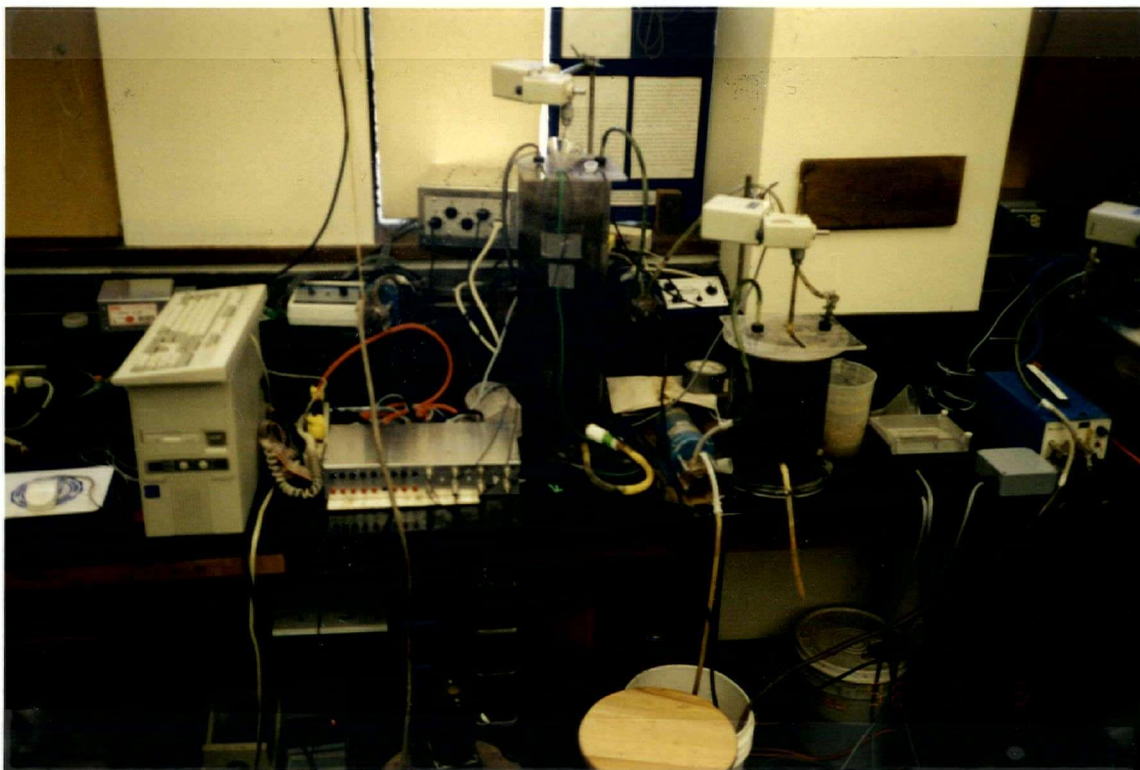
Figure 4. 2 Two Stage Sequencing Batch Reactor
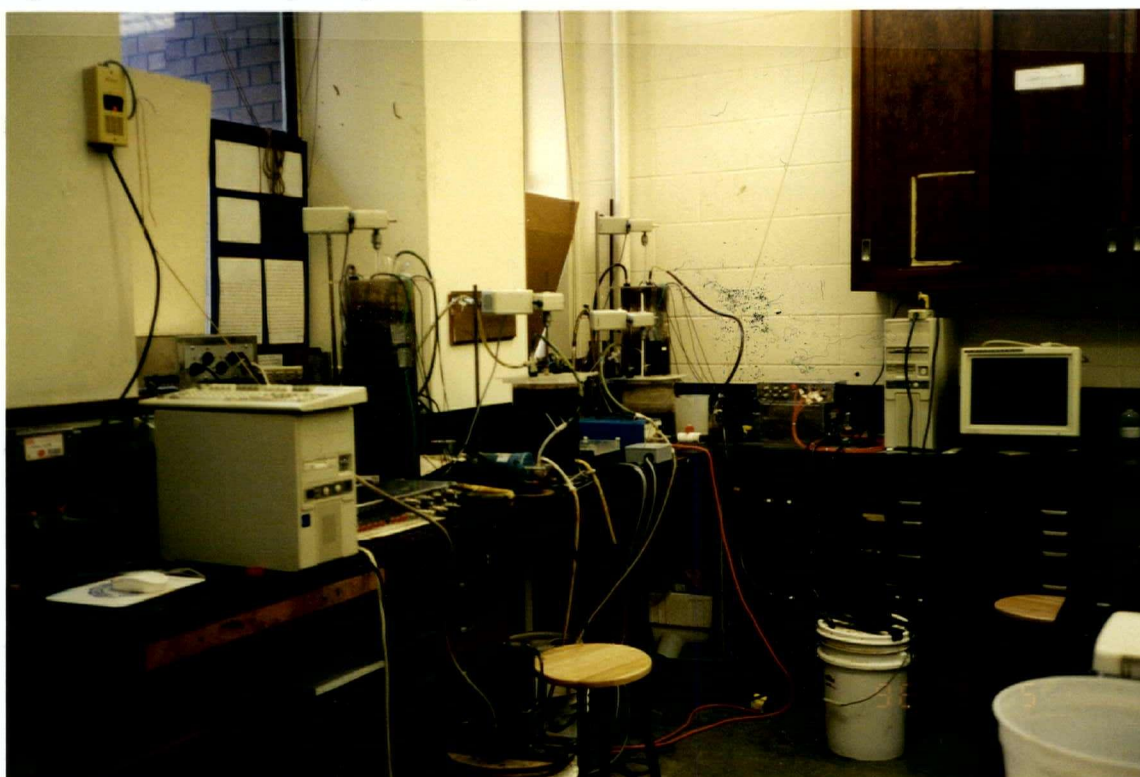
Figure 4. 3 Two Stage Sequencing Batch Reactor

measurements could be used for controlling the system. A DO probe was also inserted into the A/O reactor to monitor the changes in DO concentration throughout the cycle.

Due to the low readings that are usually generated by both the pH and the ORP probes (in the range of -400 to +300mV), amplifiers were built for each probe to enhance and stabilize the signals coming into the computer. The amplifiers were placed in a junction box. The voltage readings (output) that were obtained by the amplifiers were passed on to an analog to digital card that was installed in the computer. Each probe was designated to an interface point on the digital card. This digital card converted the signals from the amplifiers into binary code which could be processed by the computer.

A solid state relay box was built for this project. The relay box had five relay outlets so that each outlet could be controlled independently by a single solid state relay. The pumps, mixers, and aerator were plugged into their designated outlet in the relay box. A commercial software known as LabTECH Control was used for data collection and process control (Laboratory Technology Co., 1994). The automated control methods were programmed into this software. An I/O control card was also installed in the computer. If the readings generated from the probes reached the "trigger" point which was preset in the program, the software would sent a signal to the relay box (I/O control card). The pumps, aerator, or mixers would then be turned on or off according to the program that was developed. Each experimental component and its description are listed in Table 4. 1. The control methods are described in more detail in section 5. 2.

Table 4. 1 Components of an Individual TSSBRs

Experimental Components	# Used	Item Description
Aerator	1	Optima, 60 Hz/50W, Max. 5.5 L/min, Min 0.5 L/min
Mixer	2	Caframo, 60 CYC/70W
ORP Probe	2	Broadley-James Code #605
pH Probe	4	WW Scientific, Cat. No. 34105-023
DO Meter	1 (for suspended growth only)	Point-4 oxygen meter, PT4 system Inc.
DO Probe	1 (for suspended growth only)	Oxyguard, PT4 system Inc.
DO Probe	1 (for attached growth only)	
Computer	1	486-DX
Input/Out Control Card	1	PCL-711 REV. A1
Analog to Digital Card	1	PCLD-711 REV. A1
Standby Power Supply	1	American Power Conv. UPS-SX
Pump Speed Controller	3	Masterflex, Model #7553-71, 50/60 Hz, 3AMP
Perstaltic Pump	3	Masterflex, 1-100RPM, Model #661063

4. 3 Ringlace

The Ringlace is made up of a material known as polyvinylidene chloride. From the manufacturer's manual, this material is water resistant and chemically stable. It has a life span of more than 10 years at temperatures up to 50°C and exposure to concentrations up to 65% for chemicals such as Nitric acid, Caustic Soda, Acetic Acid, Benzene, Acetone, and Cyclohexane.

The Ringlace is a rope type product with rings that are approximately 2 cm long extending outwards from the center all along its length. The Ringlace were often tightly stretched over a rigid frame and were immersed into the process mixed liquor. This particular type of medium is often more advantageous over other types of attached growth system because of the oscillating motion of the rope (Ringlace) which keeps the existing biomass fresh. Due to the build-up of bacteria on the Ringlace, a slight slack would have developed in the rope. This would help to create a swaying motion which would break dead biomass off the Ringlace; hence preventing the build-up of a thick anaerobic biomass layers that are often found on solid inflexible structures.

4. 3. 1 Ringlace Configuration

A rigid frame was built in order to immerse the Ringlace into the process mixed liquor. Eight Ringlace, each with a length of approximately 15 inches, were mounted on and stretched tightly over the frame. The frame was made out of three stainless steel poles and two circular PVC rings. Four rods that were parallel to each other with a two inch

separation were embedded onto the rings. The end of each Ringlace was strapped onto the PVC rods. The entire frame was then immersed into the process mixed liquor in the A/O reactor. The bottom ring was approximately 5 inches from the bottom of the reactor. The Ringlace configuration is shown in figure 4. 4.

4. 4 Experimental Procedures

One objective of this research was to compare the treatment efficiency between an attached growth system and a suspended growth system. The treatment efficiency for either method, in theory, should be the same if they were run under a real-time controlled process. However, the attached growth system should be able to reach it's "trigger" point at a faster rate (shorter HRT) as it was able to retain more bacteria in the system; hence preventing the wash-out effect (a longer SRT). The Solids Retention Time (SRT) for the A/O reactor was determined to be 10 days based on Ra's work (1997). Therefore, each concentration was run for twenty days so that two SRTs and several HRTs could be completed. The SRT in the anoxic reactor wasn't controlled because the growth rate for denitrifiers were relatively low. It should be noted that the sludge from each reactor was completely separated in this research during the operations. However, wasted sludge from the A/O reactor was often added to the anoxic reactor manually for the purpose of providing an external carbon source to improve upon the treatment efficiency of nitrate.

Figure 4. 4 Configuration of Ringlace

Samples of wasted sludge from both the A/O and anoxic reactors were collected for Total Nitrogen (TN) and Total Phosphorus (TP) analysis. Influent, A/O effluent, and final effluent samples were collected everyday for a period of 20 days for each run. The treatment efficiency of each parameter such as D.O., BOD_5, COD, TOC, NH_4-N, NO_x-N, PO_4^{-3}, Total Solids (TS), Total Suspended Solids (TSS), Total Volatile Solids, (TVS), Total Volatile Suspended Solids (TVSS), as well as the Hydraulic Retention Time (HRT) were compared between the two systems. Before sampling for each run, a time period of approximately 25 days was allowed for the bacteria to acclimatize to the new wastewater concentration.

In order to control the loading rate, 2L of wastewater was changed every cycle. However, the concentration of the wastewater was changed for the three different runs tested. The different concentrations that were used in this research was based on the Total Organic Carbon (TOC) concentrations of the swine wastewater. The influent was monitored closely due to the approximate range of concentration that was wanted for this experiment. The concentrations of 500, 2,000, and 4,000 mg/L were used to prove if the pH pattern can be duplicated under different conditions.

4. 4. 1 Calculation for SRT and HRT

SRT for the attached growth system was difficult to calculate because some of the bacteria were immobilized on the Ringlace. Therefore, the SRT for the attached growth system was only based on the amount of suspended bacteria. The following equation was

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used for the calculation of SRT for both the attached growth and suspended growth system:

$$SRT = \frac{VX}{Q_w X_w}$$

where,   V = Process volume, (L)

X = Mixed Liquor Volatile Suspended Solids (MLVSS), (mg/L)

Q_w = Sludge wastage flowrate, (m³/day)

X_w = Wasted sludge MLVSS, (mg/L)

It should be noted that each system's HRT was not fixed because of the application of real-time control. As mentioned before, the working volume for the A/O reactor of the attached growth and suspended growth systems was 10.5L and 12L respectively. Two litres of wastewater was changed every cycle. With the above information, the following equations were used to calculate the HRT for each individual reactor:

Attached Growth System, A/O reactor: $\dfrac{V}{Q} = \dfrac{10.5L}{\dfrac{2L}{cycle} \times \dfrac{\# of\ cycle}{day}}$

where,   V = Process volume, (m³)

Q = System flowrate, (m³/day)

Suspended Growth System, A/O reactor: $\dfrac{V}{Q} = \dfrac{12L}{\dfrac{2L}{cycle} \times \dfrac{\# \, of \, cycle}{day}}$

Both Systems, Anoxic reactor: $\dfrac{V}{Q} = \dfrac{7L}{\dfrac{2L}{cycle} \times \dfrac{\# \, of \, cycle}{day}}$

Total HRT = respective HRT for A/O reactor + HRT for anoxic reactor

4. 4. 2 Track Analysis

Track analyses were necessary, especially when a real-time control process was being used. In order to know where to set the control point, it was essential to understand how each parameter (nutrient) behaved throughout the entire cycle of treatment. There were some distinct features on the pH curve that could be used as control point (such as the Nitrogen Break Point (NBP), or the Residual Carbon Manipulation Point (RCMP)). The track analyses would give an indication of where these points would occur, and therefore a better understanding of where the control point should be assigned.

For the track analyses, samples were taken every half an hour throughout the cycle, beginning from the anoxic phase to the end of the aerobic phase. Approximately 30 ml of well mixed samples were taken from both the A/O reactor and the anoxic reactor. The filtered samples were tested for TOC, nitrate, ammonia, and ortho-phosphate.

4. 5 Operating Procedures

The swine wastewater used for this project was obtained from a farm in Aldergrove, B.C. Originally, the wastewater was collected from a pond that was set outside the barn. This pond was built as a storage facility. The wastewater would later be used as irrigation water. However, there was a suspicion that disinfectant might be present in the wastewater, as the treatment efficiency for the TSSBRs system deteriorated. Disinfectant were commonly used in pig farms to protect the livestocks from attracting any diseases. The amount of disinfectants were often over-used. If a large dose of antibiotics were present in the wastewater, it could seriously damage the bacteria in the system, and hence lowering it's treatment efficiency. Therefore, instead of collecting from the pond, the swine manure that was obtained for this project came directly from the pig raising pen, where the amount of disinfectant present was minimal.

The swine manure was collected with 5 gallon buckets and was stored in the laboratory refrigerator at 4°C until required. A 0.5 mm sieve was used to screen out the larger particles before the wastewater was fed into the influent tank. The screening was necessary because of the high proportion of suspended solids that swine manure contained. A high suspended solids concentration would retard the rate of degradation by the bacteria, and hence affect the treatment efficiency. The removal of these large particles would also reduced the oxygen demand for the system (Burton, 1992).

The treatment system was originally seeded with sludge taken from the pilot-scale sewage treatment plant managed by the department of Civil Engineering of the University of British Columbia (located at the UBC south campus). 4L and 2L of sludge with a

MLSS concentration of approximately 15,000mg/L were added to the A/O and Anoxic reactor respectively at the start-up of the operation. The reactors were filled with swine wastewater until it reached the 12L mark for the A/O reactor. The system was then run at a fixed time procedure so that a pH pattern could be monitored. When a steady pH curve was observed, the fixed time procedure was replaced by the automatic process control system. Sampling began when the system showed a constant real-time operation and a steady state condition.

4. 5. 1 A/O (Anoxic/Oxic) Reactor

This reactor operated in the mode of a conventional sequencing batch reactor (SBR). The reactor went through the following five stages:

feed → anoxic phase → aerobic phase → settle → decant to anoxic reactor

The reactor was operated according to the above sequence so that a complete removal of carbon, nitrogen, and phosphorous in swine wastewater could be achieved. Table 4. 2 summarizes the operation mode for the A/O reactor. The anoxic phase was designed so that denitrification and phosphorus release could take place. The length of the anoxic phase, which included sludge settling and effluent decanting, was fixed. The total time provided for the anoxic sequence in the A/O reactor was 4 hours.

Aerobic conditions began immediately after 4 hours of the anoxic state. The aerobic phase was provided for nitrification, carbon oxidation, and phosphorus uptake.

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This was the important part of this research as real-time control was applied during the aerobic phase. The length of the aerobic phase was dependent on the concentration of the wastewater. The air and mixer would be turned OFF when the pH value reached the designated control point to allow sludge settling and effluent decanting

Table 4. 2 Operation mode for A/O reactor

Mode	Reaction Condition	Purpose	Time
Fill	- Mixer on and aeration off	- Add 2L of wastewater into the reactor	- 10 minutes
React (Anoxic)	- Continuous stirring of mixed liquor	- Denitrification - Phosphorus release	- 3 hrs 21 minutes
React (Aerobic)	- Continuous stirring - Aeration on	- Nitrification - Phosphorus uptake - Carbon oxidation	- Depends on the concentration of wastewater
Settle	- Aeration and stirring off	- Solids settling	- 26 minutes
Decant	- Aeration and stirring off	- Drawing effluent into anoxic reactor	- 3 minutes

4. 5. 2. Anoxic Reactor

Because the effluent from the A/O reactor was decanted after the completion of the aerobic phase, the nitrates that were produced in the aerobic cycle had to be removed before the wastewater could be discharged into the environment. The purpose of this anoxic reactor was to achieve a relatively complete removal of the nitrate (denitrification) without a supplement of an additional carbon source such as methanol. This reactor was designed so that the residual carbon content from the A/O effluent along with endogenous respiration of the bacteria would accomplish the necessary reduction of nitrate. This reactor responded according to the action of the A/O reactor. In another words, if the pH value in the A/O reactor has reached the designated "trigger" point, the entire system would be shut down to allow sludge settling. The anoxic reactor would then respond by discharging the effluent after the sludge has been settled for 20 minutes (see figure 4. 5 for the operation cycle of TSSBRs).

4. 6. Analytical Procedures

All the analytical procedure used were in accordance with the Standard Methods Manual for Water and Wastewater (A. P. H. A., 1995).

Anoxic
phase

Aerobic
phase

Influent
feeding

a) Operation Cycle of
A/O Reactor

Sludge
settling

sludge

Effluent transfer

Final effluent
discharge

Draw

Feeding

sludge

b) Operation Cycle of
Anoxic Reactor

Anoxic
phase

Sludge
Settling
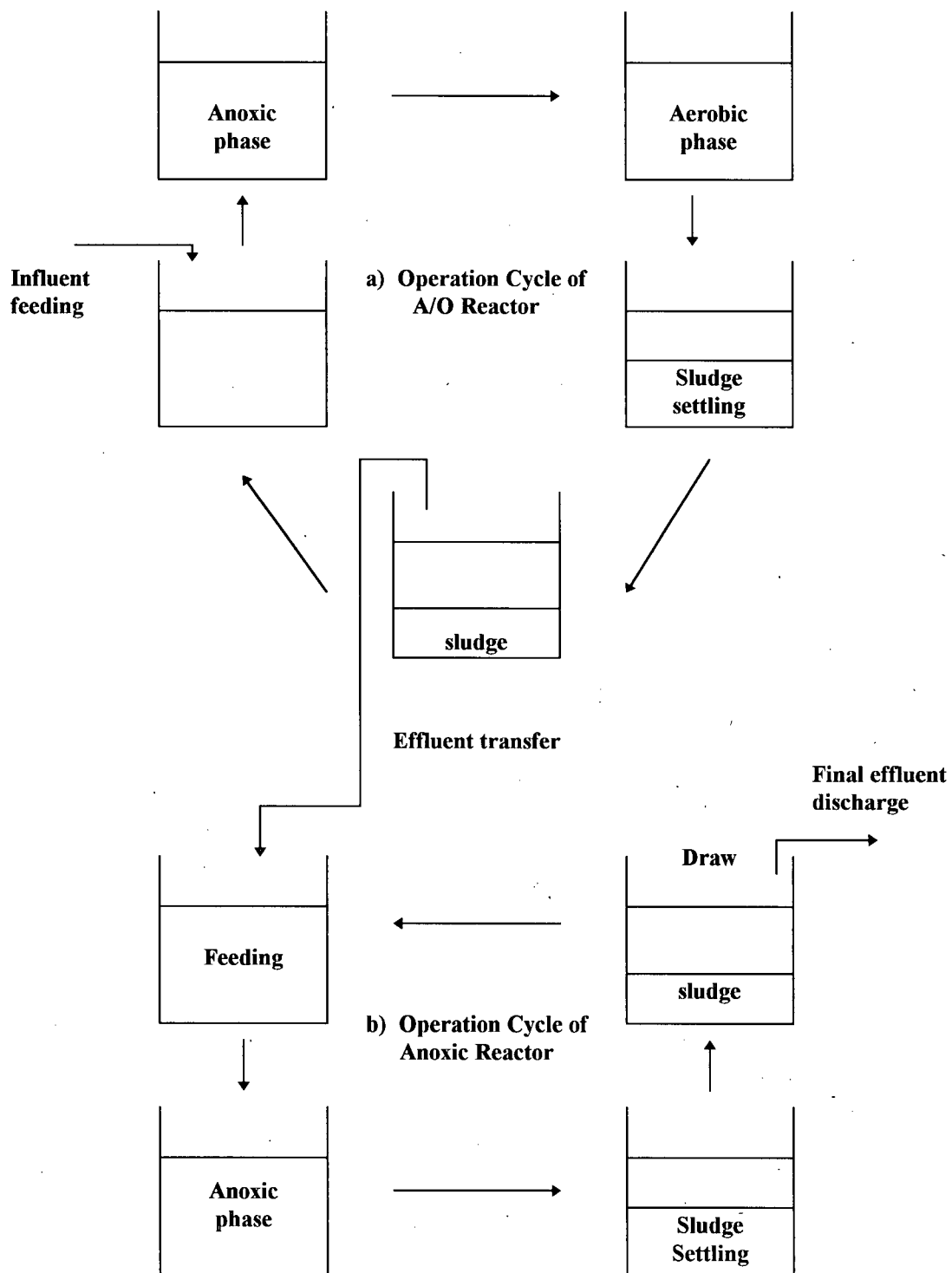
Figure 4. 5. Schematic of the Operation Cycle of TSSBRs

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4. 6. 1 Solids Determination

Total Solids (TS), Total Volatile Solids (TVS), Total Suspended Solids (TSS), and Total Volatile Suspended Solids (TVSS) were analyzed in this research. The Gooch Crucible Method was used for the Total Solids Determination. The Suspended Solids and Volatile Suspended Solids were first filtered through Whatman glass mircrofibre filter paper. After filtration of the sample they were put in a Blue Line drying oven at a temperature of 105°C for 24 hours. TVS and TVSS were determined by igniting the dried residue at 550°C for 1 hour in a Lindberg muffle furnace in accordance with Standard Methods.

4. 6. 2 Chemical Oxygen Demand

During the determination of the chemical oxygen demand (COD), organic matter is converted to carbon dioxide and water regardless of the biological nature of the substance. Sulfuric acid, mercuric sulfate, and potassium dichromate were used to digest the sample in this COD test. A catalyst solution (composed of silver sulfate and sulfuric acid) was also added before the digestion process began. After samples were put through this digestion procedure, absorbency readings were recorded on a colorimeter. The colorimeter was a Brinkmann PC 800 (600nm). Only the supernatant samples were analyzed for their COD values.

4. 6. 3 BOD₅

BOD_5 was done according to Standards Methods. The right proportion of wastewater and nutrient solution was added to the BOD_5 bottles. The nutrient solution contains 1ml of buffer solution, 1 ml of $MgSO_4$, 1 ml of $CaCl_2$, and 1 ml of $FeCl_3$ for every 1L of water. A YSI model 59 DO probe was used to analyze the samples.

4. 6. 4 TOC

A Shimadzu analyzer, model TOC-5050 was used for the measurement of TOC. This was done by measuring the Total Carbon (TC) content then subtracting the Inorganic Carbon (IC) concentration of the sample. The TC measurement was done by heating the sample at $680^{\circ}C$. The TC compounds in the sample is combusted or decomposed to become CO_2. The CO_2 converted becomes the TC concentration of the sample. The IC measurement was done by injecting the sample into the IC vessel where it will be acidified by H_3PO_4. This will then be converted into CO_2. This CO_2 will in turn be the IC concentration of the sample.

4. 6. 5 Orthophosphate

Orthophosphate values were measured by an auto analyzer technique. The model of auto analyzer was a Technicon Manifold (#2) made by Pulse Instruments Ltd., Canada. Technicon Industrial Systems has a technique which is in accordance with Standard

Methods. The automated procedure for the determination of orthophosphate depends on the well known chemistry whereby ammonium molybdate reacts in an acid medium to form molybdophosphoric acid which is then reduced to the molybdenum blue complex by reaction with ascorbic acid.

4. 6. 6 Nitrate

This procedure utilizes the reaction in which nitrate is reduced to nitrite by an alkaline solution of hydrazine sulfate containing a copper catalyst. The stream is then treated with sulfanilamide under acidic conditions to yield a diazo compound which couples with N-1-naphthylethylenediamine dihydrochloride to form a soluble dye which is measured colorimetrically. The final product measured represents the nitrite ion originally present plus that formed from the nitrite.

4. 6. 7 Ammonia

This procedure utilizes the Berthelet Reaction in which the formation of a green colored compound believed to be closely related to indophenol occurs when the solution of an ammonium salt is added to sodium phenoxide followed by the addition of sodium hypochlorite. A solution of potassium sodium tartrate is added to the sample stream to eliminate the precipitation of the hydroxides of heavy metals which may be present.

4. 6. 8 TKN and TP of Sludge Samples

All the sludge samples were sent over to the environmental engineering laboratory of the Civil Engineering Department for analysis.

CHAPTER 5

RESULTS AND DISCUSSION

5. 1 Track Analysis

Track studies were done for each run to determine if the pH and the nutrient profiles in the A/O reactor were the same for all concentrations. Track analysis also renders information about when the control point should be set for this automated process. The success of the real-time control operation was dependent on the results of the track analysis.

5. 1. 1 Attached Growth

5. 1. 1. 1 TOC = 2,000mg/L

As shown in figure 5. 1, both the pH and the ORP curves had their own distinct features throughout the entire cycle. Even though the curves were slightly different, all the control points identified in the cycle were nearly identical in terms of the biological and chemical state of the wastewater. This proved that it would be possible to use either pH or ORP as a control parameter in a real-time control process.

From the nutrient profile in figure 5. 2, it was shown that the NO_x-N that was produced from nitrification during the aerobic phase was quickly denitrified to N_2 during

pH and ORP Curves
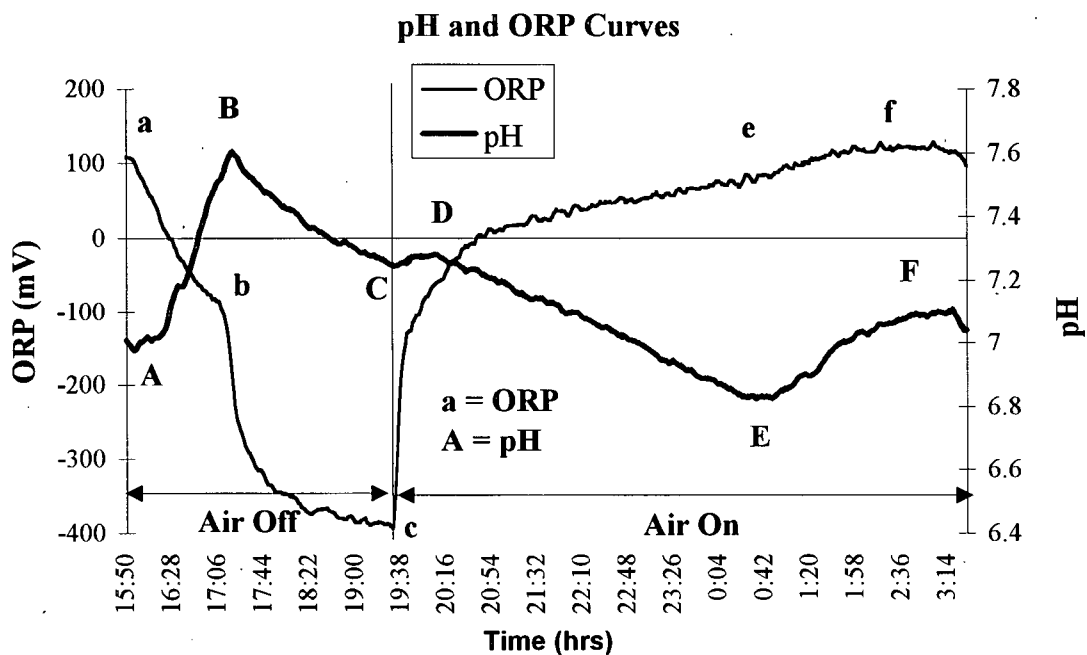
Figure 5. 1 pH and ORP patterns throughout one cycle during the 2,000 mg/L run for the attached growth system
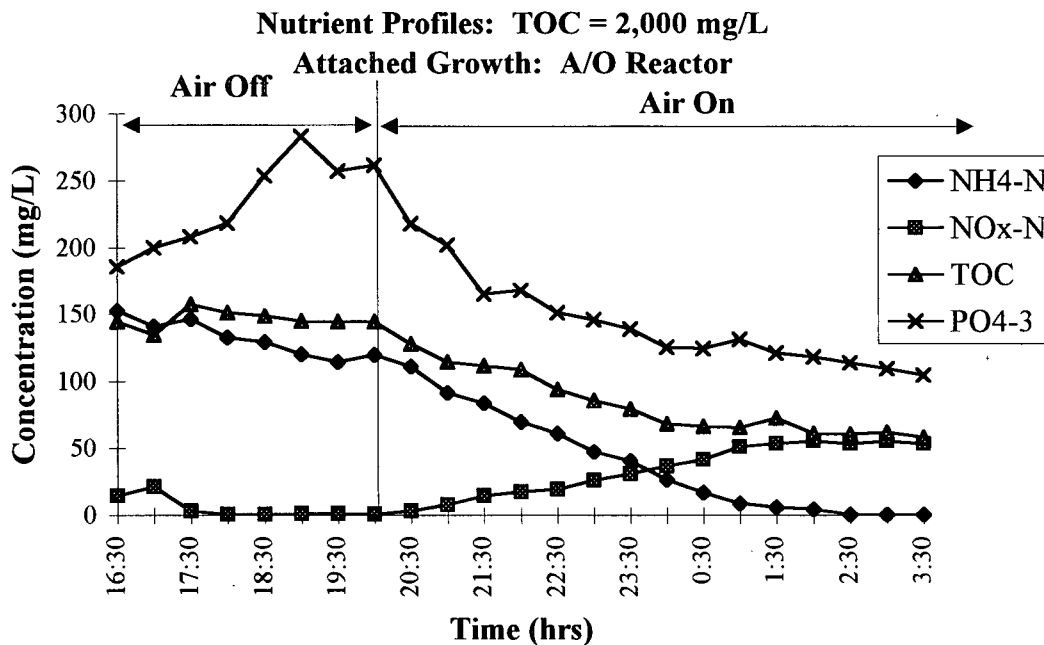
Figure 5. 2 Nutrient profiles throughout one cycle during the 2,000 mg/L run for the attached growth system

the anoxic period using the carbon that was provided by the feed. There was a slight decrease in the ammonia concentration during the anoxic phase. This decrease could have been caused by the uptake of nitrogen in the form of NH_4 by anaerobes for the purpose of cell synthesis. Also the introduction of air into the A/O reactor due to the high mixing speed, especially near the top of the reactor, could have caused nitrification at a very slow rate, which in turn decreased the concentration of ammonia. It could be illustrated that there were two rates in phosphorus release. Phosphorus release occurred more rapidly after the complete removal of nitrate. This led to the conclusion that nitrate is an inhibitor to phosphorus release. Hascoet et al., 1995, discovered that the rate of rapid P release decreased with a presence of high nitrate concentration. Phosphorus release would not occur rapidly without a carbon substrate, even with the absence of nitrate (Iwema et al., 1985). But according to figure 5. 2, there was an abundance of carbon substrate in the wastewater, therefore phosphorus release proceeded quickly. Phosphorus removal followed the rule of "luxury uptake". The bacteria were exposed to alternating anaerobic/aerobic conditions. By stressing the bacteria during the anaerobic phase (releasing of phosphorus), the bacteria would be able to absorb phosphorus above their normal levels during the aerobic phase.

As the cycle switched to the aerobic mode, ammonia-N, soluble phosphorus, and TOC concentration began to decrease. At the same time, nitrate concentration began to increase as ammonia was being converted into nitrate through nitrification.

5. 1. 1. 1. 1 pH Time Profile

- Point A. The end of the aeration phase and the beginning of the anoxic stage. The rise in pH was due to the production of OH^- from denitrification (Yu et al., 1997).

- Point B. This breakpoint is known as the "nitrate apex" for the pH curve (Al-Ghusain et al., 1995). This point indicated the end of denitrification (figure 5. 1). Sometimes, the apex could not be seen because of rapid denitrification during settling or/and influent loading. The decrease of pH from point B to point C was caused by the formation of volatile fatty acids during anaerobic fermentation.

- Point C. This is the introduction of oxygen into the system. The slight rise in the pH to point D was thought to be by air-stripping of carbon dioxide from the system and the rapid consumption of VFA that was produced during the anoxic phase (Ra et al., 1997). This was represented by the rapid consumption of TOC at the beginning of the aeration phase in figure 5. 2. It should be mentioned that carbon dioxide is acidic in its nature. By removing CO_2 from the system meant an increase in the system pH.

- Point D. The initial carbon dioxide stripping and the consumption of VFA have been exhausted. This point indicated the beginning of nitrification. Since the alkalinity of the wastewater was strongly related to it's ammonia concentration, the removal of ammonia from the system meant a decrease in pH (the release of H^+ ion from nitrification (Yu et al., 1997)).

- Point E. This point was known as the Nitrogen Break Point (NBP). It signified the end of nitrification. No more hydrogen ion was produced due to the end of nitrification. The complete removal of ammonia indicated the stoppage of alkalinity consumption of the wastewater; hence the end of further pH decrease. Aeration could be terminated if ammonia was the target nutrient. The slight rise in pH beyond this point was caused by air-stripping of carbon dioxide. As aeration began, both the heterotrophic bacteria and nitrifiers competed for oxygen to carry out their reactions. Usually the rate of oxygen consumption is higher for the heterotrophic bacteria than the nitrifiers due to their differences in metabolic pathways. The end products of carbon oxidation include CO_2. But generally, there is no air stripping of CO_2 during the nitrification period, as all the oxygen was used up by bacteria for accepting electrons (maintains the equilibrium of gas-solubility in the reactor). However, the end of nitrification increased the DO concentration in the reactor. The excess oxygen in the reactor at that moment caused a shift in the equilibrium of gas-solubility. In order to maintain this equilibrium, gases such as CO_2 that have been accumulated during the cycle were stripped by air. This would subsequently lead to an increase in pH.

- Point F. This plateau was known as the Residual Carbon Manipulation Point (RCMP) (Ra et al., 1997). After the end of nitrification, readily available organic substances might still be present in the wastewater. The plateau on the curve indicated the depletion of carbon content in the system. At this point, much of the CO_2 accumulated in the reactor have been extracted from the system and the reactor was virtually saturated. Saturation of

the system indicated the discontinuation of the dissolution of oxygen, resulted in a relatively constant pH.

5. 1. 1. 1. 2 ORP Time Profile

- Point a. The beginning of the anoxic phase. This point represented a transition from high DO concentration in the reactor to a period where nitrate oxygen dominated the system ORP. The addition of fresh wastewater into the system reduced the redox potential as the ratio of reduced materials to oxidized materials was increased. The decline of DO concentration and the sudden change of oxidation/reduction status in the reactor (from oxidizing ammonia and carbon nutrients under an aerobic condition to reducing nitrate under an anoxic state) might also contributed to the decline in the ORP curve. It should be noted that ORP is tremendously sensitive to the changes of DO in the reactor.

- Point b. This point is known as the "nitrate knee" (Peddie et al., 1989). This point coincided exactly with the "nitrate apex" of the pH curve as an indication of complete removal of nitrate (see figure 5. 1). The discontinuation of denitrification resulted in an abrupt change in the ORP slope. This point also indicated the stoppage of respiratory activity and the onset of anaerobic fermentation.

- Point c. This is the beginning of the aeration phase. This rise in the ORP curve was a rapid response to the introduction of oxygen into the system. As with point A, it was a

reflection of the sudden change in reduction/oxidation state. The rate of ORP change was dependent on the aeration rate and the strength of the wastewater.

- Point d. The introduction of oxygen had been the primary influence in the rapid rise of the ORP up to this point. As the system adapts to the aerobic environment, the change in the ORP curve had gradually decreased and the further increase was due to all the other on-going bio-chemical reactions in the reactor such as nitrification, carbon oxidation, and phosphorus release.

- Point e. This point corresponded with the NBP on the pH curve, which signified a complete removal of ammonia-N. This inflection point represented the discontinuity of nitrification. Because of the end of nitrification, the aeration rate was greater than the OUR by the bacteria. The availability of oxygen had suddenly increased. This increase in DO concentration could have caused the abrupt change of slope on the ORP curve.

- Point f. This point also coincided with the RCMP on the pH curve, which signified a complete removal of readily available carbon in the system. The plateau of the ORP curve indicated a fully oxidized state or at least the rate of oxidation was equalled to the aeration rate.

5. 1. 1. 2   TOC = 4,000 mg/L

The track analysis of the TOC = 4,000 mg/L run is shown in figure 5. 3 and 5. 4. There were two major differences between the 2,000mg/L and 4,000mg/L pH curve. During the anoxic phase, the pH for the 4,000mg/L pH curve decreased sharply until it reached a plateau (see figure 5. 3). The "nitrate apex" during the 4,000 mg/L run was not observed.   There could be a couple reasons for this behaviour.   First of all, the concentration of the influent was much higher, meaning that the wastewater wasn't diluted as much with tap water as compared to the other runs.   This could affect the pH (decreased) of the influent entering the A/O reactor.   The pH decrease in the influent can affect the system's pH.   Secondly, the increased carbon content for this run along with the decreased nitrate concentration produced from the aerobic phase· promoted rapid denitrification within the A/O reactor.   The decreased concentration in nitrate meant a smaller amount of OH⁻ released through denitrification.   This would subsequently lead to a smaller increase in pH.   Also, a smaller amount of nitrate meant less carbon was utilized for denitrification.   Therefore, more readily available organic content was reserved for the production of volatile fatty acids and the hydrolysis of organic matter into soluble form. The high nitrate removal rate led to a much quicker development of an anaerobic condition.   The above combinations produced a favourable environment for phosphorus release by the bacteria.   The production of volatile fatty acids caused the pH to drop rapidly.

There was a slight increase in the concentration of ammonia during the anoxic phase after the influent had been fed.   There was a strong possibility that ammonification,

pH and ORP curves
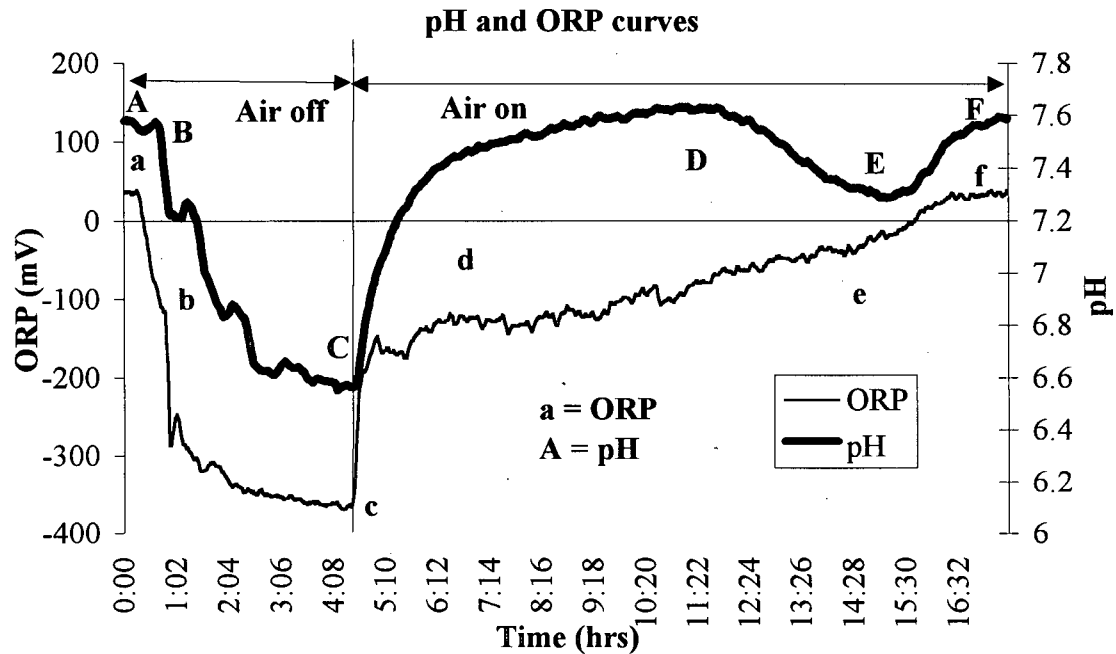
Figure 5. 3 pH and ORP patterns throughout one cycle during the 4,000 mg/L run for the attached growth system

Nutrient Profiles: TOC = 4,000 mg/L
Attached Growth: A/O Reactor
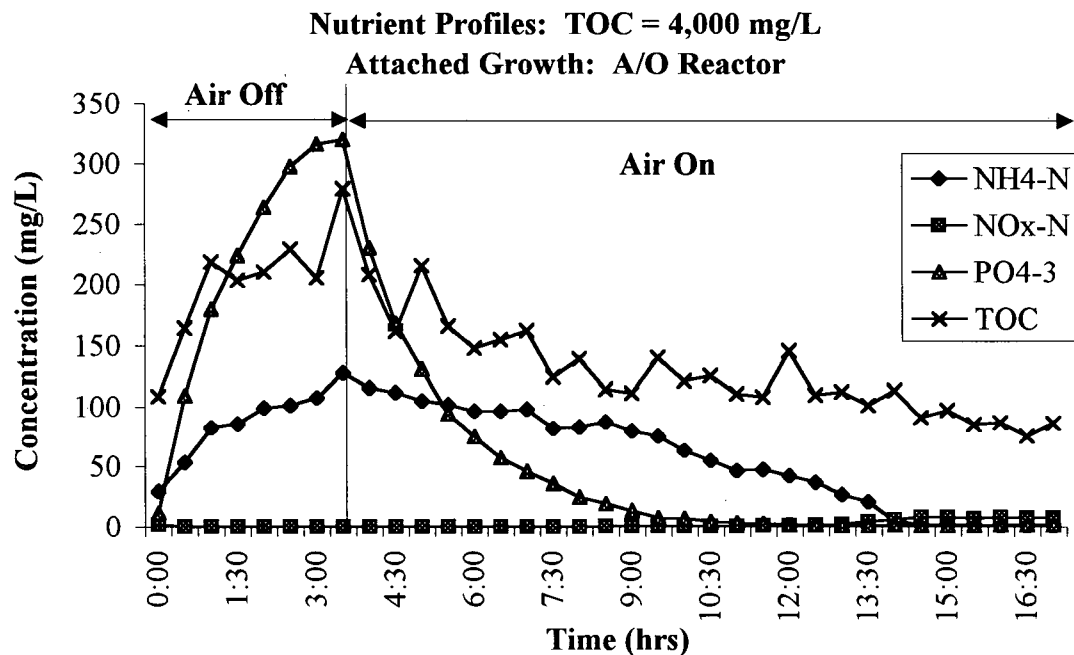
Figure 5. 4 Nutrient profiles throughout one cycle during the 4,000 mg/L run for the attached growth system

the degradation of organic nitrogen compounds, might have been the cause of this increase.

During the aerobic phase, the time that it took from point C to point D was much longer for the 4,000 mg/L curve. The initial rise was caused by carbon dioxide stripping from the system and the rapid consumption of VFA that was produced during the anoxic cycle. Comparing figures 5. 2 and 5. 4, there was a larger increase in TOC during the anoxic cycle in figure 5. 4, meaning a larger production of VFA during the 4,000 mg/L run. As the cycle switched to the aerobic phase, there was a rapid consumption of the acids produced, meaning an increase in pH. The pH continued to rise until most of the acids were exhausted. In the case of the 2,000 mg/L run, because a smaller amount of VFA was produced, there was a smaller increase occurred during the aerobic cycle as less VFA were provided for consumption.

In figure 5. 4 it can be seen that the ammonia concentration decreased as aeration began. The interesting fact was that the nitrate concentration did not increase; meaning that the removal of ammonia did not go through the nitrification mechanism. Ammonia could have been removed by air-stripping; but since the pH in the reactor was within 6.5 to 7.7, the possibility of this was very unlikely. Simultaneous nitrification and denitrification might be occurring within the A/O reactor. This scenario was also observed by Hong, 1997. In his research, he concluded that a significant amount of the unaccounted for nitrogen loss was obtained with complete nitrification when the system was operated with an under aeration procedure. He concluded that simultaneous nitrification and denitrification was the reason for the nitrogen loss. Because of the high concentration of the influent in this run, there was an imbalance between the oxygen

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provided and the oxygen uptake rate (OUR) by the bacteria. Some of the bacteria did not receive the necessary oxygen to carry out their appropriate reactions. Therefore, some of the facultative heterotrophic bacteria used nitrate as an electron acceptor, which in turn created simultaneous nitrification denitrification during the aerobic phase in the A/O reactor. This could explain the unaccounted loss of nitrogen in the A/O reactor.

Ammonia could also have been consumed by bacteria for cell synthesis (assimilation) as a nitrogen element. As mentioned before, there was an imbalance between the oxygen provided and the OUR by the bacteria. Autotrophs are the primary agents that are responsible for nitrification, and a comparatively larger amount of energy is required to convert carbon dioxide to more complex organic forms suitable for cell synthesis. Because of their longer metabolic pathways, they were not as aggressive when competing with heterotrophs for oxygen to carry out their oxidation reactions. In this case of under-aeration, much of the available oxygen would be consumed by the heterotrophic (carbon oxidation) bacteria. Along with the oxygen requirement, heterotrophic bacteria also needed nutrients to synthesize new cells. Nitrogen in the form of NH_4-N was provided for the bacteria to utilize in this scenario.

The consumption of ammonia by bacteria would decrease the pH of the system (the release of H^+ ions) as the alkalinity of the wastewater was strongly related to its ammonia concentration. Denitrification, however, works exactly the opposite as it releases the OH^- ion (which increases the pH of the system). A constant pH would be produced if the rate of ammonia removal equals to the rate of denitrification. The above reasons explained the relatively constant pH that was produced after the initial pH rise from point C to point D during the aerobic phase.

Bacteria needed oxygen in order to carry out their oxidation reactions. The high loading rate in this research created an imbalance between the aeration rate and the nutrients available in the wastewater. At the beginning of the aerobic cycle where nutrients were plentiful, the uptake of oxygen by bacteria was occurring at a very high pace. Therefore, some bacteria weren't able to obtain the necessary oxygen requirement to carry out nitrification, resulting in simultaneous nitrification denitrification. This was represented by the dissolved oxygen concentration of approximately 0 mg/L at the beginning of the aerobic phase (see section 5. 3. 1. 2 for more details). However, the reaction rate decreased as the nutrients in the system decreased. As the cycle reached point D, there was a balance between the aeration rate and the OUR. At this point, simultaneous nitrification and denitrification have stopped. Therefore, this point indicated the beginning of nitrification alone. Since the alkalinity of the wastewater was strongly related to it's ammonia concentration, the removal of ammonia solely from the system meant a decrease in pH.

5. 1. 1. 3 TOC = 500 mg/L

The pH pattern for this particular concentration was very similar to the 2,000 mg/L curve. As shown in figure 5. 5 and 5. 6, the cycle for this run was much shorter at approximately 8 hours than the previous runs discussed. This is rather obvious as the concentration of the influent for this run was much lower, while other factors such as aeration rate and SRT remained the same. Therefore, the HRT for this run was relatively short compared to other runs.
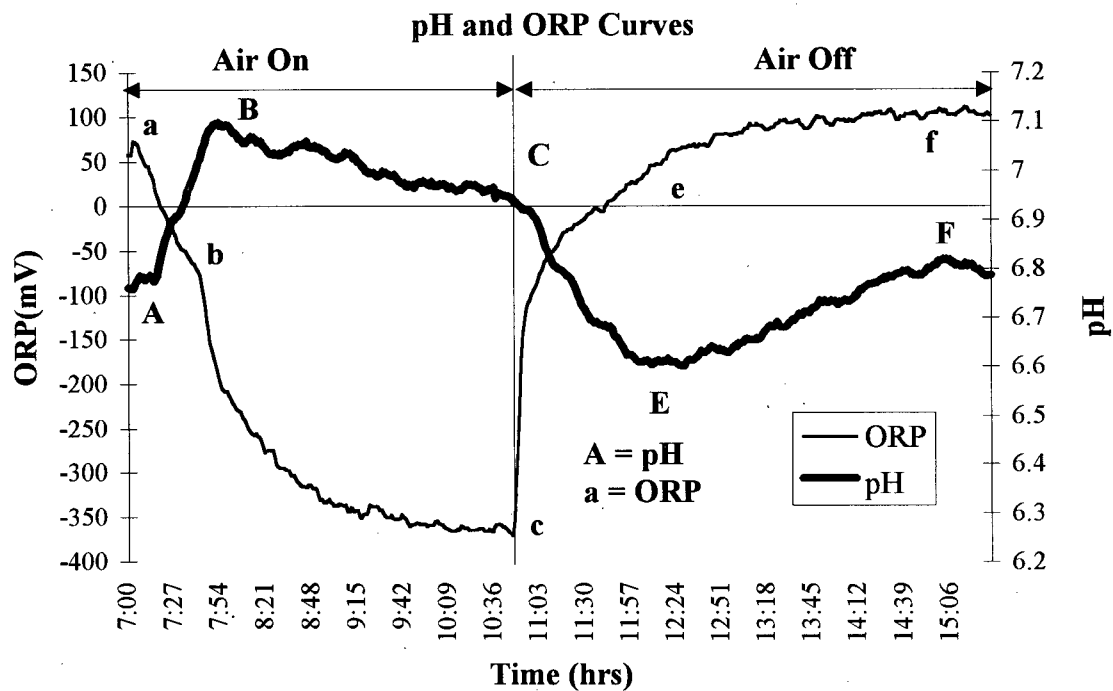
Figure 5. 5 pH and ORP patterns throughout one cycle during the 500 mg/L run for the attached growth system
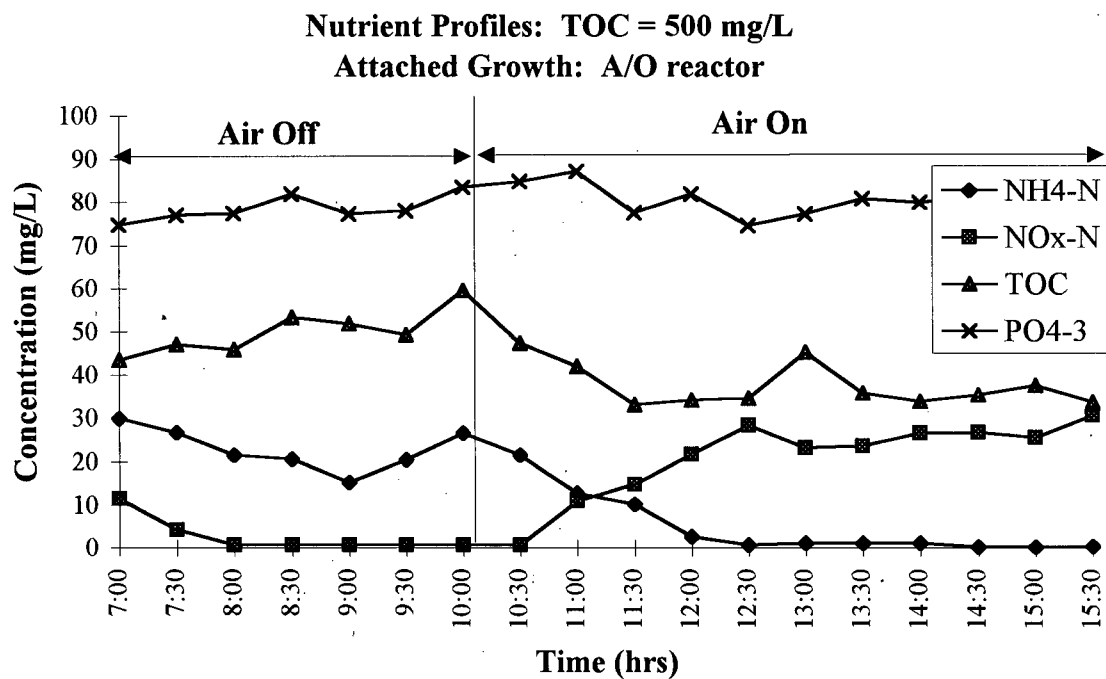
Figure 5. 6 Nutrient profiles throughout one cycle during the 500 mg/L run for the attached growth system

As shown by figures 5. 5 and 5. 6, the nitrate apex on the pH curve and the nitrate knee on the ORP curve again correlated with the complete removal of nitrates. After complete denitrification during the anoxic phase, the pH curve did not decrease as much as the 2,000 mg/L curve. It was found from the 2,000 mg/L track analysis that much of the phosphorus release occurred after complete denitrification. Because of the low loading rate, much of the fresh carbon content from the influent was used for denitrification. As denitrification was completed, there wasn't enough readily available organic matter in the system for the production of VFA and the hydrolysis of organic matter into soluble form. This action hindered the bacteria's ability to release phosphorus into the solution. This was illustrated well by figure 5. 6 as it could be seen that there was hardly any increase in the TOC and PO_4^{-3} concentration during the anoxic phase. The lack of production in VFA caused the pH to only drop slightly.

As mentioned before, there wasn't enough VFA produced during the anoxic cycle due to the lack of carbon source from the influent. Therefore when the cycle switched to the aerobic mode, there was no sharp increase in the pH pattern as there was no VFA available for bacterial consumption. Also because of the low loading rate, the pH value decreased immediately after aeration began. The nitrifiers were able to utilize the oxygen that was supplied and carried out their oxidation reactions immediately after aeration had commenced. This was probably due to the excess oxygen that was provided as there was an imbalance between the substrates in the mixed liquor and the aeration rate. Therefore, nitrification and carbon oxidation occurred at the same time.

The NBP was found on the pH curve but not on the ORP curve in figure 5. 5. It was found in other researches that ORP was very sensitive to the changes of DO

concentration and was affected greatly by factors like pH and solids concentration (Ra, 1997, Saune et al., 1996). During the 500 mg/L run, the aeration rate supplied was much greater than the OUR, resulting in excess oxygen present in the reactor. As noted in figure 5. 6, the ammonia concentration in the A/O reactor was relatively low compared to other runs. The combined effect of a low ammonia loading and a high aeration rate caused nitrification to occur rapidly. The NBP on the ORP curve was masked by this rapid reaction by the bacteria. The pH curve, on the other hand, stayed relatively constant because the measurement of pH in the reactor was based on the concentration of the hydrogen ion in the solution, and not a direct measurement of the overall bio-chemical reaction state in the system like ORP.

Examining figures 5. 5 and 5. 6 closely, it could be determined that the complete removal of carbon from the A/O reactor in this run had not occurred at point F as suggested previously. The higher carbon concentration of the other two runs caused the complete carbon removal point to be at the RCMP. However, the low concentration of TOC and the high aeration rate caused the carbon content to be stripped quickly from the system. The inability of the system to provide additional carbon during the anaerobic phase caused the depletion of readily available organics in a very short time after the cycle switched to the aerobic mode. From this observation, it could be suggested that both the pH and the ORP curve beyond the NBP were not strictly caused by the complete removal of carbon from the system. The changes in the pH curve beyond the NBP was mainly caused by the soluble-gas composition in the wastewater (see Section 5. 1. 1. 1. 1 for more details). The changes in the ORP curve were an effect of changes in the DO concentration within the A/O reactor. Even though the pH curve was reproducible from

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cycle to cycle and would be very stable in a real-time control process, the RCMP would not be a good representative as a complete carbon removal point because of its inability to characterize this point under different circumstances.

Both the advantage and disadvantage of using pH over ORP as a real-time control parameter were demonstrated in this run. The pH patterns produced in the aerobic phase of the cycle were uniform, despite the system being operated under different conditions. The ORP, on the other hand, was not able to detect the NBP under conditions of excessive aeration, which might lead to control failure. Since the distinct features such as NBP would always be detected on the pH profile, the use of pH as a control parameter would be more stable in a real-time system.

ORP, on the other hand, possesses an advantage over pH as a control parameter in terms of operating costs. When the NBP on the ORP curve was not observed as in this case, the suspicion of over-aeration could be justified. This would, in turn, be an indication to adjust the aeration rate accordingly to create a balance between the aeration rate and the oxygen demand. The above condition could be detected by monitoring the ORP pattern closely, and thereby optimizing the treatment process by reducing the aeration costs.

5. 1. 2 Suspended Growth

5. 1. 2. 1 TOC = 2,000 mg/L

The track analysis of the 2,000 mg/L run for the suspended growth system is very similar to the attached growth system. All the distinct features that were described in the attached growth section correlated exactly the same way with the suspended growth system. From figure 5. 7 and 5. 8, it could be seen that the pH rose very sharply after influent feeding. This again was due to rapid denitrification where the OH ion was released to the solution. The nitrate knee on the ORP curve coincided with the nitrate apex on the pH curve. Both represented the end of denitrification. The pH curve dropped rapidly after the nitrate knee due to the beginning of anaerobic fermentation. During anaerobic fermentation, VFA were produced; hence the drop in pH. As the cycle switched to the aerobic mode, pH rose slightly as the bacteria consumed the VFA that were produced during the anaerobic condition. Nitrification occurred immediately after the anoxic phase. The pH drop after the initial rise in the aerobic phase was due to the consumption of ammonia in the wastewater. It should be noted that ammonia is strongly related to the wastewater's alkalinity. The pH curve again coincided with the ORP curve on both the NBP and RCMP. There was no major difference on the shape of the curves between the attached growth and suspended growth system; meaning that there was no significant difference in the bio-chemical reactions taking place within the reactor. Therefore, it can be concluded that the implementation of Ringlace into a SBR system does not affect the operation of the real-time control process.

pH and ORP Curves: TOC = 2,000 mg/L
Suspended Growth: A/O reactor
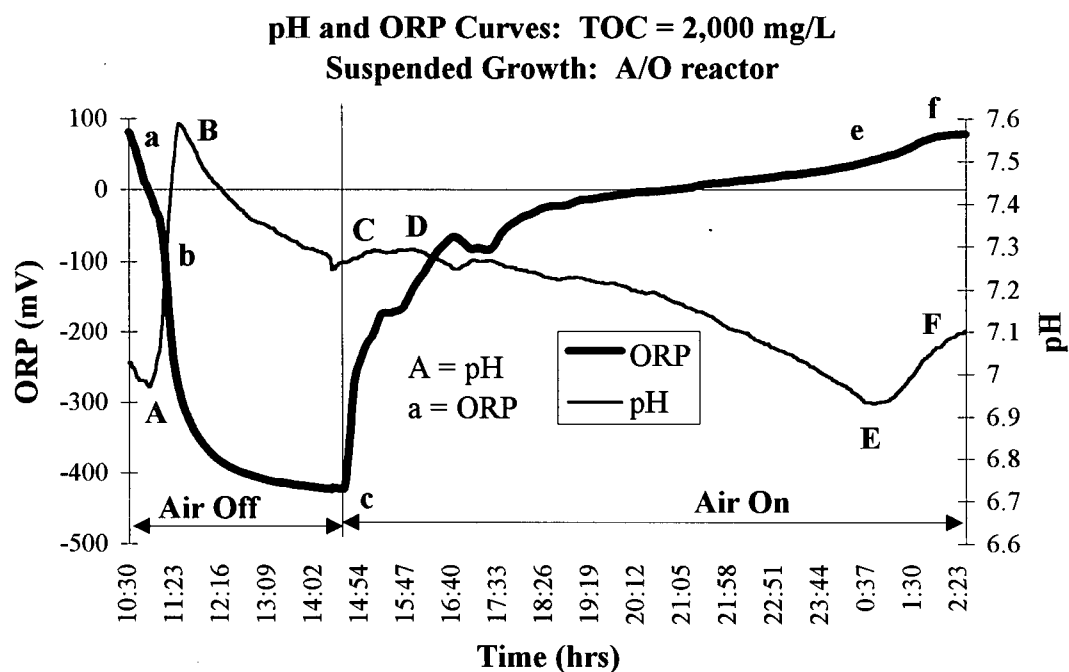
Figure 5. 7 pH and ORP patterns throughout one cycle during the 2,000 mg/L run
for the suspended growth system

pH and ORP Curves: TOC = 2,000 mg/L
Suspended Growth: A/O Reactor
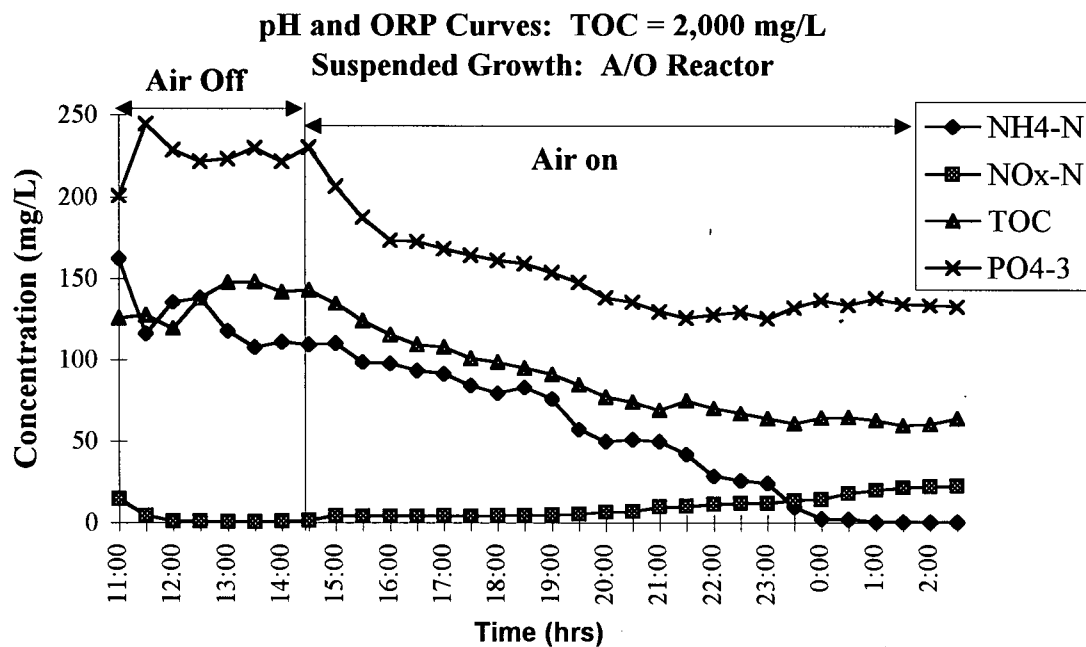
Figure 5. 8 Nutrient profiles throughout once cycle during the 2,000 mg/L run for
the suspended growth system

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There was a slight difference in the method of phosphorus release between the attached growth and suspended growth system according to the track analysis. Rapid denitrification and phosphorus release occurred simultaneously at the beginning of the anoxic phase. It was found, however, in the 2,000 mg/L run for the attached growth system that nitrate was an inhibitor to phosphorus release. There could be a couple reasons that could describe this behaviour. It was found that nitrate reduction (denitrification) and phosphorus release can take place at the same time if there is an abundance of carbon substrate available for consumption (Gerber et al., 1986), as was illustrated by the TOC concentration at the beginning of the anoxic phase. Also in figure 5. 8, it can be seen that nitrate still wasn't completely removed after the initial release of phosphorus. It was found in many other researches that P-uptake could also occur under anoxic conditions (Comeau et al, 1986, Gerber et al., 1986, Kerrn-Jespersen et al., 1993). It was concluded in their studies that a fraction of the polyP organisms are able to use nitrate as an electron acceptor in the absence of oxygen. It could very well be conceded in this case that some of the polyP bacteria in the reactor were utilizing nitrate as an electron acceptor to uptake phosphorus until the nitrate concentration reached zero. This explained why the phosphorus concentration decreased after the initial rise in the anoxic phase.

The lack of uptake during the aerobic phase was caused by an insufficient release of phosphorus during the anoxic phase. As mentioned before, a carbon source was needed to facilitate primary release of phosphorus. The lack of VFA production caused a partial release of phosphorus during the anoxic phase, which in turn resulted in a limited phosphorus uptake by the bacteria in the aerobic phase.

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The proportion of bacteria in the attached growth system (AGS) could be very different than the suspended growth system (SGS). There was a larger population of bacteria for the AGS because the SRT of 10 days was based only on the suspended bacteria in the reactor. The bacteria attached onto the Ringlace were not taken into account when calculating the system's SRT. The SRT for the SGS, on the other hand, was calculated basing on the entire bacterial population. The larger population could have diversified the species within the reactor. In any AGS, the organic loading rate is very important to the population of different types of bacteria on the medium (Metcalf & Eddy, 1991). Heterotrophic bacteria have a faster growth rate than autotrophs and can usually out-compete autotrophic bacteria for oxygen under conditions where carbon content is plentiful (EPA Nitrogen Control Manual, 1993). Due to the above reasons, heterotrophic bacteria were usually the most active bacteria in the reactor. As the slime developed onto the medium, heterotrophic bacteria became the most likely candidate to be attached onto the outer layer of the slime. Since the external layer of the slime was constantly exposed to alternating anoxic/oxic conditions, it is plausible that the outer surface of the biofilm consisted of mostly facultative, heterotrophic bacteria. The inner layer of the slime, however, was not exposed to oxygen due to the thick biomass that usually existed on the Ringlace. The inner slime, therefore depended heavily on diffusion to obtain any electron acceptors or nutrients for cell synthesis. But oxygen was usually taken up by heterotrophic bacteria at the surface of the biofilm. Nitrate, therefore, was the next alternative for the bacteria in the inner layer as the electron acceptor. With this in mind, simultaneous nitrification and denitrification could occur even under conditions of excess aeration. Therefore, the bacteria in the AGS should be able to adapt better to the

anoxic/oxic conditions than the bacteria in the SGS because there was a portion of bacteria that stayed constantly under an anaerobic or anoxic condition (anaerobic layer on the biofilm), whereas some bacteria in the suspended growth system might not be able to adjust to the changing anoxic/oxic conditions because they are obligative bacteria - bacteria that can only survive under one set of condition (either aerobic or anaerobic, not both). Due to the above reasons, there should be more bacteria in the AGS that were capable of denitrification. And because of the high ammonia loading of swine manure, a high nitrate concentration was usually produced after the aerobic cycle. The population of denitrifiers in the SGS might not be able to handle the high nitrate concentration. In order to compensate for that, a portion of the Bio-P bacteria in the SGS might switch to nitrate as the electron acceptor to assist in denitrification.

5. 1. 2. 2 TOC = 4,000 mg/L

The track analysis for the SGS's 4,000 mg/L run was very similar to the AGS (figure 5. 9 and 5. 10). As illustrated by figure 5. 9, the nitrate knee was not seen on the ORP curve. From results of previous runs and other researches, it was found that the nitrate knee usually occurs between -100 to -200 mV. Influent loading often results in a sharp drop in the ORP curve since influent feeding reduces the redox potential by increasing the ratio of reduced materials to oxidized materials in the reactor. It would be very possible that the sharp drop in ORP curve overlapped with the nitrate knee. However the exhaustion of nitrate could be proved by the pH curve in figure 5. 9 despite the inability of the ORP curve to detect the nitrate knee. The pH curve dropped sharply at

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Figure 5. 9 pH and ORP patterns throughout one cycle during the 4,000 mg/L run for the suspended growth system
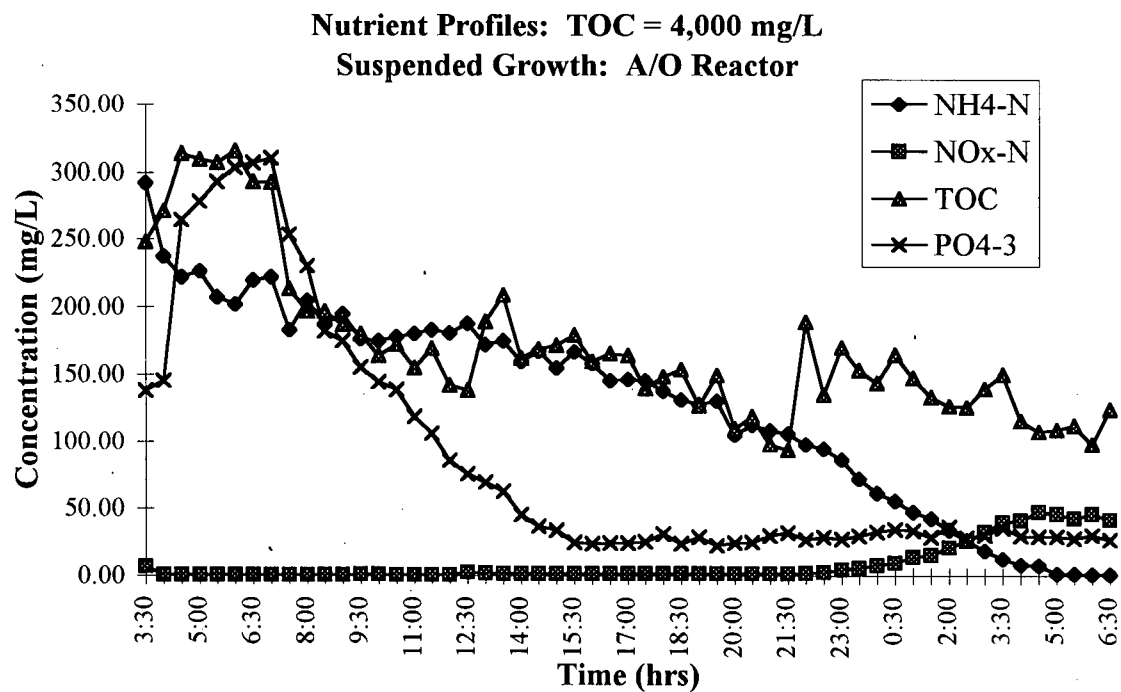
Figure 5. 10 Nutrient profiles throughout one cycle during the 4,000 mg/L run for the suspended growth system

the beginning of the anoxic cycle. This was attributed to the new high strength wastewater entering the A/O reactor. As influent feeding stopped and the mixed liquor was completely mixed, the pH curve rose slightly because of rapid denitrification. Referring to figure 5. 9 and 5. 10, the disappearance of nitrate occurred at the same time as the nitrate apex, proving that pH is a valuable tool to monitor any anoxic or anaerobic activities. The drop beyond the nitrate knee was due to the production of VFA under anaerobic condition. Also from these figures, it could be seen that rapid denitrification occurred in the A/O reactor upon the cessation of aeration. At the end of the aerobic phase, the nitrate concentration was up to approximately 30 mg/L. It should be noted that samples were not taken during settling or influent feeding. As the anoxic cycle began (after influent feeding), the nitrate concentration decreased down to approximately 10 mg/L. This indicated that denitrification could have occurred during settling and influent feeding as no air was provided during that period of time. One factor that might have enhanced the denitrification rate after the aerobic period and before influent feeding was the increased residual carbon content that remained in the A/O reactor. As mentioned before, the RCMP does not represent a complete removal of carbon if the organic loading was low. However, the high organic content in the influent caused the complete removal of carbon to be at the RCMP for this run. Because the control point was set in between the NBP and RCMP, there would be more carbon remaining in the system after the air was shut off. Denitrifiers would use these soluble carbon content for denitrification during settling. Denitrification continued during influent feeding as fresh carbon source was added to the A/O reactor.

The pH, ORP, and nutrient profiles behaved in the same manner as the 4,000 mg/L run for the AGS. It could be seen that simultaneous nitrification denitrification occurred as the nitrate concentration did not increase even though the ammonia concentration decreased during the aerobic phase. The release of OH⁻ from denitrification caused the pH to stay constant even though ammonia was also being consumed at the same time (from point C to point D). The nitrate concentration began to increase when there was a balance between the nutrients in the mixed liquor and the oxygen uptake rate. This led to the stoppage of simultaneous nitrification denitrification. The consumption of alkalinity (NH_4-N) solely caused the drop in pH beyond point D. The NBP on the ORP curve was very apparent and matched the NBP on the pH curve. Phosphorus release and uptake occurred in the anoxic and aerobic phase, respectively.

5. 1. 2. 3 TOC = 500 mg/L

The track analysis for the A/O reactor for this concentration are shown on figure 5. 11 and 5. 12. The pH and ORP curves displayed several distinct features that were similar to certain cycles for the same run of the AGS. During the beginning of the anoxic phase, the ORP curve dropped sharply. That was due to fresh wastewater entering the A/O reactor. Some nitrate could have been removed during the influent feeding stage when fresh carbon from the wastewater were readily available for consumption. However, the ORP value rose slightly after the initial drop and stayed at approximately -70 mV for a short period of time before dropping again. As mentioned before, the period where the ORP value stayed constant is known as the nitrate knee. Usually the nitrate knee is not as

pH and ORP Curves: TOC = 500 mg/L
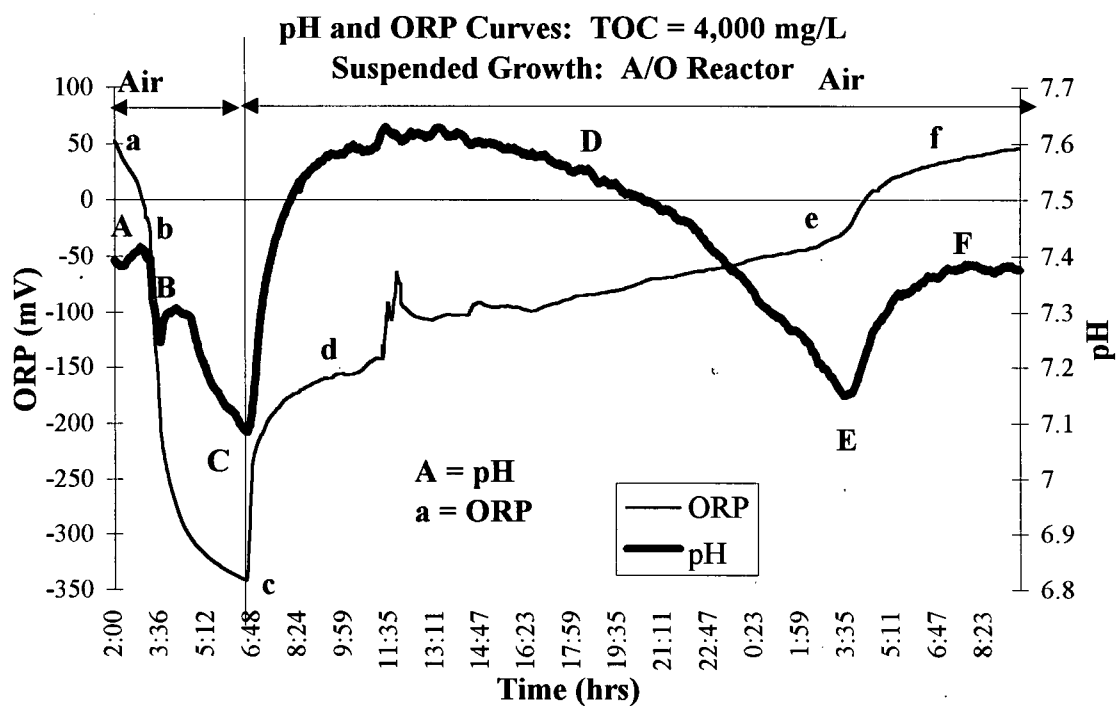## Suspended Growth: A/O reactor
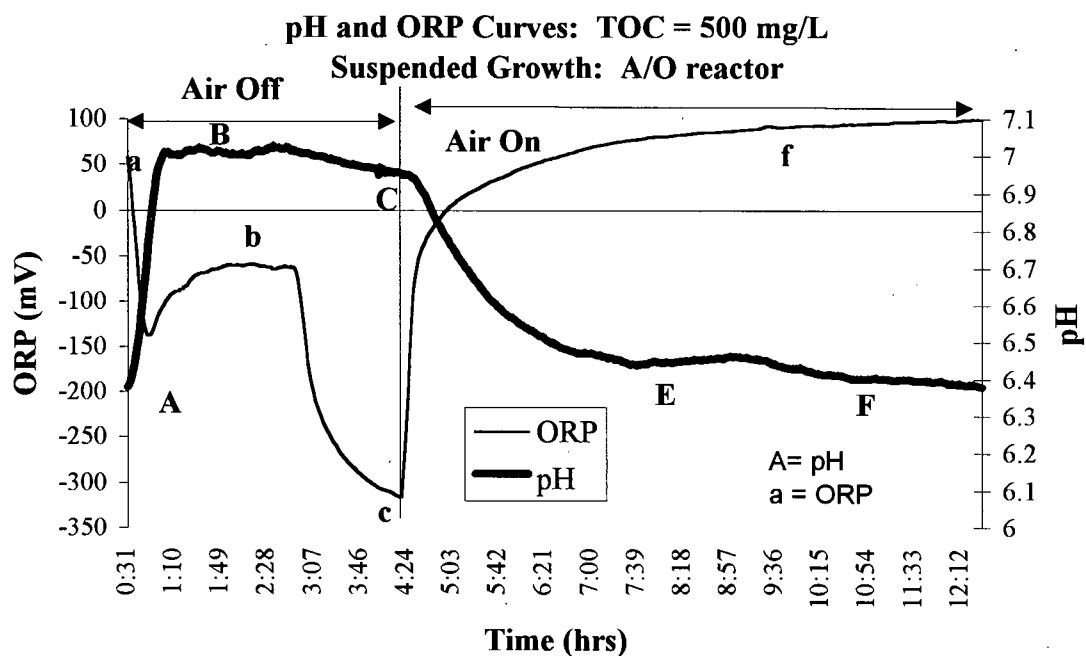
Figure 5. 11 First track analysis of pH and ORP patterns throughout one cycle
during the 500 mg/L run for the suspended growth system

Nutrient Profiles: TOC: 500 mg/L
Suspended Growth: A/O reactor
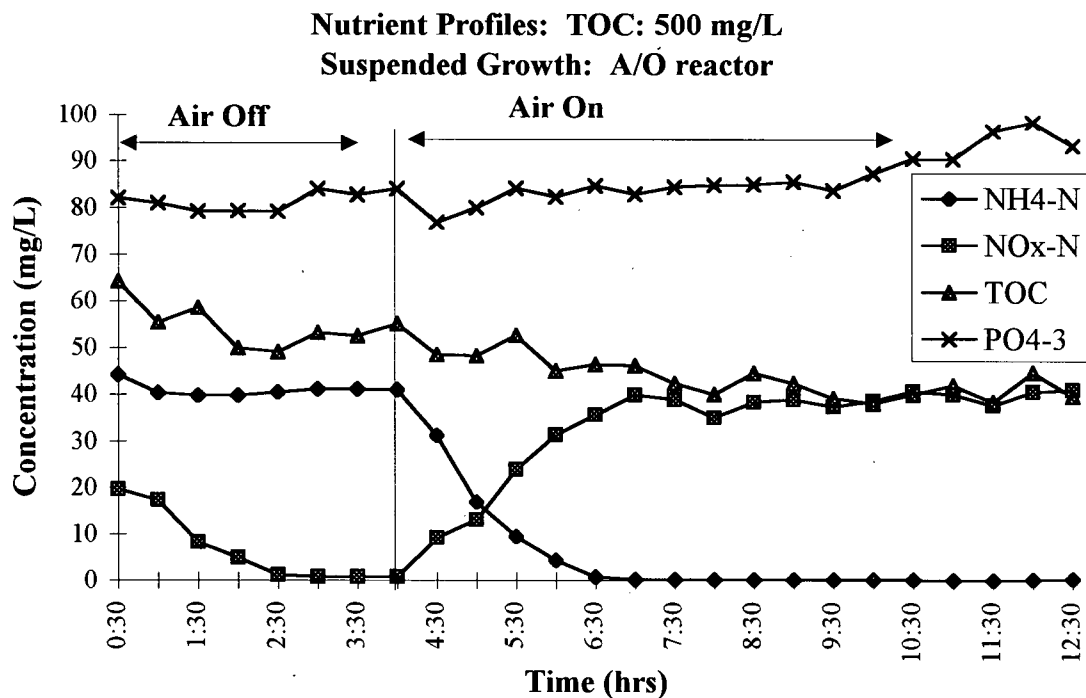
Figure 5. 12 First track analysis of nutrient profiles throughout one cycle during the
500 mg/L run for the suspended growth system

obvious as the one illustrated in figure 5. 11 because fresh carbon from the wastewater

was used for rapid denitrification once it entered the reactor. However, the TOC

concentration of the influent was merely 500 mg/L. The nitrate knee indicated that there

wasn't enough soluble carbon substrate in the influent to carry out rapid denitrification.

Instead, bacteria depended upon endogenous respiration for cell synthesis. Therefore, it

took a longer period of time for the complete removal of nitrate. As shown by figure 5.

11 and 5. 12, denitrification wasn't completed until the disappearance of the nitrate knee

on the ORP curve. It could be seen that there was a slight decrease in TOC concentration

during the anoxic phase. That was attributed to the use of readily biodegradable materials

for denitrification. The prolonged period for complete denitrification was probably a good

indication of the degree of phosphorus release during the anoxic phase. It was discovered

by other runs that nitrate was an inhibitor to phosphorus release as denitrifiers and Bio-P

bacteria competed for carbon source to carry out their appropriate reactions. An

inadequate supply of carbon substrate from the influent caused very poor phosphorus

release. As shown by figure 5. 12, instead of rapid P-release, a very slow uptake of P in

the presence of nitrate and a slight release of phosphorus after complete denitrification

were observed during the anoxic phase. This correlated well with discoveries of several

researchers (Gerber et al, 1986, Iwema et al., 1985). In Gerber's research, it was found

that with the absence of a carbon substrate and the presence of nitrate, uptake of

phosphorus was observed. He concluded that some polyP organisms were capable of

using nitrate as an electron acceptor for the oxidation of stored PHB. In Iwema's

research, it was discovered that in the absence of both a carbon substrate and nitrate, a

slow endogenous release of phosphorus could be found. It is shown by this track analysis

that insufficient carbon in the wastewater caused the lack of P-release during the anoxic phase. Insufficient P-release could also be described by the pH curve. The initial rise of the pH curve was due to the higher pH influent entering the reactor. The release of OH⁻ ion from denitrification using fresh carbon substrate could also contributed to the rise in pH. However, the pH did not drop immediately after the initial rise like all the other runs. Nitrate still wasn't completely removed from the system. A slower rate of denitrification occurred after the initial removal, as represented by the pH curve. The pH value rose slightly at a slower rate after the steep initial rise, indicating the OH⁻ ion was being released at a slower rate. That was due to the use of endogenous respiration instead of fresh carbon substrate by the bacteria for denitrification. It could be seen that nitrate removal was slower after the initial first hour and a half. There was an abrupt slope change on the pH curve after the complete removal of nitrate. The slight decrease in pH was caused by the hydrolysis of lipids or similar materials into VFA. However, not enough VFA were produced to enhance P-release during the anoxic phase. The ammonia concentration, on the other hand, stayed relatively constant during the anoxic phase of this track analysis.

Some interesting features on the pH and ORP curves were also shown during the aerobic phase. The NBP was not seen on the ORP curve as in the case of the 500 mg/L run of the AGS. This was caused by excessive aeration. There wasn't a balance between the aeration rate and the biomass concentration and the nutrient levels in the system. Plenty of oxygen was available as air was introduced into the reactor. Most of the oxygen in this case was utilized by nitrifiers because the carbon content was practically depleted during the anoxic phase. The rapid rise in the ORP curve at the beginning of the aerobic

phase was due to the introduction of air. Due to rapid nitrification, the NBP was masked within this rise. A decrease in aeration rate would cause the nitrification rate to decrease, which would in turn lead to the appearance of NBP on the ORP curve. The NBP on the pH curve was not very distinguishable either. It could be seen that the slope change after the NBP was very minute, making it difficult to notice the NBP. The rise in a normal pH curve after the NBP is very complex and not yet completely understood. But this could have been caused by an insufficient supply of carbon from the wastewater. The major cause for the decrease in pH at the beginning of the aerobic phase was due to the consumption of ammonia in the wastewater. However, the production of CO_2 by heterotrophic bacteria could also contribute to the decrease in pH during a cycle with an adequate supply of carbon from the influent. Generally, there was no CO_2 stripping by air during the nitrification period because all the oxygen was used up by bacteria for electron acceptors, maintaining an equilibrium in gas-solubility within the reactor. This led to an accumulation of CO_2 in the solution, which decreased the pH of the system. At the point where ammonia is completely removed, excess oxygen appears in the reactor. Dissolution of oxygen continues until the reactor is totally saturated. The excess oxygen caused a shift in the equilibrium of gas solubility. In order to maintain that equilibrium, some gases such as CO_2 will be stripped by air. During this run, however, there was little CO_2 accumulated in the solution because heterotrophic reactions were limited due to the lack of carbon source available. The little amount of CO_2 generated was stripped immediately after the NBP, causing the slight rise in pH. However, the slope change wasn't large enough for the computer to recognize. Therefore, the cycle continued until the computer

recognized another point further down the cycle that matched the control point that was originally set.

There was a slight increase in the phosphorus concentration by the end of the cycle. The excess air and the lack of nutrients in the system might have put too much stress upon the Bio-P bacteria. Also the microorganisms might have been using endogenous respiration for survival during the aerobic cycle. Without a replenishment of carbon substrates, cell lysis could have occurred. That would release nutrients such as phosphorus and VFA that were stored in the bacterial cells into the solution. This was probably the reason why there was a slight decrease in pH by the end of the cycle.

The second track analysis for this run is shown in figures 5. 13 and 5. 14. This pH curve resembled a normal pH pattern. Due to the low organic loading rate, the RCMP did not represent a complete removal of carbon. This point occurred before the NBP. The nitrate apex was very clear and coincided with the nitrate knee on the ORP curve during the anoxic phase. As shown by the nutrients profiles, a certain amount of TOC was used immediately for denitrification at the beginning of the anoxic phase. In fact, denitrification using fresh carbon could have taken place during the influent feeding stage. Even though samples weren't taken during the influent feeding stage, it could be seen that the nitrate concentration after the aerobic phase was higher than the nitrate concentration at the beginning of the anoxic cycle, suggesting that fresh carbon was used for denitrification while influent was entering the reactor. It could be demonstrated that the organic loading was higher for this cycle because complete removal of nitrate occurred faster than the other track analysis. The NBP was more apparent on the pH curve and coincided with the ORP curve during the aerobic phase. There was basically no differences between the two

pH and ORP Curves: TOC = 500 mg/L
Suspended Growth: A/O reactor

Figure 5. 13 Second track analysis of nutrient profiles throughout one cycle during
the 500 mg/L run for the suspended growth system

Nutrient Profiles: TOC = 500 mg/L
Suspended Growth: A/O reactor
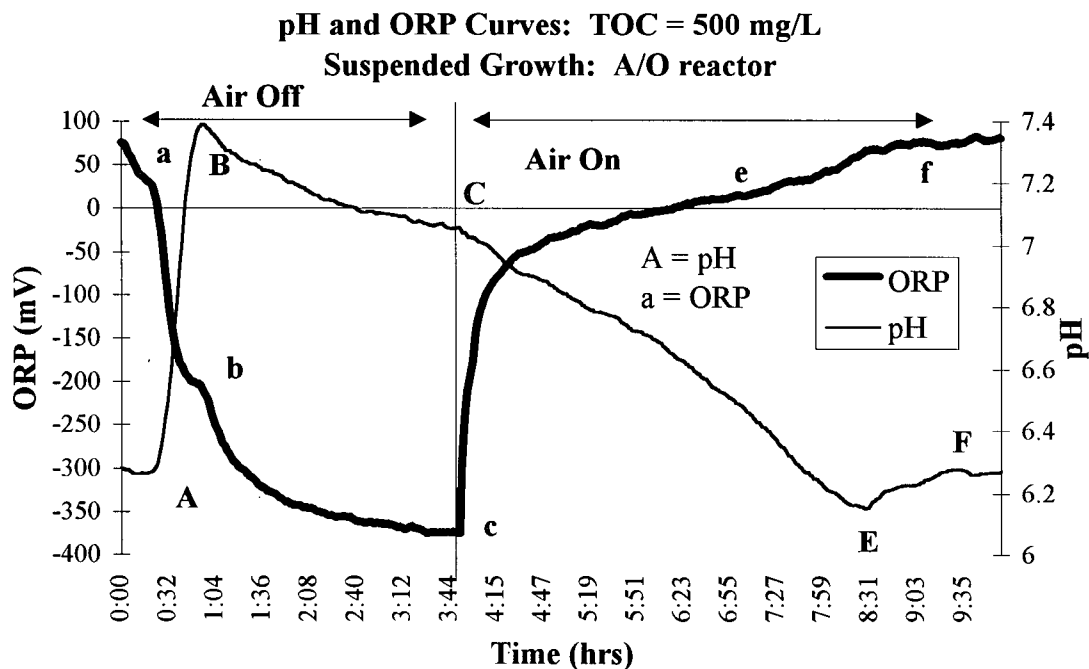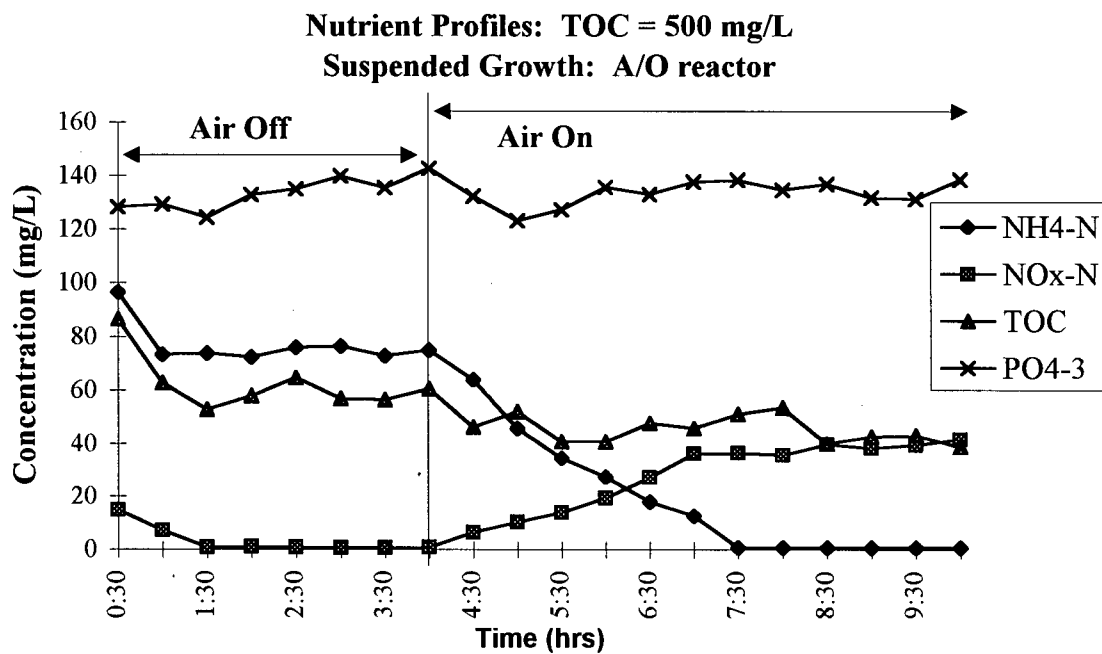

Figure 5. 14 Second track analysis of nutrient profiles throughout one cycle during
the 500 mg/L run for the suspended growth system

track analysis. All the parameters measured were very similar to the other track analysis. The organic loading for this cycle might have been slightly higher, and therefore created more reactions by heterotrophic bacteria. These reactions caused an accumulation of CO_2 in the system, which led to a more visible rise in pH after the NBP when air stripping of CO_2 occurred.

5. 2 Designation of Control Point Using pH as a Control Parameter

Points E and F on the pH profile and points e and f on the ORP profile could be used for real-time control. These points represent a complete removal of a nutrient and have an abrupt and constant slope change that could be easily set-up for a real-time control process. Point F, in most cases, would be preferred over point E for control purposes because at point F most of the carbon content in the A/O reactor has been exhausted. If the control point was set at point E at which there is a complete removal of ammonia, there might still be an accumulation of carbon in the A/O reactor, which might lead to a high concentration of TOC in the final effluent.

The point that was used for control in this research was in between point E and F, and not point F itself because a small amount of organic carbon must be left in the A/O effluent for denitrification in the anoxic reactor. Subsequently, the organic carbon that was transferred to the anoxic reactor along with endogenous respiration by the bacteria would be used for denitrification. There was no control for the anoxic phase in the A/O reactor. A set-time of approximately 4 hours was provided based on previous studies (Fernandes et al., 1991). Once aeration begins, the computer would recognize each point

that was programmed into the software step by step. Aeration would stop once the target point had been reached. Settling, drawing, and feeding would then occur accordingly.

It is not recommended to extend aeration beyond point F on the pH curve or point f on the ORP curve. Aeration beyond this point might cause a complete removal of organic carbon, which would retard the rate of denitrification in the anoxic reactor. Continual aeration and the lack of nutrients remaining in the reactor might also lower the treatment efficiency of the system by placing too much stress on the bacteria. Cell lysis would occur and nutrients would be released back into the solution. Aeration and mixing energy could also be saved as it was unnecessary to continue aeration after point F (most of the nutrients had been removed).

It was difficult to use the actual pH value for control because the changes between the actual pH value were minimal (often from 6.5 to 7.8). It was suggested that the mV pH readings should be used for control purposes due to a larger slope change over time (0 to -100 mV, see figure 5. 15).

The flow chart for the control strategy is illustrated in figure 5. 16. The average mV pH values from 4 pH probes inserted in the A/O reactor was scanned at an interval of 1 minute. They would then be processed into calculating it's moving average. A moving average was a calculation in which the last "x" readings would be used as an average value. This tabulation would give a more accurate and stable control operation. In this research, the "x" was denoted to be 10. The slope of the mV pH curve along with it's moving average were then calculated according to the value obtained above. However, the scanning interval was set at 8 minutes so that a significant change of slope could be detected. The "x" that was used for the moving average in this case was equated to 5 due
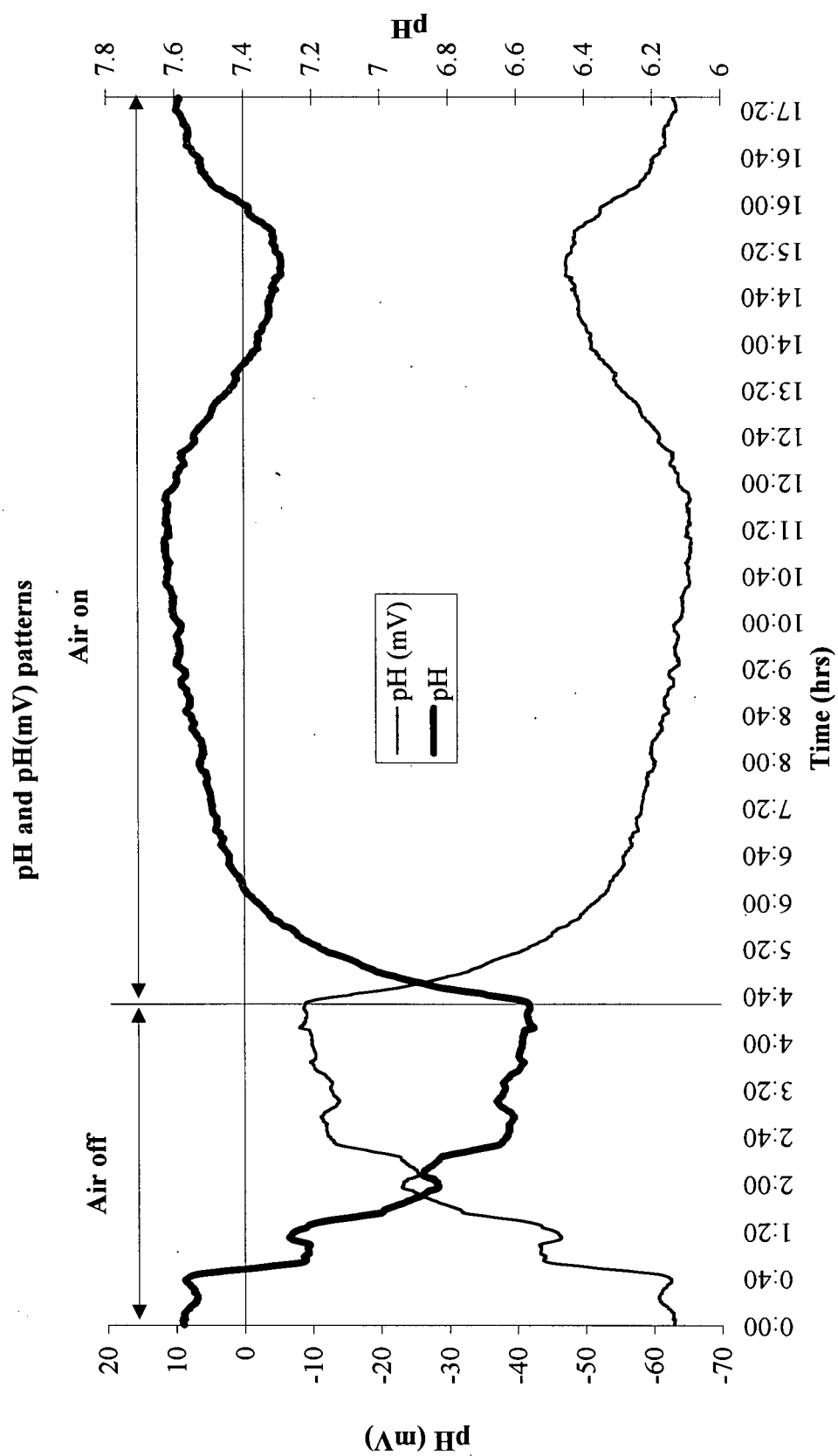
Figure 5. 15 Variations of pH and pH(mV) during one cycle
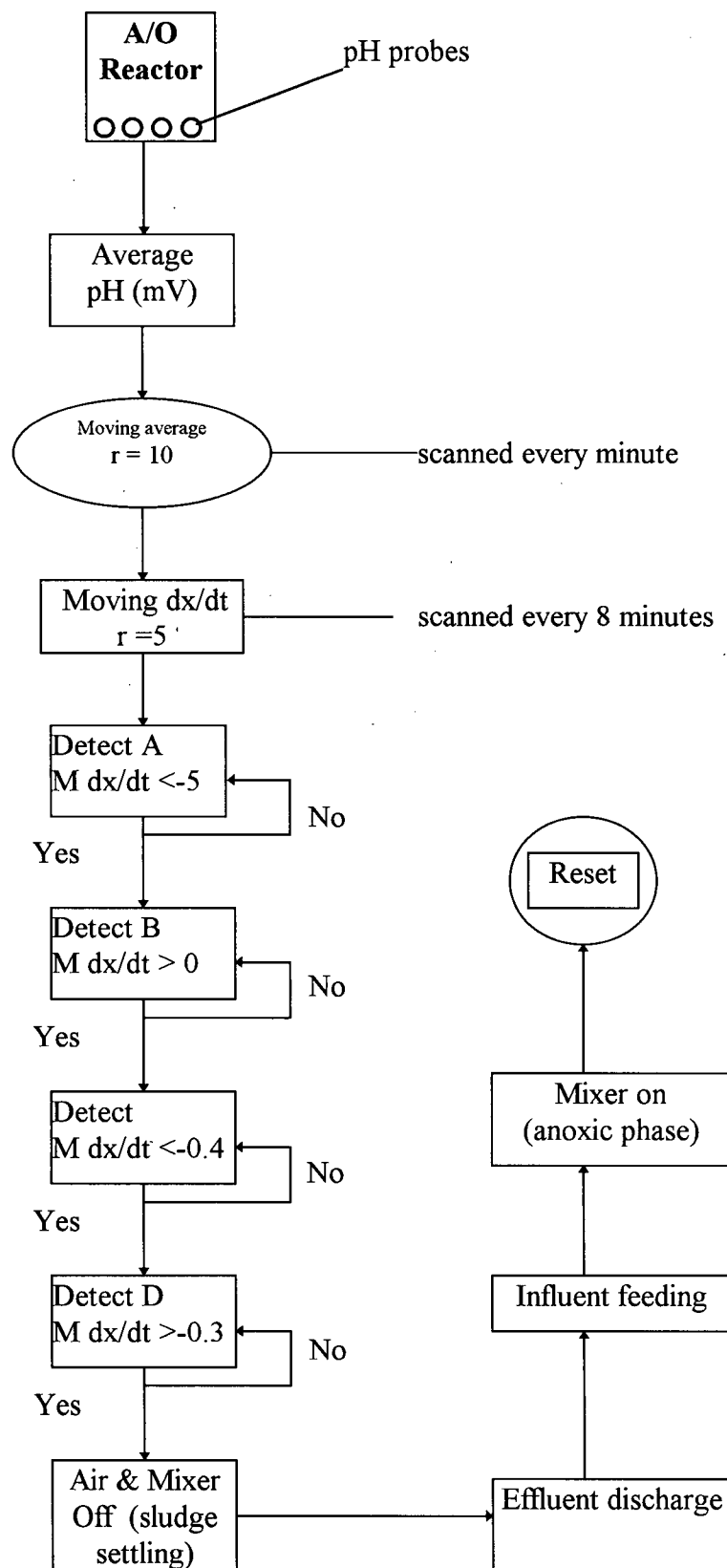
Figure 5. 16 Flow chart for control strategy

to the change of the scanning interval. The computer would then proceed to the next stage to recognize four distinct points on the curve (see figure 5. 17):

1) when the slope was smaller than 0.5

2) when the slope passed above the x-axis

3) when the slope returned below the x-axis and passed beneath the -0.5 mark

4) when the slope became larger than -0.4

When the computer recognized the last stage, it would automatically send the signals to the relay box. The computer would then carry out the appropriate actions that were programmed into the software. To avoid any inaccuracy in the control process, the control strategy was programmed to recognize each feature step by step until occurrence of the final designated control point.

5. 3 System Performance

5. 3. 1 Attached Growth

5. 3. 1. 1 TOC = 2,000 mg/L

The pH, ORP and DO patterns generated from this real-time control process were consistent and uniform throughout this run despite the fluctuations in influent concentrations (see figure 5. 18 and 5. 19). At the beginning of the aerobic cycle, the DO
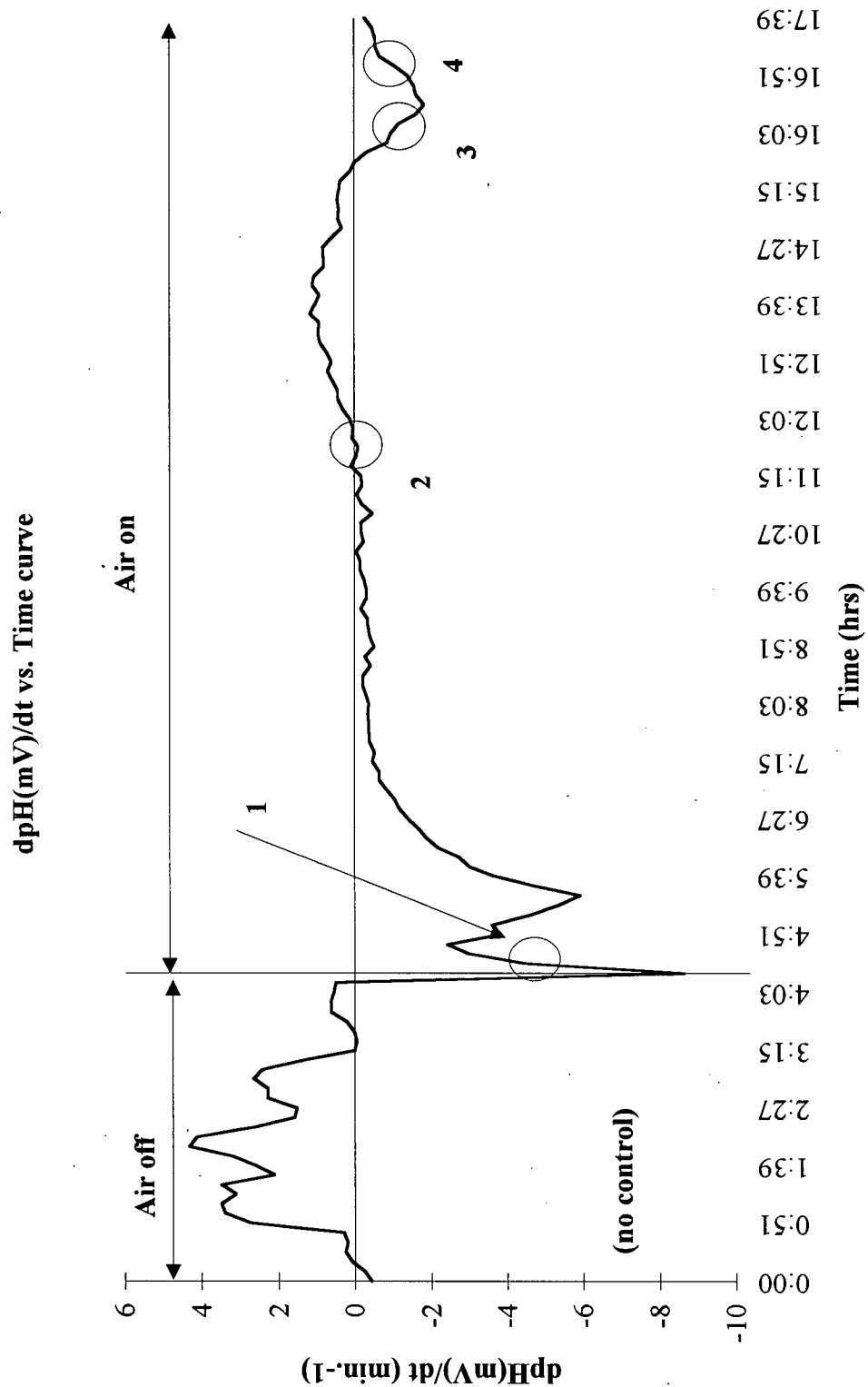
Figure 5. 17 Four distinct points on the dpH(mV)/dt curve recognized by the computer
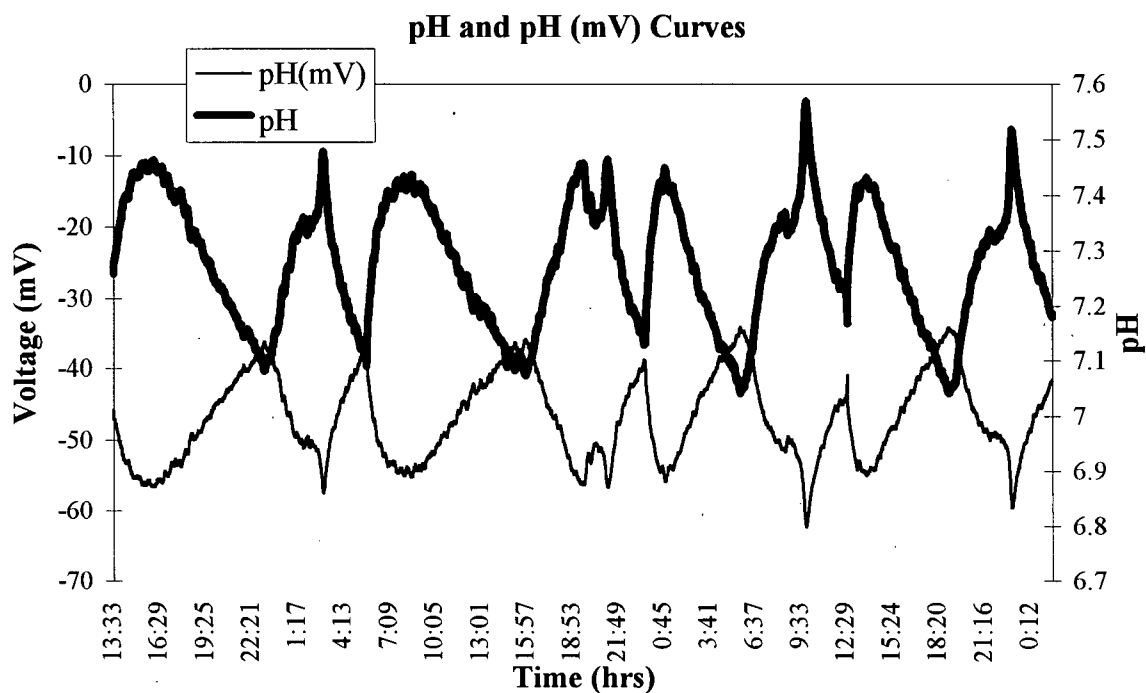
pH and pH (mV) Curves

Figure 5. 18 Consistent pH and pH(mV) curves of the 2,000 mg/L run for the attached growth system

ORP and DO Patterns
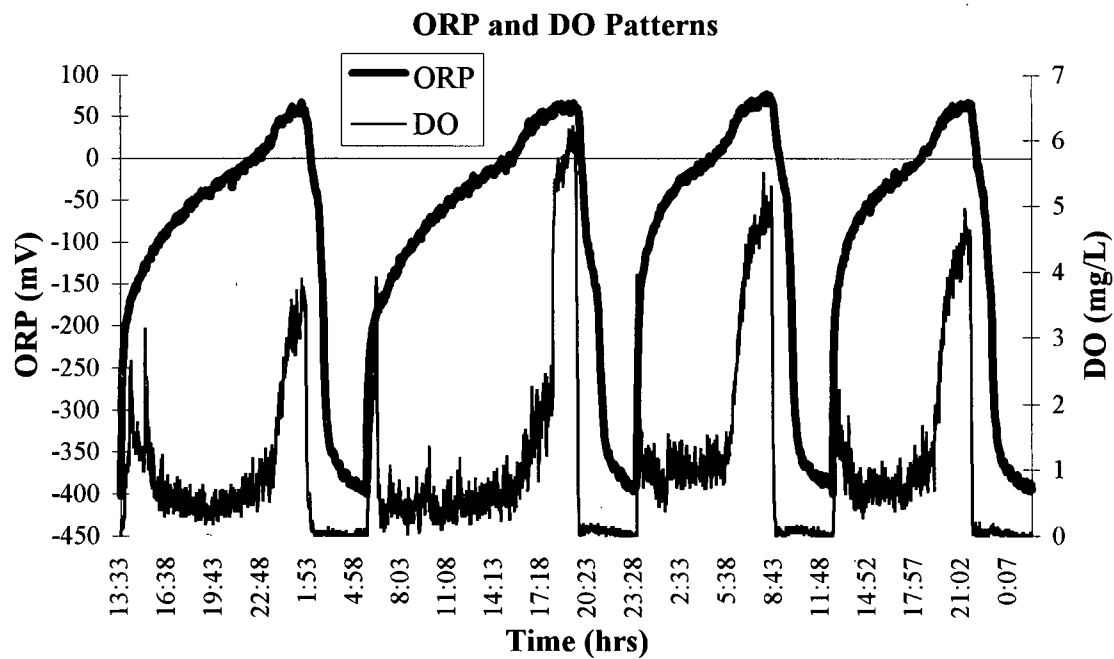
Figure 5. 19 Consistent DO and ORP curves of the 2,000 mg/L run for the attached growth system

stayed steady at a very low concentration despite the introduction of oxygen into the A/O reactor. This was attributed to the OUR by both the nitrifiers and carbon oxidizing bacteria being greater than the aeration rate - meaning that the oxygen was being consumed once it became available. This was demonstrated by the reduction in ammonia and TOC concentrations during the aerobic cycle. As illustrated in figures 5. 18 and 5. 19, the DO "jump" coincided with the NBP on both the pH and ORP curves. The "jump" in DO concentration was caused by the lowered oxygen uptake rate of the bacteria. This point represented the end of nitrification, meaning that the nitrifiers no longer needed the oxygen to carry out their oxidation reactions as there was no ammonia left in the reactor (rate of oxygen consumption decreased). This resulted in excess oxygen being present in the reactor, which in turn caused the increase in DO concentration. As mentioned before, the ORP curve was very sensitive to the changes in DO concentration in the reactor. It was reported in another research that an increase in DO concentration from 0 to 2 mg/L would cause an increase in the ORP value of 200 mV (Saune et al., 1996). This could explain the abrupt increase in the ORP curve after the NBP. Because of the decreased oxygen consumption rate, the DO concentration continued to climb until the reactor reached saturation. The plateau on the DO curve after the "jump" corresponded with the RCMP on the ORP and pH curve. Oxygen was not utilized by the bacteria as the nutrients were exhausted from the system. This created the plateau on the DO curve. One possible set back in using DO as a control parameter is that it does not have the ability to represent any anaerobic activities, as illustrated in figure 5. 19. The oxygen concentration was constantly near 0 mg/L during the anoxic phase.

The success of a real-time control system was based on the consistency of the patterns produced and the complete removal of pollutants in the wastewater despite the time difference for each cycle. The HRT of the system was not constant because the length of the aerobic phase depended upon when the computer recognized the "trigger" point. Figure 5.20 reflects a flexible HRT in the A/O reactor. The influent was diluted to a concentration of approximately 2,000 mg(TOC)/L. The HRT for this run was averaged to be 2.56 days in the A/O reactor, and 1.71 days in the anoxic reactor. The total HRT for the entire system, therefore, was approximately 4.27 days.

Table 5.1 summarizes the performance of the real-time control process for this run. The C/N ratio in this project was based on TOC/NH_4-N because the pH pattern in a real-time system changes according to the concentration of the above. The average influent concentrations of TOC and NH_4-N were 2117.4 mg/L and 631.01 mg/L, respectively. The relative C/N ratio, therefore, was 3.36. This was considerably lower than the C/N ratio used by Ra, 1997, which was approximately 12.2 (extremely high). The higher C/N ratio meant that less nitrate would be produced during the aerobic cycle, while more carbon would be available for denitrification and phosphorus release. The higher C/N ratio would probably lead to a better treatment efficiency. In his research, he was able to obtain removal efficiencies of 97.9%, 98.6%, and 95% in TOC, NH_4-N, and PO_4^{-3} respectively, while maintaining an average final NO_x-N effluent concentration of 2.7 mg/L.

During this run, the concentrations of the ammonia influent, A/O effluent, and final effluent were 631.01 mg/L, 0.78 mg/L, and 0.82 mg/L respectively. The removal of ammonia occurred in the A/O reactor only. There was no increase in the ammonia
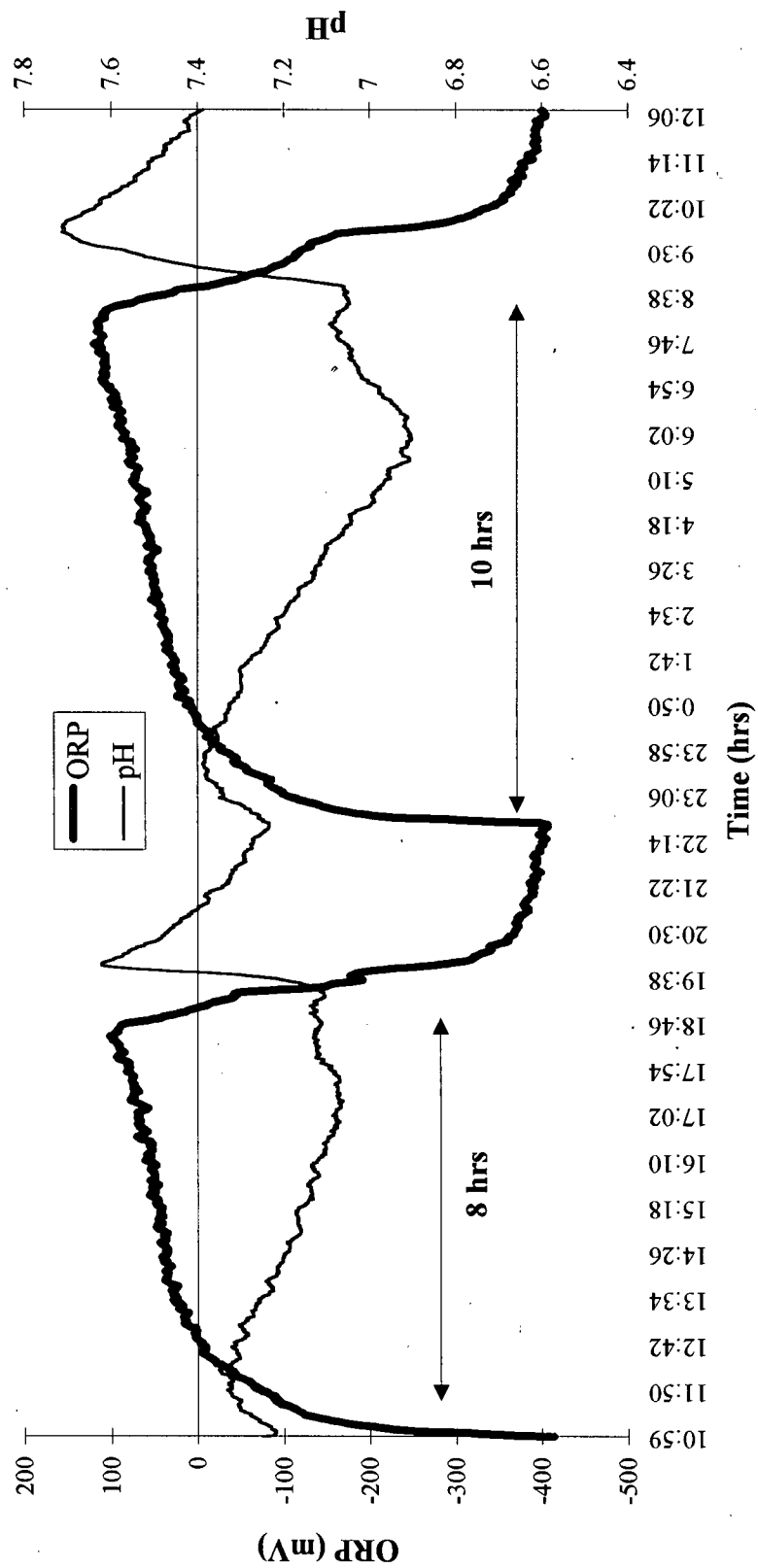
Figure 5. 20 Differences in cycle length

Table 5.1 Summary of system performance for the 2,000 mg/L run (attached growth system)

	Influent (mg/L)			Attached Growth A/O (mg/L)			Effluent (mg/L)			% removal
	Average	Min. - Max.	Std.	Average	Min. - Max.	Std.	Average	Min. - Max.	Std.	
TOC	2117.4	1300 - 3624.5	593.39	100.13	76.2 - 144.3	16.34	69.66	62.35 - 80.12	4.44	96.47
NH_4-N	631.01	512.35 - 805.57	58.26	0.78	0 - 2.89	0.85	0.82	0 - 3.89	1.05	99.87
NO_x-N	0.7	0.31 - 1.12	0.25	66.25	11.72 - 104.89	27.56	29.03	6.85 - 55.92	14.79	54.16
PO_4^{-3}	216.55	172.45 - 303.82	41.71	108.16	75.2 - 127.57	17.44	113.68	90.36 - 127.56	10.43	45.43
BOD_5	2060.25	1570-5 - 3173.57	437.47	13.56	6.42 - 20.49	4.11	6.35	4.80 - 8.31	1.32	99.68
COD	5492.21	4480.42 - 7165.53	861.52	192.13	169.69 - 215.21	13.28	163.96	141.68 - 190.7	15.85	96.94
TS	4036.94	2244 - 6917.5	1828.77	1333.06	1060 - 1620	160.29	1184.72	1090 - 1282.5	70.77	65.71
TVS	2610.28	1445 - 4607.5	1312.47	543.33	410 - 677.5	97.48	403.33	320 - 500	65.76	81.53
TSS	3120.28	1440 - 5670	1493.56	106.11	0 - 255	107.7	66.11	0 - 160	57.32	97.33
TVSS	2307.5	1220 - 4642.5	1118.97	54.72	0 - 127.5	55.94	31.11	0 - 72.5	25.98	98.42
DO	0	-	-	5.83	4.28 - 6.96	0.83	6.18	4.32 - 6.99	0.62	-

concentration in the anoxic reactor. The removal efficiency for ammonia was approximately 100%. This did not come as a surprise as the control point was set beyond the NBP. Also nitrification did not require any carbon source because nitrifiers are autotrophs, which require carbon dioxide as an energy or carbon source. Therefore, there was no need for concern over the amount of carbon in the solution. The NH_4-N profile with time is illustrated in figure 5. 21.

The real-time control system also achieved a very high removal efficiency of over 90% in COD, BOD_5, and TOC. The TOC profile with time is illustrated in figure 5. 22. This result was expected because the control point was set at the point between the NBP and the RCMP. As illustrated in table 5. 1, the A/O and final effluent had average TOC concentrations of 100.13 mg/L and 69.66 mg/L respectively. The decrease in TOC concentration was the result of the subsequent use of the residual carbon in the A/O effluent by the denitrifiers in the anoxic reactor for the complete removal of nitrate. The influent BOD_5/COD ratio was approximately 0.38, while the BOD_5/TOC ratio was roughly 0.98. With regard to the organic carbon content, the swine wastewater had a similar carbon composition to typical wastewater in which the BOD_5/COD ratio varies from 0.4 to 0.8 and BOD_5/TOC varies from 1 to 1.6 (Metcalf and Eddy, 1991). The effluent, on the other hand, had a BOD_5/COD and BOD_5/TOC ratio of 0.039 and 0.091 respectively. This result suggested that most of the easily biodegradable carbon was removed from the solution, while the inorganic and long-chained organic carbon were not treated by the system.

The NO_x-N concentration in the influent was negligible. However, because of nitrification, much of the NH_4-N was converted into NO_x-N in the A/O reactor. The NO_x-

NH4-N Concentrations

Attached Growth: TOC = 2,000 mg/L
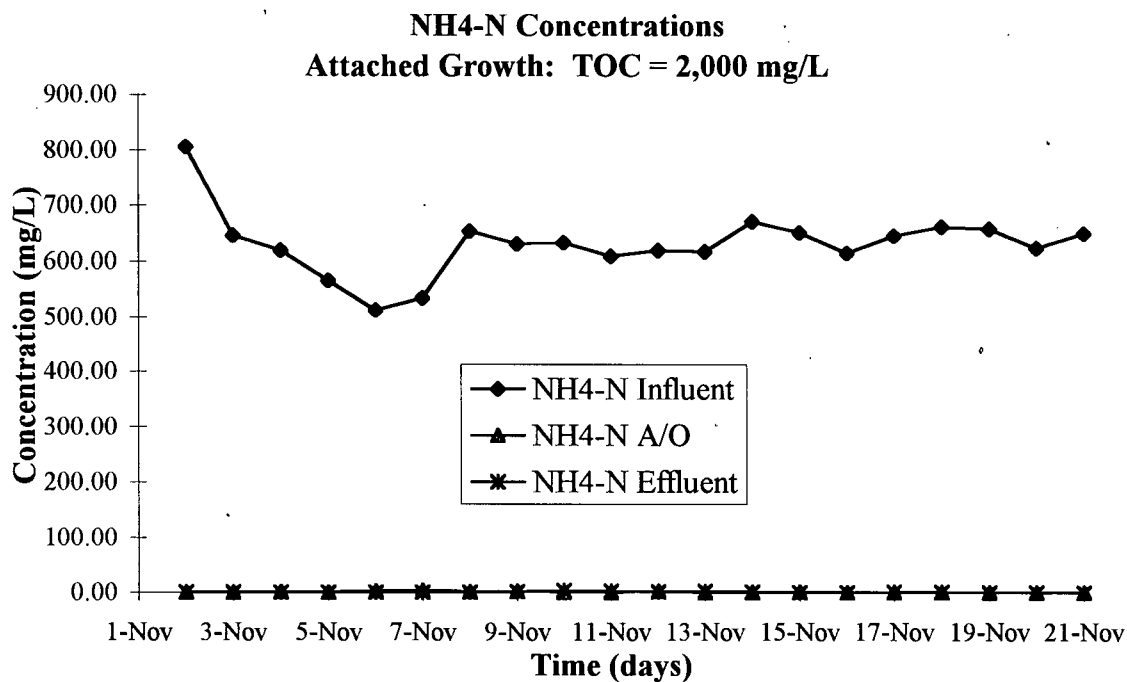
Figure 5. 21 NH₄-N profile with time for the 2,000 mg/L run (attached growth
system)

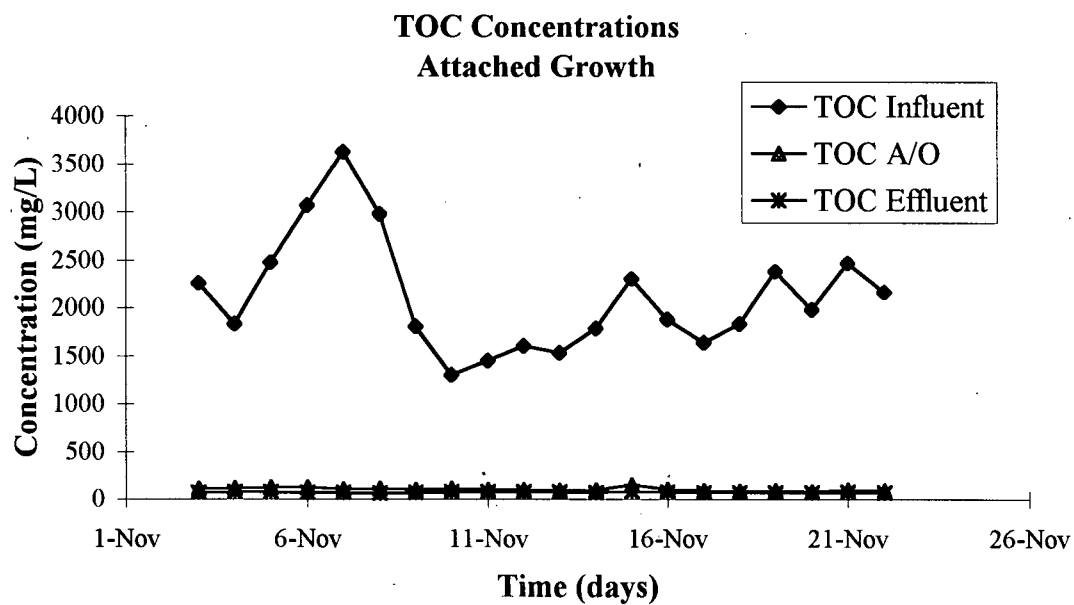**TOC Concentrations**

Attached Growth

Figure 5. 22 TOC profile with time for the 2,000 mg/L run (attached growth
system)

N was subsequently transferred to the anoxic reactor. The anoxic reactor was provided to prevent nitrate from discharging into the environment. The anoxic reactor was able to remove up to 55% of the nitrate from the A/O effluent using either the carbon source available in the A/O effluent or endogenous respiration by the bacteria. Incomplete removal of nitrate was probably caused by the high production rate of nitrate in the A/O reactor while having an insufficient supply of soluble carbon from the A/O effluent. This is illustrated by the track analysis done on the anoxic reactor through one cycle of this run (figure 5. 23). It can be seen that there was no visible decrease in the TOC concentration from the beginning to the end of the cycle. This indicated that there wasn't enough soluble carbon in the A/O effluent for the bacteria in the anoxic reactor to consume. Therefore, the bacteria relied heavily on endogenous respiration to carry out denitrification. However, using stored carbon in the bacteria's cell as an energy source often takes longer than using external carbon source. The time that was given for denitrification in the anoxic reactor, therefore, might not have been enough to reach the point of complete nitrate removal.

The NO_x-N profile with time is shown in figure 5. 24. On November 6th and 7th, the nitrate production in the A/O reactor was relatively low. The nitrate concentrations in the A/O effluent were 11.72 mg/L and 11.97 mg/L respectively. The average concentration of the A/O effluent excluding the two days above was 72.3 mg/L. The low production of nitrate in the A/O reactor was the result of a high loading rate. The TOC concentration of the influent on November 6th and 7th were 3066 and 3624.5 mg/L respectively (see figure 5. 22). These concentrations were the highest in this run. The high loading rate caused an imbalance between the aeration rate and the oxygen
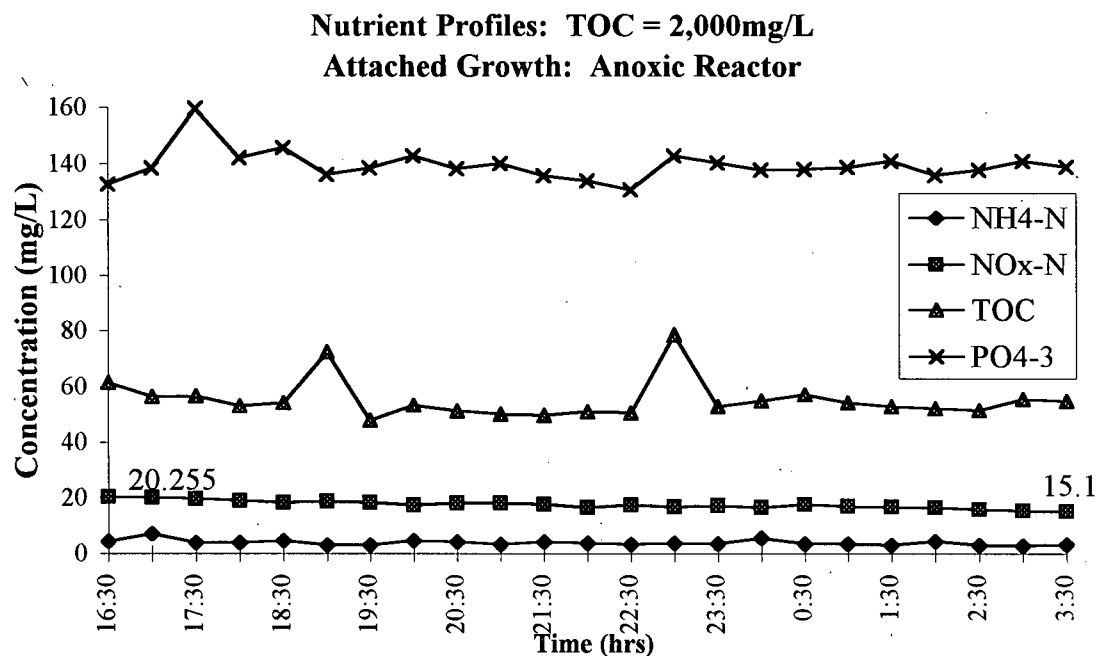
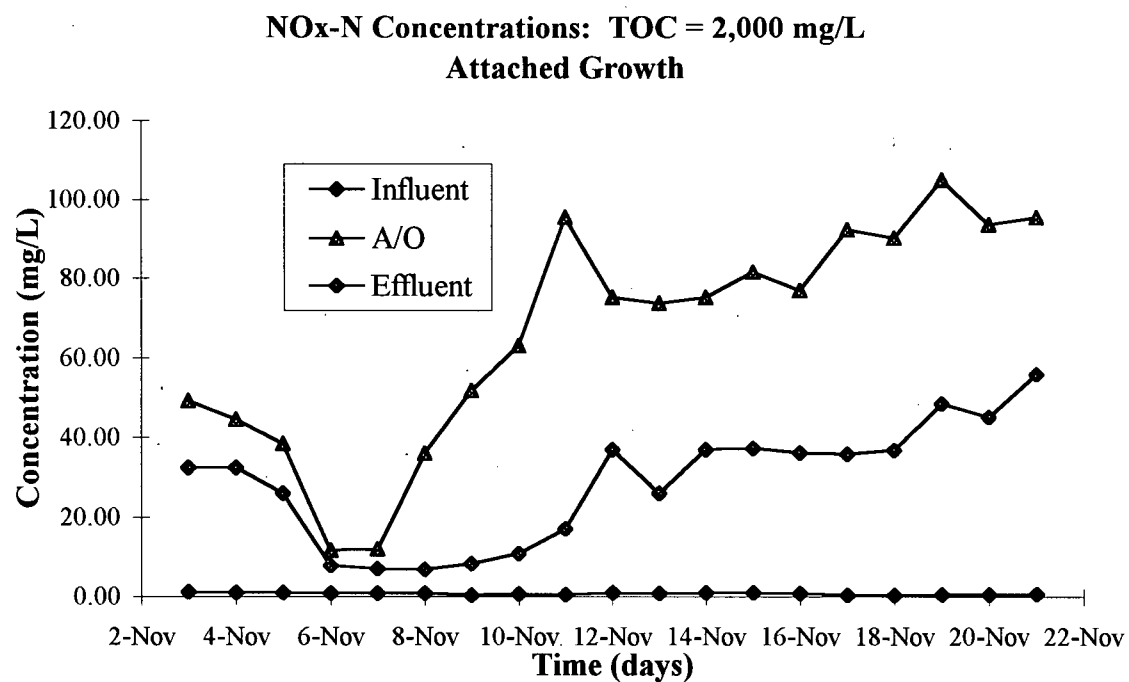**Figure 5. 23 Nutrient profiles of one cycle for the anoxic reactor**

Figure 5. 24 NO$_x$-N profile with time for the 2,000 mg/L run (attached growth system)

consumption rate, which resulted in simultaneous nitrification and denitrification during the aerobic cycle. Examining the ORP and pH curves on these days (figure 5. 25), it could be seen that the pH curve rose rapidly at the beginning of the cycle. The pH value didn't drop until 1 to 2 hours later. This was a characteristic of a high loading cycle where simultaneous nitrification and denitrification occurred (see section 5. 1. 1. 2 for more details). Because part of the nitrate was denitrified within the A/O reactor, the nitrate concentration in the A/O effluent was very low. This would in turn lead to a low concentration in the final effluent.

Sludge from the A/O reactor was added to the anoxic reactor to enhance nitrate removal by providing an alternative carbon source. The addition of sludge increases the biomass in the anoxic reactor, therefore providing more bacteria for denitrification. By providing sludge from the A/O reactor, endogenous respiration rate would increase. The bacteria in the transferred sludge might have stored extra carbon sources in their cells and might have extracted this stored energy to keep normal bio-chemical activity going when an external carbon source is not available. Also, there might be organic matter attached to the sludge that the bacteria in the anoxic reactor could hydrolyse. The extra carbon available would increase the denitrification rate in the anoxic reactor. Sludge was drawn from the A/O reactor and was added to the anoxic reactor on November 7th. The extra carbon content kept the nitrate concentration low until all the carbon had been consumed. Unfortunately no more sludge was added to the anoxic reactor for the rest of this run. Therefore, the NO_x-N concentration was relatively high at the end of the run.

The treatment efficiency for ortho-phosphate was only 45.43% for this run. The profile for PO_4^{-3} with time is shown in figure 5. 26. The low treatment efficiency was a
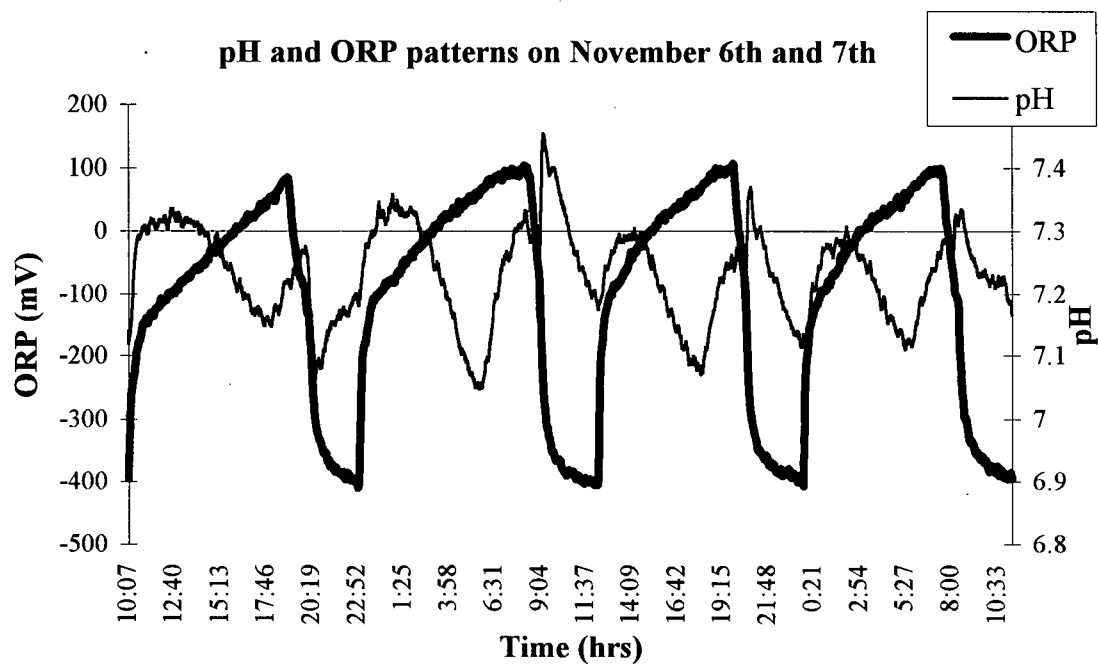
Figure 5. 25 pH and ORP curves for November 6th and 7th
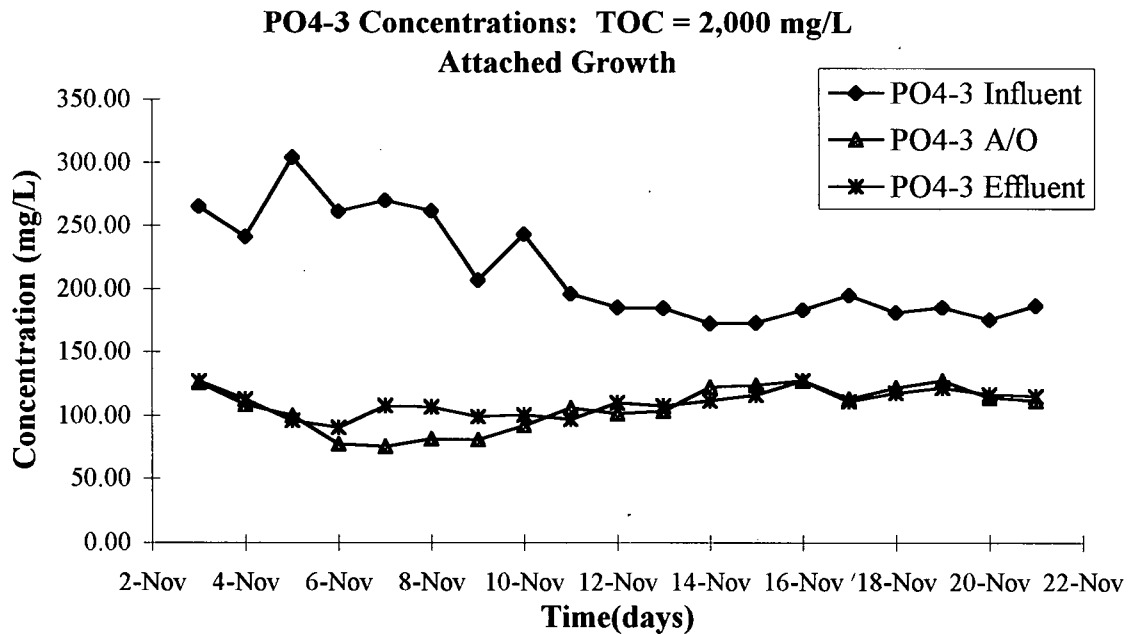
Figure 5. 26 PO_4^{-3} profile with time for the 2,000 mg/L run (attached growth system)

consequence of a relative low loading rate. The presence of a high nitrate concentration would result in an increase in the rate of carbon consumption, and a decrease in the rate of P release (Iwema et al., 1985). From the track analysis in figure 5. 1, section 5. 1. 1. 1, it could be seen that the rate of phosphorus release was greater after the nitrate knee, meaning that the presence of nitrate inhibited P release. Because of the high production rate of NO_x-N during the aerobic phase, much of the time in the anoxic period would be denoted for denitrification. The high nitrate concentration also forced the bacteria to consume much of the readily available carbon from the influent for denitrification. The high NO_x-N concentration along with an inadequate carbon content caused an insufficient release of phosphorus during the anoxic phase. This would in turn lead to ineffective phosphorus removal during the aerobic phase.

November 6th and 7th were the days with the best treatment efficiency for phosphorus. As mentioned before, these two days had the highest loading rate of this run. The lack of nitrate produced in the aerobic cycle during these days left sufficient time and carbon in the anoxic phase for efficient phosphorus release. It was found that the amount of phosphorus uptake increases with an increased VFA potential (Lie et al., 1997). In figure 5. 25, it could be seen that the pH curve dropped sharply during the anoxic phase. This was an indication of a high amount of VFA produced. The above explanations improved the bacteria's ability to uptake phosphorus during the aerobic phase; hence a better treatment efficiency.

It was also illustrated by figure 5. 23 that the phosphorus concentration was relatively constant in the anoxic reactor. Because the sludge was completely separated during the transfer of liquid from the A/O reactor, no Bio-P bacteria existed in the anoxic

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reactor. But even if there were any suspended Bio-P bacteria that escaped to the anoxic reactor during the liquid transfer, there wasn't enough carbon substrate in the A/O effluent for phosphorus release to take place.

The solids levels in influent, A/O effluent, and final effluent are represented in figures 5. 27 and 5. 28. The average TS, TVS, TSS, and TVSS removal efficiencies were 65.71%, 81.53%, 97.33% and 98.42% respectively. While high removal efficiencies were achieved for TVS, SS, and VSS, a relatively lower TS removal was observed. This might be the result of the influent wastewater containing a high level of non-organic solids in colloidal form. A high suspended solids removal rate suggested good settleability of sludge for the system.

5. 3. 1. 2 TOC = 4,000 mg/L

The success of a real time control system is marked by the consistency of the curves produced from cycle to cycle. The pH, ORP, and DO curves in the A/O were consistently obtained throughout this run (see figure 5. 29 and 5. 30). The advantage of the real-time process was again demonstrated as the time for each cycle was different due to fluctuations in the concentration of the influent. By using this real time control process, the entire system's performance, which included treatment efficiencies, energy savings, and cycle time was optimized.

As shown in figure 5. 30, the DO parameter also correlated with both the pH and the ORP curves during the aerobic phase. The DO profile was very similar to the ones produced in the 2,000 mg/L run. The DO "jump" and the DO plateau after the "jump"

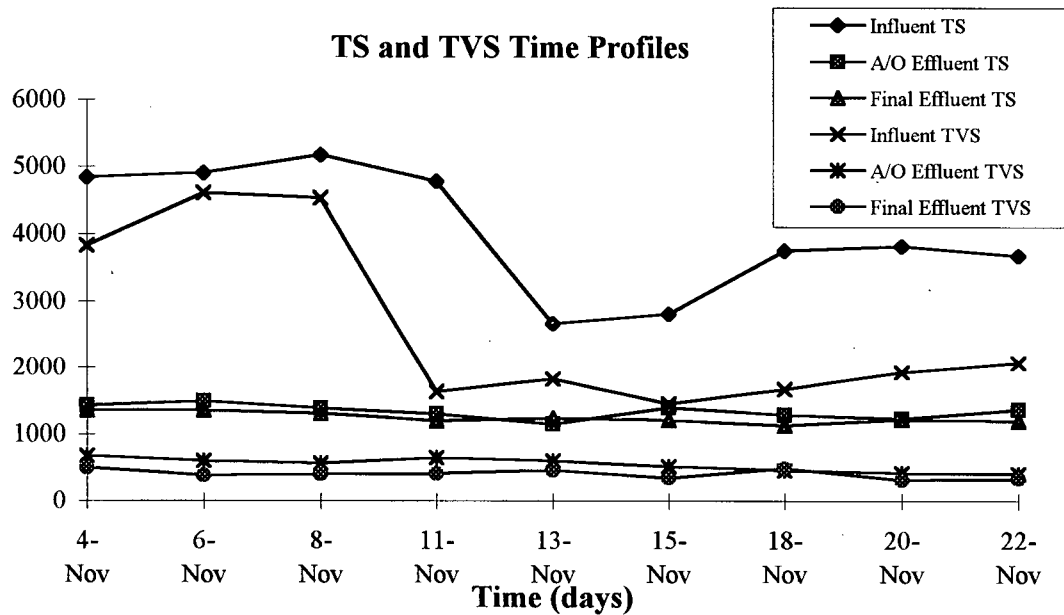**Figure 5. 27 TS and TVS profiles with time for the 2,000 mg/L run (attached growth system)**
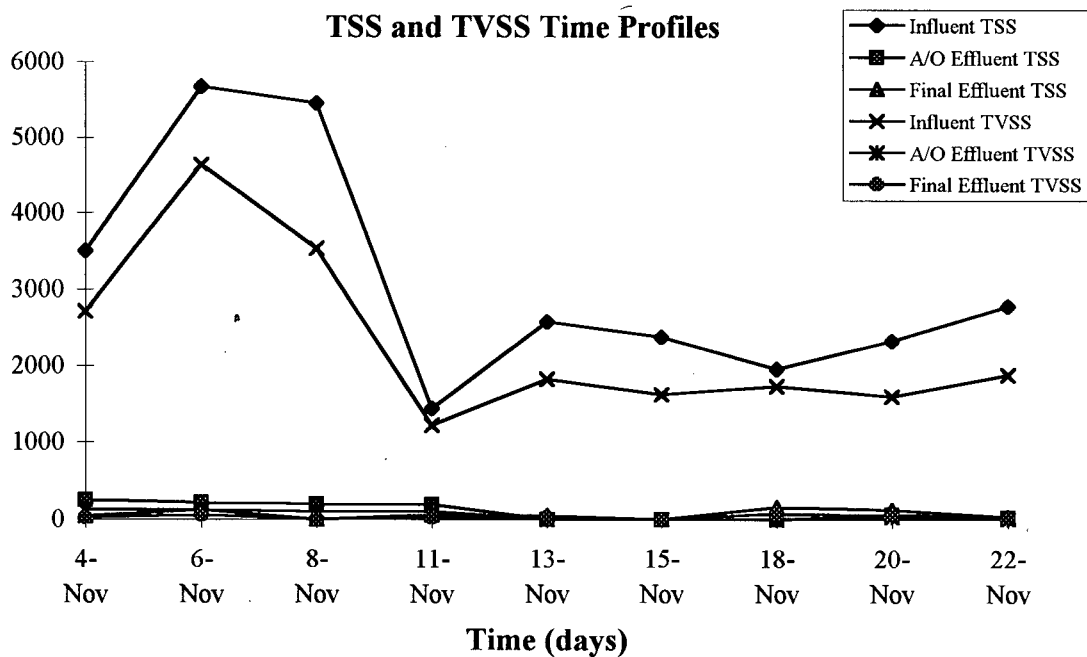

Figure 5. 28 TSS and TVSS profiles with time for the 2,000 mg/L run (attached growth system)

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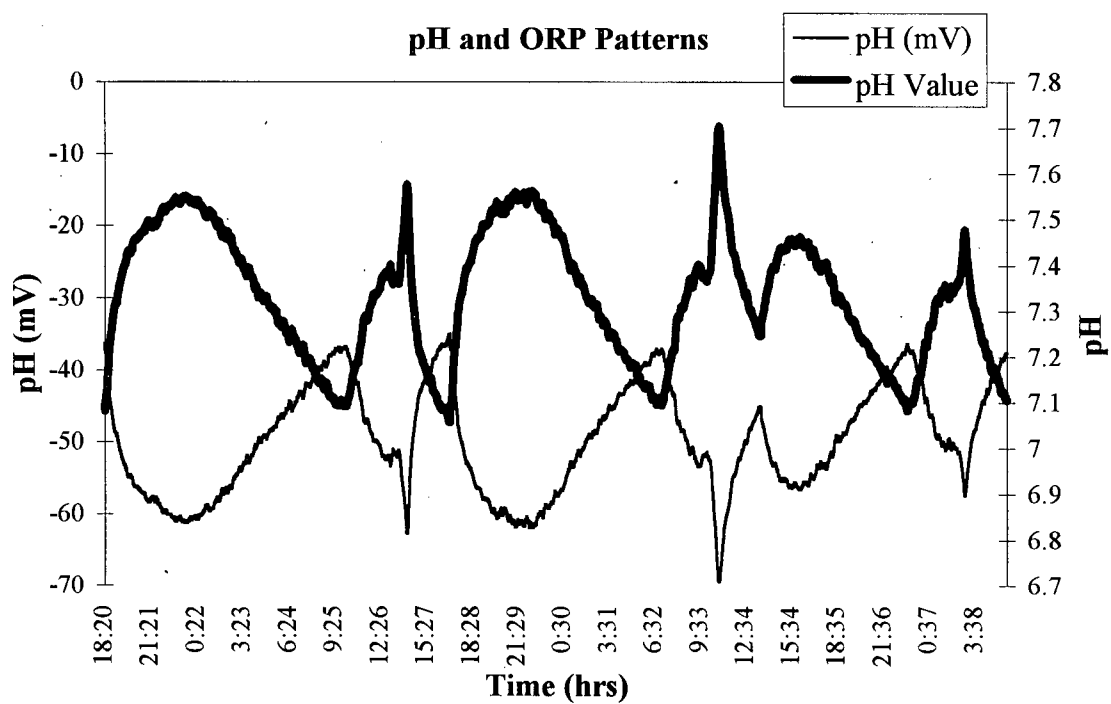
Figure 5. 29 Consistent pH and pH(mV) curves obtained from the 4,000 mg/L run (attached growth system)
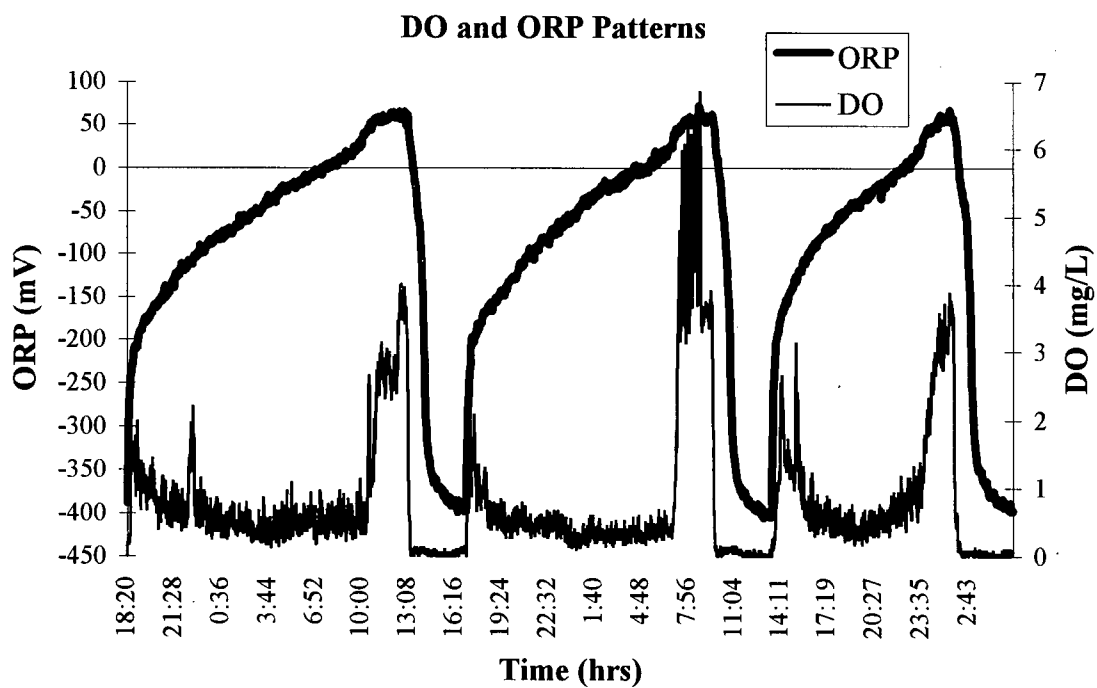
Figure 5. 30 Consistent DO and ORP curves obtained from the 4,000 mg/L run (attached growth system)

coincided with both the NBP and the RCMP respectively on the pH and ORP curves. At the beginning of the anoxic phase, the excess oxygen from the aerobic phase was consumed immediately. The DO concentration stayed constantly at 0 mg/L throughout the anoxic state of the cycle. The differences between the pH and ORP curves from the 2,000 mg/L run have already been discussed in the track analysis sections in this report.

The HRT of the system was not constant because the length of the aerobic phase depended on when the computer recognized the "trigger" point. For this run, the average HRTs were approximately 4.3 and 2.9 days for the A/O and anoxic reactor, respectively. The total HRT for the entire system, therefore, was approximately 7.3 days. The HRT for this run was much longer than the 2,000 mg/L run because of the higher concentration of the wastewater. Yet, the SRT and the aeration rate were kept constant throughout this research. The combined effect of the above created an imbalance between the aeration rate and the oxygen uptake rate. Therefore, it took the bacteria a longer period of time to remove all the desired nutrients. The advantage of a real-time system was most apparent for this reason. If the system was to run under a fixed time procedure, like for example 10 hours for the aerobic phase, there should be enough time to remove all the nutrients for the 2,000 mg/L run. However, the aerobic phase for this run averaged approximately 15 to 17 hours per cycle. Without the real-time control system there would be an incomplete removal of nutrients. This would lead to a low treatment efficiency of the system and the discharging of pollutants into the environment.

Table 5. 2 summarizes the performance of the real-time control process. The removal efficiency of any treatment system heavily depends on the characteristics of the wastewater. The average influent concentration of TOC and NH_4-N was 3842.23 mg/L

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Table 5. 2 Summary of system performance for the 4,000 mg/L run (attached growth system)

	Attached Growth									
	Average Influent (mg/L)	Min. - Max. (mg/L)	Std.	Average A/O (mg/L)	Min. - Max. (mg/L)	Std.	Average Effluent (mg/L)	Min. - Max. (mg/L)	Std.	% removal
TOC	3842.23	2821 - 5115.5	591.88	134.49	98.35 - 235.9	34.92	96.27	81.9 - 142.45	13.23	97.48
NH$_4$-N	795.38	722.2 - 926.4	50.33	1.87	0.89 - 6.42	1.27	7.28	3.46 - 12.52	3	99.09
NO$_x$-N	0.33	0.23 - 0.45	0.06	12.14	4.52 - 36.12	8.19	4.04	0.97 - 8.16	2.36	60.86
PO$_4^{-3}$	400.2	292.77 - 466.39	45.24	8.38	1.84 - 22.32	6.94	21.26	10.58 - 49.87	10.05	94.7
BOD$_5$	5475	3937.5 - 7462.5	1014.61	16.28	9.3 - 26.19	4.87	18.58	10.23 - 32.37	6.23	99.65
COD	10869.58	8221.95 - 14086.57	1721.18	235.1	202.95 - 302.74	34.53	205.98	195.95 - 330.96	10.67	98.07
TS	7095.56	5565 - 9625	1408.19	1520.28	1295 - 1715	130.91	1436.39	1352.5 - 1545	59.12	79.07
TVS	4914.44	3792.5 - 6937.5	1112.63	531.94	445 - 597.5	58.24	446.94	395 - 505	44.15	90.43
TSS	5103.89	3070 - 7280	1536.73	81.67	15 - 190	51.78	100	35 - 210	55.34	97.84
TVSS	4348.06	2890 - 6040	1008.91	81.67	15 - 190	51.78	97.22	35 - 210	53.74	97.65
DO	0	-	-	4.65	2.54 - 7.24	1.36	5.54	3.68 - 7.83	1.16	-

and 795.38 mg/L respectively. The relative TOC/NH_4-N ratio, therefore, was 4.83. This ratio was higher than the one in the 2,000 mg/L run. The higher C/N ratio suggested that the predominant reactions at the beginning of the aerobic phase would be carbon oxidation as much of the available oxygen would be consumed by heterotrophic bacteria first. This would create an insufficient supply of oxygen in the A/O reactor, which would in turn lead to simultaneous nitrification denitrification.

During this run, the concentrations of ammonia influent, A/O effluent, and final effluent were 795.38 mg/L, 1.78 mg/L, and 7.28 mg/L respectively. The removal efficiency for ammonia was approximately 99.09%. This again was the expected result as the control point was set beyond the NBP. The NH_4-N profile with time is illustrated in figure 5. 31. The removal of ammonia occurred in the A/O reactor only. The increase in the ammonia concentration in the anoxic reactor was not caused by ammonification. Because the bacteria mainly obtained their energy from endogenous respiration, sludge from the A/O reactor was often added to the anoxic reactor to enhance the denitrification rate. Sludge wastage from the A/O reactor usually occurred after the wastewater had been fed during the anoxic phase. Therefore, high strength wastewater was added to the anoxic reactor as part of the sludge. Because the anoxic reactor was designed for the removal of nitrate, the ammonia in the wastewater from the sludge remained in the reactor until it was discharged.

The removal efficiencies for TOC, BOD_5, and COD were also over 90% for this run. The TOC profile with time is illustrated in figure 5. 32. This result also didn't come as a surprise as the control point was set at the point between the NBP and the RCMP. The residual carbon in the A/O effluent was subsequently used by denitrifiers in the anoxic

NH4-N Concentrations
Attached Growth (4,000 mg/L)
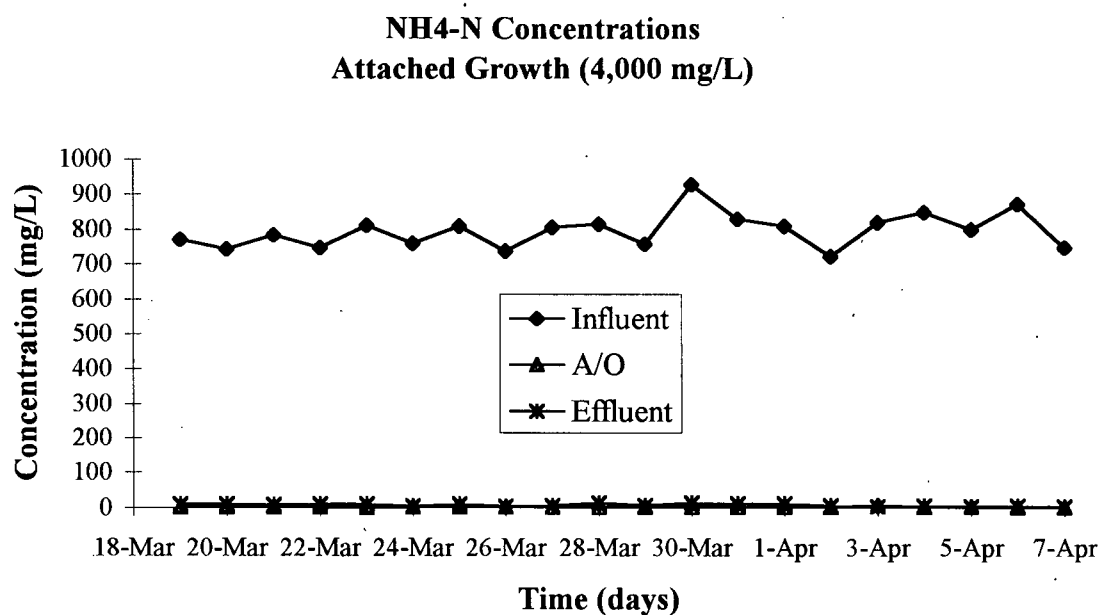
Figure 5. 31 NH₄-N profile with time for the 4,000 mg/L run (attached growth system)

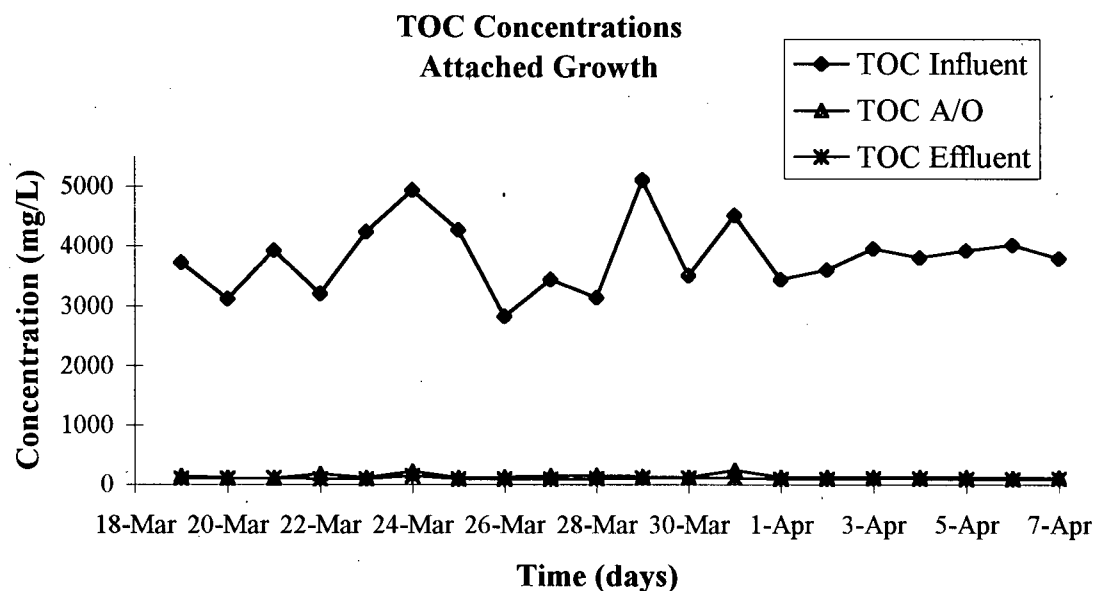**TOC Concentrations**
Attached Growth

Figure 5. 32 TOC profile with time for the 4,000 mg/L run (attached growth system)

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reactor for the removal of nitrate; hence a complete removal of organic carbon. The influent BOD_5/COD ratio was 0.5. This ratio was slightly higher than the 2,000 mg/L run. The higher ratio suggested that the influent contained a higher proportion of bio-degradable matter and a lower suspended solids level. The presence of a low proportion of suspended solids in the influent would enhance the rate of bio-degradation and reduced the oxygen demand by the bacteria (Manning et al., 1985, Burton, 1992). Therefore, the system would be more effective when the wastewater contained a high BOD_5/COD ratio. The TOC concentrations of the A/O and final effluent were 134.49 mg/L and 96.27 mg/L respectively. These results were also slightly higher than the 2,000 mg/L run. The higher concentrations of influent COD and TOC implied that there would be more biologically resistant organic matter in the wastewater. Also examining the effluent BOD_5/COD (0.090) and BOD_5/TOC (0.19) for this run suggested that the easily bio-degradable materials in the wastewater were utilized by the bacteria for cell synthesis while much of the long-chained carbon remained in the system.

The NO_x-N concentration in the A/O effluent should have been higher than in the 2,000 mg/L run because of the higher ammonia loading rate. However, the higher loading rate caused an imbalance between the nutrients levels and the aeration rate. This resulted in simultaneous nitrification/denitrification during the aerobic phase. The above behaviour reduced the amount of NO_x-N in the A/O effluent. The anoxic reactor was able to remove up to 60% of the nitrate from the A/O effluent using either the carbon source available in the A/O effluent or endogenous respiration by the bacteria. The A/O effluent contained an average NO_x-N concentration of 12.14 mg/L, while the final effluent carried an average NO_x-N concentration of 4.04 mg/L. Maintaining a final effluent nitrate concentration

under 5 mg/L displayed the effectiveness of this system as this concentration is lower than the drinking water discharge limit which is set at 8 mg/L. The time profile for NO_x-N is shown in figure 5. 33.

The track analysis of the anoxic reactor during one cycle is shown in figure 5. 34. It can be seen that soluble TOC from the A/O effluent was immediately used up by bacteria for denitrification as there was a sharp drop in the concentration of TOC at the beginning of the cycle. It should be noted that the nitrate concentration in the A/O effluent was minimal to begin with. Therefore, only a small amount of carbon was needed by the bacteria in the anoxic reactor to completely remove nitrates from the system. If there were a method to control the loading rate into the A/O reactor that would cause simultaneous nitrification and denitrification, the NO_x-N results for this run suggested that the anoxic reactor could be eliminated while still achieving a relatively low nitrate concentration before discharge. As mentioned before, sludge from the A/O reactor was added to the anoxic reactor to enhance nitrate removal by providing an alternative carbon source.

The system was able to achieve a remarkable treatment efficiency of approximately 95% removal of soluble phosphorus. The influent phosphorus concentration fluctuated slightly, but the A/O and final effluent constantly produced the same results. The A/O effluent phosphorus concentration was lower than 10 mg/L. The system was able to perform at a level better than the 2,000 mg/L run because of the readily available organic matter that was present in the wastewater. The PO_4^{-3} time profile is shown in figure 5. 35. The results of this run confirmed that of Iwema et al., 1985. Based on his research, when nitrate was absent but a carbon substrate was added, the rate of P release was rapid until

NOx -N Concentrations
Attached Growth (4,000 mg/L)

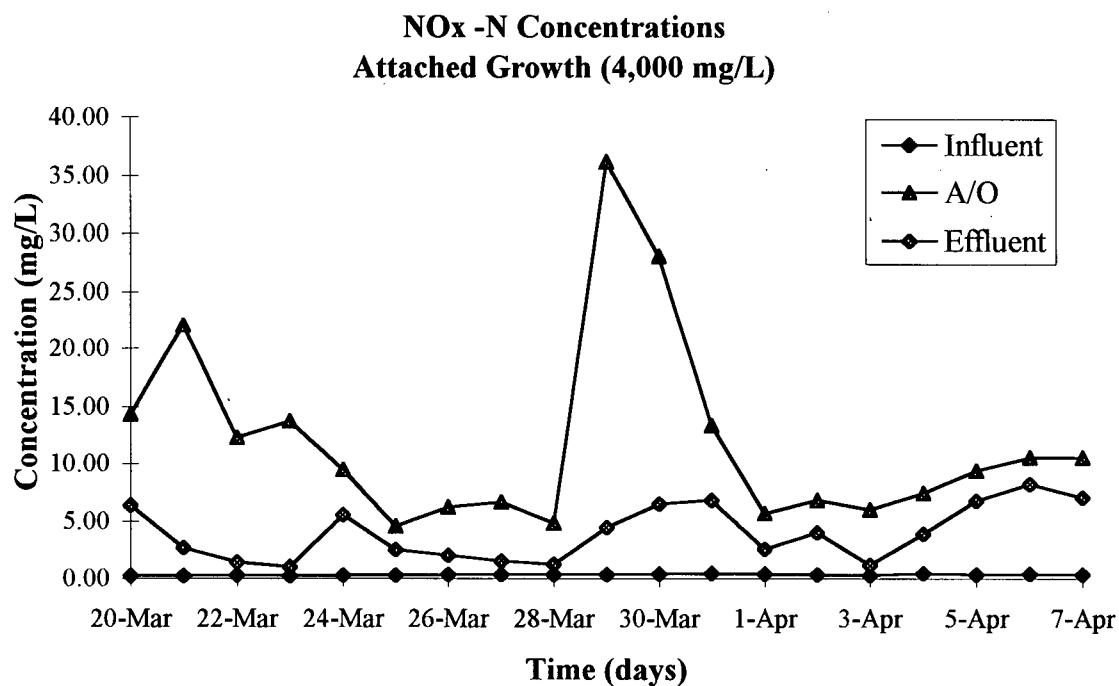**Figure 5. 33 NO$_x$-N profile with time for the 4,000 mg/L run (attached growth system)**
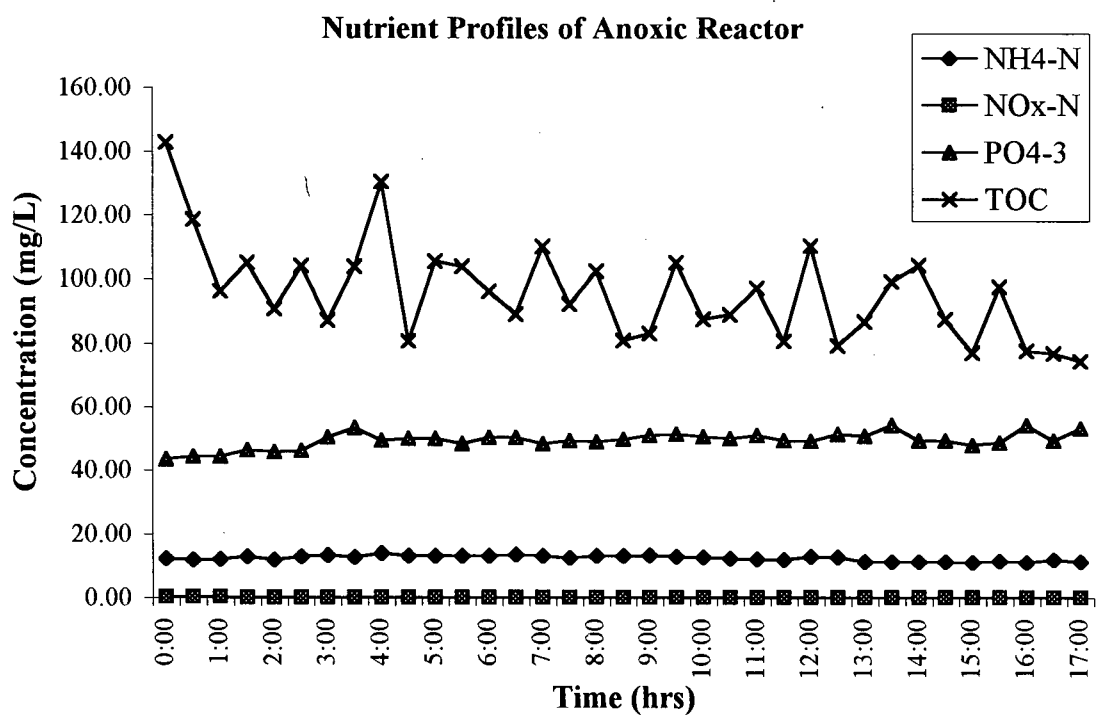
Nutrient Profiles of Anoxic Reactor

Figure 5. 34 Nutrient profile of once cycle for the anoxic reactor

PO4-3 Concentrations
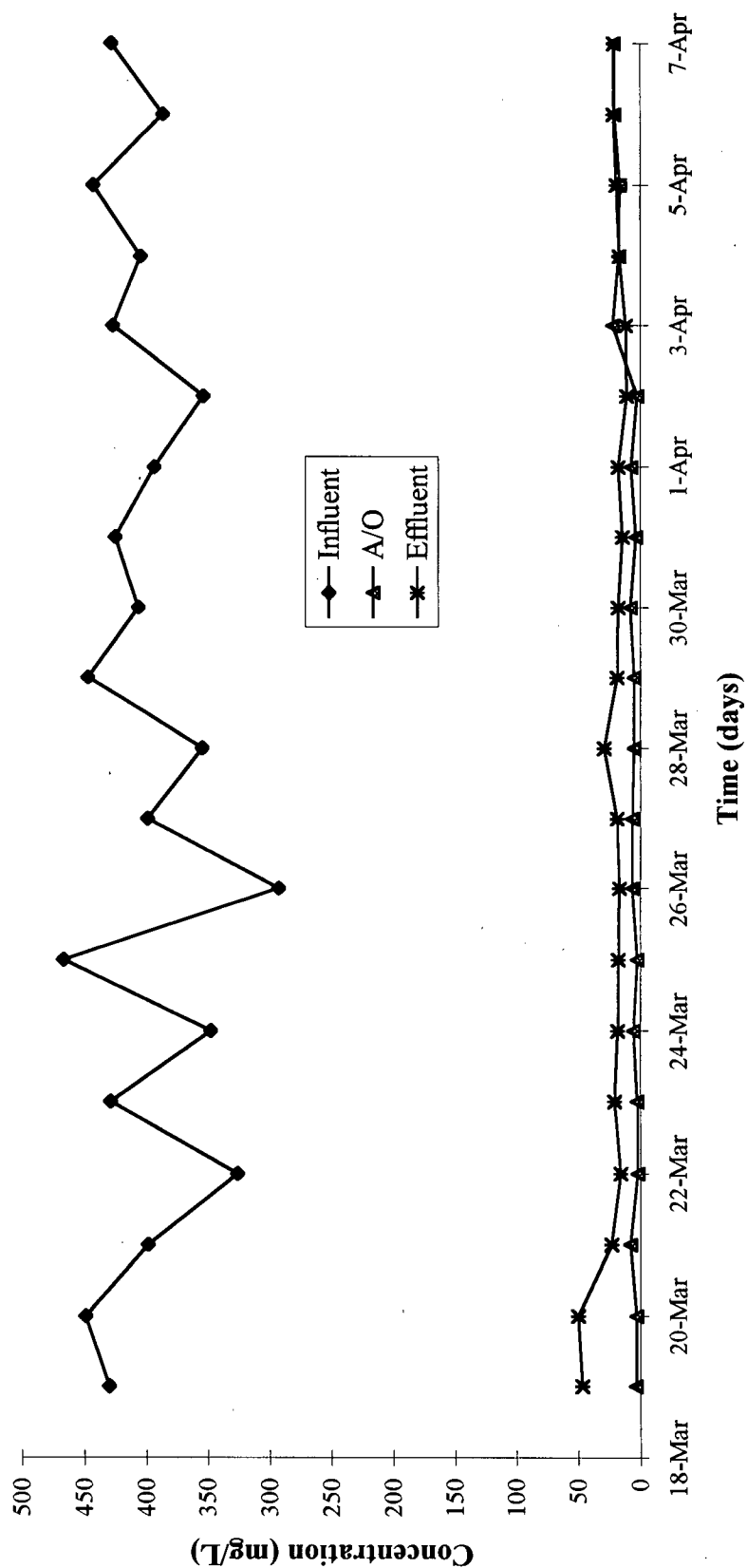**Attached Growth (4,000 mg/L)**

Figure 5. 35 PO₄⁻³ profile with time for the 4,000 mg/L run (attached growth system)

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the substrate was exhausted, after which time the release was reported to have stopped. This led to the conclusion that a higher substrate level, along with the absence of nitrate, would lead to greater P release, which was exactly the case in the 4,000 mg/L run. Also, anaerobic fermentation occurred after a complete removal of nitrate during the anoxic phase of the cycle. The combination of a higher concentration in total organic carbon from the influent and the quick development of anaerobic conditions promoted the production of volatile fatty acids (pH decrease) and the hydrolysis of organic matter. The bacteria were able to use the suddenly increased carbon content to enhance phosphorus release. This would subsequently lead to a higher phosphorus uptake rate by the bacteria in the aerobic phase. The phosphorus concentration in the final effluent was greater than the A/O effluent because sludge wastage from the A/O reactor occurred after the influent had been fed. Since the influent contained a high concentration of phosphorus, the effluent concentration from the anoxic reactor was strengthened when the sludge from the A/O reactor was added to the anoxic reactor, as the anoxic reactor was only targeted to remove nitrate from the system.

According to figure 5. 34, there was a slight increase in the concentration of ortho-phosphate in the anoxic reactor. As mentioned before, the sludge from the A/O reactor was completely separated from the anoxic reactor. However, sludge from the A/O reactor could have gotten into the anoxic reactor as a result of sludge addition to enhance the denitrification rate. In another words, Bio-P bacteria could have existed in the anoxic reactor. At the beginning of the cycle, the phosphorus concentration was approximately 43 mg/L. By the end of the cycle, the concentration reached approximately 53 mg/L. In the absence of a carbon substrate and nitrate, a slow endogenous release of phosphorus

could be observed (Iwena et al., 1985). Because of the low nitrate concentration entering the anoxic reactor and the rapid use of the carbon substrates from the A/O reactor for denitrification, the particular situation described above occurred. That led to the slight increase of phosphorus by the end of the cycle.

The solids levels in the influent, A/O effluent, and final effluent are represented in figures 5. 36 and 5. 37. The average TS, TVS, TSS, and TVSS removal efficiencies were 79.07%, 90.43%, 97.84% and 97.65% respectively. While high removal efficiencies were achieved for TVS, SS, and VSS, a relatively lower TS removal was observed. This might be the result of the influent wastewater containing a high level of non-organic solids in colloidal form. The higher removal efficiency of TS and TVS during this run comparing to the 2,000 mg/L run was probably due to the higher influent BOD_5/COD ratio. A higher BOD_5/COD ratio suggested that the influent contained a greater proportion of bio-degradable materials and a lower solids ratio. The influent, in another words, did not contain as high of proportion of non-organic solids in colloidal form than the 2,000 mg/L run.

5. 3. 1. 3 TOC = 500 mg/L

The DO patterns produced from this run were uniform and consistent despite the fluctuation in influent concentrations. This actually demonstrated the downfall of using DO as a control parameter. The DO parameter could not detect any deficiency in the system because of it's inability to monitor any anaerobic respiratory activity. From figures 5. 38 and 5. 39, it can be seen that both the pH and ORP curves were very consistent

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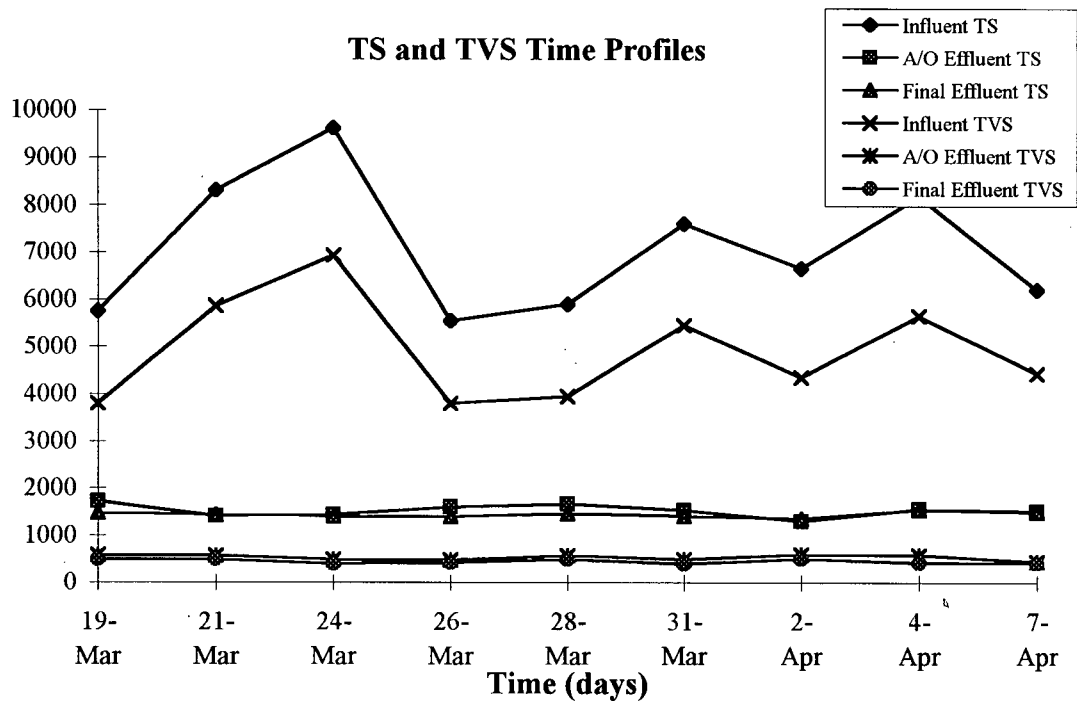**Figure 5. 36 TS and TVS profiles with time for the 4,000 mg/L run (attached growth system)**
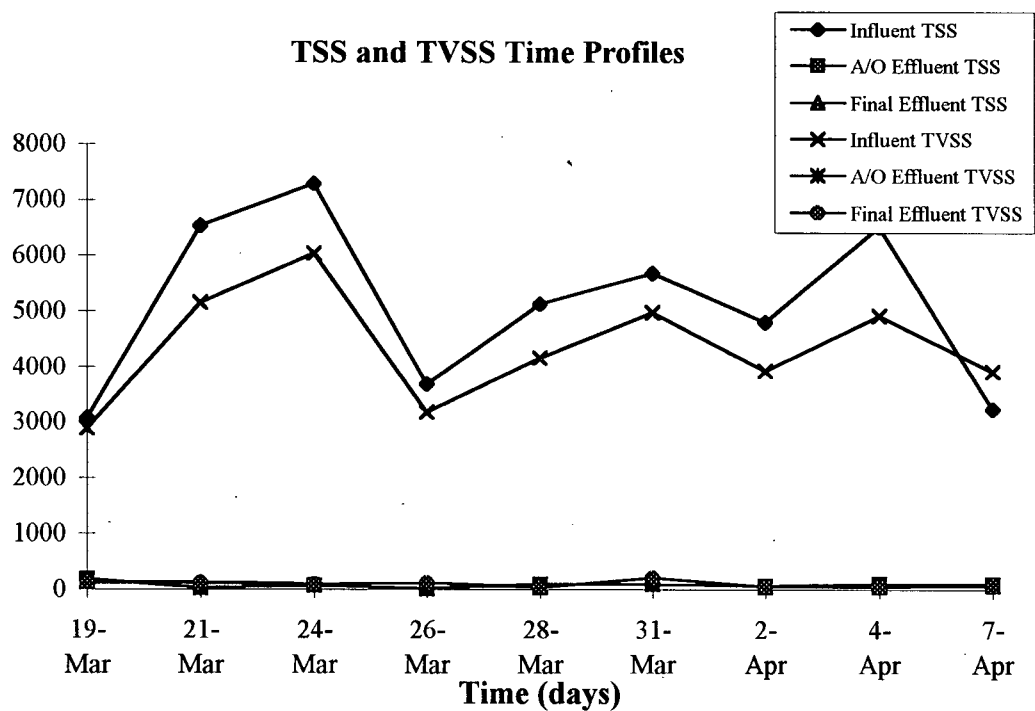

Figure 5. 37 TSS and TVSS profiles with time for the 4,000 mg/L run (attached growth system)
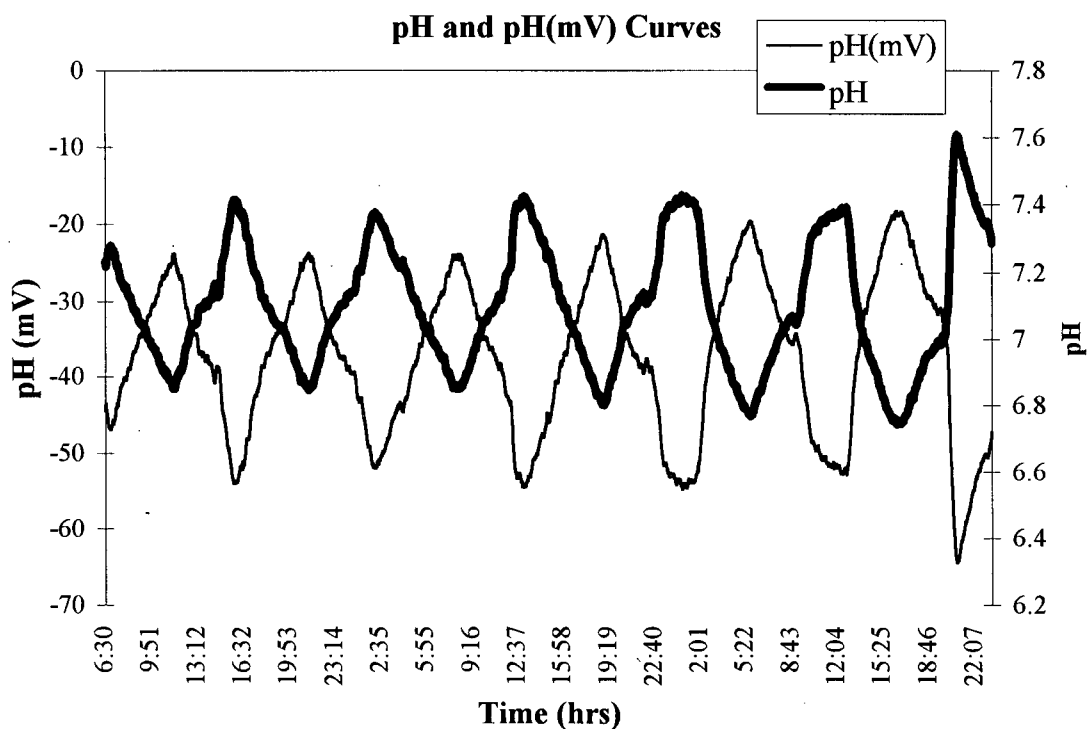
117

Figure 5. 38 Consistent pH and pH(mV) curves obtained from the 500 mg/L run (attached growth system)
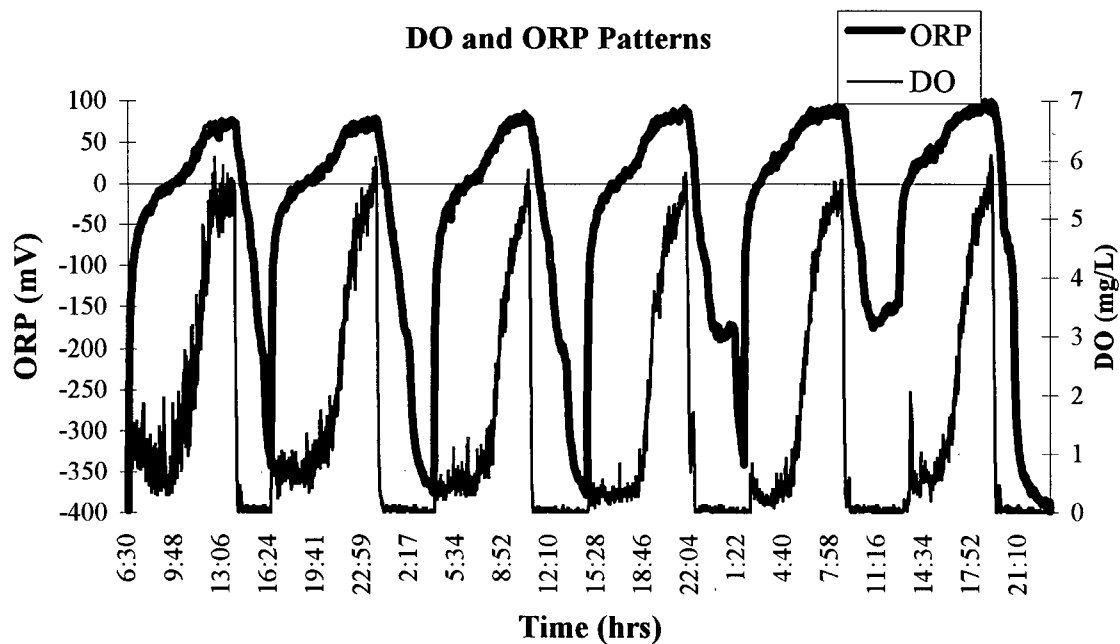

Figure 5. 39 Consistent DO and ORP curves obtained from the 500 mg/L run (attached growth system)

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during the aerobic phase. However, because this system was operated in a batch mode, any disruption in the anoxic phase could affect the overall system performance. In the 4th cycle in figure 5. 39, the nitrate knee was very apparent on the ORP pattern, and the curve did not reach the bottom plateau like the other cycles. The nitrate apex could barely be detected on the pH curve (see figure 5. 38), and no volatile fatty acids were produced during the anoxic cycle as the pH value did not decrease after the nitrate knee as it did in the other runs. This was the result of an inadequate supply of fresh organic carbon into the system. In order to carry out denitrification bacteria must either obtain a carbon substrate from the wastewater or use endogenous respiration. The conversion of ammonia during the aerobic phase left the system with a high concentration of nitrate. That is why the system was designed to be fed after the aerobic period. However, because this run was designed so that a low organic loading rate could be applied, not enough carbon was provided by the feed for rapid denitrification. The bacteria, therefore, must have relied on endogenous respiration for denitrification. However, the rate of denitrification using endogenous respiration was much slower than using an external carbon source. Therefore, the 4 hours that were provided for the anoxic phase in the A/O reactor was not enough for a complete removal of nitrate. That was the reason why the pH and ORP curves during the 5th cycle in figures 5. 38 and 5. 39 did not produce a nitrate apex or nitrate knee. The accumulation of nitrate could affect the overall performance of the system. A high nitrate concentration could be toxic to the bacteria or inhibit their performances. In some research, phosphorus release was found to be initiated when the nitrate concentration dropped below the detection limit (Malnou et al., 1984, Ra et al. 1997), while other research found that nitrate was an inhibitor to P-release (Hascoet et al.,

1985, and Iwema et al., 1985). In either scenario, it would not be a good idea to maintain a high concentration of nitrate in the A/O reactor. An external carbon source like methanol or a high loading rate should be employed in a system if the nitrate knee or apex cannot be observed on either the ORP or pH curve. It should be noted that real-time control should be extended to the anoxic phase and influent loading to prevent incomplete denitrification and phosphorus release.

The success of a real-time control system was based on the consistency of the patterns produced and the complete removal of pollutants in the wastewater despite the fluctuations in the influent concentrations. From figure 5. 40 it can be seen that the aerobic phase for the first three cycles were uniform and consistent. However, the next four cycles on the graph, the pH did not rise after the NBP, making it difficult to notice the NBP. Unfortunately, no track analysis for the attached growth system was done when this occurred. Therefore, it was difficult to explain what had transpired during those cycles. Fortunately, a track analysis was done when this occurred for the suspended growth system. Please refer to section 5. 1. 2. 3 for more details about this incidence. Nonetheless, this pattern was very similar to the normal pattern and the real-time control process was able to recognize the set point and carry out the appropriate actions.

Figure 5. 41 reflects a flexible HRT in the A/O reactor. The influent was diluted to a concentration of approximately 500 mg(TOC)/L. The HRT was much shorter than in the two runs discussed previously. It was rather obvious that the HRT for this run would be considerably lower as the loading rate for this run was considerably less than the others. The HRT in the A/O reactor was approximately 1.82 days, while the HRT for the anoxic reactor averaged 1.21 days. The total HRT for the entire system was therefore 2.03 days.
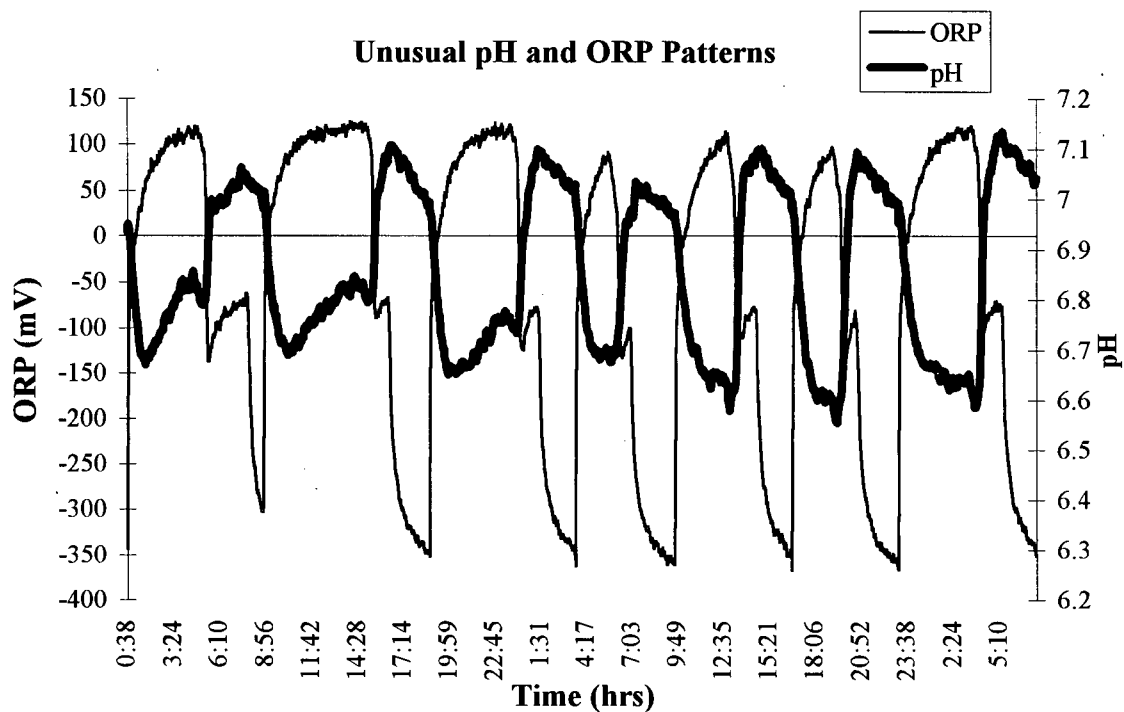
Figure 5. 40 Unusual ORP and pH curves produced during the 500 mg/L run (attached growth system)
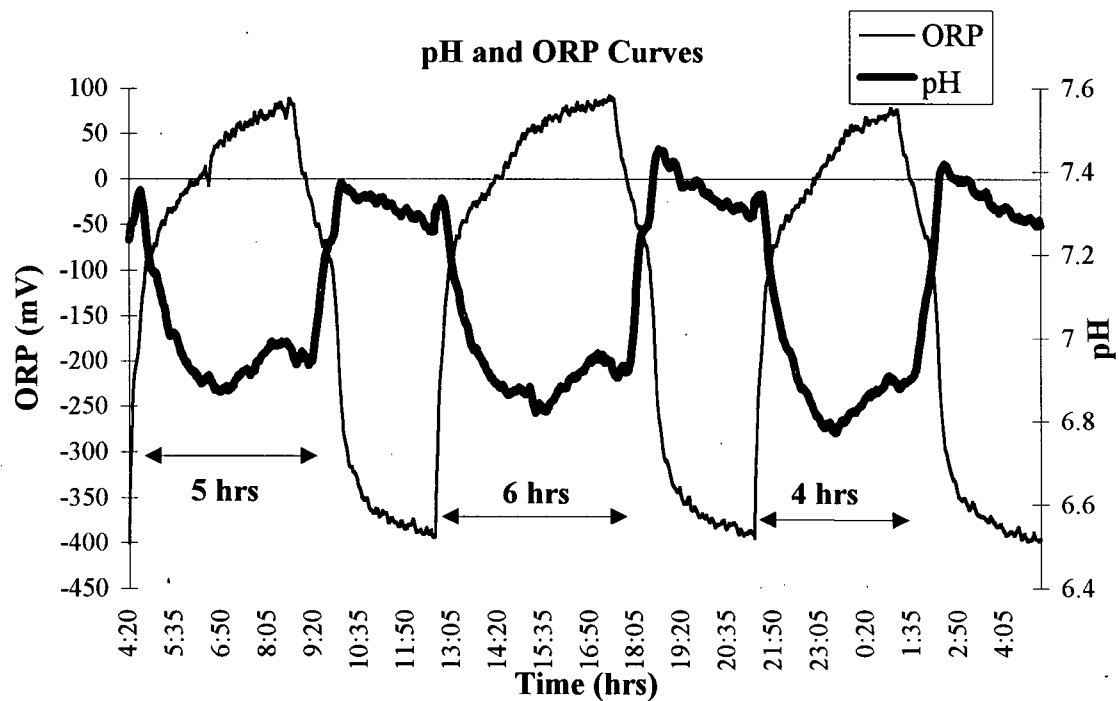

Figure 5. 41 Differences in cycle length

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Table 5. 3 summarizes the performance of the real-time control process for this run. The average influent concentrations of TOC and NH_4-N were 665.42 mg/L and 270.66 mg/L respectively. The relative C/N ratio, therefore, was 2.46. This run had the lowest C/N ratio of all the runs. The low C/N ratio indicated that the time to achieve complete carbon oxidation would be shorter. In fact, carbon oxidation could be finished before a complete removal of ammonia. Supposedly much of the available oxygen should have been consumed by heterotrophic bacteria when the cycle switched to the aerobic mode. However, because the aeration rate was so much greater than the OUR, nitrification and carbon oxidation occurred at the same time (see section 5. 1. 1. 2). Nonetheless, carbon oxidation still occurred at a greater rate and was finished within a very short period of time. Ammonia conversion, on the hand, continued after the completion of carbon oxidation. The low C/N ratio also indicated an insufficient supply of carbon in the fresh wastewater. During the aerobic phase, most of the ammonia was converted to nitrate. When the cycle switched back to the anoxic phase, the bacteria were supposed to use the fresh carbon from the wastewater for denitrification and phosphorus release. The lack of carbon would cause incomplete denitrification and hamper P-release by the bacteria.

During this run, the concentrations of the ammonia influent, A/O effluent, and final effluent were 270.66 mg/L, 1.29 mg/L, and 0.87 mg/L respectively. The removal of ammonia occurred in the A/O reactor only. There was no increase in the ammonia concentration in the anoxic reactor. The removal efficiency for ammonia was approximately 100%. The ammonia results found in this run matched the previous two runs. This was a remarkable result as the concentrations for the three different runs were

Table 5. 3 Summary of system performance for the 500 mg/L run (attached growth system)

	Attached Growth									
	Average Influent (mg/L)	Min. - Max. (mg/L)	Std.	Average A/O (mg/L)	Min. - Max. (mg/L)	Std.	Average Effluent (mg/L)	Min. - Max. (mg/L)	Std.	% removal
TOC	665.42	323.8 - 998.4	165.96	44.78	28.74 - 75.53	11.03	31.88	23.67 - 46.28	6.54	95
NH$_4$-N	270.66	219.79 - 333.79	24.26	1.29	0.32 - 3.32	0.65	0.87	0.31 - 5.98	1.26	99.38
NO$_x$-N	0.75	0.68 - 0.88	0.056	45.52	28.85 - 65.26	11.1	12.05	5.52 - 19.29	3.64	70.87
PO$_4^{-3}$	123.92	99.5 - 162.37	20.04	108.15	93.23 - 127.58	11.38	110.54	97.55 - 126.22	11.03	9.64
BOD$_5$	646.16	494 - 886.5	109.99	10.31	5.46 - 12.99	2.34	8.16	4.11 - 18.84	3.97	98.72
COD	2082.05	1661.9 - 2834.82	415.24	132.77	82.16 - 236.21	49.49	99.03	69.9 - 145.18	22.72	95.15
TS	1651.94	1135 - 2410	100.37	730.28	482.5 - 865	27.53	699.72	585 - 835	109.76	55.81
TVS	1119.72	685 - 1795	97.34	335.28	245 - 415	19.08	277.22	215 - 362.5	47.24	72.76
TSS	570.56	335 - 1020	40.53	6.11	0 - 25	6.24	18.37	0 - 50	32.02	97.06
TVSS	302.5	167.5 - 510	56.71	14.17	0 - 100	46.5	9.17	0 - 25	16.01	97.43
DO	0	-	-	4.28	3.16 - 6.27	0.82	5.05	3.31 - 6.7	0.8	-

so far apart. Without a real time control system it would be difficult to establish this kind of efficiency, and it would not be possible to reduce energy savings because aeration and mixing would be carried on in the aerobic phase even after the point of complete removal of all the nutrients every cycle. With these results, it can be concluded that real-time control ensures a complete removal of ammonia from the system. The NH_4-N profile with time is illustrated in figure 5. 42.

High organic removal efficiencies of 95%, 99.38%, and 95.15% were achieved for TOC, BOD_5, and COD respectively. The TOC profile with time is illustrated in figure 5. 43. The result of this run showed no significant difference compared to the other runs, in which all achieved an organic removal efficiency of over 90%. Again, the advantage of real-time control was perfectly illustrated. The fluctuations of influent concentrations did not affect the system performance in organic removal. The influent BOD_5/COD ratio was approximately 0.31. This ratio was the lowest of the three runs. As mentioned before, a high BOD_5/COD ratio represents a high proportion of bio-degradable materials in the wastewater. The low ratio suggested a high proportion of solids in the wastewater which could retard the rate of degradation by the bacteria. The low ratio also suggested that there wasn't enough easily bio-degradable matters in the wastewater for the bacteria to consume. This led to a poor system performance.

The NO_x-N production in the A/O reactor was higher than in the 4,000 mg/L but was lower than in the 2,000 mg/L run. One would imagine that the 4,000 mg/L run should have produced more nitrate due to it's higher loading of ammonia. But as mentioned before, simultaneous nitrification/denitrification occurred in the aerobic phase during the 4,000 mg/L run. Therefore, the production of NO_x-N reduced as nitrate was

NH4-N Concentrations
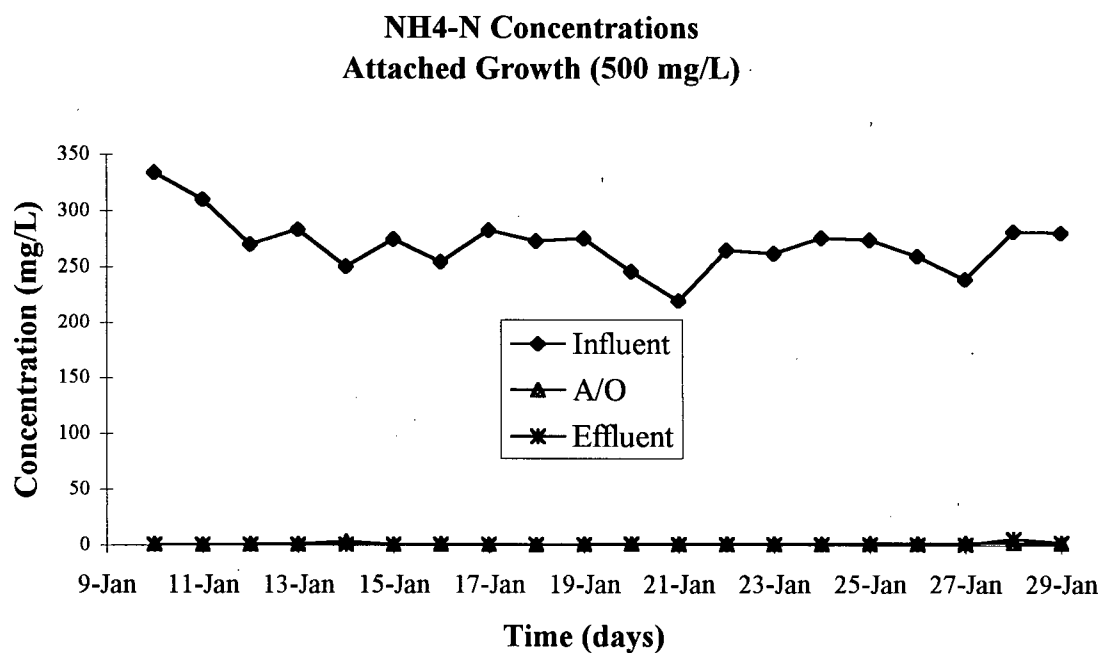## Attached Growth (500 mg/L)

Figure 5. 42 NH₄-N profile with time for the 500 mg/L run (attached growth system)
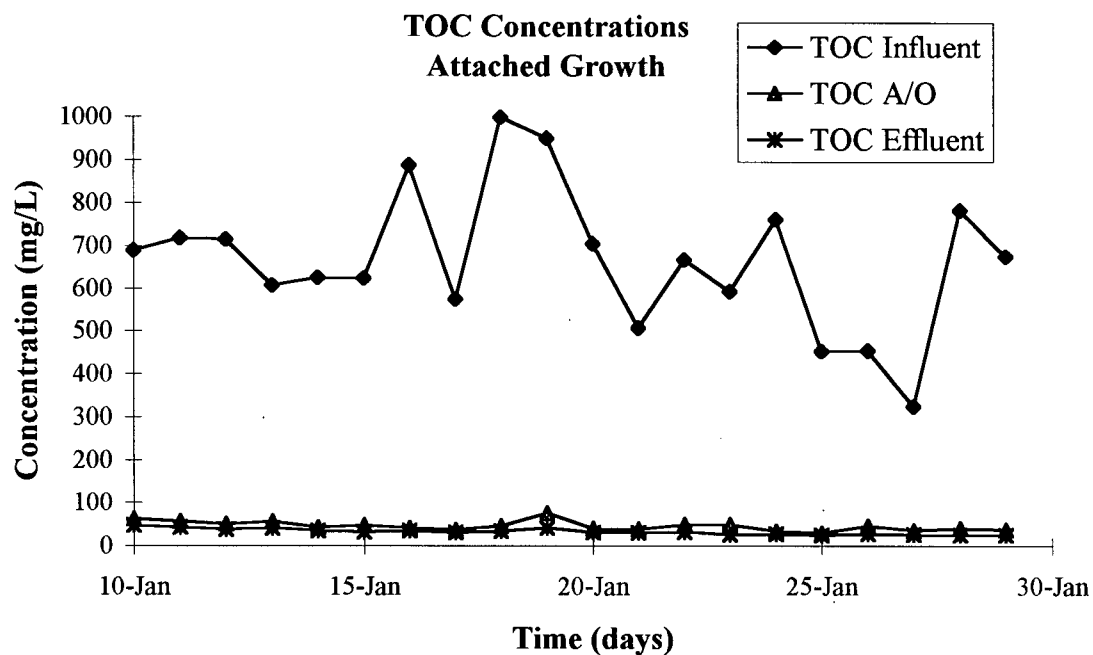

Figure 5. 43 TOC profile with time for the 500 mg/L run (attached growth system)

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converted to N_2 immediately once it had been transformed from ammonia during the aerobic phase. As for the 2,000 mg/L run, the combined effect of a higher ammonia loading rate and the balance of aeration rate and OUR which caused the disappearance of simultaneous nitrification/denitrification resulted in a higher production of nitrate than the 500 mg/L run. Figure 5. 44 illustrates the time-profile of NO_x-N.

The anoxic reactor was able to remove up to 70% of the nitrate from the A/O effluent using either the carbon source available in the A/O effluent or endogenous respiration by the bacteria. This is illustrated by the track analysis done on the anoxic reactor through one cycle of this run (figure 5. 45). As mentioned before, the complete removal of carbon occurred before the RCMP for this run. Therefore, the A/O effluent contained virtually no soluble carbon that could be taken up by bacteria to carry out denitrification. The bacteria in the anoxic reactor relied solely on endogenous respiration for the removal of nitrate. Endogenous respiration, however, occurred at a very slow rate and stored carbon in bacterial cells would be depleted at some point due to continual use. Because of the slow rate of endogenous respiration and the relatively quick cycle time in the A/O reactor due to the low loading rate, not enough time was provided for the bacteria to remove all the nitrate in the anoxic reactor. However, the constant addition of sludge to the anoxic reactor to replenish the carbon source in the system (enhance denitrification rate) was enough to maintain a relatively low concentration of NO_x-N in the final effluent. It should be noted that sludge addition occurred most frequently in this run as denitrification was only based on endogenous respiration.

The treatment efficiency for ortho-phosphate was only 9% for this run. This removal efficiency was the lowest of the three runs. The low treatment efficiency was due

NOx -N Concentrations
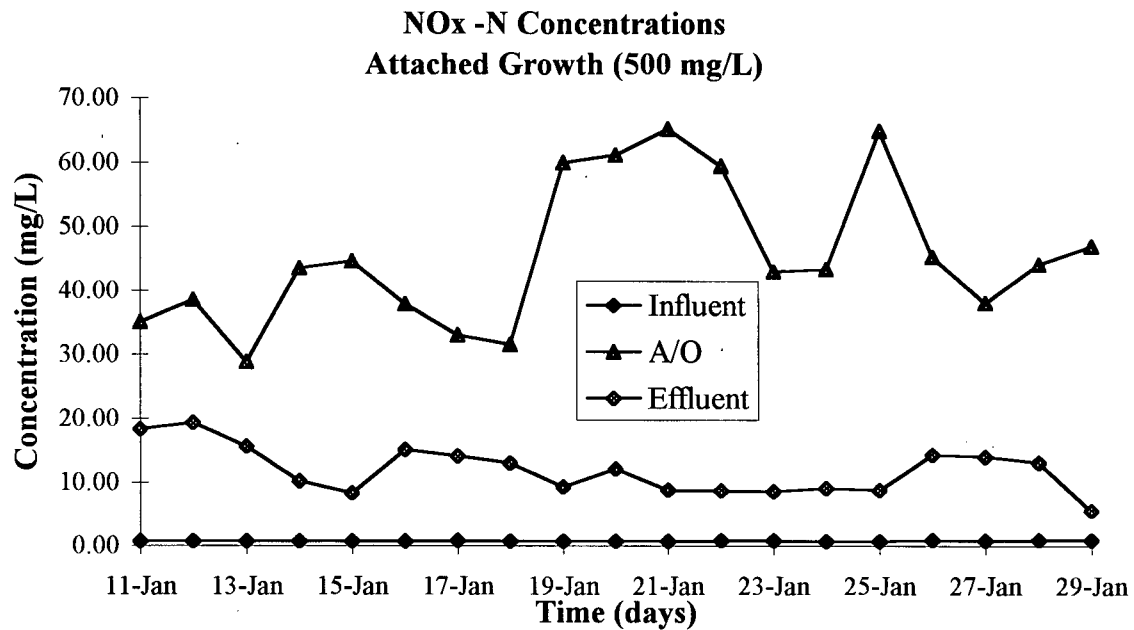**Attached Growth (500 mg/L)**

Figure 5. 44 NO$_x$-N profile with time for the 500 mg/L run (attached growth system)
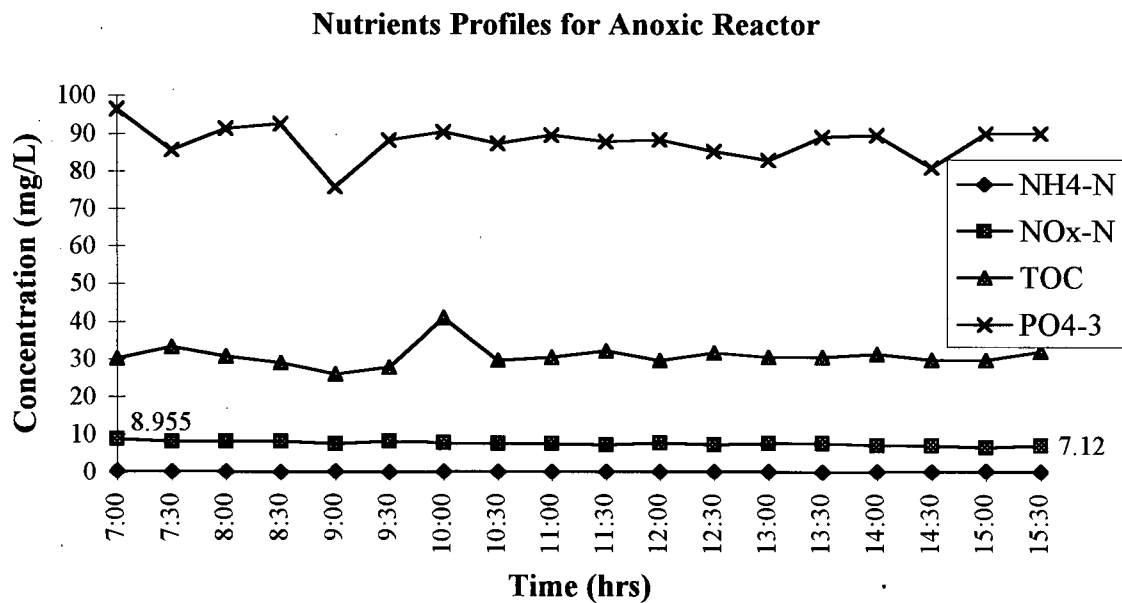
Nutrients Profiles for Anoxic Reactor

Figure 5. 45 Nutrients profiles of one cycle for the anoxic reactor

to the same reasons as the 2,000 mg/L run. During the anaerobic stage, the hydrolysis of the poly-P compounds in the cells provided the energy requirements for the Bio-P bacteria. This resulted in a release of phosphorus into the solution (Comeau et al., 1986). The phosphorus release which occurs in the presence of VFAs is called the primary release, while the release in the absence of VFA is known as the secondary release (Barnard, 1994). The primary release of phosphorus actually required the bacteria to obtain energy to transfer VFA into their cells for storage in the form of PHB. Secondary release of phosphorus is not related to the storage of organic compounds and phosphorus. Therefore, Bio-P bacteria will not uptake the phosphorus in the subsequent aerobic phase. Insufficient carbon supply in the wastewater and a lack of activity by the fermentation bacteria during the anaerobic phase could lead to secondary release. The profile for PO_4^{-3} with time is shown in figure 5. 46.

Much of the easily bio-degradable materials was reserved for denitrification. Once denitrification was finished, the rate of phosphorus release was supposed to increase. However, because of the low organic loading, much of the carbon was used up after denitrification. Some bacteria were able to use the remaining carbon for P-release, but most of the bacteria were forced to carry out secondary phosphorus release. Subsequently, there was very little phosphorus uptake during the aerobic phase, as much of the phosphorus release was carried out through the secondary release mechanism. It should be noted that the rate of secondary phosphorus release was relatively slow compared to primary release. Therefore there wasn't a major increase in soluble phosphorus concentration in the A/O effluent. Nonetheless, the system's poor

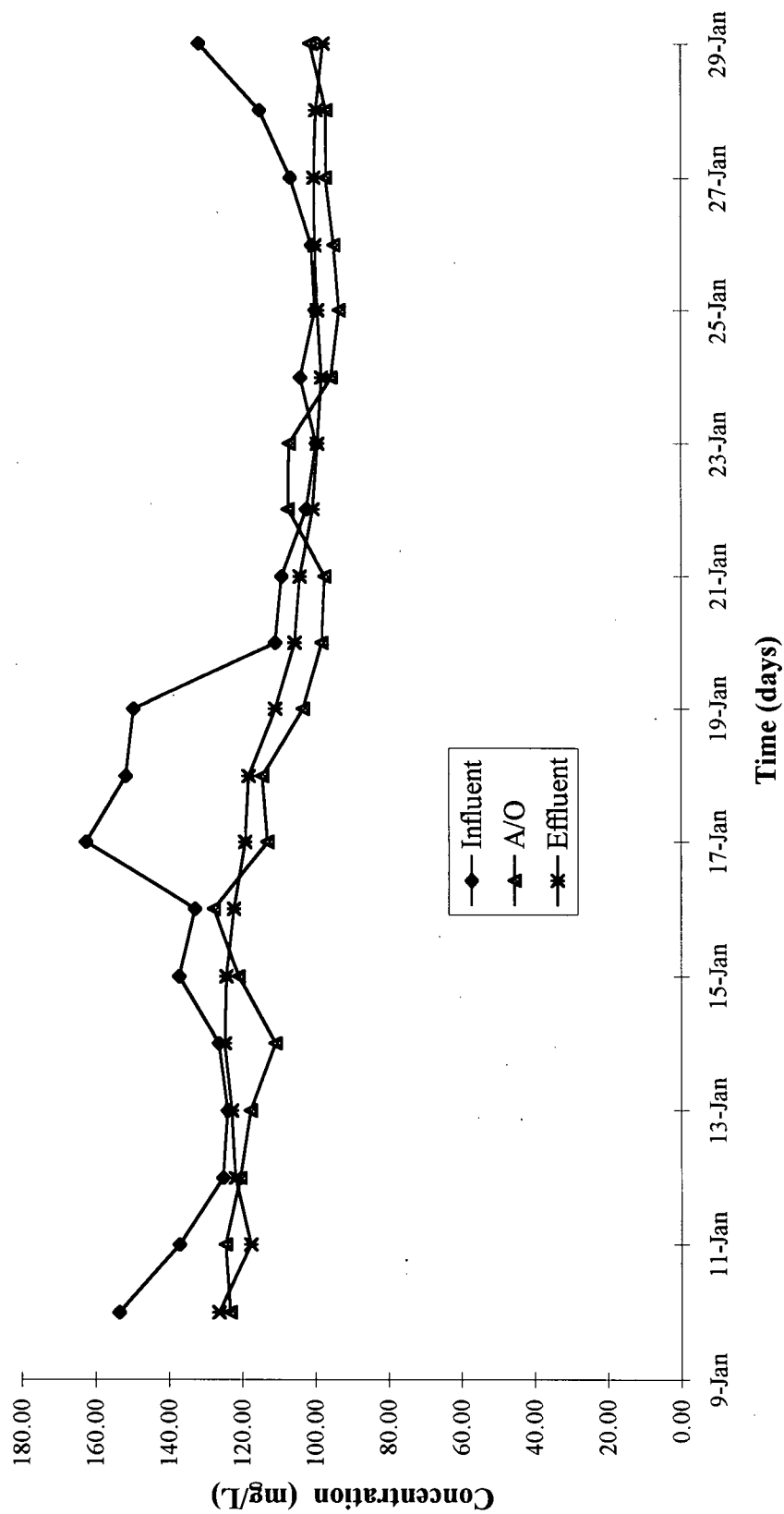**PO4-3 Concentrations**
Attached Growth (500 mg/L)

Legend: Influent, A/O, Effluent

Concentration (mg/L)

Time (days)

Figure 5. 46 PO$_4$$^{-3}$ profile with time for the 500 mg/L run (attached growth system)

performance in phosphorus removal was mainly a consequence of an insufficient supply of carbon in the wastewater.

It is also illustrated in figure 5. 45 that there was no phosphorus release in the anoxic reactor. Sludge was constantly added to the anoxic reactor to maintain a low concentration of nitrate. This action led to an increase of Bio-P bacteria in the anoxic reactor. However, it was found that when nitrate was present, but no carbon substrate added, the phosphorus concentration stayed at a constant level until all nitrate was consumed (Iwema et al., 1985). This confirmed the findings of this research. There wasn't enough carbon in the A/O effluent for complete denitrification in the anoxic reactor. Since there was no substrate to promote phosphorus release and nitrate was an inhibitor to P release, the phosphorus level stayed constant in the anoxic reactor.

The solids levels in the influent, A/O effluent, and final effluent are represented in figures 5. 47 and 5. 48. The average TS, TVS, TSS, and TVSS removal efficiencies were 55.81%, 72.76%, 97.06% and 97.43% respectively. This BOD_5/COD ratio of 0.31 was the lowest of the three runs, meaning that the influent was made up of a high proportion of colloidal solids that could not be removed by a biological process. The highest removal efficiency of total solids was found when the BOD_5/COD ratio was high, which was during the 4,000 mg/L run, followed by the 2,000 mg/L run, then the 500 mg/L run.
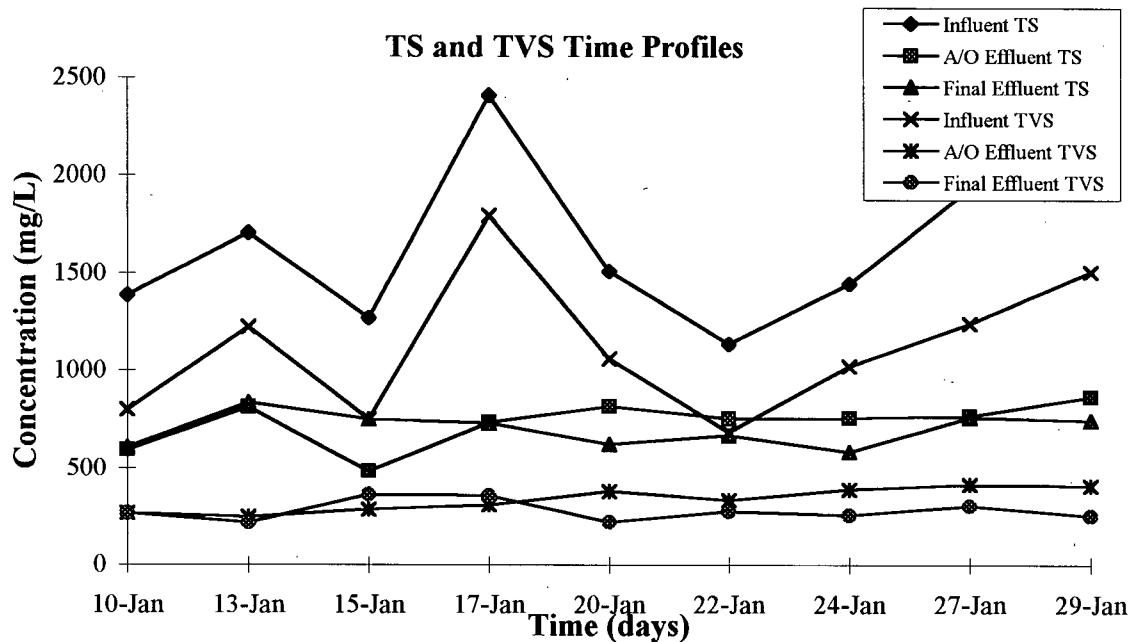
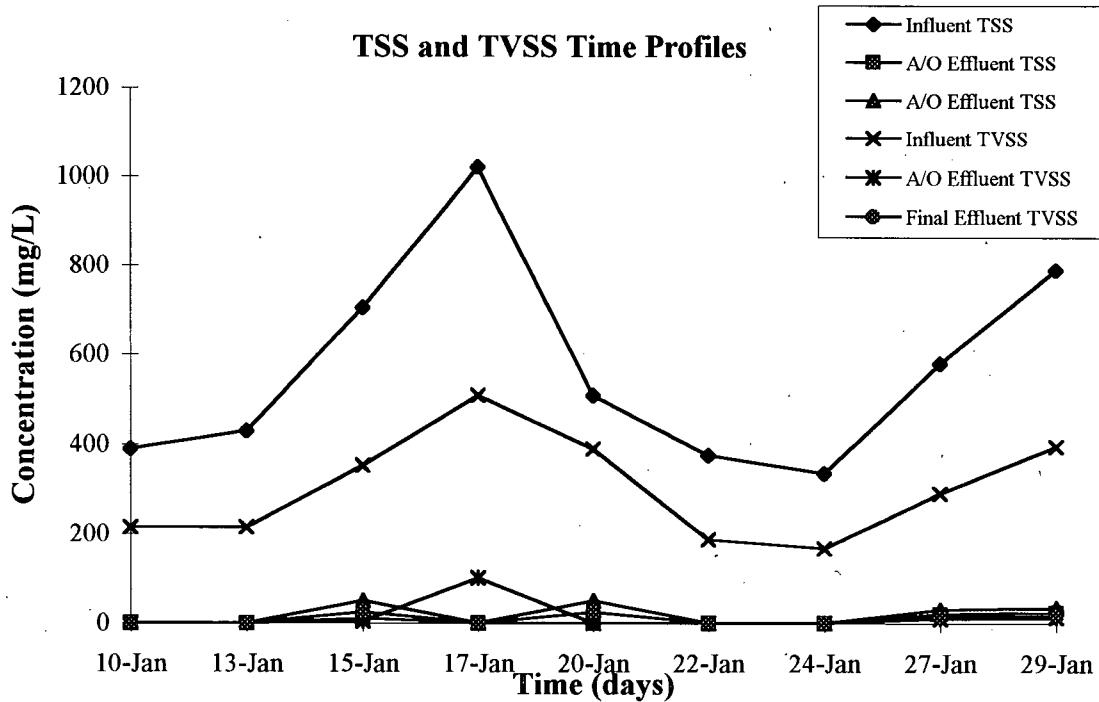**Figure 5. 47 TS and TVS profiles with time for the 500 mg/L run (attached growth system)**

Figure 5. 48 TSS and TVSS profiles with time for the 500 mg/L run (attached growth system)

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5. 3. 2 Suspended Growth

5. 3. 2. 1 TOC = 2,000 mg/L

The pH, ORP, and DO patterns for this run are illustrated in figures 5. 49 and 5. 50. There were no differences on any of these curves when compared to the graphs generated from the AGS. The patterns produced were very uniform and consistent throughout this run. The advantage of the real-time control system was very apparent as the lengths of the cycle were different from each other. The conclusion is that the real-time control process was not affected by the Ringlace and could be operated under either suspended or attached growth conditions. The graphs generated from the SGS seemed to be more stable than the ones from the AGS even though they were operated under the same conditions. All the parameters including pH, ORP, and DO were originally a direct measurement of the electro-activities in the mixed liquor. A calibration of the voltage reading to its actual pH, ORP, or DO value was done before the start of the experiment. This calibration was programmed into the software for conversion. These voltage readings could be affected by many factors such as the solids concentration, electrical noise from the surrounding environment, or the dissolved oxygen concentration. The probes were inserted in between the Ringlace for the AGS. Sludge could easily attach onto the probes, which could make the values inaccurate. Also the frame on which the Ringlace was attached could have emitted electrical interference. Rust on any part of the frame could have interfere with the readings. Nonetheless, any electrical noise in the AGS did not significantly interfere with the operation of the real-time control system.

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pH and pH(mV) Curves

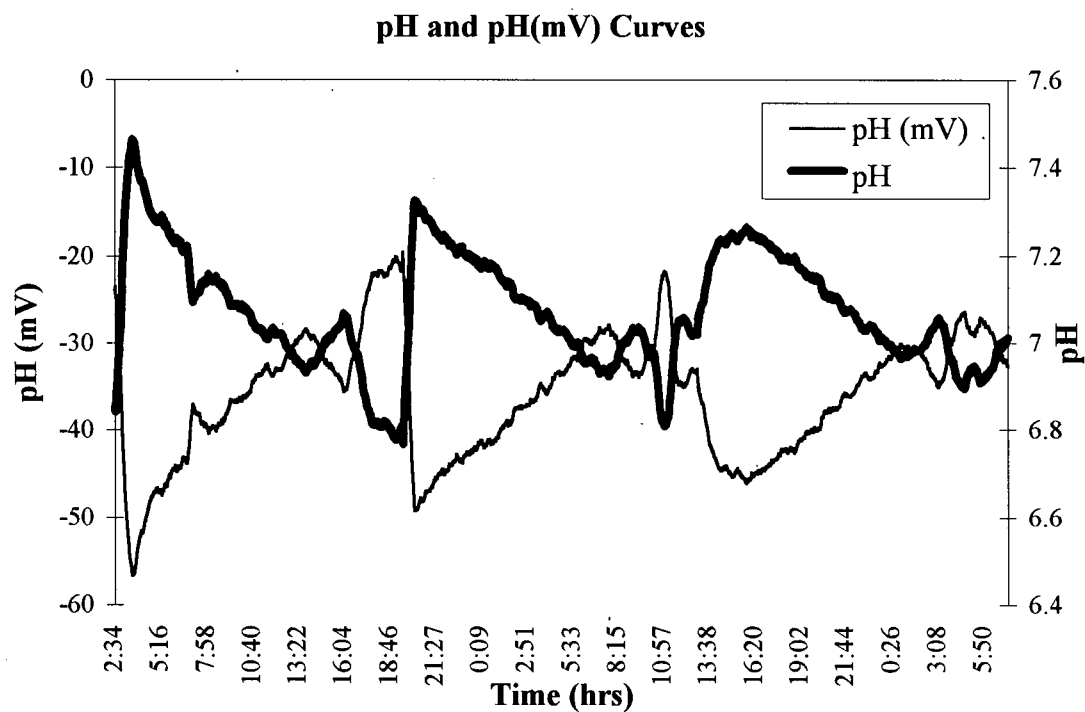
Figure 5. 49 Consistent pH and pH(mV) curves obtained from the 2,000 mg/L run (suspended growth system)
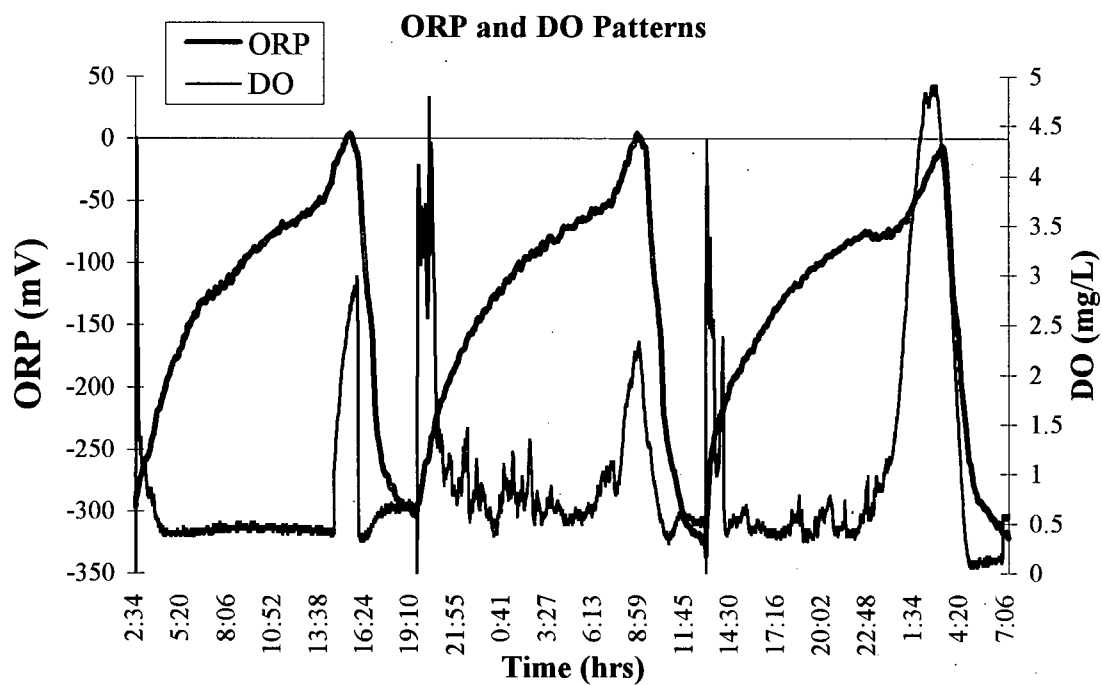
ORP and DO Patterns

Figure 5. 50 Consistent DO and ORP curves obtained from the 2,000 mg/L run (suspended growth system)

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The only major difference that the real-time system generated between the AGS and SGS was the HRT. The average HRTs were approximately 3.18 days and 1.86 days in the A/O and anoxic reactor, respectively. The total HRT therefore was approximately 5 days. Referring back to section 5. 3. 1. 1, the HRT of the AGS for the 2,000 mg/L run was 4.27 days in total. The longer HRT was attributed to the amount of bacteria in the system. The HRT of the system heavily depends on the SRT. A larger amount of bacteria retained in the system could shorten the hydraulic retention time as there are more organisms to carry out the necessary reactions. The SRT of 10 days for the SGS was based on the entire suspended bacterial population. Sludge wasting was done for the AGS based on the suspended bacteria population only because it was very hard to control the amount of bacteria attached on the Ringlace. Based on this, the AGS should have a shorter retention time than the SGS, and is proven by this result.

Table 5. 4 summarizes the performance of the suspended growth system for the 2,000 mg/L run. The average influent concentrations of TOC and NH_4-N were 1910.75 mg/L and 617.7 mg/L, respectively. The relative TOC/NH_4-N ratio, therefore, was 3.09. This ratio was slightly lower than the one obtained from the 2,000 mg/L run of the attached growth system. This was, however, higher than the ratio from the 500 mg/L run of the AGS. It was suggested that the lower C/N ratio meant that nitrification might finished before the complete removal of carbon from the system. Referring back to figure 5. 3. 1. 1, it could be seen that the TOC concentration was approaching a plateau by the end of the cycle. It could therefore be determined that the RCMP could still represent a complete removal of carbon if the wastewater contains a TOC/NH_4-N ratio of over 3.

Table 5. 4 Summary of system performance for the 2,000 mg/L run (suspended growth system)

	Suspended Growth									
	Average Influent (mg/L)	Min. - Max. (mg/L)	Std.	Average A/O (mg/L)	Min. - Max. (mg/L)	Std.	Average Effluent (mg/L)	Min. - Max. (mg/L)	Std.	% removal
TOC	1910.75	1322.5 - 3173	558.07	87.01	74.57 - 106.08	9.8	72	59.69 - 85.13	6.2	95.94
NH$_4$-N	617.7	497.63 - 782.79	58.82	0.54	0 - 2.88	0.89	3.5	0 - 15.80	4.6	99.4
NO$_x$-N	0.48	0.31 - 1.08	0.22	60.12	33.75 - 96.16	20.83	23.58	1.54 - 58.60	19.15	64.71
PO$_4^{-3}$	218.67	184.13 - 269.91	24.88	113.56	80.86 - 154.61	19.89	118.79	91.2 - 133.97	11.73	44.78
BOD$_5$	1958.28	1699.5 - 2833.13	322.74	10.46	5.58 - 16.68	3.42	6.78	4.62 - 10.29	1.4	99.68
COD	4973.81	2992.38 - 6640.34	1095.66	185.45	159.19 - 215.21	18.34	177.01	155.69 - 229.21	21.07	95.31
TS	4044.44	2667.5 - 5170	925.47	1328.89	1147.5 - 1482.5	103.79	1238.89	1127.5 1347.5	74.37	67.93
TVS	3110.83	1822.5 - 3945	733.05	476.11	342.5 - 595	78.22	416.39	325 - 565	70.73	85.91
TSS	3083.89	1907.5 - 4040	772.1	58.33	0 - 180	57.34	22.78	0 - 50	19.22	99.18
TVSS	2376.39	1297.5 - 3535	766.98	29.17	0 - 90	28.67	11.39	0 - 25	9.61	99.43
DO	0	-	-	5.78	4.32 - 6.83	0.81	6.11	4.66 - 6.99	0.62	-

The suspended growth system under the real-time control was also able to achieve a treatment efficiency for ammonia removal of approximately 100%. The NH_4-N profile with time is illustrated in figure 5. 51. The A/O effluent had an average concentration of 0 mg/L. There was a slight increase in the anoxic reactor of ammonia concentration beginning on November 6th which decreased back to approximately 0 mg/L on November 13th. Sludge from the A/O reactor was added to the anoxic reactor for the enhancement of nitrate removal. There were two ways of wasting sludge from the A/O reactor. Sludge could be wasted before or after the influent had been fed into the A/O reactor. The difference between the two methods is if the sludge was wasted after the influent had been fed, a portion of the influent would also be transferred into the anoxic reactor. This would in turn increase the ammonia concentration. There was no increase in ammonia in the anoxic reactor for the 2,000 mg/L run in the AGS because the sludge was wasted from the A/O reactor before the influent was fed.

A high organic removal efficiency of over 95% was achieved in the SGS. This result was very similar to the AGS. The TOC profile with time is illustrated in figure 5. 52. The A/O and final effluent had average TOC concentrations of 87.01 mg/L and 72 mg/L respectively. The AGS had a higher A/O effluent concentration but a lower final effluent concentration for the same run. This indicated that more soluble carbon remained in the AGS's A/O effluent for denitrification in the anoxic reactor. As mentioned before, there was a lower TOC/NH_4-N ratio for the suspended growth system for this run specifically. The lower ratio created a quicker rate of carbon removal. More soluble organics were utilized from the system even though the effluent was discharged at the same time for both systems. This created a smaller gap between the A/O and the final

NH4-N Concentrations

Suspended Growth: TOC = 2,000 mg/L
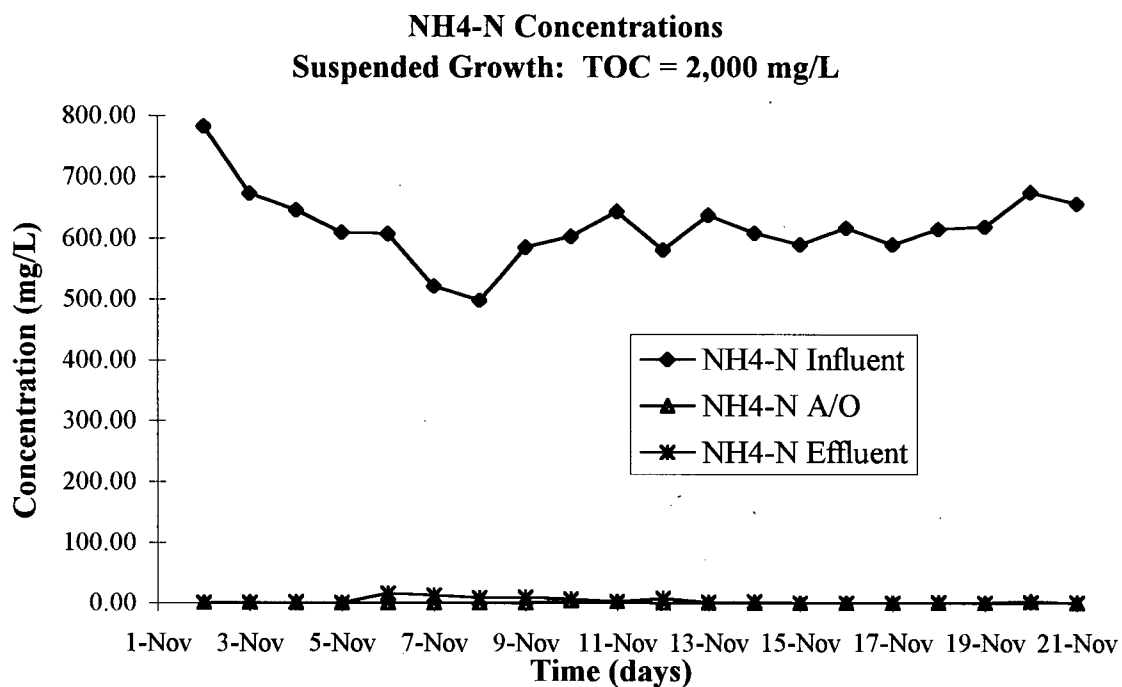
Figure 5. 51 NH₄-N profile with time for the 2,000 mg/L run (suspended growth system)

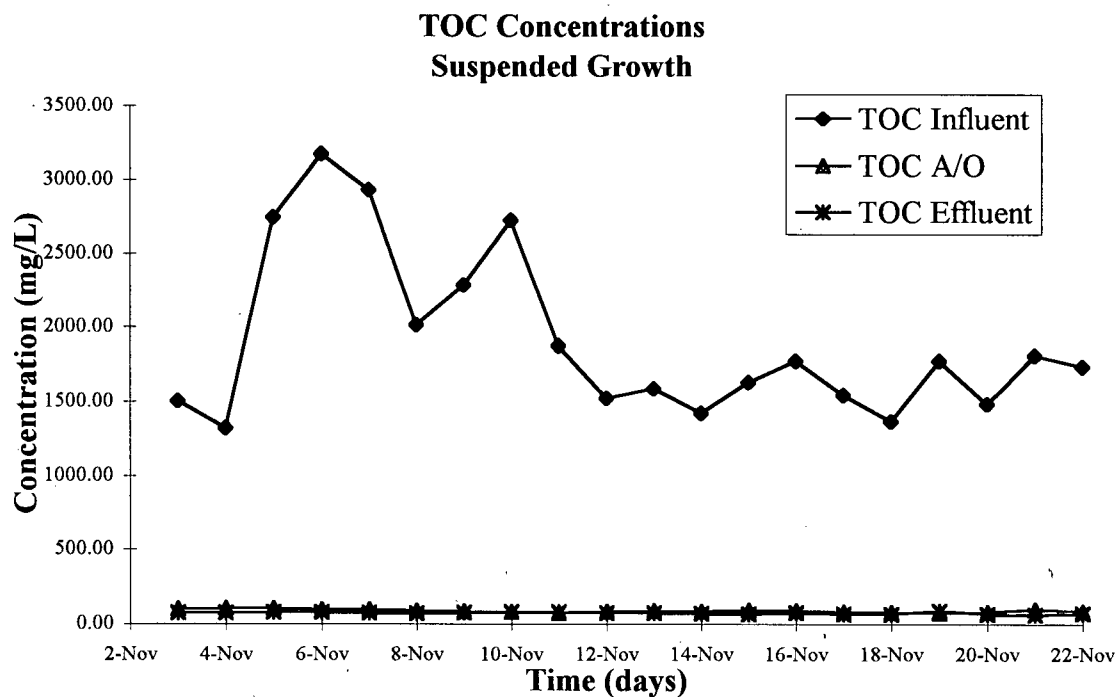**TOC Concentrations**

Suspended Growth

Figure 5. 52 TOC profile with time for the 2,000 mg/L run (suspended growth system)

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effluent concentration. The influent BOD_5/COD ratio was approximately 0.39. This ratio was nearly identical to the ratio obtained from the same run from the AGS. This would be an indication that there was no significant difference between the wastewater used for the AGS and SGS.

The NO_x-N production in the A/O reactor was very similar to the AGS for the 2,000 mg/L. A relatively high NO_x-N concentration was shown in the A/O effluent because of the high ammonia loading. The NO_x-N profile with time is illustrated in figure 5. 53. The higher concentration produced in the first two days of the run was due to the higher ammonia concentration in the wastewater (see figure 5. 51). The decrease in ammonia concentration led to a decrease in nitrate production in the following few days. On November 2nd, the TOC concentration of the influent was 1503 mg/L. The ammonia concentration, on the other hand, was 782.79 mg/L. The TOC/NH_4-N ratio was 1.92. This ratio suggested that nitrification and carbon oxidation could occur at the same rate during the aerobic phase. The difference could also be seen if the ORP and pH curve were examined closely. Figure 5. 54 and figure 5. 55 illustrate two cycles with different loading rates during the same run. Figure 5. 54 was a cycle recorded on November 2nd. The pH curve dropped sharply once aeration began. This pattern resembled the pH curve found during the 500 mg/L run, where the influent loading was relatively low. Figure 5.55 was a cycle recorded on November 11th. There was a slight difference between this curve and the one described above. The slight rise at the beginning of the cycle indicated that there was an excess of carbon in the system in comparison to ammonia. The TOC/NH_4-N ratio on that day was 3.10. This might have led to simultaneous nitrification/denitrification at

NOx-N Concentrations
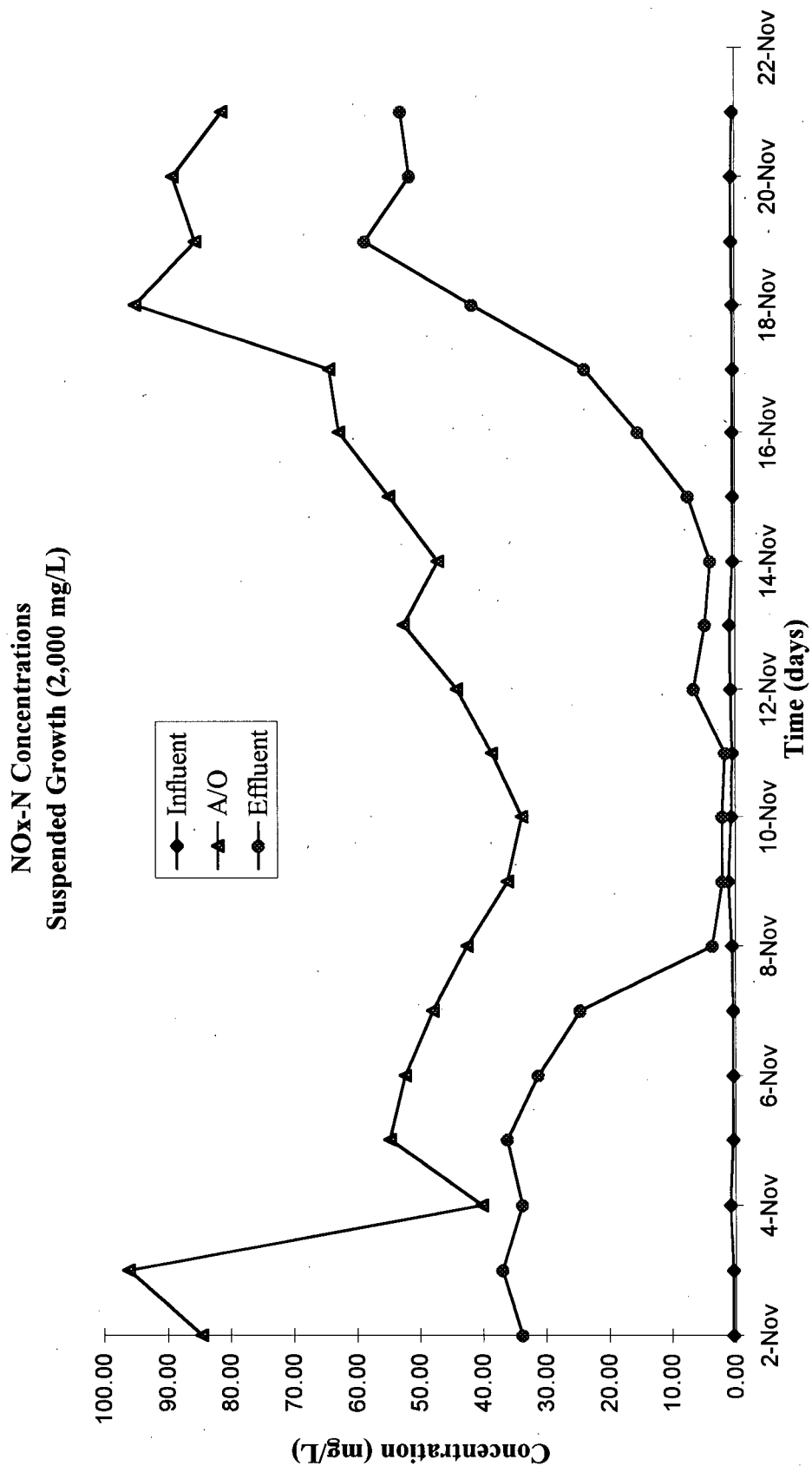**Suspended Growth (2,000 mg/L)**

Figure 5. 53 NO$_x$-N profile with time for the 2,000 mg/L run (suspended growth system)
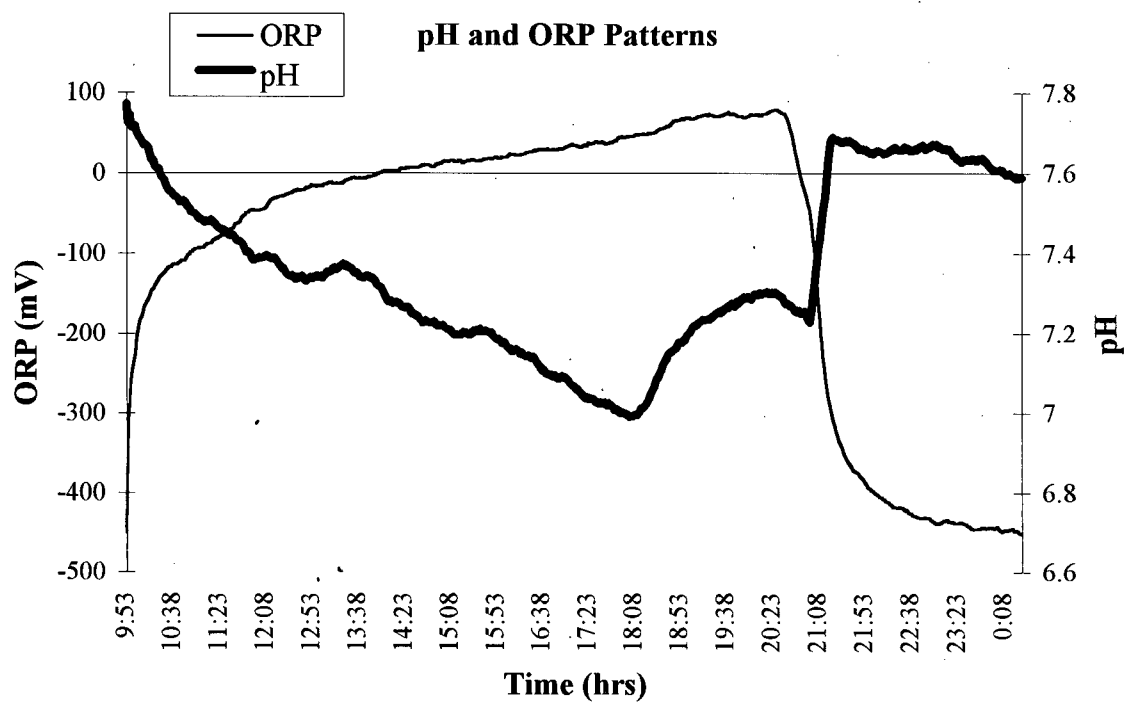
139

Figure 5. 54 pH and ORP patterns recorded on November 2nd
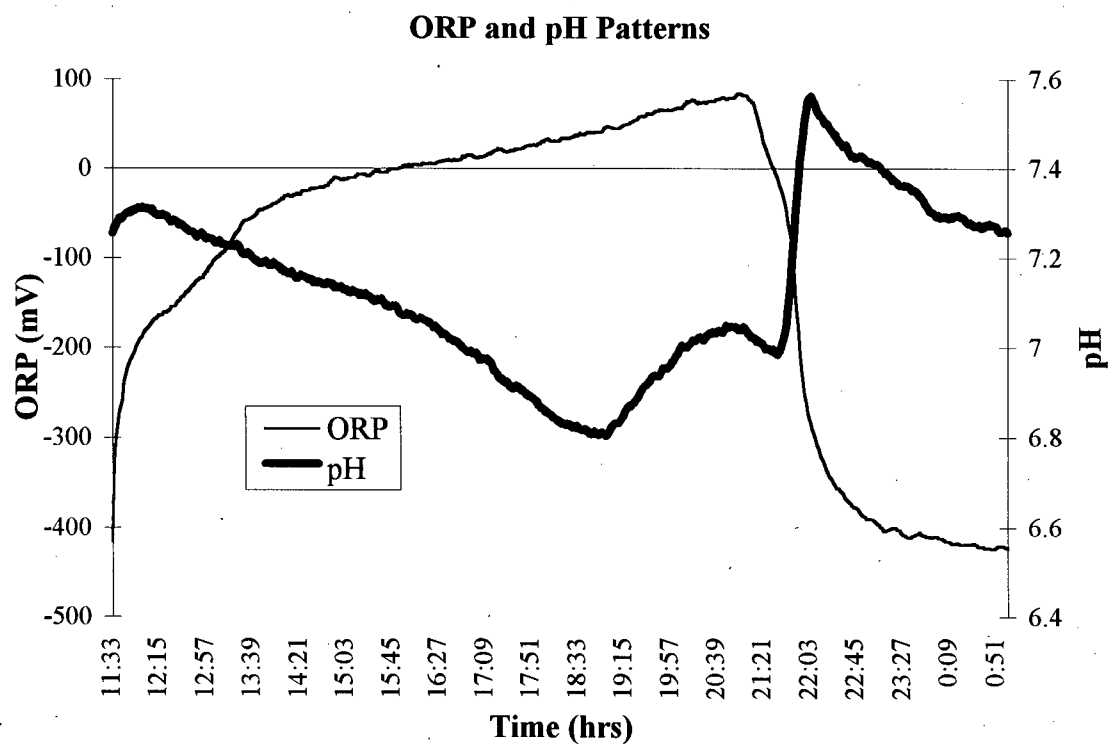

Figure 5. 55 pH and ORP patterns recorded on November 11th

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the beginning of the aerobic phase. Simultaneous nitrification/denitrification created a lower production of nitrate during these few days.

The anoxic reactor was able to remove up to 65% of the nitrate from the A/O effluent using either the carbon source available in the A/O effluent or endogenous respiration by the bacteria. The nitrate concentration in the final effluent from November 8th to 15th was lower compared to other days of this run. As mentioned before, the nitrate production from the A/O reactor was lower during these days. Also sludge from the A/O reactor was added to the anoxic reactor on November 7th to enhance denitrification. Treatment efficiency was improved until November 15th when much of the stored carbon in the bacterial cells was depleted due to continual consumption. This is illustrated by the track analysis done on the anoxic reactor through one cycle of this run (figure 5. 56). This track analysis was done on November 14th. It could be seen from the nutrients profile that the TOC concentration did not decrease throughout the entire cycle. This meant that denitrification was mainly carried out through endogenous respiration. The nitrate level was kept at a lower concentration for a longer period of time for the SGS than the AGS. As mentioned before, there were two ways of wasting sludge from the A/O reactor. This A/O sludge was wasted after the influent had been fed. When this sludge was added to the anoxic reactor, the rate of denitrification was increased based on both the soluble carbon content from the influent that was transferred over as part of the sludge and a greater amount of biomass that utilized endogenous respiration for the removal of nitrate. The carbon content therefore in the SGS's anoxic reactor was able to last longer than the AGS. The combined effect of the depletion of stored carbon content and the higher
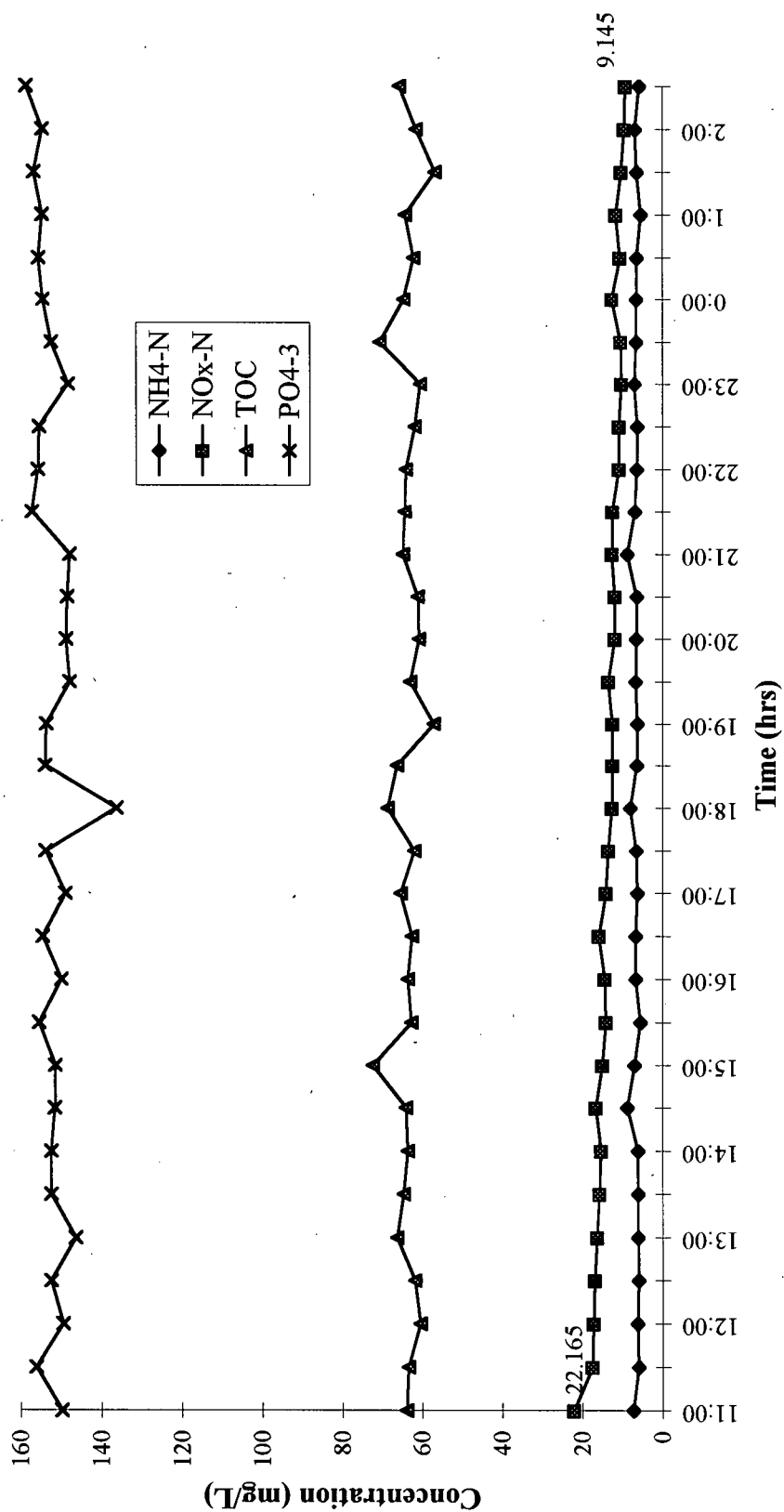
Figure 5. 56 Nutrients profiles of one cycle for the anoxic reactor

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production of nitrate in the A/O reactor led to an increase in nitrate concentration after November 15th.

The treatment efficiency for ortho-phosphate was approximately 45% for this run. The profile for PO_4^{-3} with time is shown in figure 5. 57. There was no difference between this and the attached growth system. Insufficient P-release in the anoxic phase due to a high nitrate concentration and an inadequate supply of soluble carbon caused a decrease of P-uptake during the aerobic phase. The system achieved the best ortho-phosphorus treatment efficiency on November 6th to 10th as illustrated by figure 5. 57. Referring back to figure 5. 52, these days had the highest organic loading rate of the entire run. The high concentration of organic matter in the influent promoted the production of VFA during the anaerobic phase. This led to a better P-release rate; hence a better uptake rate during the aerobic phase. As found previously from the other runs, nitrate was an inhibitor to P-release. From November 6th to 10th, the nitrate concentration from the A/O reactor was found to be the lowest of this run. The combination of the above caused the system to have the best PO_4^{-3} treatment efficiency during these days.

The solids levels in the influent, A/O effluent, and final effluent are represented in figures 5. 58 and 5. 59. The average TS, TVS, TSS, and TVSS removal efficiencies were 66%, 86%, 99% and 99% respectively. The influent BOD_5/COD ratio was 0.39, which was approximately the same as the AGS's influent. Comparing the suspended solids concentration of the A/O effluent between the two systems, the SGS seemed to maintain itself at a lower level. The reason for this could have been attributed to the Ringlace. The tube that was used to draw effluent out to the anoxic reactor was placed near a Ringlace in the AGS. During effluent decanting in the A/O reactor, some bacteria attached onto the

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PO4-3 Concentrations
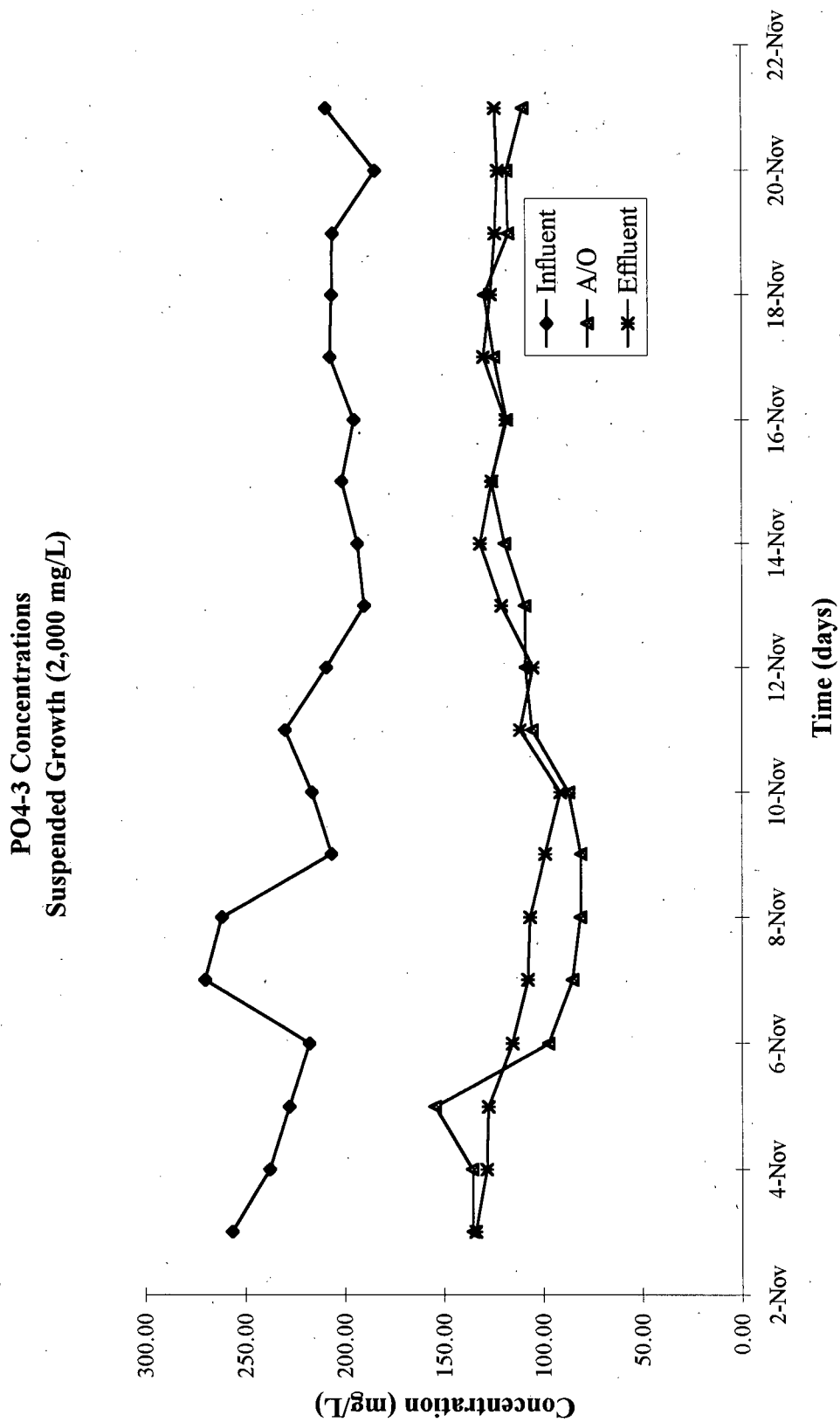## Suspended Growth (2,000 mg/L)

Figure 5. 57 PO_4^{-3} profile with time for the 2,000 mg/L run (suspended growth system)
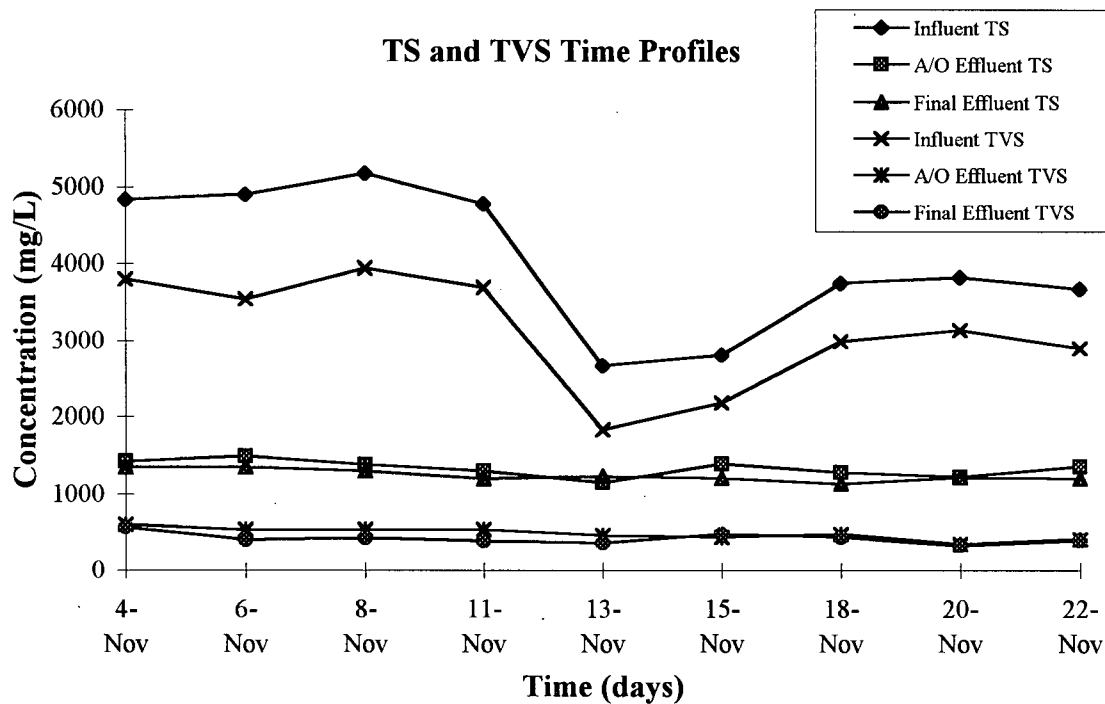
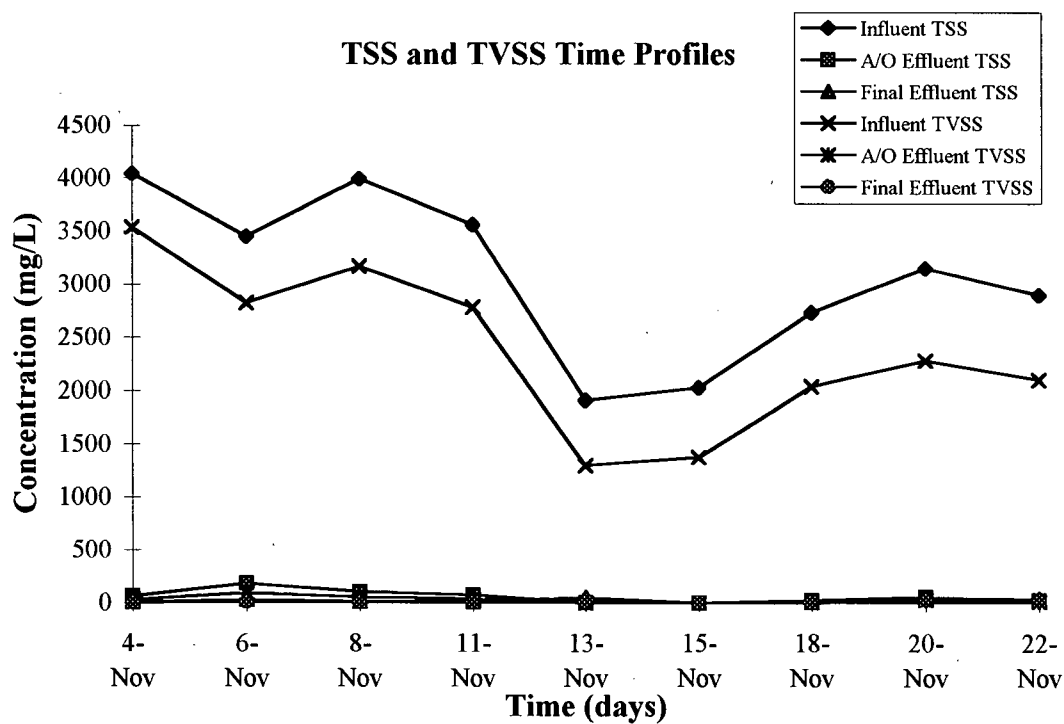**Figure 5. 58 TS and TVS profiles with time for the 2,000 mg/L run (suspended growth system)**

Figure 5. 59 TSS and TVSS profiles with time for the 2,000 mg/L run (suspended growth system

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Ringlace might not be able to resist the vacuum suction of the pump; hence detaching itself from the Ringlace. Subsequently they were transferred to the anoxic reactor. This would lead to a higher suspended solids concentration in the A/O effluent.

5. 3. 2. 2 TOC = 4,000 mg/L

The pH, ORP, and DO curves generated from this run were very consistent and uniform. The curves are illustrated in figures 5. 60 and 5. 61. These curves were very similar to the curves from the AGS. It can be seen in figure 5. 61 that the DO concentration was directly related to the ORP curve of the system. The peak value (RCMP) on the ORP curve decreased as the peak value on the DO curve (RCMP) decreased. Sludge accumulated in the A/O reactor until the excess was wasted from the reactor on a weekly basis. As sludge concentration increased, the total oxygen uptake rate increased, resulting in a downward shift of the DO curve. The sludge increase in the reactor also resulted in a decline of the ORP pattern since ORP had an inverse relation to the concentration of solids in solution (Ra, 1997). This result indicated that control using the absolute value of ORP cannot be used for the ORP curve because the ORP value shifts from cycle to cycle. For example, a control strategy can be programmed so that aeration shuts down at +50 mV, a use of absolute value of ORP as control during the aerobic phase. However, this control strategy might not work properly for all cycles because the ORP values shift every cycle according to the solids and DO concentrations in the A/O reactor. The pH curve, on the other hand, stayed relatively constant throughout this research. This is because pH measures the hydrogen ion concentration in solution rather
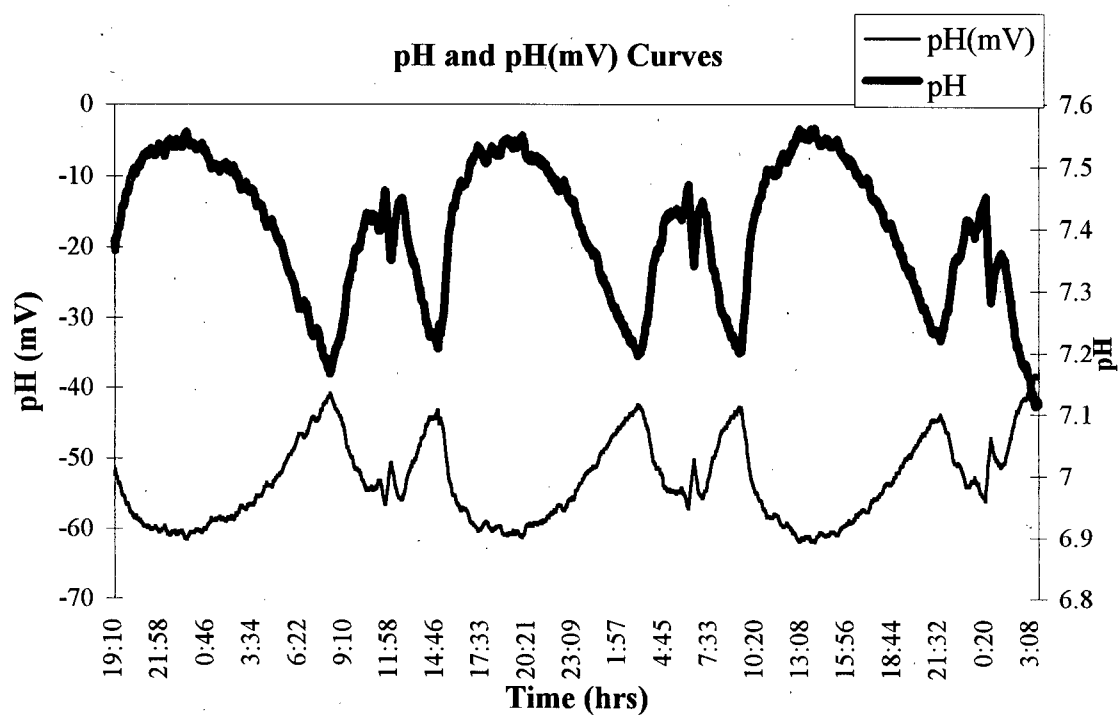
Figure 5. 60 Consistent pH and pH(mV) curves obtained from the 4,000 mg/L run (suspended growth system)
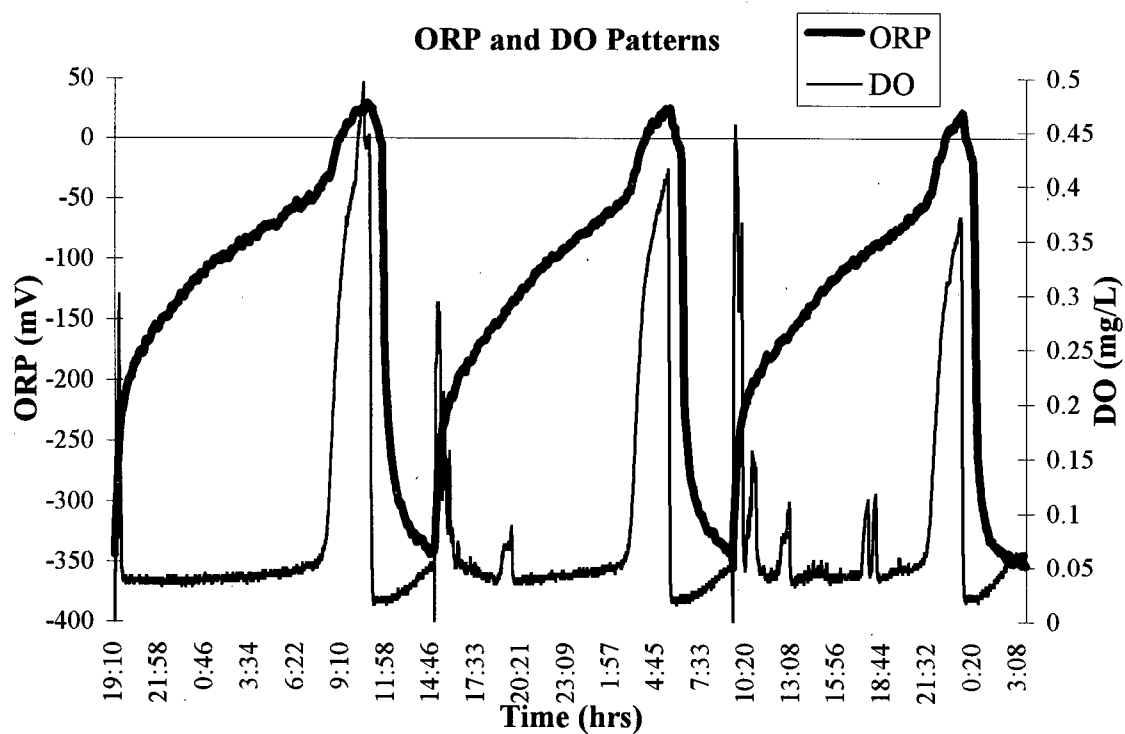

Figure 5. 61 Consistent DO and ORP curves obtained from the 4,000 mg/L run (suspended growth system)

than the overall bio-chemical reaction state in the system. This allows the use of specific pH values as a control parmaeter, e.g., pH = 6 and 8 for the present study.

The fluctuations in the influent concentrations influenced the length of each cycle due to the use of real-time control. The average HRTs were 5.0 and 2.92 days for the A/O and anoxic reactor, respectively. The total HRT therefore was 7.92 days. The longer retention time compared to the AGS was due to the lesser amount of biomass in the reactor. On the other hand, the HRT was obviously longer than in the 2,000 mg/L run because the influent concentration was higher. This demonstrated again that the real-time control system can be applied into any type of system under any influent loading rate.

Table 5. 5 summarizes the system's performance for this run. There were no major differences between the attached growth and suspended growth systems in terms of treatment efficiency besides their hydraulic retention times. The average influent TOC and NH_4-N concentrations were 3543.58 mg/L and 834.65 mg/L. The TOC/NH_4-N ratio was therefore 4.25. It was suggested in previous sections that the RCMP would represent a complete removal of carbon when the influent TOC/NH_4-N ratio was over 3. This explains why nitrate in the A/O reactor was removed during the settling phase (see section 5. 1. 2. 2). The residual carbon in the solution was used up by the bacteria in the A/O reactor before the effluent was decanted to the anoxic reactor. The residual carbon in the solution allowed rapid denitrification to take place. The C/N ratio for this run was also lower than in the 4,000 mg/L run for the AGS. This explains the higher nitrate concentration present in the reactor after the aerobic phase (see track analysis section for the 4,000 mg/L for both the AGS and SGS). The higher C/N ratio meant a higher carbon and a lower ammonia concentrations in the influent. The higher carbon and lower

Table 5. 5 Summary of system performance for the 4,000 mg/L run (suspended growth system)

	Suspended Growth									
	Average Influent (mg/L)	Min. - Max. (mg/L)	Std.	Average A/O (mg/L)	Min. - Max. (mg/L)	Std.	Average Effluent (mg/L)	Min. - Max. (mg/L)	Std.	% removal
TOC	3543.58	3095.5 - 4312	314.73	107.18	84.75 - 169.45	20.37	84.09	70.45 - 102.7	7.89	97.7
NH$_4$-N	834.65	815.28 - 981.08	64.85	1.54	0.43 - 2.95	0.81	1.31	0.61 - 5.01	0.97	99.04
NO$_x$-N	0.29	0.09 - 0.49	0.11	4.1	0.88 - 12.22	3.62	2.33	0.53 - 9.86	2.31	36.72
PO$_4^{-3}$	423.54	215.55 - 504.25	61.32	16.18	3.25 - 22.04	4.33	20.98	9.42 - 27.53	4.32	94.94
BOD$_5$	5451.14	4792.5 - 6030	361.01	17.74	9.3 - 26.76	5.98	12.04	5.16 - 22.65	5.94	99.78
COD	10336.43	8572.07 - 13467.81	1732.82	219.66	185.45 - 272.98	30.06	186.24	166.19 - 215.21	14.68	98.17
TS	7171.94	5402.5 - 9340	1179.09	1403.89	1297.5 - 1525	74.45	1398.89	1285 - 1500	70.24	79.96
TVS	5107.5	3505 - 6652	1020.4	457.78	405 - 607.5	61.03	440.28	380 - 495	33.11	91.01
TSS	4987.78	3135 - 7060	1133.23	127.78	65 - 205	41.69	111.11	35 - 225	63.38	97.73
TVSS	4256.11	2635 - 5875	978.94	118.33	65 - 195	40.47	102.78	35 - 225	53.1	97.5
DO	0	-	-	4.16	1.9 - 6.89	1.24	5.72	3.28 - 8.53	1.28	-

ammonia concentrations would cause a longer period of simultaneous nitrification/denitrification during the aerobic phase because it would take a longer time to establish a balance between the carbon and ammonia concentrations in the reactor. In another words, the rate of carbon oxidation was greater than the rate of nitrification at the beginning of the aerobic phase. As the carbon concentration decreased, the heterotrophic bacteria were not as aggressive in competing for oxygen with the nitrifiers. Therefore, all the nitrifiers in the reactor would have their chance to obtain oxygen for cell synthesis. It should be noted that facultative bacteria prefer the use of oxygen as an electron acceptor. However, this period was prolonged because of the higher C/N ratio, which meant a longer period of simultaneous nitrification/denitrification. Also a lower C/N ratio meant a higher concentration of ammonia in the influent. It is obvious that there would be more nitrate produced. The combined effect of the above caused a high production of nitrate during the aerobic phase.

The influent, A/O effluent, and final effluent ammonia concentrations were 834.65 mg/L, 1.54 mg/L, and 1.31 mg/L respectively. A 99% ammonia removal efficiency was achieved. This result is similar to all the results found in this research. There was no increase in the ammonia concentration in the anoxic reactor. This result was again expected because the control point was set beyond the NBP, which ensured a complete removal of ammonia. The small standard deviation in the A/O effluent suggested the effectiveness of the real-time control system. The time profile of ammonia is illustrated in figure 5. 62.

Figure 5. 63 shows the profile of TOC over time. The influent TOC concentration was 3543.58 mg/L, while the A/O and final effluent concentrations were 107.18 mg/L and

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NH4-N Concentrations
Suspended Growth (4,000 mg/L)
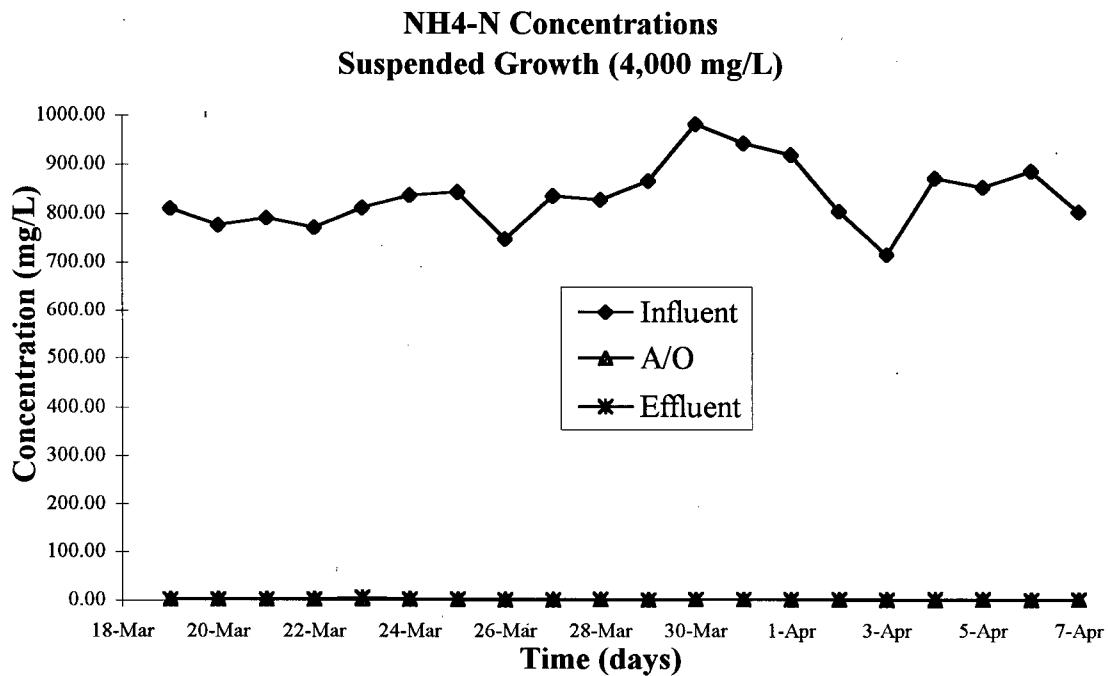
Figure 5. 62 NH₄-N profile with time for the 4,000 mg/L run (suspended growth system)

TOC Concentrations
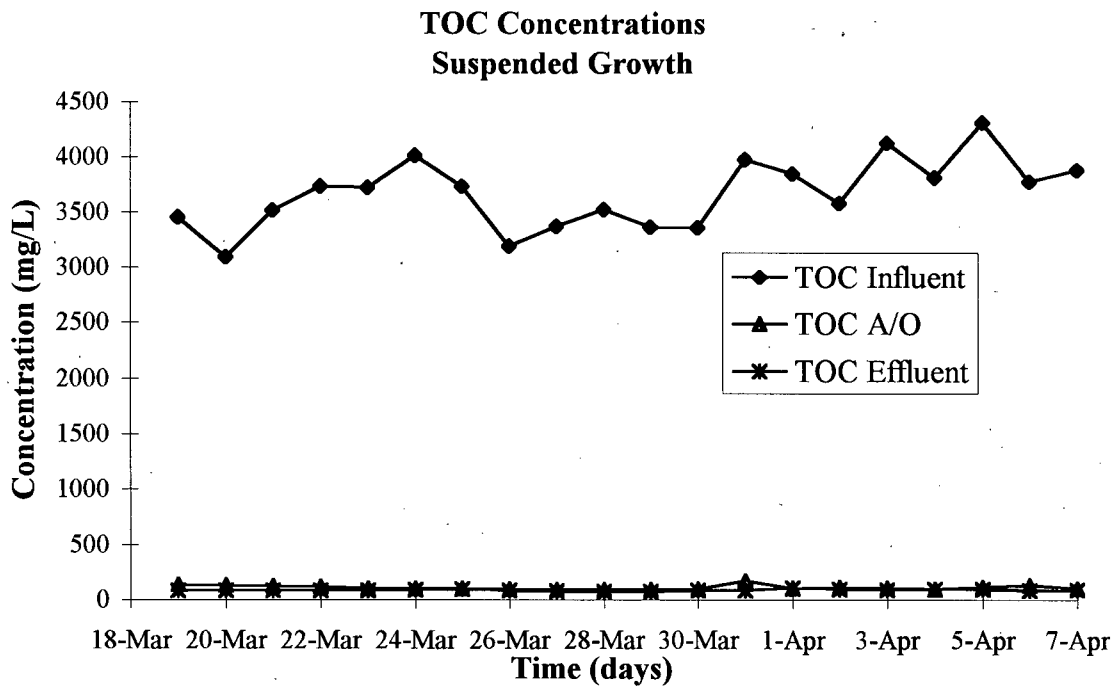## Suspended Growth

Figure 5. 63 TOC profile with time for the 4,000 mg/L run (suspended growth system)

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84.09 mg/L respectively. This gives a treatment efficiency of over 95%. The decrease in TOC concentration from the A/O to final effluent was attributed to the consumption of organic carbon for denitrification in the anoxic reactor. This result shouldn't be a surprise because the control point was set in between the NBP and RCMP, which ensured virtually a complete removal of carbon before the effluent was discharged. The averaged influent BOD_5/COD ratio was 0.53, while the influent BOD_5/TOC ratio was 1.54. The effluent ratios of BOD_5/COD and BOD_5/TOC were 0.064 and 0.14 respectively. These averages suggested that a biological treatment system heavily depends on easily bio-degradable matter in the wastewater for success. There was no significant difference between the results found in this run and the run by the attached growth system.

The nitrate concentration in swine wastewater is negligible. However, the conversion of ammonia into nitrate forced the system to also remove nitrate before discharging the effluent into the environment. The nitrate concentration in the A/O effluent was considerably lower than the nitrate concentration for the 2,000 mg/l run. Simultaneous nitrification/denitrification during the aerobic phase, rapid denitrification using available residual carbon during the settling stage, and quick denitrification during the influent feeding stage using fresh feed carbon caused the low nitrate concentration in the A/O effluent. The removal efficiency of nitrate from the anoxic reactor was only 36%. This was relatively low compared to the 2,000 mg/L run. However, the A/O effluent nitrate concentration only averaged 4.04 mg/L. This was actually an incredible removal efficiency because the ammonia concentration of the influent was approximately 834 mg/L. The final effluent nitrate concentration was 2.33 mg/L. The results found in this run were very similar to the AGS. The results suggest that the anoxic reactor was not

needed for the removal of nitrate (only for this run). The time-profile for nitrate is shown in figure 5. 64.

The track analysis for nitrate in the anoxic reactor is shown in figure 5. 65. The nitrate concentration was less than 10 mg/L at the beginning of the cycle. The nitrate concentration was reduced to 0 mg/L before the end of the cycle. As shown by figure 5. 65, the cycle for the track analysis was relatively longer than for other cycles due to the high concentration of the wastewater entering the A/O reactor. Since the anoxic reactor responds to the action of the A/O reactor, the HRT of the anoxic reactor was extended. There was plenty of time for complete denitrification using endogenous respiration. In general, the high influent concentration of this run led to a longer retention time in the anoxic reactor. This could also be a factor in the low concentration in the final effluent.

The influent, A/O effluent, and final effluent ortho-P concentrations were 423.54 mg/L, 16.18 mg/L, and 20.98 mg/L respectively. The treatment efficiency of the system was approximately 95%. The time-profile for soluble phosphorus is shown in figure 5. 66. The high treatment efficiency was attributed to the high soluble carbon concentration in the wastewater. The high organic loading rate led to the production of volatile fatty acids during the anaerobic phase. Bio-P bacteria respond to VFA in the influent wastewater under anaerobic conditions by releasing stored phosphorus (Metcalf & Eddy, 1991). The VFA are an important substrate for Bio-P bacteria during competition with heterotrophs. The VFA were stored in the form of PHA and were utilized during the subsequent aerobic phase to uptake phosphorus. The high concentration of VFA led to a maximum release of phosphorus during the anaerobic phase, and subsequently a higher uptake rate during the aerobic phase. There was no significant difference between the AGS and SGS for this

NOx-N Concentrations
Suspended Growth (4,000 mg/L)

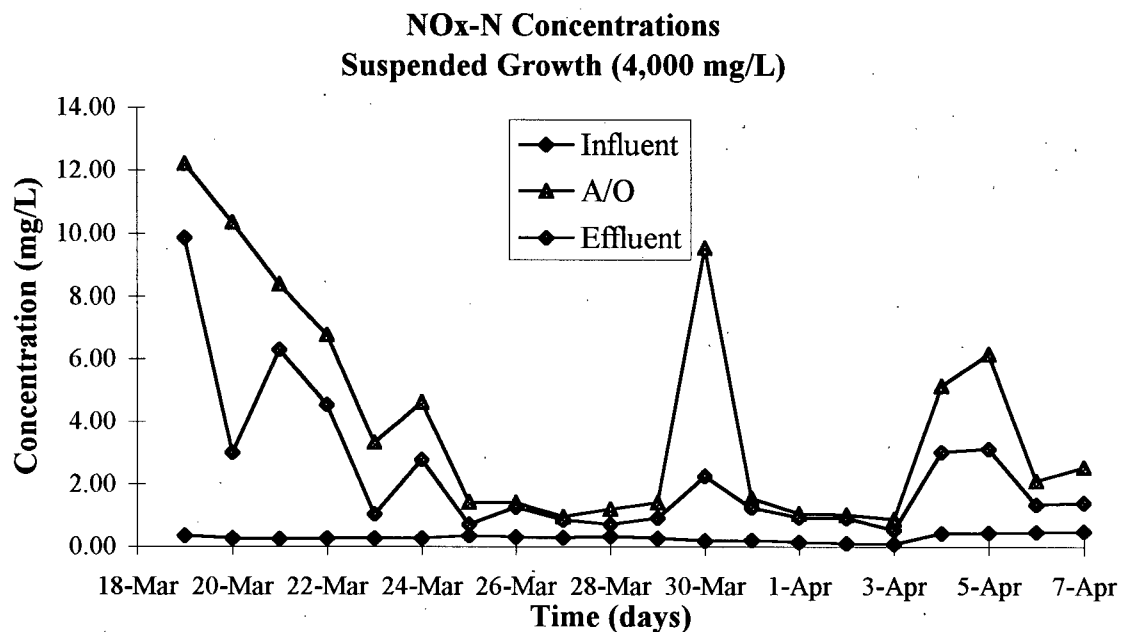**Figure 5. 64 NO$_x$-N profile with time for the 4,000 mg/L run (suspended growth system)**

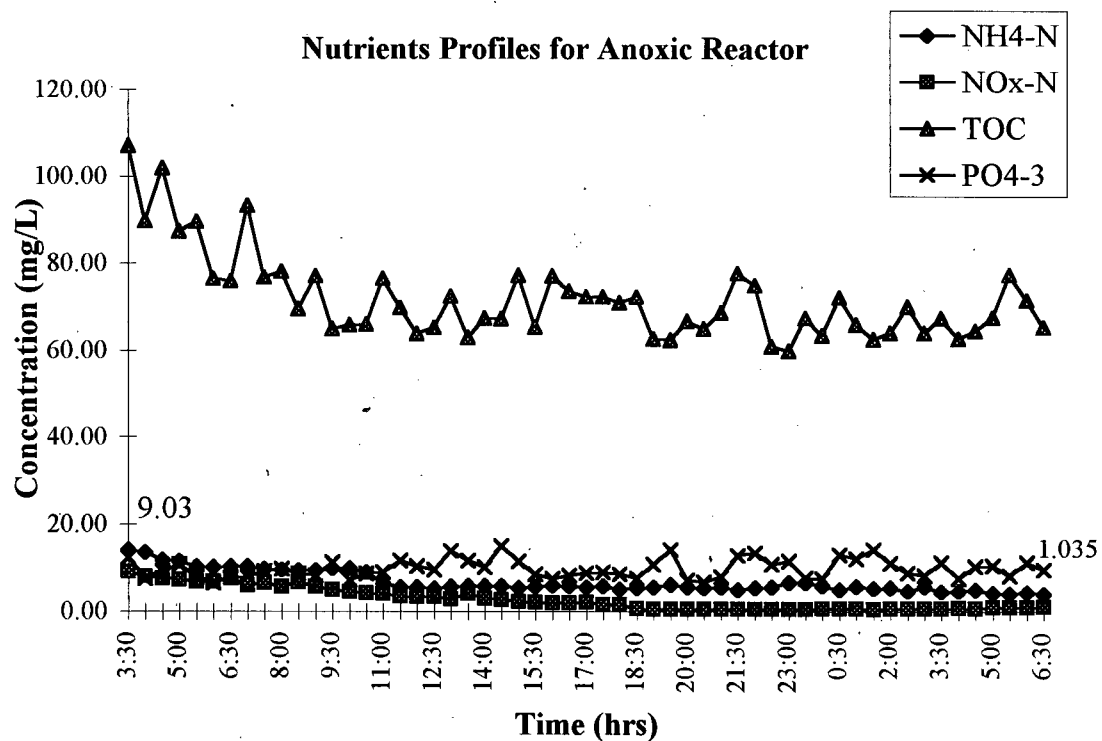**Nutrients Profiles for Anoxic Reactor**

Figure 5. 65 Nutrients profiles of one cycle for the anoxic reactor

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PO4-3 Concentrations
Suspended Growth (4,000 mg/L)

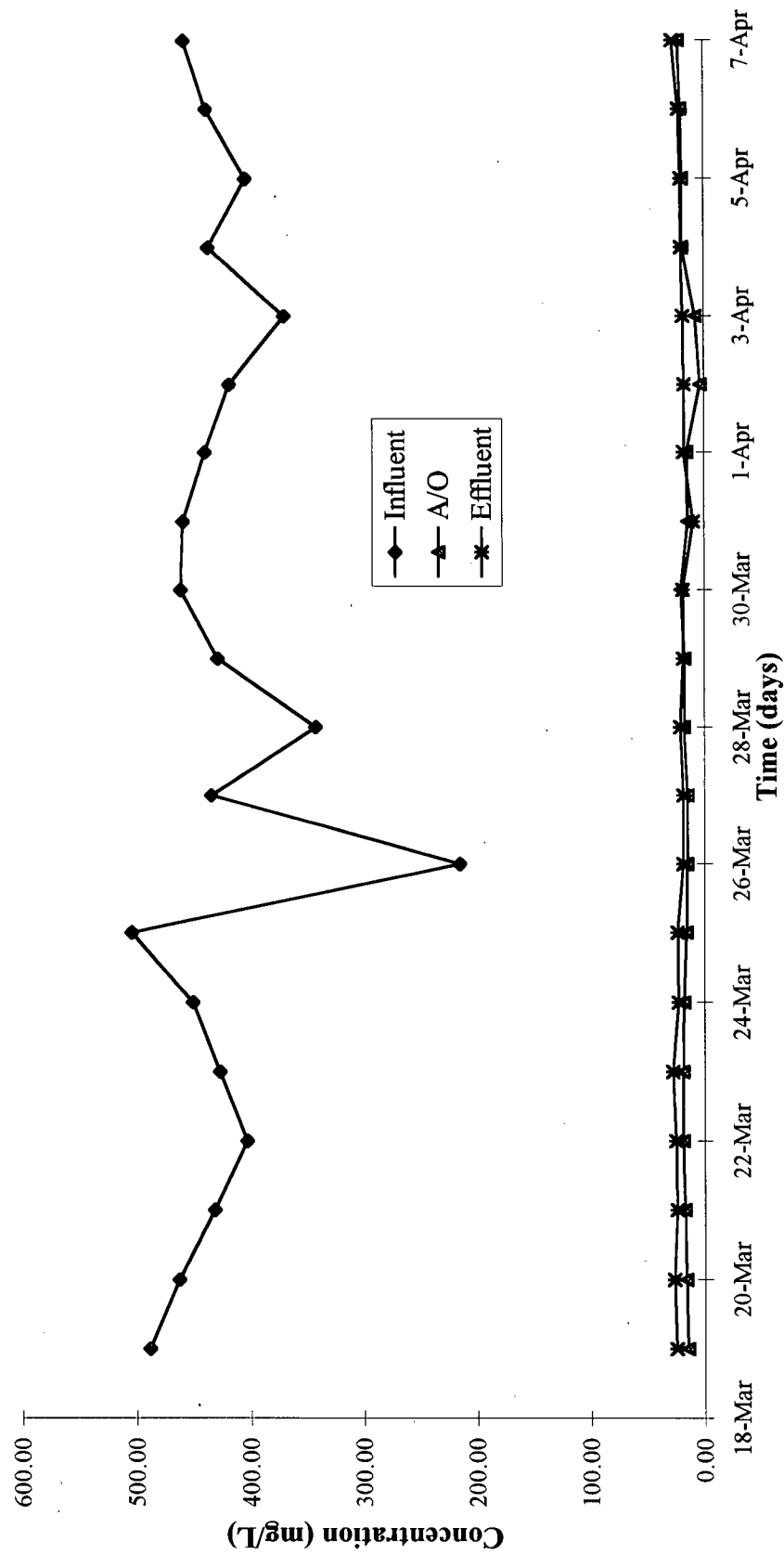**Figure 5. 66 PO$_4$$^{-3}$ profile with time for the 4,000 mg/L run (suspended growth system)**

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run. There was no phosphorus release or uptake in the anoxic reactor because it was purely designed to remove nitrate. Therefore, there weren't any poly-P bacteria in the anoxic reactor to carry out any reactions.

The solids levels in the influent, A/O effluent, and final effluent are represented in figures 5. 67 and 5. 68. The average TS, TVS, TSS, and TVSS removal efficiencies were 80%, 91%, 97% and 97% respectively. These results were approximately the same as the AGS. The slightly lower removal efficiency for TS was due to the high inorganic solids in the wastewater. The treatment efficiency was higher for TS and TVS compared to the 2,000 mg/L run. This was probably due to the higher influent BOD_5/COD ratio. The influent BOD_5/COD ratio for this run was 0.53, while it was 0.39 for the 2,000 mg/L run. The lower BOD_5/COD ratio suggested a higher proportion of inorganic solids in the wastewater that could not be removed by a biological process.

5. 3. 2. 3 TOC = 500 mg/L

Some of the pH, ORP, and DO patterns obtained from this run are illustrated in figures 5. 69 and 5. 70. All the curves shown on these graphs were consistent and uniform. They all resembled their normal real-time control patterns (pattern of the second track analysis of this run). Patterns resembling the first track analysis (unusual patterns) of this run were also found. They are shown in figure 5. 71. Unfortunately, the DO patterns weren't recorded for these cycles. Cycles for this run varied from time to time, depending on the concentrations of the influent. As demonstrated by figure 5. 71, some cycles did not receive enough carbon for rapid denitrification during the anoxic phase. That resulted

TS and TVS Time Profiles

Legend:
- ◆ Influent TS
- ▣ A/O Effluent TS
- ▲ Final Effluent TS
- ✕ Influent TVS
- ✱ A/O Effluent TVS
- ⊗ Final Effluent TVS

Figure 5. 67 TS and TVS profiles with time for the 4,000 mg/L run (suspended growth system)

TSS and TVSS Time Profiles

Legend:
- ◆ Influent TSS
- ▣ A/O Effluent TSS
- ▲ Final Effluent TSS
- ✕ Influent TVSS
- ✱ A/O Effluent TVSS
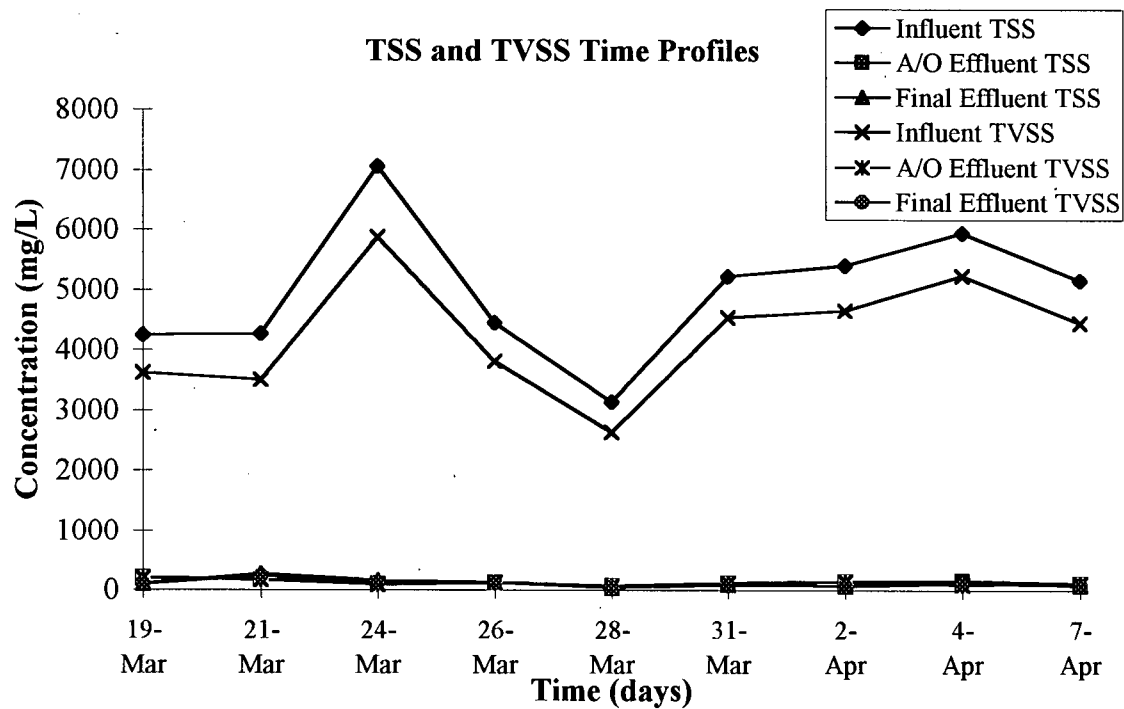- ⊗ Final Effluent TVSS

Figure 5. 68 TSS and TVSS profiles with time for the 4,000 mg/L run (suspended growth system)
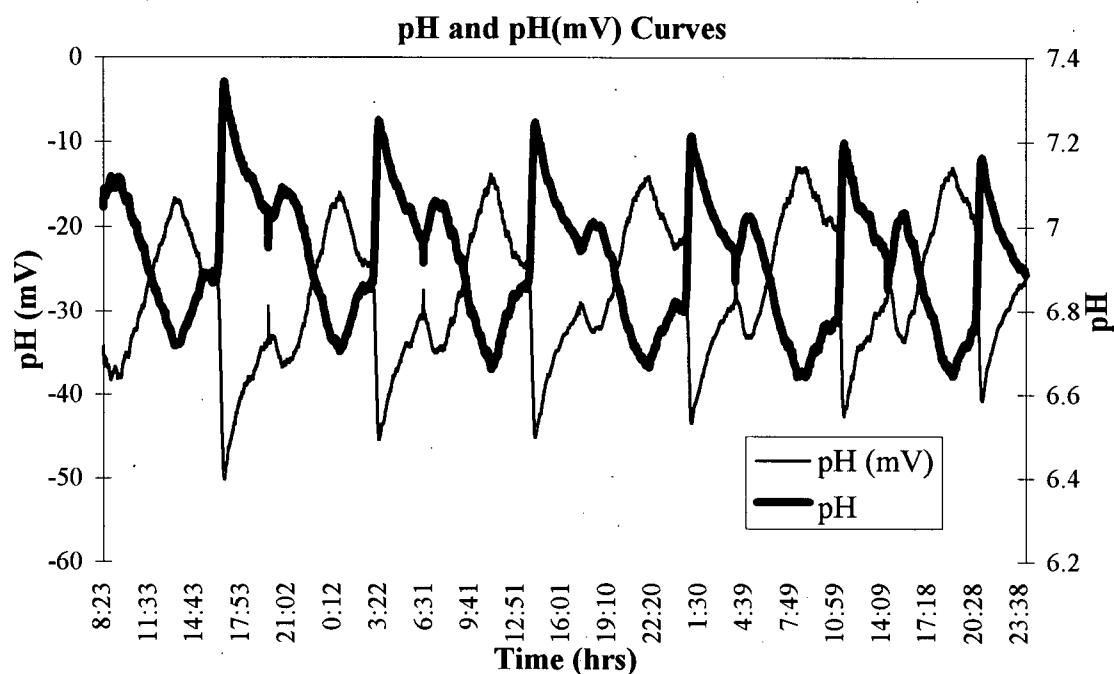
157

Figure 5. 69 **Consistent pH and pH(mV) curves obtained from the 500 mg/L run (suspended growth system)**
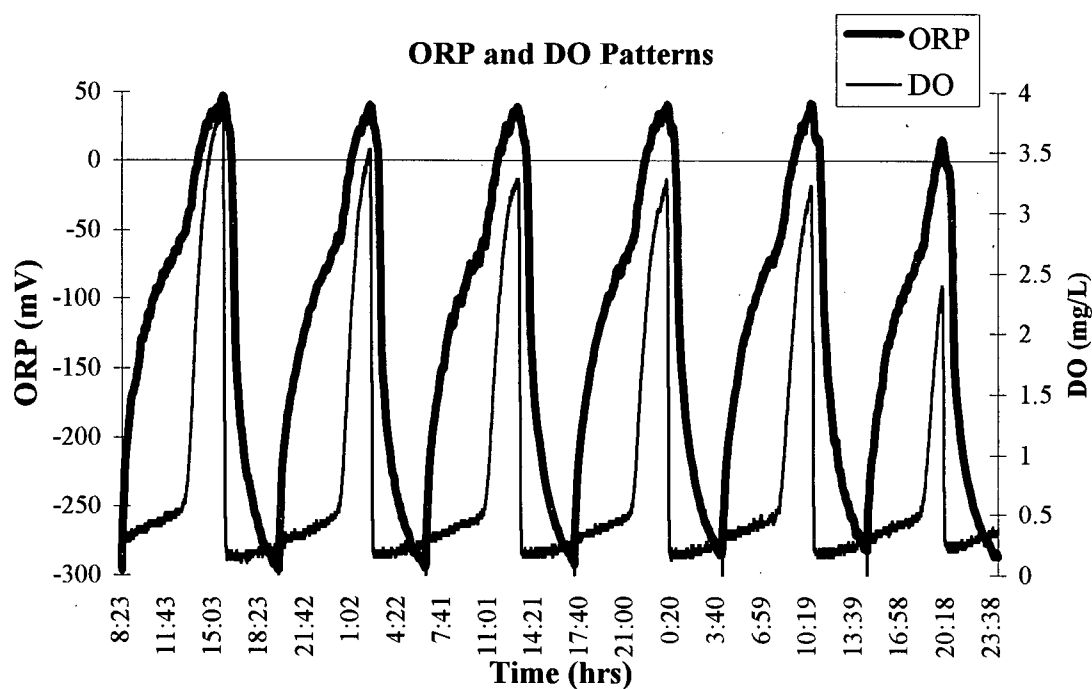

Figure 5. 70 **Consistent DO and ORP curves obtained from the 500 mg/L run (suspended growth system)**
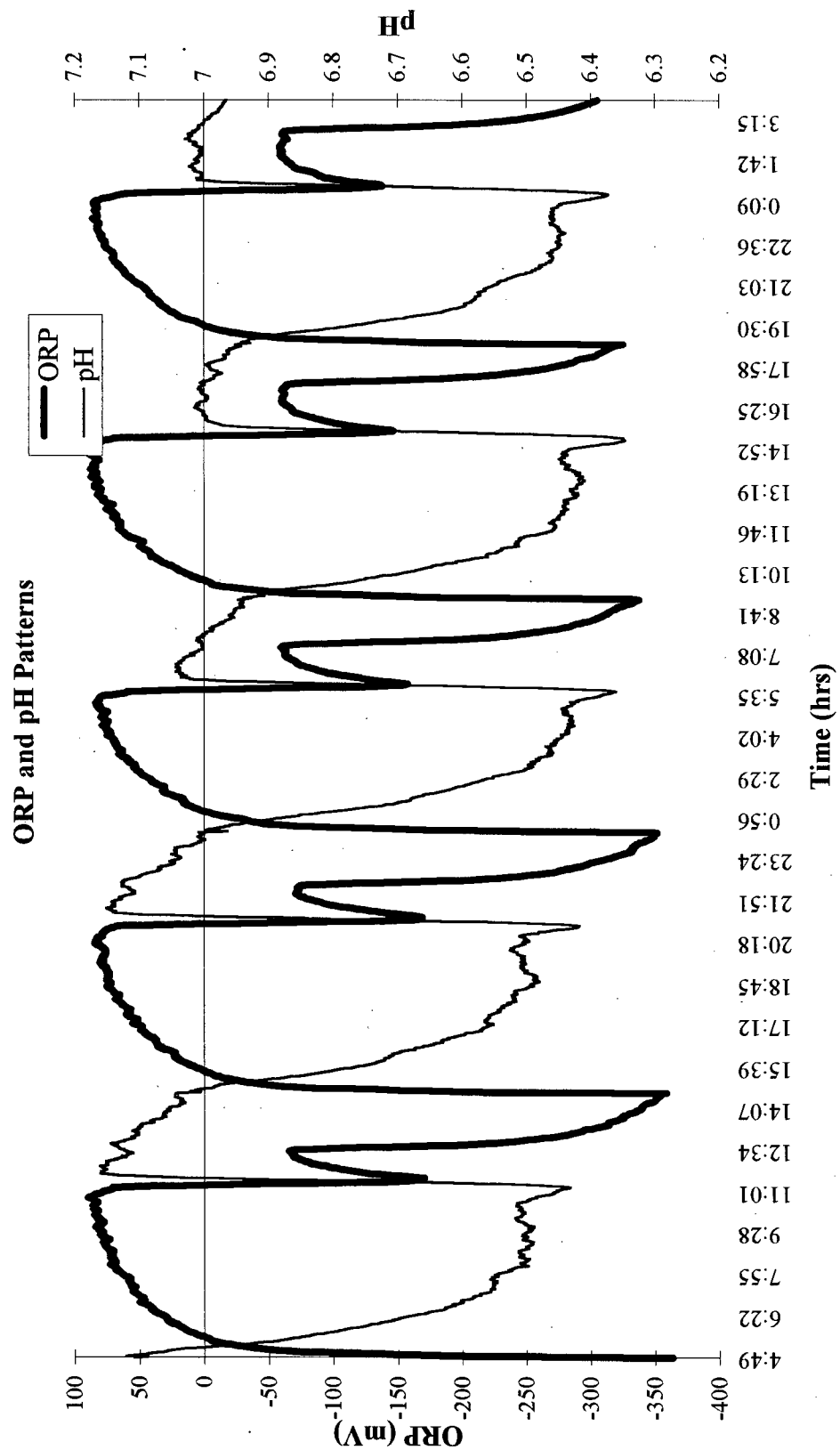
158

Figure 5. 71 Unusual pH and ORP patterns

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in the nitrate knee being more distinguishable than others. Even though the aerobic phases for each cycle weren't uniform throughout this run, the patterns were similar enough that the real-time process was still able to recognize the control point and terminated the cycle after the NBP. In reality, aeration can be shut off immediately after the NBP for this run because during a low organic loading cycle, the RCMP no longer represents a complete removal of carbon. This point is shifted to before the NBP in the aerobic phase. Nonetheless, the real-time control process ensured a complete removal of ammonia and allowed a flexible HRT despite the shortage of a carbon substrate for the complete removal of all the nutrients from the wastewater.

The downfall of using DO as a control parameter was again illustrated in this run. The DO value did not yield any information during the anoxic phase. Therefore, it cannot indicate that the system is operating under any kind of deficiency, except a lack of DO. But in fact, there is an insufficient supply of carbon in the wastewater for denitrification and phosphorus release. By using either the pH or ORP curve as a monitoring tool, methanol or other carbon substrates can be added to the system to enhance denitrification or phosphorus release when the nitrate knee is as visible as are some of them shown in this run. Again, there was no significant difference between this system and the AGS. The AGS basically responded the same way under the same conditions.

The HRTs for the A/O and anoxic reactors were 2.27 and 1.33 days, respectively. The total HRT was therefore 3.6 days. The HRT of this system for this run was longer than the AGS because the population of bacteria that existed within the reactor was less than the AGS. The advantage of a real-time control system can be shown by figure 5. 72 as the length of each cycle was different. The HRT for this run was obviously shorter than
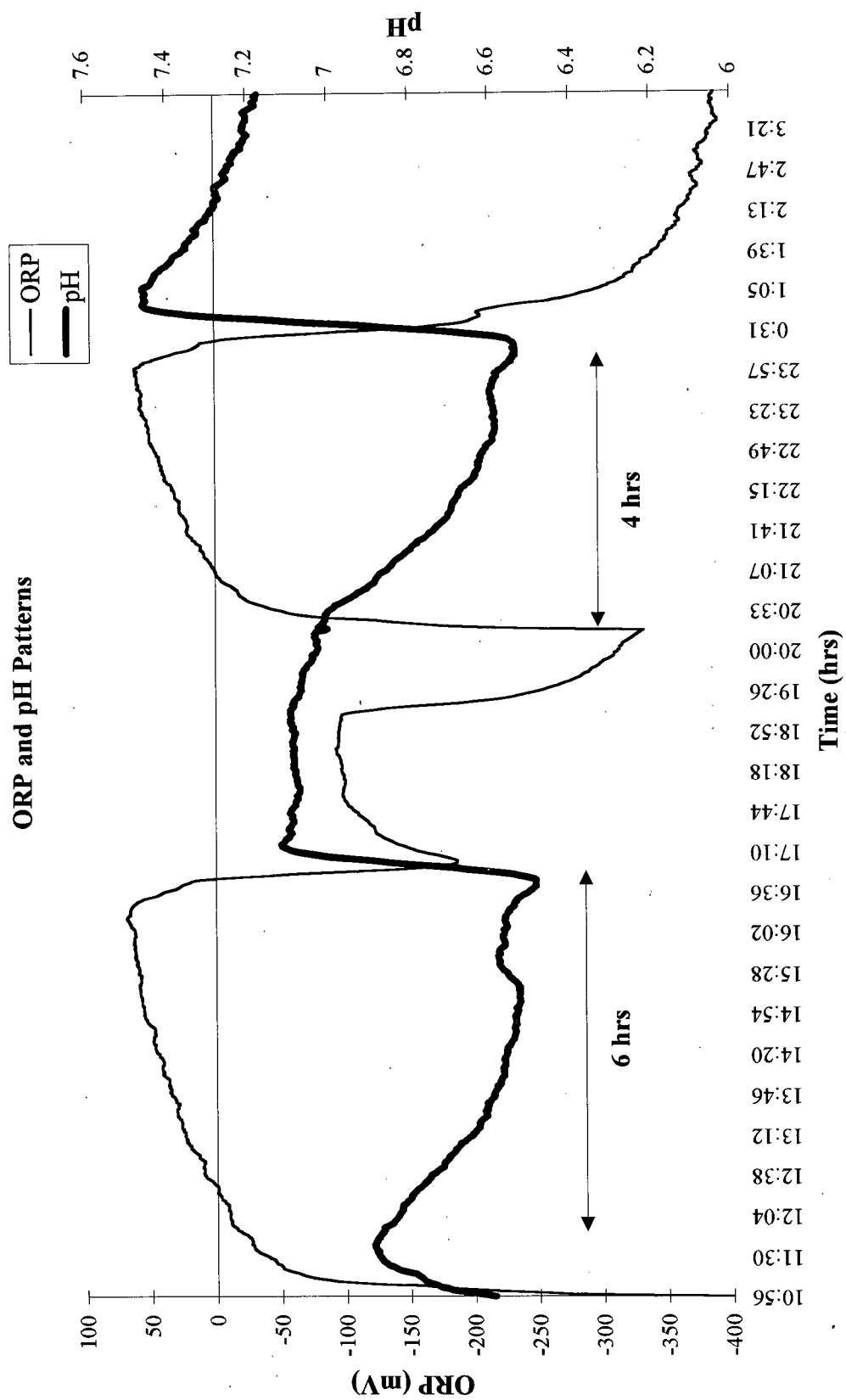
Figure 5. 72 Differences in cycle length

the two runs previously discussed for this system. The concentration of the influent was much stronger for the other two runs; hence it took a longer period of time to complete all the necessary reactions, using the same amount of bacteria.

Table 5. 6 summarizes the results for this run. The average influent TOC and ammonia concentrations were 623.54 mg/L and 268.66 mg/L respectively. The C/N ratio was therefore 2.32. This ratio is the lowest of all the runs. There should be no surprise that the system did not have enough carbon to carry out the appropriate reactions such as denitrification and phosphorus release, based on this C/N ratio. This ratio indicated that the RCMP would have occurred before the NBP, as indicated by the track studies. The C/N ratio for this run was even lower than the 500 mg/L run of the AGS. This created more cycles with a longer denitrification time (nitrate knee) during the anoxic phase in the A/O reactor, as more nitrate was produced while the carbon supply was reduced. The phosphorus concentration can definitely be affected by this ratio because if there wasn't enough carbon for denitrification, there would be even less carbon available for phosphorus release.

The time profile for ammonia is shown in figure 5. 73. The influent, A/O effluent, and final effluent ammonia concentrations were 268.66 mg/L, 1.67 mg/L, and 4.43 mg/L respectively. The removal efficiency was over 98%. There was no significant difference between this and the AGS. The real time control process ensures a complete removal of ammonia regardless of the influent concentration. Again, the small standard deviations of the A/O and final effluent indicated the effectiveness of a real-time control system.

The TOC time profile is shown in figure 5. 74. The influent, A/O effluent, and final effluent concentrations were 623.54 mg/L, 47.02 mg/L, and 40.32 mg/L respectively.

Table 5. 6 Summary of system performance for the 500 mg/L run (suspended growth system)

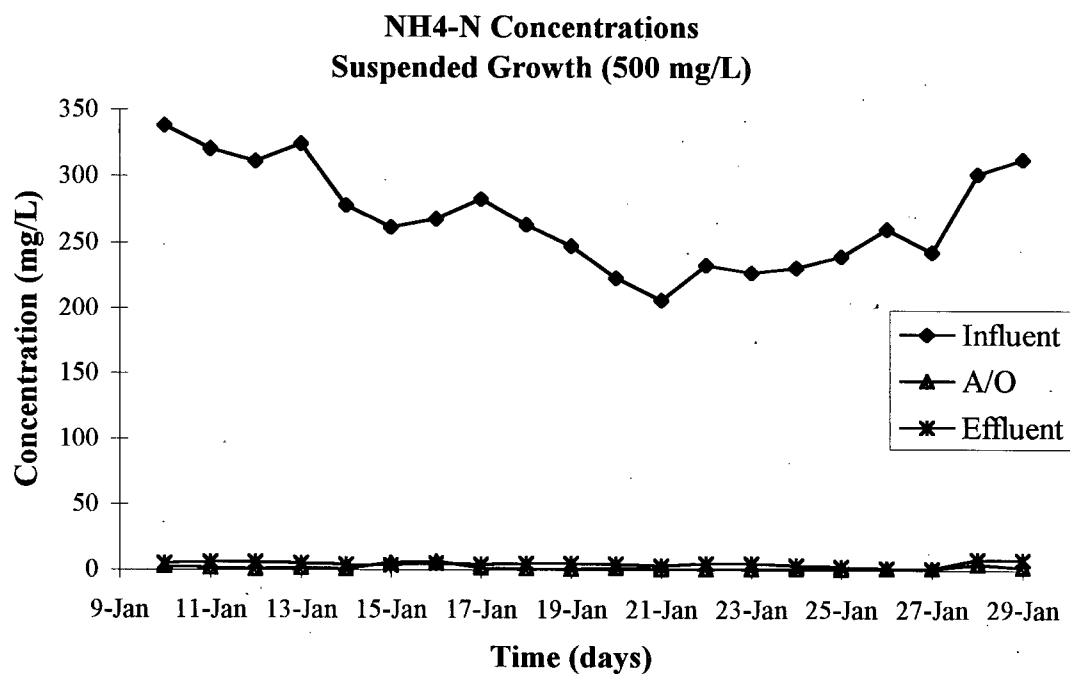|	Influent			Suspended Growth			Effluent			% removal
	Average (mg/L)	Min. - Max. (mg/L)	Std.	Average A/O (mg/L)	Min. - Max. (mg/L)	Std.	Average Effluent (mg/L)	Min. - Max. (mg/L)	Std.	
TOC	623.54	328.55 - 872.05	146.95	47.02	34.82 - 74.4	9.6	40.32	29.22 - 89.59	14.81	93.4
NH_4-N	268.66	206.15 - 337.7	38.56	1.67	0.32 - 6.29	1.73	4.43	1.07 - 8.26	1.78	98.37
NO_x-N	0.79	0.70 - 0.95	0.076	53.77	33 - 73.97	9.19	19.26	4.64 - 30.14	7.81	55
PO_4^{-3}	121.66	99.58 - 148.6	14.89	106.15	83.95 - 127.32	11.6	105.81	93.31 - 121.26	8.6	12.21
BOD_5	610.06	488.25 - 907.5	123.33	14.23	9.24 - 18.06	3.49	14.3	6.33 - 25.35	6.49	97.53
COD	2169.58	1626.88 - 3009.88	439.06	133.88	101.42 - 199.45	30.58	122.9	89.16 - 229.21	40.37	94.22
TS	1646.39	1400 - 1815	70.07	966.67	710 - 1295	34.34	781.39	695 - 922.5	20.7	52.2
TVS	1132.22	957.5 - 1322.5	89.34	503.89	267.5 - 817.5	27.89	303.33	175 - 417.5	17.5	72.65
TSS	743.89	355 - 1040	39.37	30	0 - 105	31.91	2.78	0 - 25	11.79	99.66
TVSS	372.78	177.5 - 520	19.18	15	0 - 52.5	15.95	1.39	0 - 12.5	5.89	99.66
DO	0	-	-	4.46	3.05 - 6.36	0.89	4.82	3.8 - 6.23	0.71	-

Figure 5. 73 NH₄-N profile with time for the 500 mg/L run (suspended growth system)
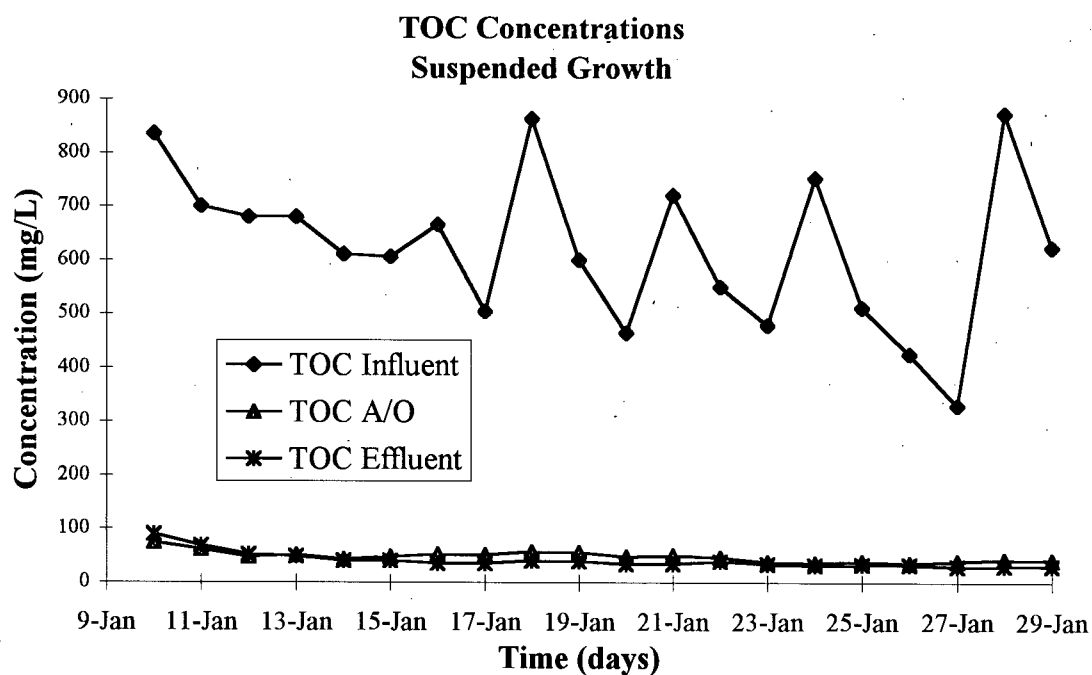

Figure 5. 74 TOC profile with time for the 500 mg/L run (suspended growth system)

The real-time control process also ensures a complete removal of nitrate regardless of the influent concentrations. The real-time process achieved an organic removal efficiency of over 90% in all the runs in this research. As mentioned before, the only difference that was found between this and the AGS was the HRT. There were no significant difference between the other parameters measured such as treatment efficiency of ammonia, phosphorus, carbon, etc. By examining the A/O and final effluent's concentration, it can be seen that virtually no soluble carbon was transferred to the anoxic reactor for denitrification. The low organic loading caused the carbon to be completely removed from the system before the effluent was discharged. The influent BOD_5/COD ratio was 0.28. This ratio turned out to be lower than the 500 mg/L run of the AGS. This in fact confirmed why there were more cycles with a longer denitrification period in the anoxic phase than the AGS. Bacteria thrive on the easily biodegradable materials contained in the wastewater. The lower BOD_5/COD ratio suggested that the proportion of biodegradable materials in the wastewater available for consumption have been reduced, meaning that the bacteria relied more on endogenous respiration for cell synthesis.

The time profile for nitrate is illustrated in figure 5. 75. The nitrate production from the A/O reactor was fairly constant throughout this run. The high nitrate production was attributed to a low influent C/N ratio. There was no simultaneous nitrification/denitrification in the A/O reactor during the aerobic phase due to the low carbon content in the reactor. Therefore, all the ammonia from the influent was converted to nitrate. As mentioned before, there was virtually no soluble carbon passed on to the anoxic reactor for denitrification. However, sludge was constantly added to the anoxic reactor to enhance the rate of denitrification. The nitrate removal efficiency in the anoxic
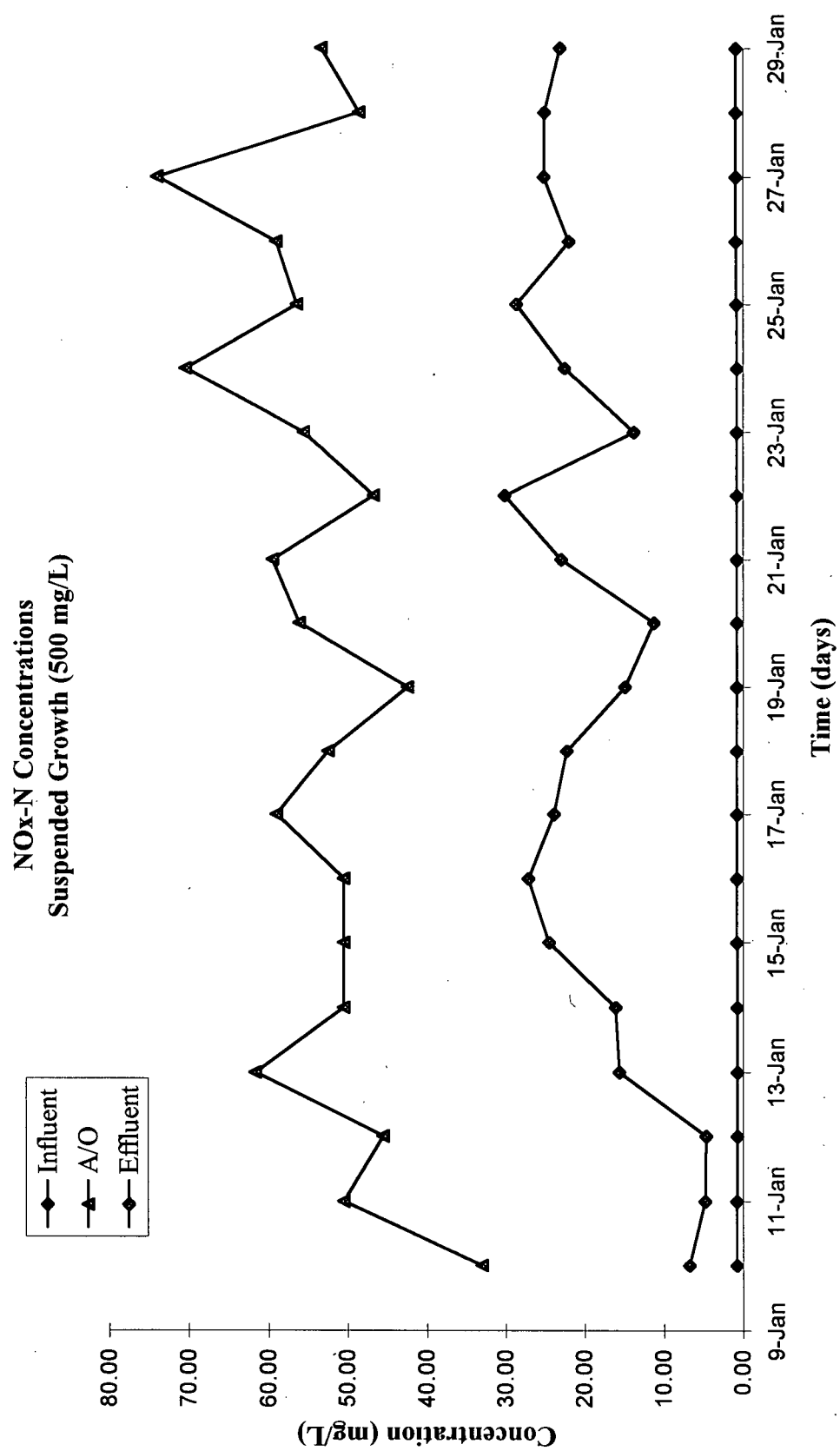
Figure 5. 75 NO$_x$-N profile with time for the 500 mg/L run (suspended growth system)

reactor was approximately 55%. This percentage was lower than the AGS. As mentioned before, the lack of carbon supplied by the influent was the downfall for this run. The bacteria were using endogenous respiration throughout the cycle to keep themselves alive. Therefore when the bacteria were transferred to the anoxic reactor, they were not as healthy as in the AGS because they had exhausted much of their stored carbon. The AGS had a larger population of bacteria within the A/O reactor that could carry out reactions using endogenous respiration. Therefore, when the sludge was wasted from the A/O, there would be a higher proportion of bacteria that still hadn't exhausted their stored carbon, which subsequently could be used for denitrification in the anoxic reactor. Comparing the two track analysis on the anoxic reactor in figures 5. 76 and 5. 77, it can be seen that the first track analysis, which corresponds to the first track analysis for the A/O reactor, removed nitrate at a slower rate. That could be the result of having bacteria that had less stored carbon within them to carry out denitrification.

The time profile for phosphorus is shown in figure 5. 78. The phosphorus removal efficiency was approximately 12%. This shouldn't come as a surprise as there wasn't enough carbon in the system for phosphorus release during the anoxic phase. The lack of production in VFA, a slow nitrate removal rate, and an insufficient carbon supply during the anoxic phase caused a minimal phosphorus release. Subsequently that would affect the uptake rate by the bacteria in the aerobic stage. It is shown that the additional biomass in the AGS did not affect the rate of removal for phosphorus as the treatment efficiencies for these two systems were approximately equal.

The solids levels in the influent, A/O effluent, and final effluent are represented in figures 5. 79 and 5. 80. The average TS, TVS, TSS, and TVSS removal efficiencies were

Nutrients Profiles of Anoxic Reactor

Figure 5. 76 First track analysis of nutrients through one cycle for the anoxic reactor

Nutrients Profiles of Anoxic Reactor

Figure 5. 77 Second track analysis of nutrients through one cycle for the anoxic reactor

PO4-3 Concentrations
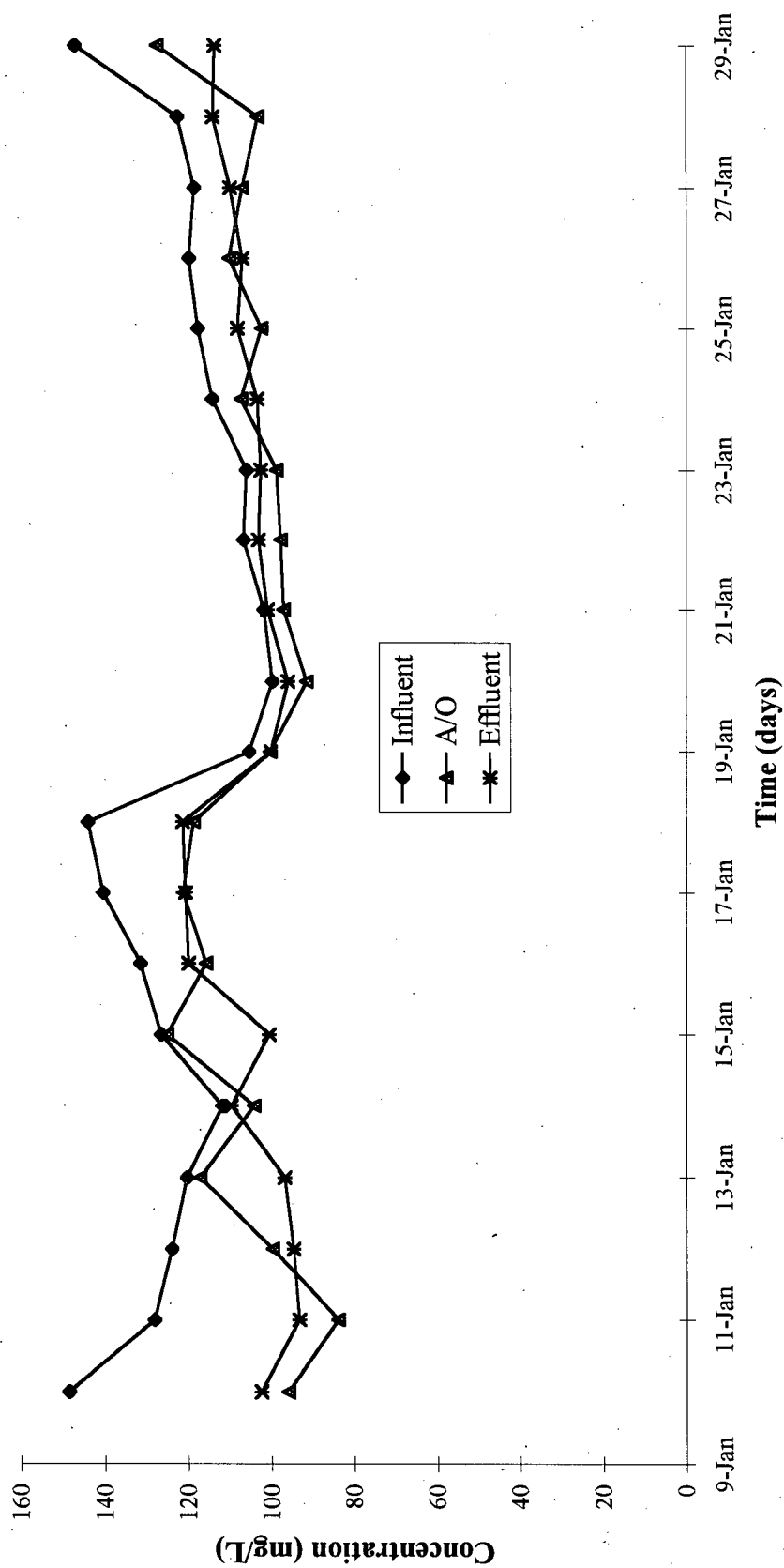**Suspended Growth (500 mg/L)**

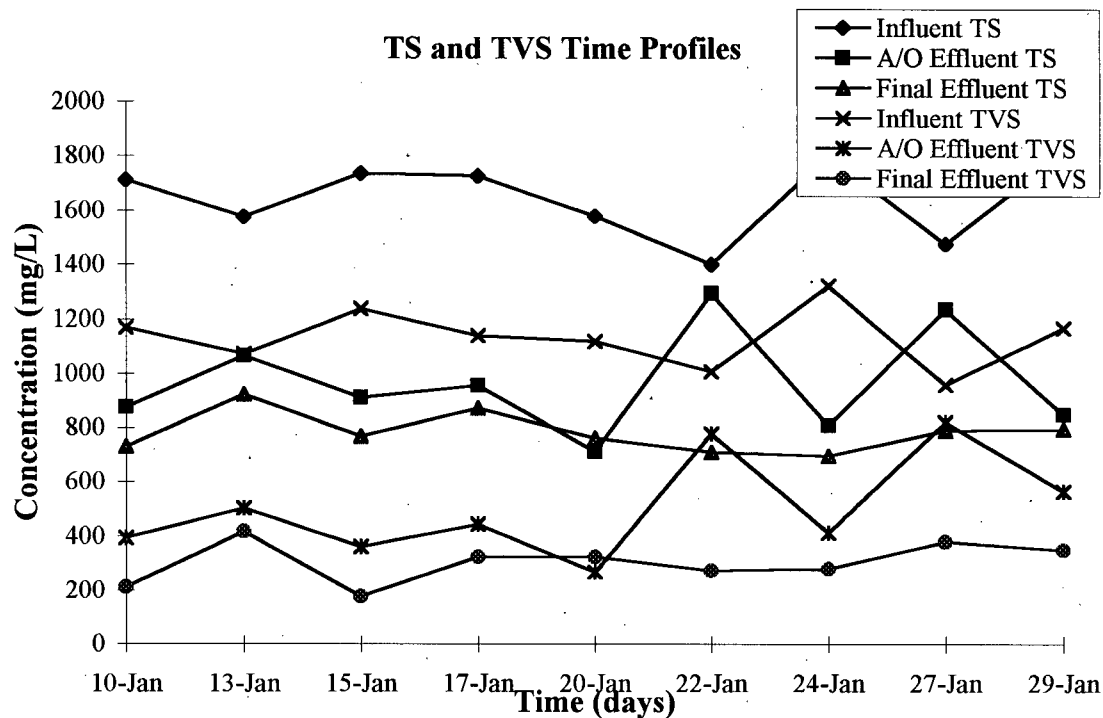**Figure 5. 78 PO$_4$$^{-3}$ profile with time for the 500 mg/L run (suspended growth system)**
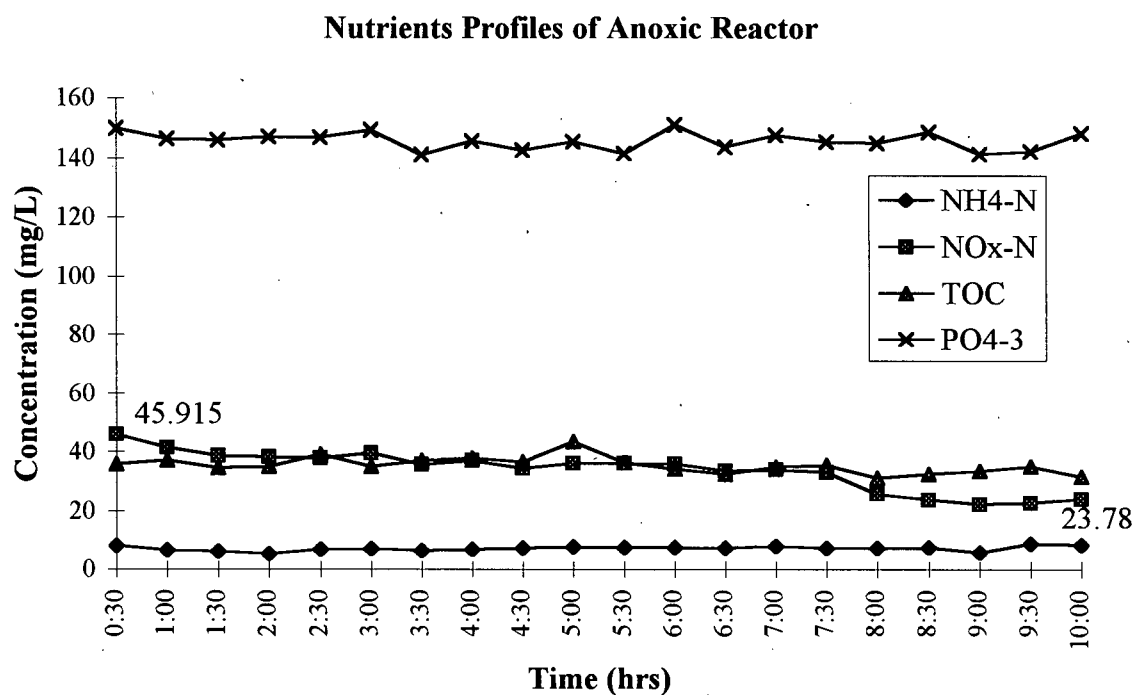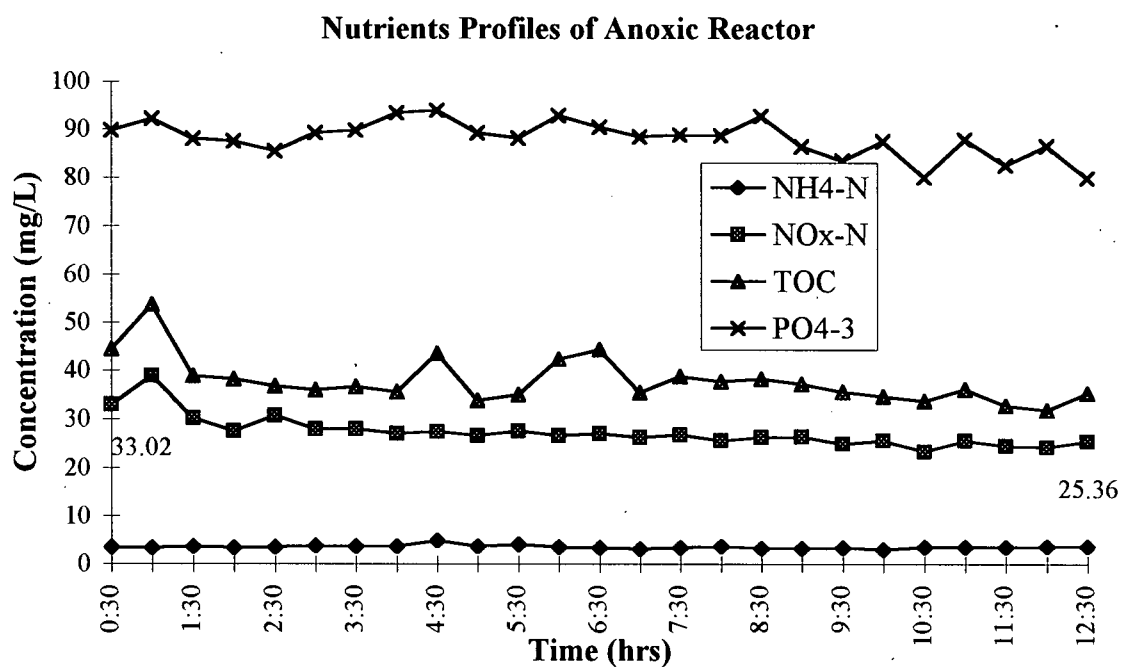
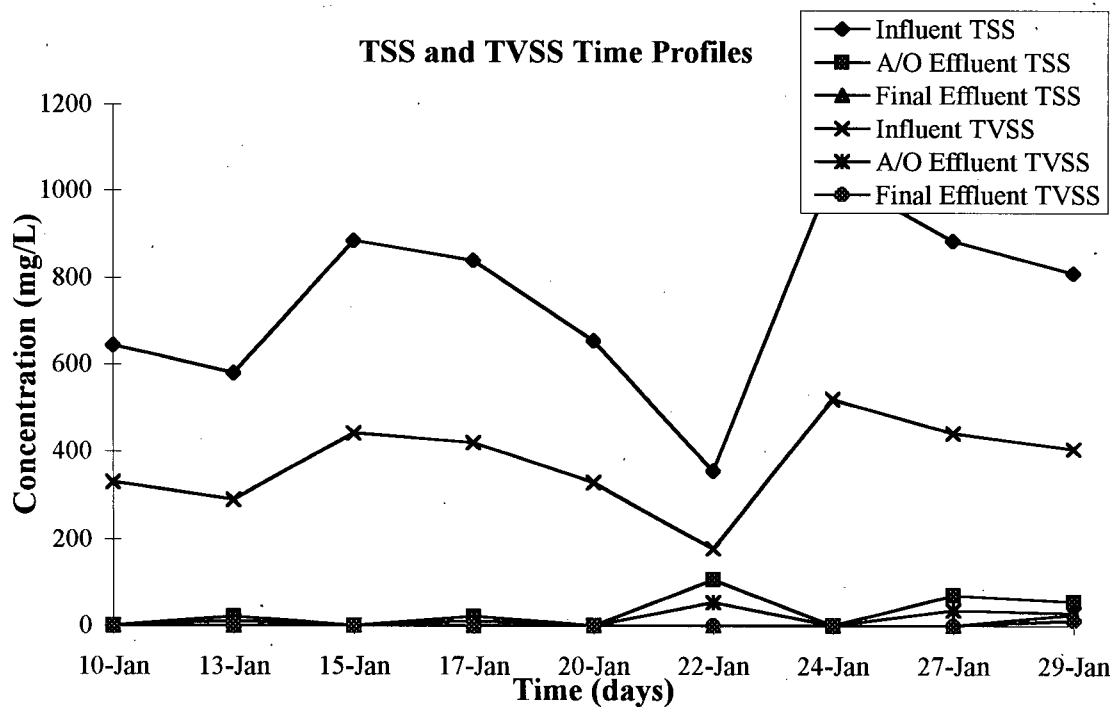**Figure 5. 79 TS and TVS profiles with time for the 500 mg/L run (suspended growth system)**

Figure 5. 80 TSS and TVSS profiles with time for the 500 mg/L run (suspended growth system)

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52%, 72%, 99%, and 99% respectively. These results were approximately the same as the AGS. The slightly lower removal efficiency for TS and TVS compared to the 2,000 mg/L and 4,000 mg/L runs was due to the high proportion of inorganic solids in the wastewater. The influent BOD_5/COD ratio of 0.28 suggested a high proportion of inorganic solids in the wastewater that could not be removed by a biological process.

5. 4 Sludge

The sludge that was wasted from the A/O and anoxic reactor was tested to determine its nitrogen and phosphorus content. All the sludge tested had a very similar average in %N, regardless of the influent concentration. The average value of %N in the sludge was 6.36% ± 0.35%. The nitrogen content was slightly lower than that observed by Turk (1986) of 9.3 to 12% (based on MLVSS) in systems treating a high strength ammonia waste, but very comparable to the UBC BPR pilot plant (6-7% gN/gMLSS) (Hong, 1997). Even though the %N in the sludge did not increase with a higher nitrogen influent concentration, the amount of sludge produced in the system increased. In another words, sludge was wasted more often from the A/O reactor when the organic loading was high in order to maintain a 10 days SRT within the reactor (TOC = 4,000 mg/L > 2,000 mg/L > 500 mg/L). The higher sludge production rate was the cause of a faster metabolism rate by the bacteria in order to consume the increased levels of nutrients in the reactor. This increased the rate at which bacteria reproduces itself. There were no significant differences between the two systems tested.

The phosphorus content taken up from the solution during the aerobic phase was stored in the sludge and physically removed from the system through wasting. Therefore, the phosphorus content in the sludge is a good indication of the extent of BPR. The %P in the sludge varied slightly for different runs. The %P in the sludge from the A/O reactor was approximately 3.03, 3.34, and 3.65% for the 500 mg/L, 2,000 mg/L, and 4,000 mg/L run, respectively. These results indicate that more phosphorus was absorbed by the bacteria during the aerobic phase for the runs with the higher loadings. This confirmed the results that were previously discussed. A sufficient supply of carbon in the wastewater promotes phosphorus release, which leads to a higher absorption rate by the bacteria during the aerobic phase. Also, more sludge was produced during the higher influent concentration runs. The combination of the above led to an overall increase of phosphorus content in the sludge. There was no major differences between the AGS and SGS.

5. 5 Statistical Analysis

Statistical analysis was done to determine if there were any significant differences between the results from the attached growth and suspended growth systems. The student's t-test was used to examine the relationships between the two systems. The paired t-test was used to compare the process performance between the two systems at the same influent concentrations. The paired t-test is designed to factor out variations in the experimental material. The influent wastewater, the experimental material, was very likely to vary daily. But since the two systems were operated in parallel, treating a

common wastewater, both systems were subject to the same daily fluctuations. Therefore, the comparisons on process performance between the two systems were not subject to the influence of the variations in the feed. However, the t-test using unequal variances was used for the comparison among the three different runs for each system. The changes in influent concentration were relatively large and had a major effect on the process performance. Therefore it would be difficult to use the paired t-test to determine the differences among the different runs. The following table represented the results of the statistical analysis using a 95% confidence level.

As shown in the following tables, the difference in removal percentage between the two systems of the same run is frequently very small. Some parameters exhibited no significant difference between the AGS and SGS. The difference of variance between the two systems on some of the parameters might have caused the paired t-test to reject the null hypothesis (AGS equal to SGS) and accept the alternative hypothesis (AGS $\neq$ SGS).

A comparison between different concentrations of the same system frequently resulted in significant differences between the two runs. This does not come as a surprise as the influent concentration was very different. The large t-value suggested that the null hypothesis of the three different runs being equalled was rejected. Instead, the researcher's hypothesis that one run was different than the others was accepted.

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Table 5. 7 The results of the paired t-test on the removal efficiency between the two different systems for the TOC = 2,000 mg/L run

	TOC = 2,000 mg/L Treatment efficiency after the A/O reactor		TOC = 2,000 mg/L Treatment efficiency after the anoxic reactor	
	Difference in Removal % between AGS and SGS	t-value/ Significant?	Difference in Removal % between AGS and SGS	t-value/ Significant?
NH_4-N	-0.04%	-0.88/No	0.48%	2.60/Yes
TOC	-0.14%	-0.39/No	0.47%	1.86/Yes
PO_4^{-3}	0.74%	0.41/No	1.8%	1.06/No
NO_x-N	-	-	-10.55%	-1.54/No

Table 5. 8 The results of the paired t-test on the removal efficiency between the two different systems for the TOC = 4,000 mg/L run

	TOC = 4,000 mg/L Treatment efficiency after the A/O reactor		TOC = 4,000 mg/L Treatment efficiency after the anoxic reactor	
	Difference in Removal % between AGS and SGS	t-value/ Significant?	Difference in Removal % between AGS and SGS	t-value/ Significant?
NH_4-N	-0.04%	-1.00/No	-0.75%	-5,000,000/Yes
TOC	-0.63%	-5.69/Yes	-0.25%	-2.27/Yes
PO_4^{-3}	1.83%	4.29/Yes	-0.24%	0.49/No
NO_x-N	-	-	24.08%	3.73/Yes

Table 5. 9 The results of the paired t-test on the removal efficiency between the two different systems for the TOC = 500 mg/L run

	TOC = 500 mg/L Treatment efficiency after the A/O reactor		TOC = 500 mg/L Treatment efficiency after the anoxic reactor	
	Difference in Removal % between AGS and SGS	t-value/ Significant?	Difference in Removal % between AGS and SGS	t-value/ Significant?
NH_4-N	0.76%	0.80/No	0.13%	5,000,000/Yes
TOC	0.81%	5.70/Yes	1.6%	2.27/Yes
PO_4^{-3}	-0.59%	-1.12/No	-2.57%	-0.21/No
NO_x-N	-	-	15.07%	1.22/No

Table 5. 10 The results of the independent t-test on the removal efficiency between two different runs for the AGS

	Attached Growth System Treatment efficiency after the A/O reactor		Attached Growth System Treatment efficiency after the anoxic reactor	
	Difference in Removal % between 2,000 mg/L run and 4,000 mg/L run	t-value/ Significant?	Difference in Removal % between 2,000 mg/L run and 4,000 mg/L run	t-value/ Significant?
NH_4-N	0.1%	2.51/Yes	0.78%	5,000,000/Yes
TOC	1.43%	4.84/Yes	-0.98%	4.68/Yes
PO_4^{-3}	-49.85%	-5,000,000/Yes	-48.12%	-5,000,000/Yes
NO_x-N	-	-	-6.64%	1.50/No

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Table 5. 11 The results of the independent t-test on the removal efficiency between two different runs for the SGS

	Suspended Growth System Treatment efficiency after the A/O reactor		Suspended Growth System Treatment efficiency after the anoxic reactor	
	Difference in Removal % between 2,000 mg/L and 4,000 mg/L run	t-value/ Significant?	Difference in Removal % between 2,000 mg/L run and 4,000 mg/L run	t-value/ Significant?
NH_4-N	0.1%	2.51/Yes	-0.45%	2.79/Yes
TOC	-1.92%	-7.15/Yes	-1.7%	-5,000,000/Yes
PO_4^{-3}	-48.76%	-5,000,000/Yes	-50.16%	-5,000,000/Yes
NO_x-N	-	-	27.99%	4.49/Yes

Table 5. 12 The results of the independent t-test on the removal efficiency between two different runs for the AGS

	Attached Growth System Treatment efficiency after the A/O reactor		Attached Growth System Treatment efficiency after the anoxic reactor	
	Difference in Removal % between 2,000 mg/L and 500 mg/L run	t-value/ Significant?	Difference in Removal % between 2,000 mg/L run and 500 mg/L run	t-value/ Significant?
NH_4-N	0.23%	1.29/No	-0.13%	-5.05/Yes
TOC	2.00%	4.73/Yes	1.47%	4.87/Yes
PO_4^{-3}	36.54%	5,000,000/Yes	36.94%	5,000,000/Yes
NO_x-N	-	-	-16.71%	-3.87/Yes

Table 5. 13 The results of the independent t-test on the removal efficiency between two different runs for the SGS

	Suspended Growth System Treatment efficiency after the A/O reactor		Suspended Growth System Treatment efficiency after the anoxic reactor	
	Difference in Removal % between 2,000 mg/L and 500 mg/L run	t-value/ Significant?	Difference in Removal % between 2,000 mg/L run and 500 mg/L run	t-value/ Significant?
NH$_4$-N	1.54%	5,000,000/Yes	0%	0.70/No
TOC	2.95%	5,000,000/Yes	2.6%	6.71/Yes
PO$_4$$^{-3}$	35.8%	5,000,000/Yes	35.14%	5,000,000/Yes
NO$_x$-N	-	-	9.71%	0.73/No

Table 5. 14 The results of the independent t-test on the removal efficiency between two different runs for the AGS

	Attached Growth System Treatment efficiency after the A/O reactor		Attached Growth System Treatment efficiency after the anoxic reactor	
	Difference in Removal % between 4,000 mg/L and 500 mg/L run	t-value/ Significant?	Difference in Removal % between 4,000 mg/L run and 500 mg/L run	t-value/ Significant?
NH$_4$-N	0.09%	1.29/No	-0.43%	-5.08/Yes
TOC	3.43%	5,000,000/Yes	2.45%	5,000,000/Yes
PO$_4$$^{-3}$	86.39%	5,000,000/Yes	85.06%	5,000,000/Yes
NO$_x$-N	-	-	-10.07%	-2.13/Yes

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Table 5. 15 The results of the independent t-test on the removal efficiency between two different runs for the SGS

	Suspended Growth System Treatment efficiency after the A/O reactor		Suspended Growth System Treatment efficiency after the anoxic reactor	
	Difference in Removal % between 4,000 mg/L and 500 mg/L run	t-value/ Significant?	Difference in Removal % between 4,000 mg/L run and 500 mg/L run	t-value/ Significant?
NH_4-N	1.44%	5,000,000/Yes	0.45%	3.48/Yes
TOC	4.87%	5,000,000/Yes	4.3%	5,000,000/Yes
PO_4^{-3}	83.97%	5,000,000/Yes	82.73%	5,000,000/Yes
NO_x-N	-	-	-18.28%	6.00/Yes

CHAPTER 6

CONCLUSIONS AND RECOMMENATIONS

6. 1 Conclusions

The main objective of this research was to investigate the possibility of using pH as a real-time control parameter in swine wastewater treatment. The reproducibility of the pH patterns was tested under three different organic loadings. Consistent patterns of pH must be observed under all three conditions in order to designate this research a success. The effectiveness of pH as a control parameter is also measured by the treatment removal efficiency for pollutants such as carbon, nitrogen, and phosphorus. The pH pattern was compared to the more established ORP pattern under the three different loadings for the purpose of cross-referencing. The preference for which control parameter to use for real-time process control was thereby evaluated. The differences between an attached growth system using Ringlace as a medium and a suspended growth system, both using the same real-time control process, were also examined.

The real-time control process was implemented into a bench scale two stage sequencing batch reactor (TSSBRs). The two systems, attached growth and suspended growth, were operated at a 10 days SRT at room temperature. The three different influent concentrations that were used in this research were based on the TOC concentration of 500 mg/L, 2,000 mg/L, and 4,000 mg/L. Two litres of fresh wastewater with their

designated concentration were fed into the reactor every cycle. Based on the results of the experimental studies, the following conclusions are made.

1) pH can be a very effective control parameter. The pH pattern was uniform and consistent most of the time, despite the fluctuations of influent concentrations. The results obtained from this research clearly demonstrated the ability of pH to monitor all three respiratory states, namely, aerobic, anoxic, and anaerobic. The nitrate apex was found to represent a complete removal of nitrate in the anoxic phase, while the nitrogen break point and the residual carbon manipulation point represented a complete removal of ammonia and carbon respectively during the aerobic phase. The clarity of each feature makes real-time control using pH possible.

2) The real-time control process was able to provide a flexible HRT for the system. The advantages of using a real-time control process are shown here. The use of real-time control will either ensure a complete removal of certain pollutants or reduce the overall cycle time, while maximizing energy savings. Both the attached growth and suspended systems were able to produce a constant effluent quality, despite the fluctuations in the quality of the influent for each run.

3) The control strategy developed for this research ensures a complete removal of carbon and ammonia, regardless of the influent concentrations. The quality of phosphorus and nitrate in the effluent will depend on the organic loading from the wastewater.

4) The treatment efficiency of each nutrient can be affected greatly by the organic loading rate. It was found that the optimal treatment efficiency for each nutrient was achieved during the 4,000 mg/L run. A sufficient supply of carbon can promote rapid

denitrification and primary phosphorus release during the anoxic phase, while creating a favourable environment for nitrification, carbon oxidation, and phosphorus uptake in the aerobic phase.

5) An anoxic reactor is not required when the organic loading from the influent is high. The high carbon content in the influent creates simultaneous nitrification/denitrification during the aerobic phase. This reduces the nitrate loading into the anoxic reactor. The nitrate concentration in the A/O effluent was usually in the range of 5 - 15 mg/L.

6) The RCMP does not represent complete removal of carbon under a low organic loading situation. The low carbon content in the influent caused much of the bio-degradable materials to be consumed earlier than usual. The RCMP was shifted before the NBP. It is therefore concluded that the control point should be set at the NBP if the C/N ratio is low, while the RCMP should be used as the control point while the C/N ratio is high.

7) When the system is operated under an inadequate supply of carbon from the wastewater, sludge wasted from the A/O reactor must be added to the anoxic reactor constantly to enhance the denitrification rate.

8) The pH pattern correlated very well with the ORP pattern. All the distinct points identified on the pH curve such as the nitrate knee, nitrogen break point, and residual manipulation break point coincided with the ORP pattern. It can be concluded that both ORP and pH can be used as a real-time control parameter.

9) The pH pattern gives out more meaningful information than the ORP pattern, especially when it relates to alkalinity. The changes in pH are well defined by denitrification and nitrification reactions, while the change in ORP cannot be directly

related to the process changes. pH also was more stable under different circumstances. The exact pH value does not fluctuate a great deal. The ORP pattern, on the other hand, is influenced by DO, solids concentration, and many other factors.

10) When the nitrogen break point disappears on the pH curve (see both 500 mg/L section), the computer was still able to recognize the control point under the current control strategy. It should be noted that this did not occur frequently. On the other hand, the NBP on the ORP curve vanishes regularly under excessive aeration. That would lead to a mis-control if the ORP was the control parameter. It can be concluded that pH may be a better controlling parameter if aeration rate is not controlled.

11) The only major difference found between the attached growth system and suspended growth system was the hydraulic retention time. The attached growth system had a shorter hydraulic retention time due to the higher biomass concentration in the reactor. The treatment efficiency of all the nutrients between the two systems were virtually the same.

6. 2 Recommendations

Further research work is recommended in the following areas.

1) The possibility of using pH and ORP as control parameters co-currently should be investigated. This will ensure no mis-control in the real-time process as each parameter will be backed up by each other.

2) Investigations should be done on loading rate control to ensure an adequate supply of carbon into the system. This will assist the system into achieving an optimal treatment efficiency for all the nutrients.

3) Further study on the possibility of using the NBP as the control point should be carried out. More soluble carbon will then be allowed to transfer over to the anoxic reactor for denitrification.

4) The possibility of reducing the system to a one reactor SBR should be investigated. This will reduce the capital cost for the entire system.

5) Further study should be conducted on the effect of aeration rate on the system. Different aeration rates should be applied for each concentration of influent in order to examine the response of the pH and ORP curves.

6) The possibility of applying an external carbon substrate to ensure complete nitrate and phosphorus removal during low organic loading situations should be investigated.

7) Pilot or full scale studies should be conducted to validate the results found in this research.

8) As mentioned before, wastewater on the farm carries disinfectants. The presence of disinfectants can damage the treatment efficiency of the system. Investigations should be conducted on the effect of antibiotics on the treatment system's performance.

CHAPTER 7

REFERENCES

Al-Ghusain, I. A., Huang, J., Hao, O. and Lim, B. S., 1995, Use of pH as control parameter for aerobic/anaerobic sludge digestion. Journal of Environmental Engineering ASCE, 121(3): pp. 225-235

American Public Health Association (A.P.H.A.), 1995, Standard methods from the examination of water and wastewater, 19th Ed., Washington, D. C.

Arun, V., Mino, T., 1989. Metabolism of carboxlylic acids located in and around the glycolytic pathway and the TCA cycle in the biological phosphorus removal process. Water science technology, 21(4/5): 363-374

Barnard, J. L., 1994. Alternative prefermentation systems. Proceedings of the Conference Seminar on Use of Fermentation to Enhance Biological Nutrient Removal. 67th Annual Water Environment Conference and Exposition. Chicago, Illinois, U. S. A., Oct. 15, pp. 13-21.

Benefield L. D., 1977. Evaluation of a comprehensive kinetic model for the activated sludge process. Journal of water pollution control federation, 49: pp. 1636-1641.

Burton C. H. (1992). A review of the strategies in the aerobic treatment of pig slurry: purpose, theory and method. Journal of agricultural engineering research, 53: pp. 249.

Charpentier, J., Florentz, M. and David, G., 1987. Oxidation-reduction potential (ORP) regulation: A way to optimize pollution removal and energy saving in the low load activated sludge process. IAWPRC, Rio, Water science technology, 19 (3/4): pp. 45-655

Charpentier, J., Godart, H., Martin, G. and Mogno, Y., 1989. Oxidation-reduction potential (ORP) regulation as a way to optimize aeration and C, N, and P removal: experimental basis and various full-scale examples. IAWPRC, Brightion, Water science technology 21(10/11): pp.1209-1223.

Comeau, Y., Hall, K. J., Hancock, R. E. W., and Oldham, W. K., 1986. Biochemical model for enhanced biological phosphorus removal. Water research, 20: pp. 1511-1521.

Dague, R. R. and Pidaparti, A. R., 1992. Anaerobic sequencing batch reactor treatment of swine wastes. 46th Purdue industrial waste conference proceedings. Lewis Publishers, Inc., Chelsea, Michigan 48118, pp. 751-760.

Environmental Protection Agency U. S., 1993. Design manual for nitrogen removal. Office of Research and Development, Office of Wastewater Enforcement and Compliance, Washington D. C., EPA/625/R-93/010.

Environmental Protection Agency U. S., 1987. Design manual for phosphorus removal. Office of Research and Development, Water Engineering Research Laboratory, Cincinnati, OH, 45268, EPA/625/1-67/001.

Fernandes F. L. (1988). An integrated treatment system for liquid swine manure. Ph.D dissertation. McGill University, Ste. Anne de Bellevue, Quebec, Canada.

Fernandes L., McKyes E., 1991. Theoretical and experimental study of a sequencing batch reactor treatment of liquid swine manure. ASAE transaction, 34(1-2): pp. 597-602.

Fernandes L., McKyes E., Warith, M., and Barrington, S., 1991. Treatment of liquid swine manure in the sequencing batch reactor under aerobic and anoxic conditions. Canadian agricultural engineering, 33: 373-379.

Fernandes L., 1994. Effect of temperature on the performance of an SBR treating liquid swine manure. Bioresource technology, 47: pp. 219-227.

Garzon-Zuniga, A. A. and Gonzalez-Martinez, S., 1996. Biological phosphate and nitrogen removal in a biofilm sequencing batch reactor. Water science technology, 34(1/2): pp. 293-301.

Gerber A., Mostert, E. S., Winter C. T., and de Villiers, R. H., 1986. The effect of acetate and other short-chain carbon compounds on the knietics of biological nutrient removal, Water SA, 12: pp. 7-12.

Gerber A., de Villiers, R. H., Mostert, E. S., and van Riet, C. J., 1987. The phenomenon of simultaneous phosphate uptake and release, and its importance in biological nutrient removal. In Biological Phosphate Removal from Wastewaters (Edited by Ramadori, R.). Pergamon Press, Oxford.

Hang-Sik, S., Hang-Bae, J., Eung-Tak, L., Sang-Min, L., and Kyeong, H., 1993. Optimal operating conditions for nutrient removal in the automatic controlled sequencing batch reactor. In: 2nd International Conference on Design and Operation of Small Wastewater Treatment Plants. Ødegaard H. (Ed.), Tapir Publishers, Trondheim, Norway, pp. 253-259.

Hascoet, M. C., Florent, M., and Granger, P., 1984. Biochemical Aspects of Biological Phosphorus Removal from Wastewater. Enhanced Biological Phosphorus Removal from Wastewater, Vol 1., IAWPRC Post Conference Seminar, pp. 33, September 24, 1984, Paris, France.

Hascoet, M. C. and Florentz M., 1985. Influence of nitrate on biological phosphorus removal from wastewater. Water S A, 11: pp. 1-8

Heduit A., and Thevenot D. R., 1992. Elements in the interpretation of platinum electrode potentials in biological treatment. Water science technology, 26(5/6): pp. 1335-1344.

Hong, W. Z., 1997, Oxidation-reduction potential and organic carbon sources as two control parameters for simultaneous nitrification and denitrification nutrient removal processes, PhD Dissertation, University of British Columbia, Vancouver, B. C., Canada, pp.1-244.

Ishizaki, A., Shibai, H., and Hirose, Y., 1974. Basic aspects of electrode potential change in submerged fermentation. Journal of agriculture and biological chemistry, 38(12): pp. 2399-2406.

Iwema, A., and Meunier A., 1985. Influence of nitrate on acetic acid induced biological phosphate removal. Water science technology, 17(11): pp. 289-294.

Jenkins, C. J. and Mavinic, D. S., 1989a. Anoxic-aerobic digestion of waste activated sludge: Part 1 - solids reduction and digested sludge characteristics. Environmental technology letter, 10: pp. 355-370.

Jenkins, C. J. and Mavinic, D. S., 1989b. Anoxic-aerobic digestion of waste activated sludge: Part 2 - Supernatant Characteristics, ORP monitoring results and overall rating system. Environmental technology letter, 10: pp. 371-384.

Kerrn-Jespersen J. P. and Henze, M., 1993. Biological phosphorus uptake under anoxic and aerobic conditions. Water research, 27: pp. 617-624.

Kjaergaard, L., 1977. The redox potential: its use and control in biotechnology. In advances in biochemical engineering, Ed. by Ghose, T. K., Fiechter, A. and N. Blakebrough , Springer - Verlag, Berlin, pp. 131.

Koch, F. A. and Oldham, W. K., 1985. Oxidation -reduction potential - A tool for monitoring, control and optimization of biological nutrient removal system". IAWPRC, Paris. Water science technology, Paris 17: pp. 259-279.

Koch, F. A., Oldham, W.K. and Wang, H. Z., 1988. ORP as a tool for monitoring and control in bio-nutrient removal system. In proceedings of joint ASCE/CSCE environmental engineering conference, July 12-19, Vancouver, B.C., pp.162-170.

Labtech Manual, 1994. Laboratory Technology Co., Wilminton, MA, USA.

Lie, E., Christensson, M., Jonsson, K., Ostgaard, K., Johansson, P., and Welander, T., 1997. Carbon and phosphorus transformations in a full-scale enhanced biological phosphorus removal process. Water research, 31(11): pp. 2693-2698.

Liu, J. X., van Groenestijn, J. W., Doddema, H. J., and Wang, B. Z., 1996. Removal of nitrogen and phosphorus using a new biofilm activated sludge system. Water science technology, 34(1/2): pp. 315-322.

Lo, C. K., Yu, C. W., Tam, N.F.Y. and Traynor, S., 1994. Enhanced nutrient removal by oxidation-reduction potential (ORP) controlled aeration in a laboratory scale extended aeration treatment system. Water research, 28(10): pp. 2087-2094.

Lo K. V., Liao P.H. and Kleeck van R.J. (1991). A full-scale sequencing batch reactor treatment of dilute swine wastewater. Canadian Agricultural Engineering, 33: pp. 193-195.

Lotter, L. H., and Murphy, M., 1985. The identification of heterotrophic bacteria in an activated sludge plant with particular reference to polyphosphate accumulation. Water SA, 11(4): pp. 179-184.

Malnou, L. H., Meganck, M., Faup, G. M., and du Rostu, M., 1984. Biological phosphorus removal: study of the main parameters. Water science technology, 16(10/11): pp. 173-185

Manning, J. F. and Irvine, R. L., 1985. The biological removal of phosphorus in a sequencing batch reactor. Journal of water pollution control federation, 57(1): pp. 87

McCarty, P. L., Beck, L., St. Amant, P., 1969. Biological denitrification of wastewater by addition of organic material. Proceedings of the 24th Industrial Waste Conference, Purdue University, pp. 1271-1285.

Metcalf and Eddy., 1991., Wastewater engineering: treatment, disposal and reuse/ 3rd. McGraw-Hill, Inc.

Matsuda, A., Tetsuo, T., and Fujii, S., 1988. Behavior of nitrogen and phosphorus during batch aerobic digestion of waste activated sludge - continuous aeration and intermittent aeration by control of DO. Water research, 22(12): pp. 1495-1501.

Munoz-Colunga, A. and Gozalez-Martinez, S., 1996. Effects of population displacements on biological phosphorus removal in a biofilm SBR. Water science technology, 34(1/2): pp. 303-313.

Osborn, D. W. and Nicholls, H. A., 1978. Optimization of the activated sludge process for the biological removal of phosphorus. Prog. Wat. Technol., 10: pp. 261-277.

Peddie, C. C., Koch, F. A., Jenkins, C. J., Mavinic, D. S., 1988. ORP as a too for monitoring and control of SBR systems for aerobic sludge digestion. In proceedings of joint ASCE/CSCE environmental engineering conference, July 12-19, Vancouver, B. C., pp. 171-178.

Peddie, C. C., Mavinic, D. S. and Jenkins C. J., 1990. Use of ORP for monitoring and control of aerobic sludge digestion. Journal of environmental engineering, 116(3): pp. 461-471.

Plisson-Saune, S., Capdeville, B., Mauret, M., Deguin, A., and Baptiste, P., 1996. Real-time control of nitrogen removal using three ORP bending points: signification, control strategy and results. Water Science Technology, 33(1): pp. 275-280.

Ra, C. S., 1997. Oxidation Reduction Potential (ORP) and pH as a Real-Time Control Parameter in Swine Manure Treatment Process, PhD Dissertation, University of British Columbia, Vancouver, B. C., Canada, pp.1-199.

Ra, C. S., Lo, K. V. and Mavinic, D. S., 1998, Real-time control of two-stage sequencing batch reactor system for the treatment of animal wastewater, Environmental Technology Letters, 19: pp. 343-356

Radjai, M., Hatch, R. T., and Cadman, T. W., 1984. Optimization of amino acid production by automatic self-tuning digital control of redox potential, biotechnology and bioengineering symp. No. 14, John Wiley & Sons, Inc.

Randall, A. A., Benefield, L. D., and Hill, W. E., 1997. Induction of phosphorus removal in an enhanced biological phosphorus removal bacterial population. Water Research, 31(11): pp. 2869-2877.

Sasaki, K., Yamamoto, Y., Tsumura, K., Sachiko, O., and Mori, Y., 1996. Development of 2-reactor intermittent aeration activated sludge process for simultaneous removal of nitrogen and phosphorus. Water science technology, 24(1/2): pp. 111-118

Schmidt, C. G. and Dague, R. R., 1993. Anaerobic sequencing batch reactor treatment of swine wastes at 20°C, 25°C, and 35°C. 48th Purdue industrial waste conference proceedings. Lewis Publishers, Chelsea, Michigan 48118, pp. 541-549.

Sen, D., Farren G. D., Rhodes R., Copithorn, and Randall C. W., 1993. Full scale evaluation of nitrification and denitrification on fixed film media (Ringlace) for design of single sludge nitrogen removal system. Anaheim Conference, pp. 137-148.

Turk, O., 1986. The Feasibility of a Shortened Pathway for Nitrogen Removal from Highly Nitrogenous Wastes. Ph. D. Dissertation, University of British Columbia, Vancouver, B. C.

Suresh, N., Warburg, M., Timmerman, M., Wells, J., Coccia, M., Roberts, M. F., and Halvorson, H. O. New Strategies for the Isolation of Microorganisms Responsible for Phosphate Accumulation Enhanced Biological Phosphorus Removal from Wastewater, Vol. 1., IAWPRC Post Conference Seminar, pp. 131, September 24, 1984, Paris, France.

Wareham, D. G., Mavinic D. S., and Hall K. J., 1994. Sludge digestion using ORP-regulated aerobic-anoxic cycles. Water research, 28(2): pp. 373-384.

Yu, R. F., Liaw, S. L., Chang, C. N., Lu, H. J., and Cheng, W. Y., 1997. Monitoring and control using on-line ORP on the continuous flow activated sludge batch reactor system. Water Science Technology, 35(1): pp. 57-66.

APPENDIX

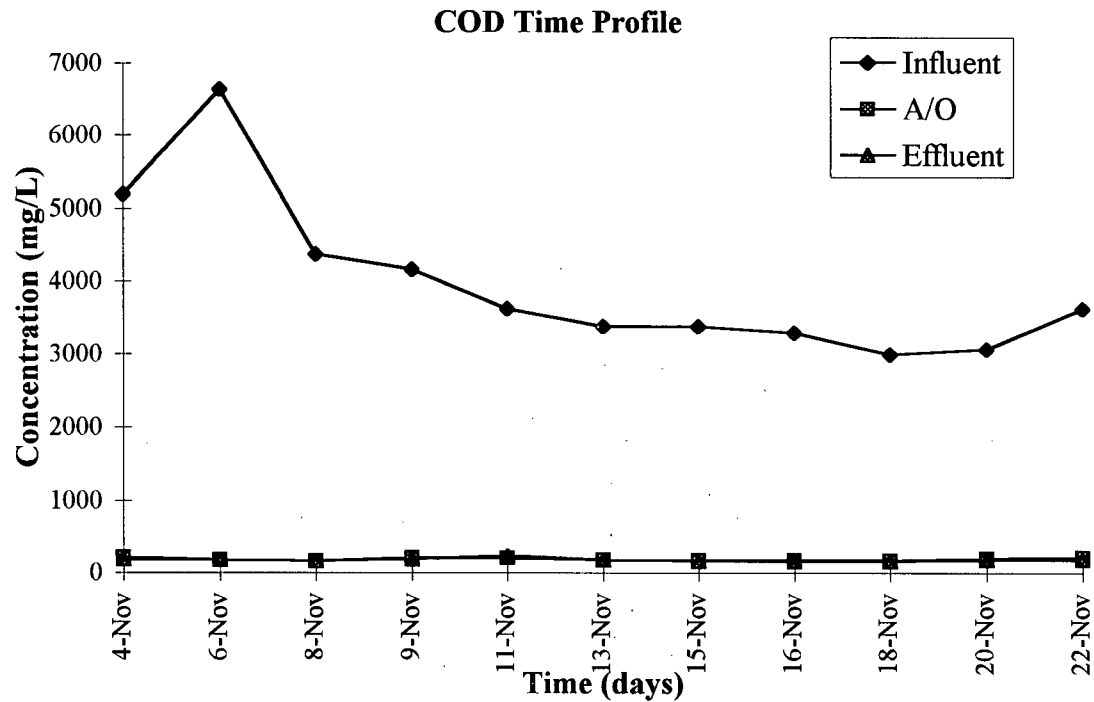Figure A1 BOD$_5$ time profile for the 2,000 mg/L run (attached growth system)

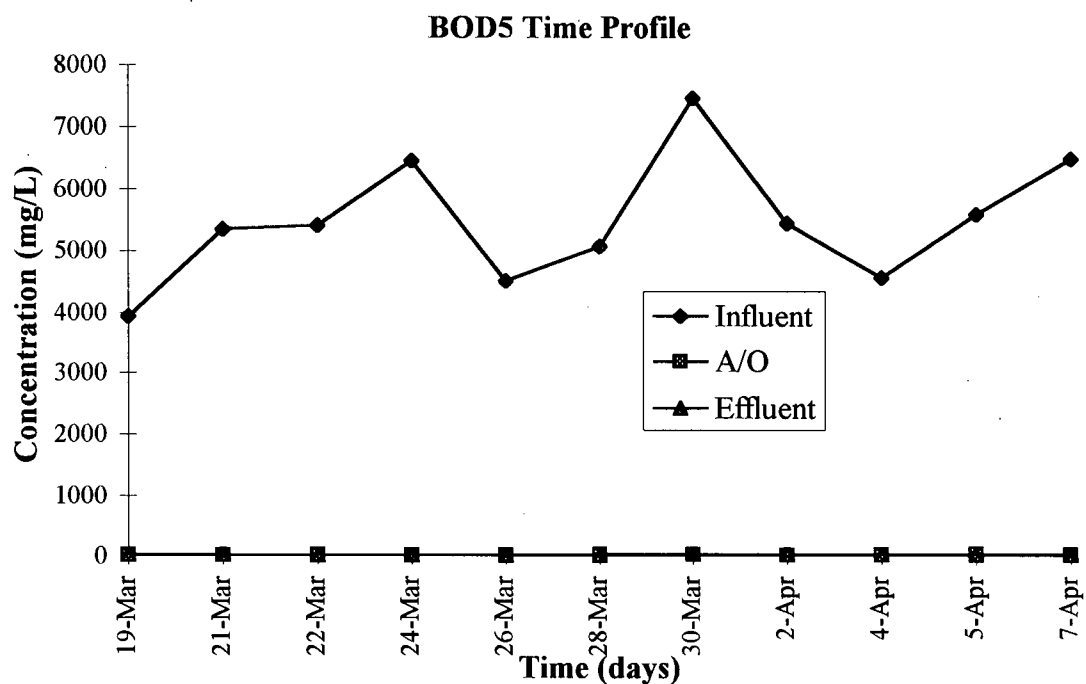Figure A2 COD time profile for the 2,000 mg/L run (attached growth system)

Figure A3 BOD$_5$ time profile for the 4,000 mg/L run (attached growth system)
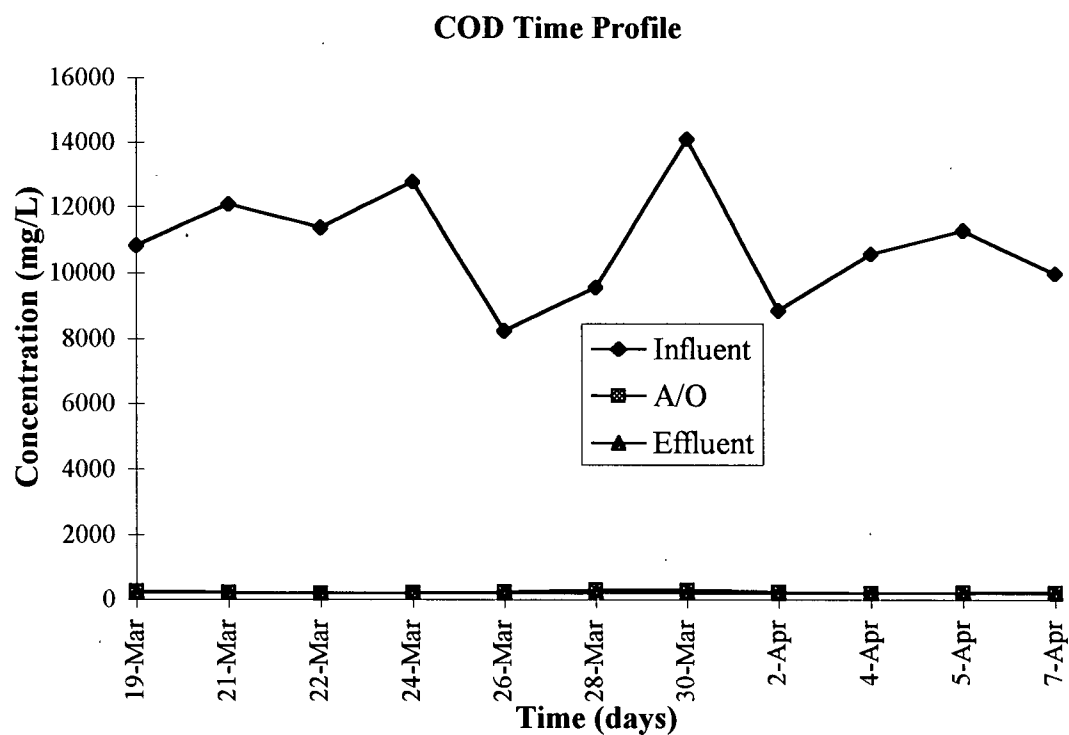
Figure A4 COD time profile for the 4,000 mg/L run (attached growth system)
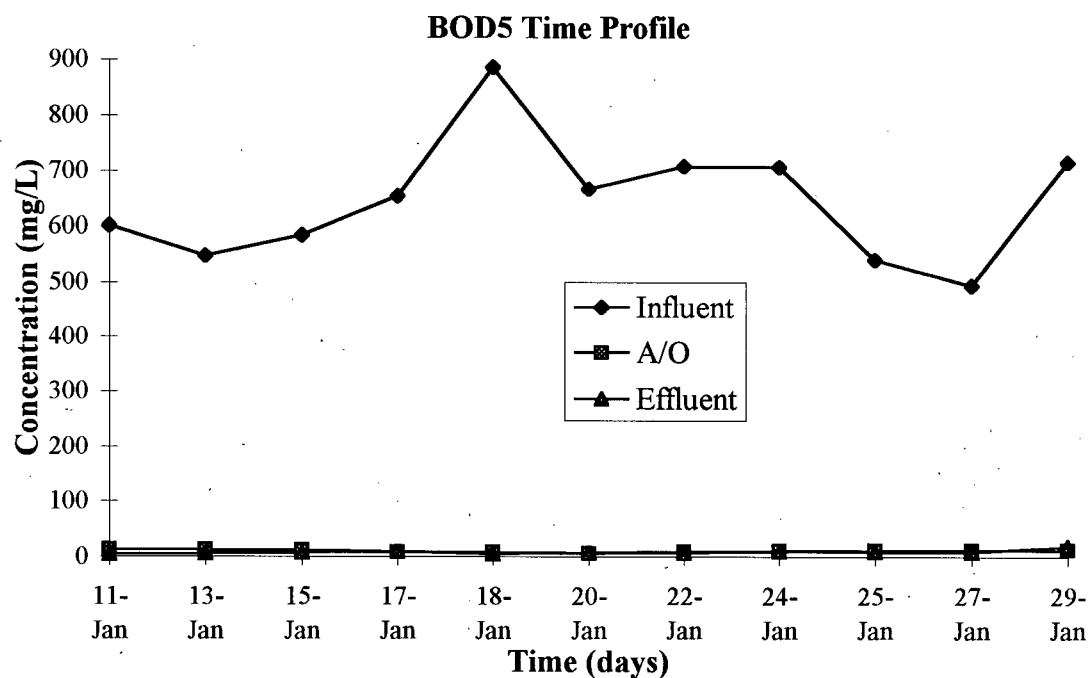
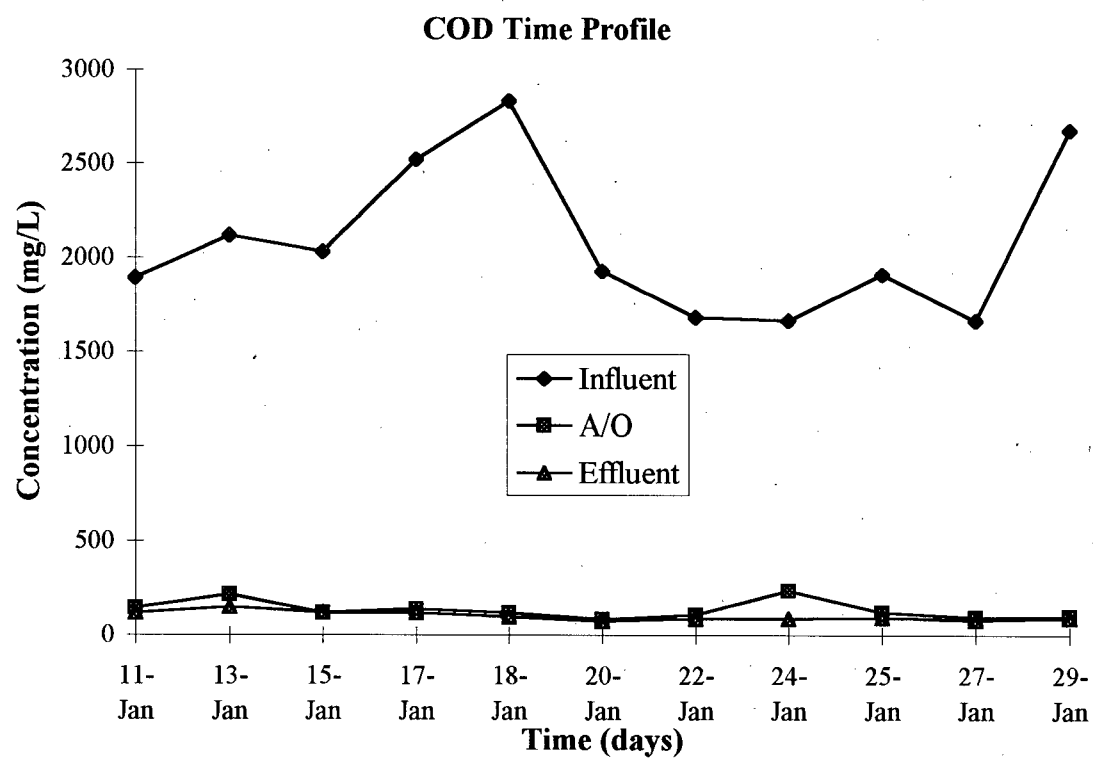Figure A5 BOD$_5$ time profile for the 500 mg/L run (attached growth system)

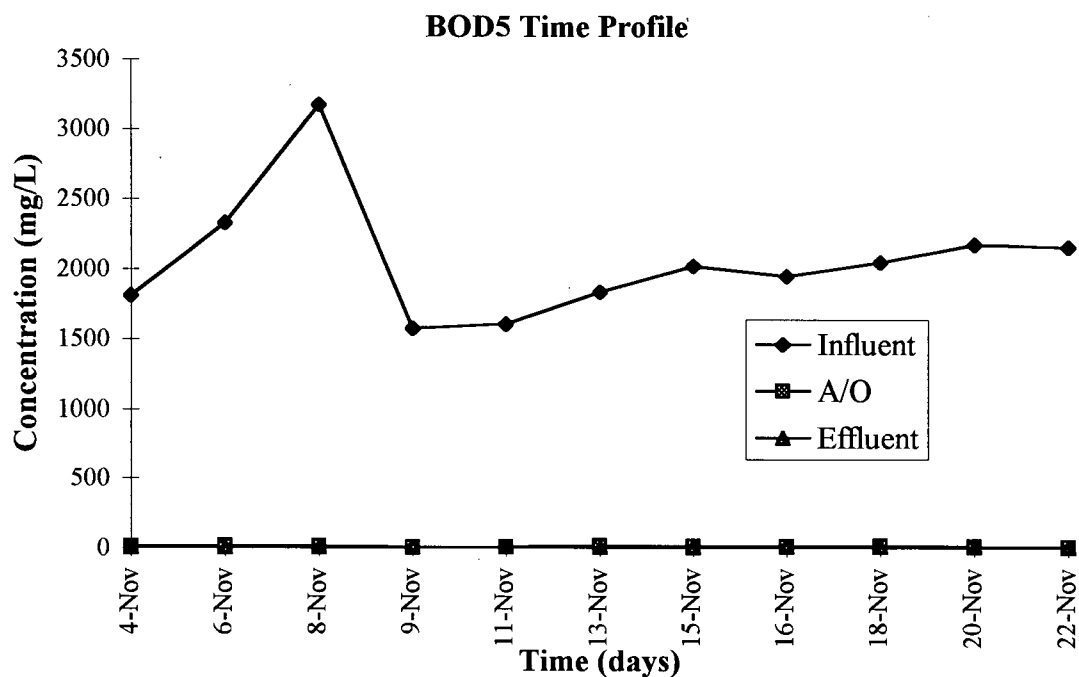Figure A6 COD time profile for the 500 mg/L run (attached growth system)

Figure A7 BOD$_5$ time profile for the 2,000 mg/L run (suspended growth system)
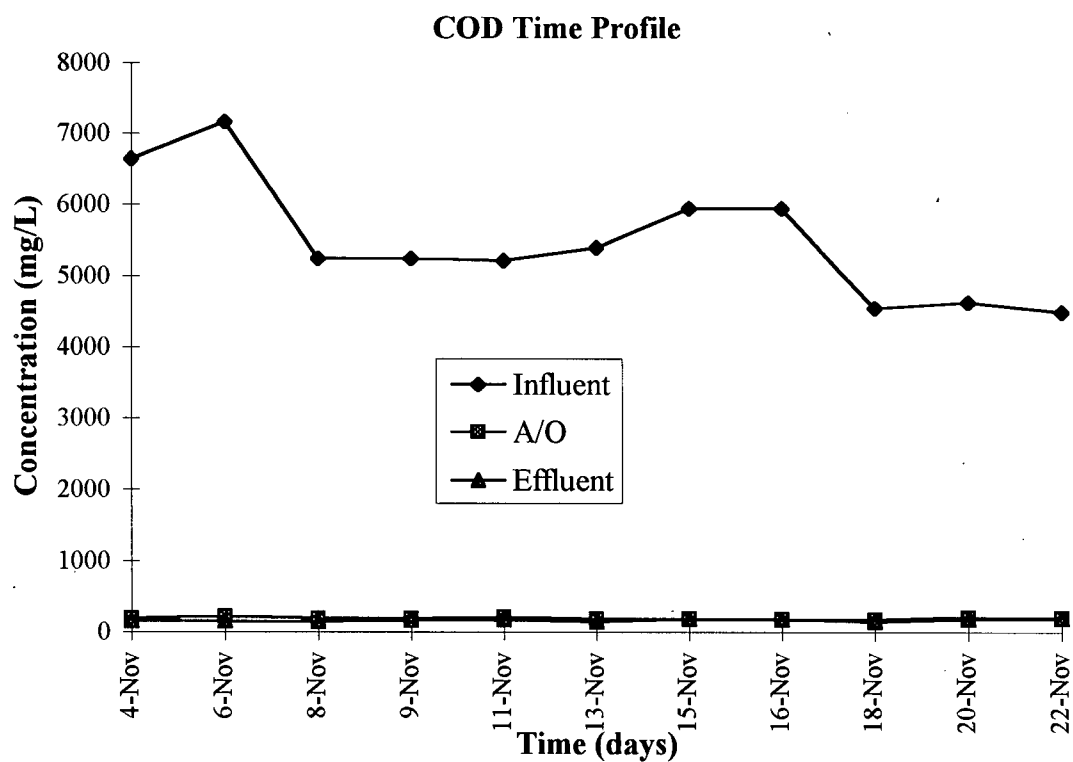
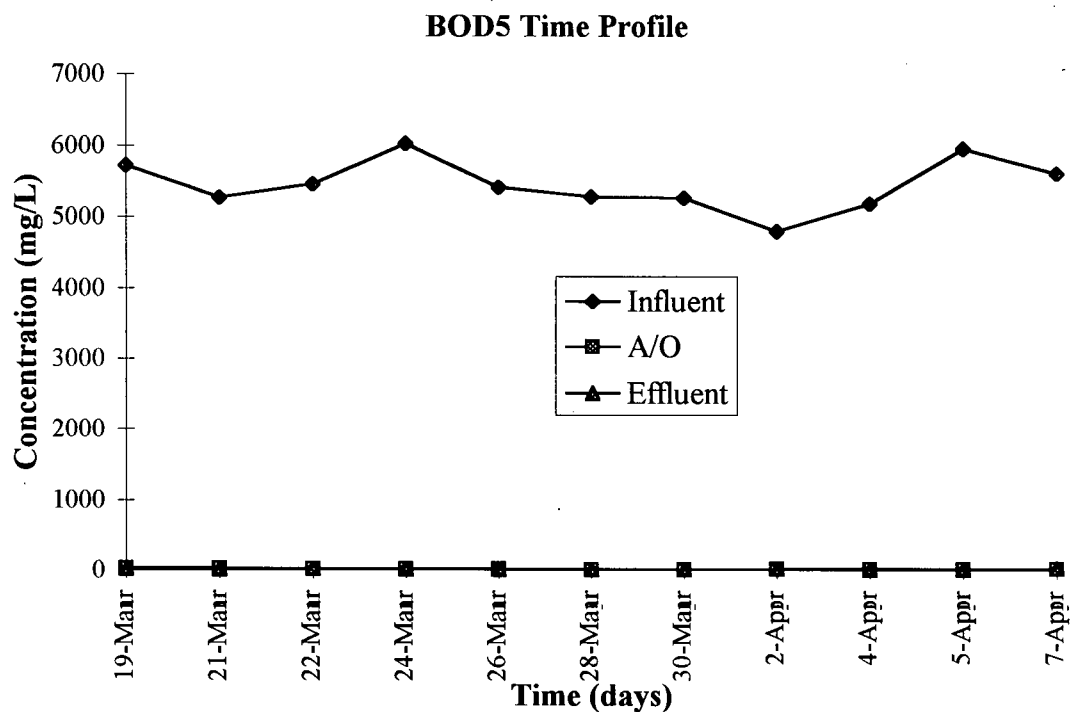Figure A8 COD time profile for the 2,000 mg/L run (suspended growth system)

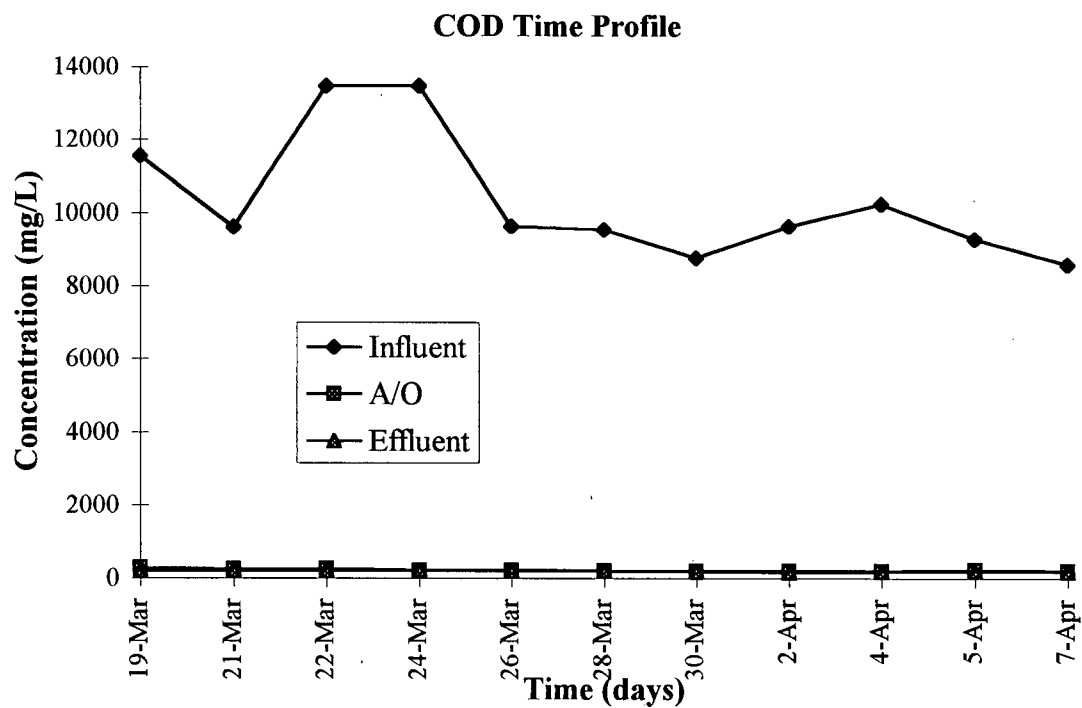Figure A9 BOD$_5$ time profile for the 4,000 mg/L run (suspended growth system)

Figure A10 COD time profile for the 4,000 mg/L run (suspended growth system)
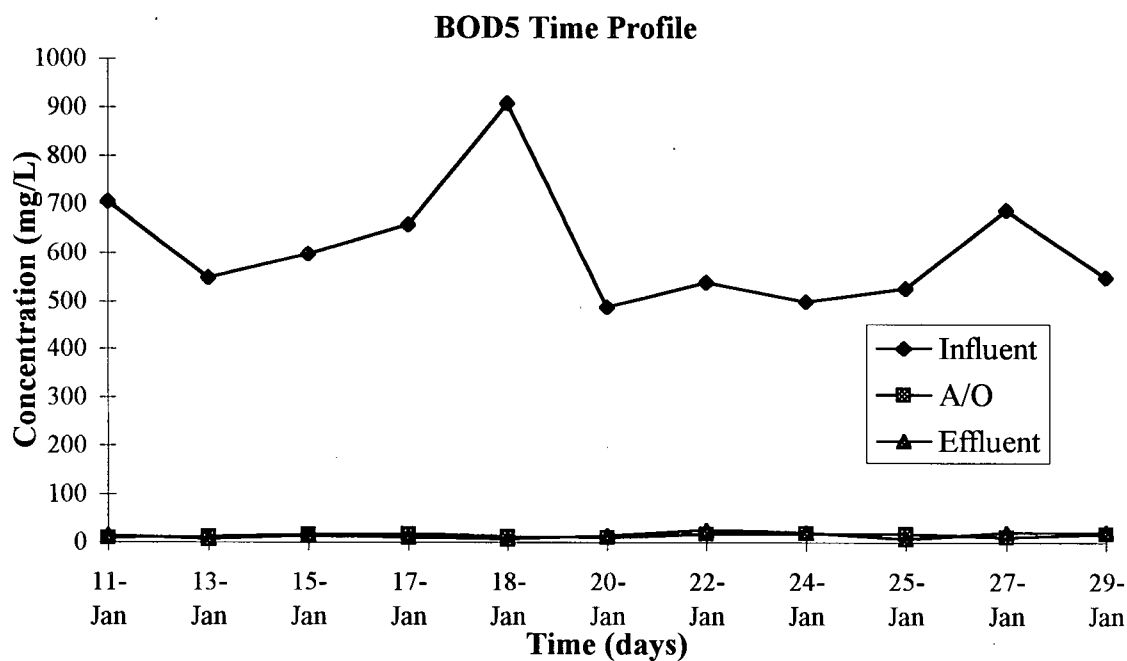
Figure A11 BOD$_5$ time profile for the 500 mg/L run (suspended growth system)

Figure A12 COD time profile for the 500 mg/L run (suspended growth system)

Figure A13 DO time profile for the 2,000 mg/L run (attached growth system)

Figure A14 DO time profile for the 2,000 mg/L run (suspended growth system)

Figure A15 DO time profile for the 4,000 mg/L run (attached growth system)

Figure A16 DO time profile for the 4,000 mg/L run (suspended growth system)

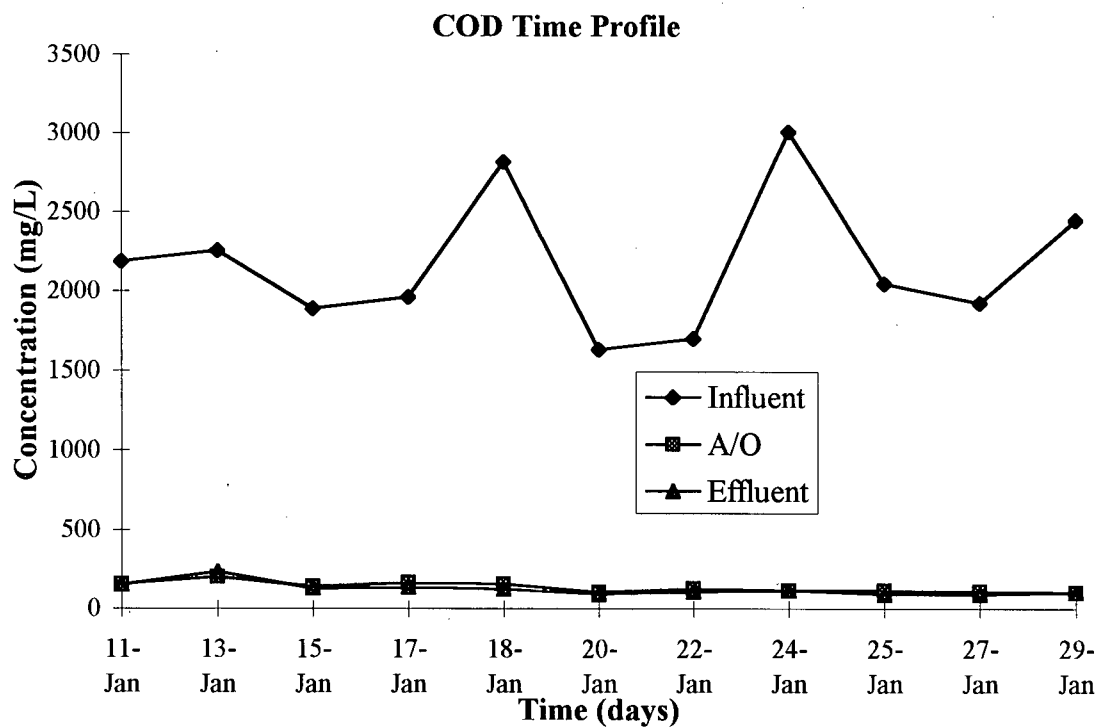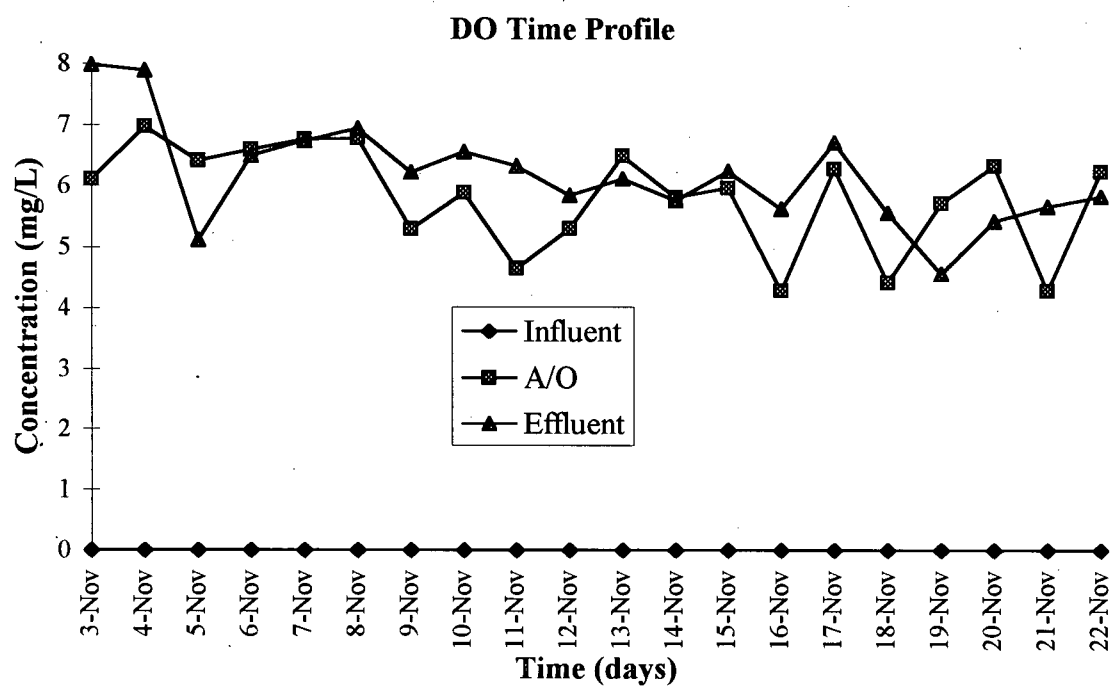Figure A17 DO time profile for the 500 mg/L run (attached growth system)

Figure A18 DO time profile for the 500 mg/L run (suspended growth system)

Figure A1 BOD$_5$ time profile for the 2,000 mg/L run (attached growth system)

Figure A2 COD time profile for the 2,000 mg/L run (attached growth system)

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BOD5 Time Profile

Figure A3 BOD$_5$ time profile for the 4,000 mg/L run (attached growth system)

COD Time Profile

Figure A4 COD time profile for the 4,000 mg/L run (attached growth system)

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Figure A5 BOD$_5$ time profile for the 500 mg/L run (attached growth system)

Figure A6 COD time profile for the 500 mg/L run (attached growth system)
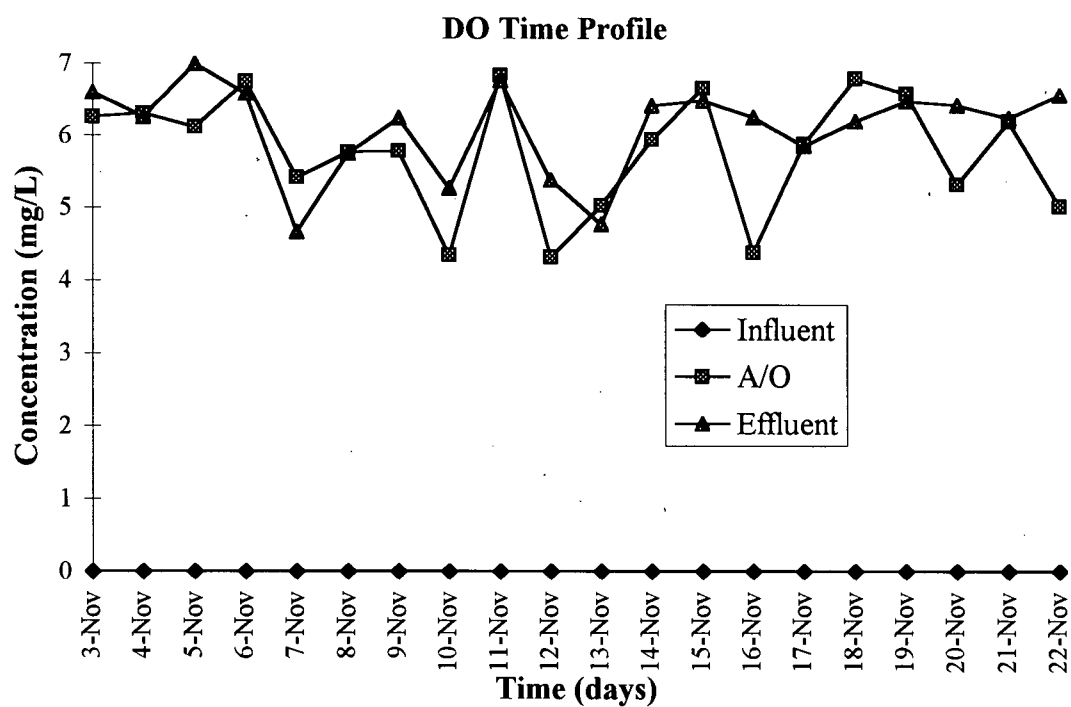
BOD5 Time Profile

Figure A7 BOD5 time profile for the 2,000 mg/L run (suspended growth system)

COD Time Profile

Figure A8 COD time profile for the 2,000 mg/L run (suspended growth system)

BOD5 Time Profile

Figure A9 BOD5 time profile for the 4,000 mg/L run (suspended growth system)
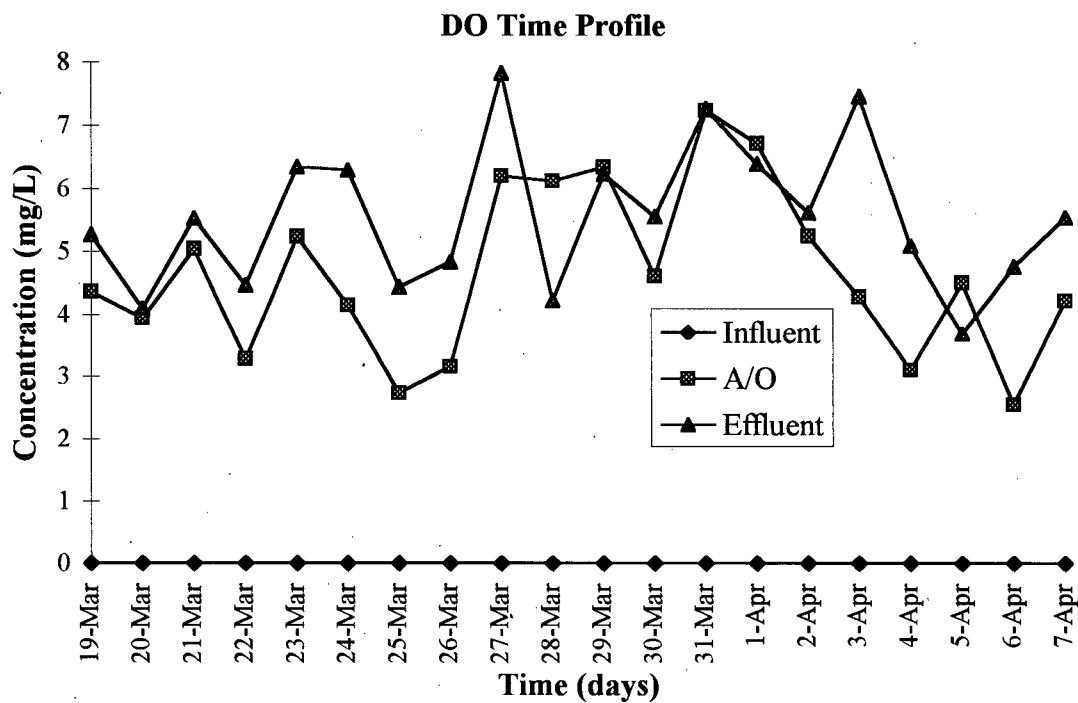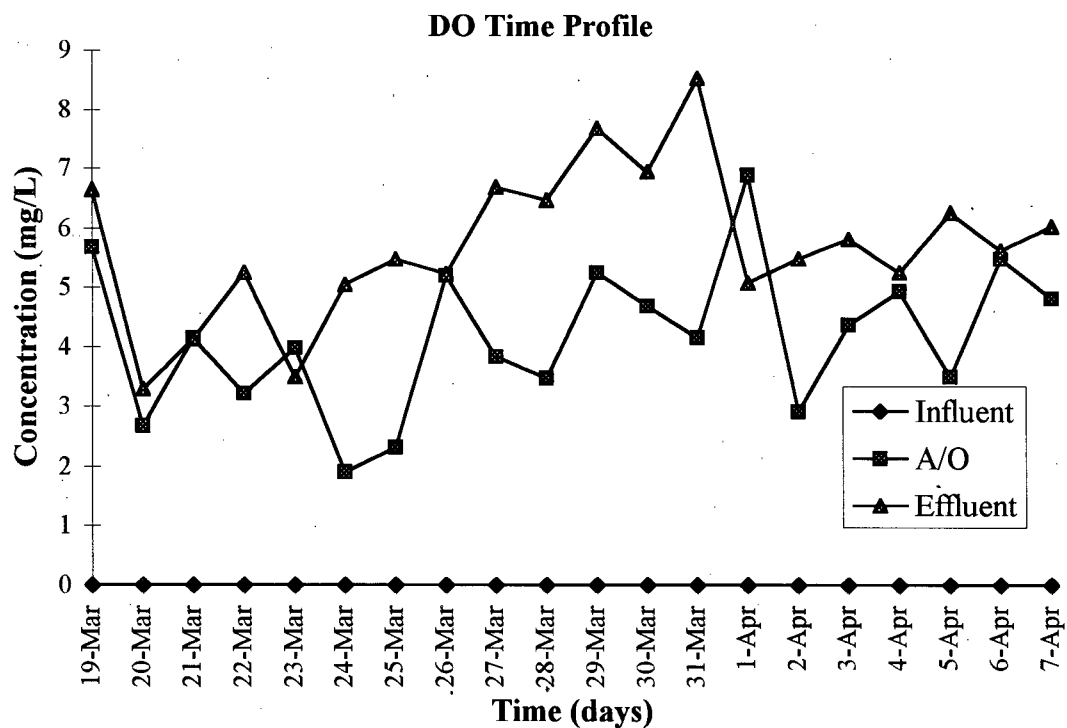
COD Time Profile

Figure A10 COD time profile for the 4,000 mg/L run (suspended growth system)

Figure A11 BOD₅ time profile for the 500 mg/L run (suspended growth system)

Figure A12 COD time profile for the 500 mg/L run (suspended growth system)

DO Time Profile

Figure A13 DO time profile for the 2,000 mg/L run (attached growth system)

DO Time Profile

Figure A14 DO time profile for the 2,000 mg/L run (suspended growth system)

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DO Time Profile

Figure A15 DO time profile for the 4,000 mg/L run (attached growth system)

DO Time Profile

Figure A16 DO time profile for the 4,000 mg/L run (suspended growth system)
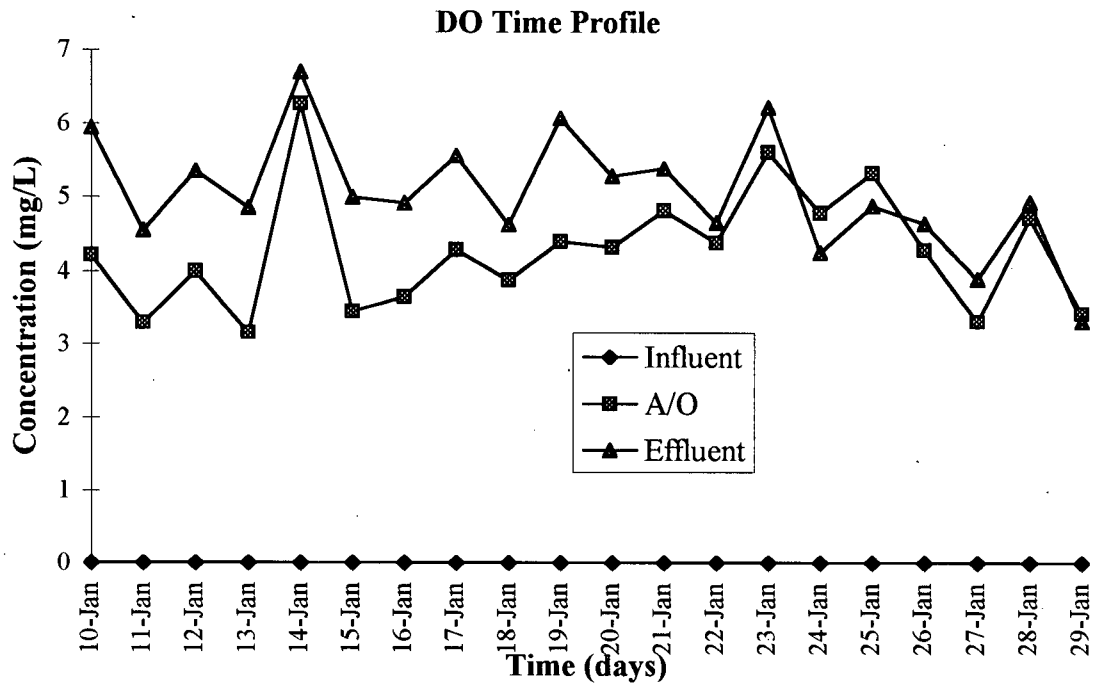
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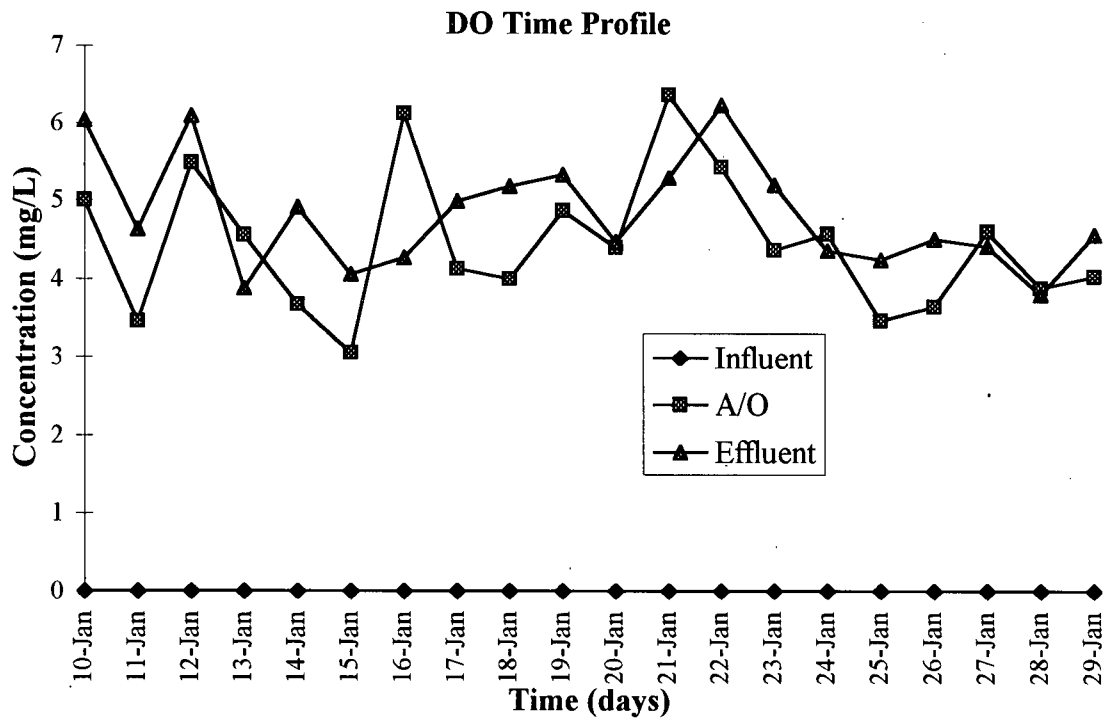**Figure A17 DO time profile for the 500 mg/L run (attached growth system)**

Figure A18 DO time profile for the 500 mg/L run (suspended growth system)