AN INTEGRATED TREATMENT SYSTEM FOR LIQUID SWINE MANURE

A Thesis Submitted to the Faculty of Graduate Studies and Research in Partial fulfillment of the Requirements for the degree of Doctor of Philosophy

by

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ABSTRACT

The present study investigated the feasibility of an integrated system that includes a microscreening unit followed by a sequential batch reactor (SBR) for the treatment of liquid swine manure.

A new mechanical device was developed to separate animal waste slurries into semi-solid and liquid fractions using an endless conveyer belt made of filter material which is continuously cleaned by an air knife. The best results for this unit were obtained with a screen of 0.1 mm mesh opening size and a hydraulic loading of 35 l/min. For influent slurries with 2 to 8% dry matter, the solids removal efficiency was between 47 to 60%, and the separated solids contained 14 to 18.5% of dry matter, respectively. Also considerable amounts of organic carbon and nutrients were removed from the influent waste liquid by the microscreening process.

The screened liquid manure was treated in a laboratory scale SBR operating on the basis of a 24 hour cycle. This treatment system proved to be highly efficient, since more than 99% of NH_3-N, 96% COD and 97% suspended solids removal were achieved in the reactors operating at 7 and 9 days hydraulic retention time and 20 days biological solids retention time. Nitrification and denitrification were

carried out by alternating aerobic/anoxic conditions within the same reactor. Anoxic operation did not affect the activity of the nitrifiers, and the nitrite plus nitrate concentration increased rapidly following the introduction of an air supply. The denitrification rate was higher during anoxic fill as compared to the anoxic react segment, because of the accessibility of the exogenous carbon. Anoxic fill also eliminated the problem of excessive production of foam. However, extending anoxic operation over 8 hrs deteriorated the solids settling process in the SBR. Hydraulic retention time (HRT) of less than 5 days and temperatures below 10°C also adversely affected the performance of the SBR. The biological solids retention time (BSRT) in the range of 10 to 30 days had a negligible effect on the quality of the effluent produced.

A mathematical model for the simulation of SBR operation was developed based on the mass balance concept. The model can predict successfully the behaviour of COD, NH_3 and NO_2-NO_3-N removal and/or formation during the fill and react periods. The best fit technique and regression analysis were used to calibrate the model cofficients. The validity of the model was investigated for different SBR operation conditions: 1) change in influent concentration; 2) change in HRT; and 3) change in BSRT. In general, the predictions were in the range of 85 - 95% of the laboratory measured values. The model is useful for design and determination of SBR operating parameters.

RESUME

Cette étude examine le potentiel d'un système intégré de traitement du purin de porc, incluant une unité de tamisage et un réacteur séquentiel biologique (RSB).

Un nouveau procédé mécanique de séparation des fractions liquides et solides du purin a été développé. Ce procédé utilise un convoyeur à courroie fait de matériel filtrant continuellement nettoyé par un jet d'air. Les meilleurs résultats ont été obtenus avec un filtre dont les ouvertures avaient 0.1 mm de largeur et avec une charge hydraulique de 35 litres par minute. Pour des boues dont le contenu en matière sèche variait de 2 à 8%, l' éfficacité dans l'élimination des solides était de 47 à 60%. Les fractions solides, séparées par le procédé, contenaient de 14 à 18.5% de matière sèche. Des quantités considérables de carbone organique, azote et ses composés ainsi que phosphore total ont aussi été éliminés du purin par le tamisage.

La fraction liquide du purin tamisé a été traitée dans le réacteur séquentiel, en laboratoire, avec des cycles d'opération de 24 heures. Ce système s'est avéré très efficace pour des temps de rétention hydraulique de 7 à 9 jours ainsi que 20 jours pour la rétention des solides biologiques. Par conséquent, plus de 99% des composés azotés NH_3 -N, 96% de la demande chimique en oxygène (DCO) et 97% des solides en suspension ont été éliminés. L'alternance de conditions aérobies et anoxiques à l'intérieur du même réacteur a permis d'obtenir la nitrification et la dénitrification. Les conditions anoxiques ne semblaient pas avoir affecté l'activité des nitrobacters, les concentrations de nitrite et nitrate ont augmenté rapidement dès que l'oxygène a été disponible. Le taux de dénitrification

était plus élevé durant le remplissage anoxique que durant la phase de réaction, à cause de la disponibilité du carbone exogène. Le remplissage anoxique a éliminé aussi la formation de mousse. Cependant, des périodes d'operération anoxique, dépassant 8 heures, affectaient la déposition des solides dans le réacteur. La réduction du temps hydraulique de rétention à moins de 5 jours et des températures de moins de $10^{o}C$ ont également diminué la performance du RSB. Des périodes de rétention des solides biologiques variant de 10 à 30 jours n'ont en que peu d'influence sur la qualité de l'effluent obtenu.

Un modèle mathématique, s'appuyant sur le principe d'équilibre dynamique des masses et simulant l'opération du réacteur séquentiel, a été développé. Le modèle peut prédire avec précision raisonable les comportements d'élimination et/ou de formation de DCO et des composés d'azote (NO_2 - NO_3 -N) durant le remplissage et la période de réaction. Des analyses de régression ont servi à comparer les valeurs mesurées et prédites. La validité du modèle a été étudiée pour différentes conditions d'opération: 1) en faisant varier le concentration de l'affluent, 2) en faisant varier le temps de rétention hydraulique et 3) en changeant le temps de rétention des solides biologiques. Le rapport des valurs prédites aux valeurs mesurées en laboratoire était de 85 à 95% en général. Ce modèle peut être utile pour le design et la détermination des paramètres d'opération d'un tel système.

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CHAPTER 1 INTRODUCTION

1.1 Statement of the Problem

In Quebec, since the mid-sixties a phenomenal expansion in the swine industry nurtured trends towards intensive production and animal confinement. As an example, Figure 1.1 illustrates the changes in livestock production in the agricultural regions of Richelieu and North of Montreal over the past twenty-five years. By 1985, the Province of Quebec contributed almost 40% of the Canadian swine farming activities (Statistique Canada, 1985). The sharp growth in the swine population, abetted by the availability of new types of grain and the rising hog market value, resulted in the accumulation of large quantities of wastes, and consequently their storage and disposal presented economic and ecological problems. According to Barrington (1985) the manure management facilities have not only been overlooked but underdeveloped in technology to meet the current needs.

Traditionally, manure generated at animal production centers is used to fertilize and amend the nearby croplands. However, in Quebec, the geographic concentrations of swine operations along with the large volumes of waste produced could not be accommodated by the land in the

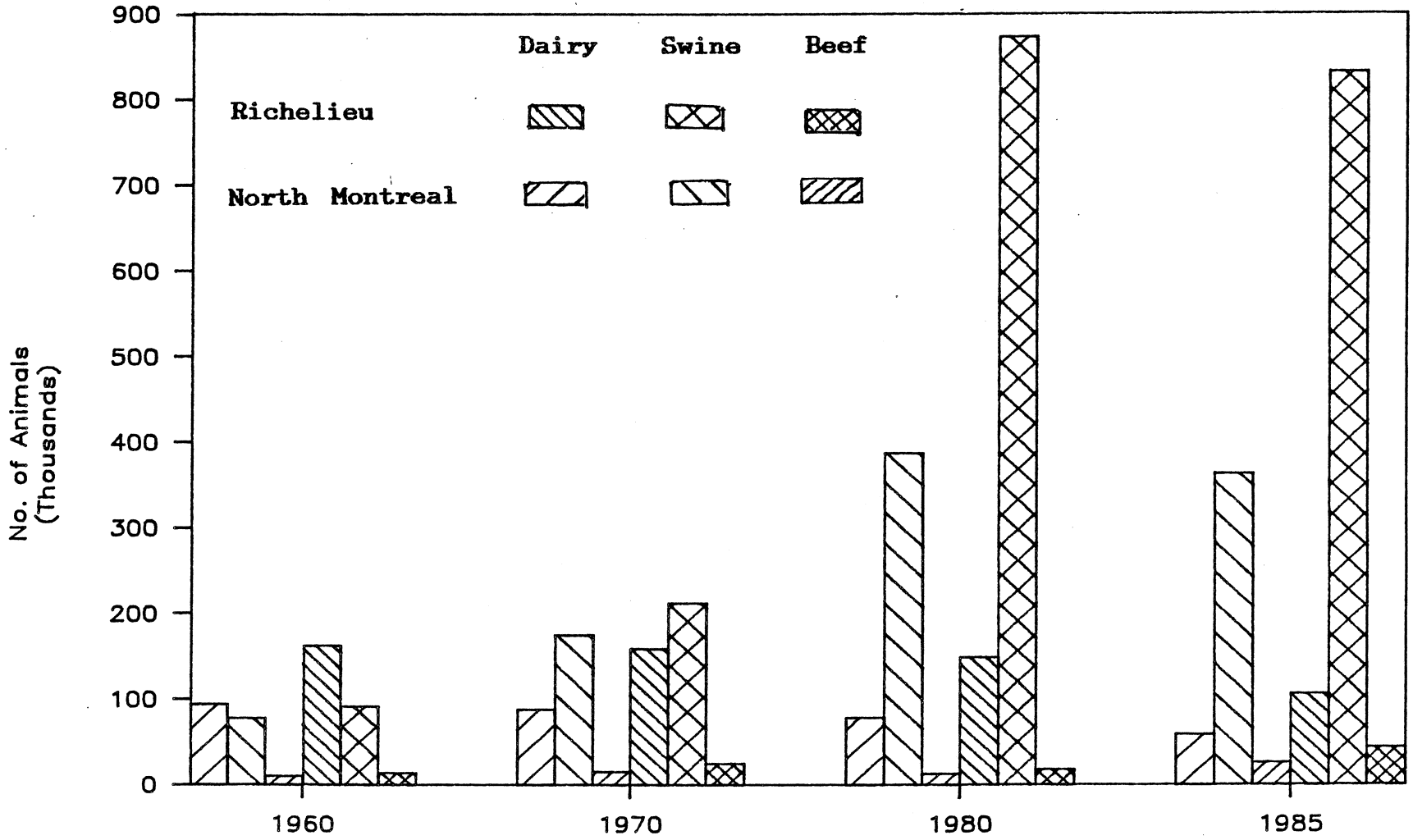


Figure 1.1 Livestock production in two regions of Quebec

Source: Agriculture Canada (1983); Statistique Canada (1985)

vicinity. Gangbazo and Buteau (1985a, b and c) reported that in several counties the animal units (A.U.) per hectare of tillable land surpass the permissible manure application rate by as much as 200%. Concurrently, the competitive position of manure as compared to chemical fertilizer is reduced, due mainly to handling costs and the increased distances of transportation.

Improper management of manure, such as high land application rates, result in severe water pollution problems when contaminants gain access to waterways and groundwater by means of run-off and percolation. Gangbazo and Blais (1987), Dupont et al. (1984), and Couillard and Cluis (1980) found intensive animal farming and manure mismanagement to be chiefly responsible for the high concentration of pollutants and the deterioration of the hydrographic basins of l'Assomption, St-Francois, Chaudiere and Yamaska rivers.

Liquid swine manure has relatively high concentrations of suspended solids, organic matter, nitrogen and phosphorus compounds (Agriculture Canada, 1980). Thus it constitutes a prime pollution potential. Consequences of water pollution are the serious depletion of dissolved oxygen in receiving waters, the stimulation of eutrophication in surface waters, the toxicity of ammonia to aquatic life, the high levels of nitrates in drinking water associated with methemoglobinemia and cancerous disease, and

the reduction of suitable water for municipal and industrial reuse (Peavy et al., 1985; Robinson, 1983; Berger, 1982; McKinney, 1970).

In an attempt to abate pollution problems associated with agricultural sources, the government agencies imposed severe regulations on existing and newly established livestock operations. In view of the 1981 Quebec legislation, swine producers operating on an insufficient land base had to give serious consideration to animal waste treatment, in order to prevent further degradation of the environment. In this context, the stabilization and control of organic matter and total nitrogen is a priority in all manure management strategies.

The application of an integrated system incorporating primary and secondary treatment is not a conventional practice for livestock waste management. The importance of mechanical separation of solids before the biological stabilization of the soluble waste is not generally recognized (Hepherd et al., 1980). Screening permits the removal of coarse particles and renders the liquid manure easier to pump and to process for further application (Holmberg et al., 1983; Bishop et al., 1981). Aerobic and anaerobic biological processes were investigated and proposed for animal waste treatment (Sweeten, 1980). Commonly, aerobic systems are designed and operated to

provide odour control and partial stabilization of the wastewater (Ghaly, 1982).

The combined removal of carbon and nitrogen in biological treatment processes (incorporating nitrification and denitrification) offers considerable economic advantage. Therefore it has received greater attention in the last few years. Yet, the available technology for animal waste treatment is inadequate to provide simultaneous reduction of carbon and nitrogen.

Primarily, the problem calls for a single stage reactor of which the operation provides diverse environmental conditions, i.e. aerobic and anoxic, in a cyclic fashion to enhance the oxidation of carbon and ammonia nitrogen, and the reduction of nitrates. Such requirements seem ideally suited to the sequencing batch reactor (SBR) which has been found to be an efficient and flexible way to treat various types of dilute wastewaters (Lo and Liao, 1986; Decreon et al., 1985; Schmidtke and Topnik, 1983; Irvine and Busch, 1979). The SBR system is basically a fully or partly aerated sequences of partially filling the reactor with the waste material, then a react phase followed by a settling period, the drawing off the supernatant and finally an idle period. The cyclic nature of this process gives temporal control over operating modes, and the aeration supply. The SBR process is still in its

infancy with respect to the treatment of agricultural wastes, and more research is required before it can have a broader practical application. To date the SBR technology has not been applied for the treatment of strong waste such as liquid swine manure.

1.2 Purpose and Scope of Study

The general objectives of this research project are: to demonstrate the feasibility of an integrated system, including microscreening and a sequential batch reactor, for the treatment of liquid swine manure; to develop the necessary design information to achieve varying degrees of nitrogen removal, without neglecting other environmental issues such as reduction of suspended solids and organic carbon, and odour control; to develop and verify a mathematical model for the combined removal of organic carbon and nitrogen in a SBR under a wide variety of process operation conditions.

This study is limited to the treatment of liquid swine manure with the total solids content between 2 and 8% For the purpose of this study, a prototype microscreening unit was developed and tested under various operating conditions. Laboratory scale reactors were used to test the effectiveness of SBR to further treat and stabilize the screened liquid swine manure.

CHAPTER 2 LITERATURE REVIEW

This chapter will provide a referenced background on the mechanical and biological treatment processes which are commonly employed in swine waste management. In addition, studies regarding the SBR treatment system will be included. Basically, the four main sections of this review contain:

(1) The characteristics of swine waste which are of interest to this study.

(2) The screening processes (i.e. solid/liquid separation).

(3) Aerobic and anaerobic biodegredation of wastewater for the stabilization of both the organic carbon and nitrogen loads.

(4) Biological treatment processes with emphasis on the sequential batch reactor.

2.1 Characteristics of Swine Waste

The knowledge of the production of swine waste and its characteristics is essential for the selection of an appropriate waste management technology. Overcash et al. (1975) reviewed the work of several researchers on voided swine excreta. These authors found manure production to

increase predominantly with animal age, weight and feed intake. On the average, the weight of manure produced is 7.6 kg per day per 100 kg of swine weight, and the moisture content is about 92% which gives it fluid characteristics. To estimate accurately the ultimate quantities and the characteristics of hog waste, one must consider not only waste excretion rates but also post-excretion changes. In this context, animal housing, floor washing and the manure collection technology exert major influences (Turnbull, 1981; Hobson and Robertson, 1977; Taiganides et al., 1964).

It is a common practice for modern swine production buildings to use liquid manure systems almost exclusively , including slotted floors and flushing gutters (Turnbull, 1984). In these systems, manure falls freely into a collection system where it is inevitably diluted with water used for cleaning and spilled by the animals in drinking. The amount of water added at this stage affects the nature and the variability of the wastewater to be handled.

The characteristics of typical wastewater from swine operations in Quebec are presented in Table 2.1 . It can be observed that the waste is rich mainly in organic compounds, nitrogen and phosphorus. Typically, the total solids content is in the order of 3 to 5% (MAPAQ, 1983).

TABLE 2.1 - QUEBEC SWINE WASTE CHARACTERISTICS

Parameters	Range of Value		
	low	average	high
dry matter (g/l)	38.0	56.0	74.0
biological oxygen demand BOD_5 (g/l)	13.4	19.8	35.0
chemical oxygen demand COD (g/l)	37.6	52.4	80.0
total nitrogen (g/l)	4.8	6.1	7.4
ammonia nitrogen (g/l)	3.1	3.9	6.0
total phosphorus (g/l)	1.4	1.9	2.4
potassium (kg/t)	15.0	23.0	31.0
magnesium (kg/t)	7.0	12.0	17.0
calcium (kg/t)	1.6	2.8	4.0
total volatile solids (%)	73.8	77.0	80.2

Source: Aquatech (1983).

2.2 Solid/Liquid Separation

There is an increasing interest in management techniques which convert semi-liquid slurries into the liquid and solid fractions that are easier to handle and process. According to Oleszkiewicz et al. (1980) and Ngoddy et al. (1971) the efficient solid/liquid separation of livestock wastewater is a necessary part of any wastewater management scheme in which water is to be reclaimed and profitably recycled.

At the present time, the physical separation of solids from animal waste slurries can be performed by a variety of conventional operation units. The principle of

solid/liquid separation is based on physical differences among the particles such as specific gravity, size and shape (McCabe and Smith, 1977).

2.2.1 Phase Separation by Specific Gravity

This separation can be attained by means of sedimentation, flotation and centrifugation. In these systems, the mechanism is dependent on the differences that exist between the buoyant forces and the drag forces exerted on the particles. Generally, particle separation due to a density difference can be mathematically expressed by Stoke's law. Solids separation by imparting a force on the particles will improve the rate and the efficiency of the process (Merkel, 1981; Metcalf and Eddy, 1979).

Extensive studies have been conducted to evaluate the performance of these aforementioned methods (Green and Kramer, 1979; Blaha, 1977; Glerum et al., 1971). The efficiency, expressed on the basis of the removal of total suspended solids (TSS) and chemical oxygen demand (COD) can be considerably high, in the order of 75 and 60% respectively. The efficiency varies with the type of unit used, hydraulic retention time, solid concentration in the influent slurry, the applied centrifugal force and temperature.

The operation of the settling tank, dissolved air flotation unit and devices that take advantage of centrifugal forces requires skilled labour and high energy inputs. The cost surveys undertaken for these units have shown conclusively that capital investment costs far exceed what can be considered feasible for the average livestock operation (Overcash et al., 1983; Pain and Hepherd, 1980; Ngoddy et al., 1971).

2.2.2 Screening Units

In recent years, mechanical separators have received greater attention. Their operation is relatively simple, less expensive, and the solid/liquid separation efficiency can be relatively high (Huijsmans and Lindley, 1984; Pos et al., 1983; Avnon, 1980). The mechanism consists of separating particles from the liquid bulk by simply trapping them onto a screening or filtering medium with a particular opening size. According to Schmidtke (1981), the performance of the screen is influenced by the screen type (opening size and geometry), the solids type (size distribution and shape) and the influent solids loading.

2.2.3 Examples of Screening Units

The following are some of the models developed and used in animal wastewater treatment.

The stationary screen functions as an inclined drainage board. The liquid is mostly drained in the upper slope of the screen and the solids are displaced by the incoming slurry. Graves and Clayton (1972) tested a stationary sloping screen with 0.5 mm bar spacing for dairy manure diluted between 2:1 to 50:1 . Almost 50% of solids were removed from the highly diluted slurry. Shutt et al. (1975) evaluated the performance of a stationary screen with swine waste that possessed 0.2 to 0.7% total solids. The best results were obtained with a 1.0 mm screen opening size and a loading rate of 0.3 l/min per mm of screen width. As much as 35% of solids, 60% BOD and 65% COD were successfully removed. However, when concentrated slurries were used, they easily blinded the screen and caused an increased moisture content in the separated solids.

Vibrating screens utilize a short, rapid reciprocating motion that causes particles in suspension to migrate to the periphery or center of the screen where they can be removed. Research conducted by Gilbertson and Nienaber (1978) and Pain et al. (1978) on vibrating screens indicate that cattle slurries with more than 6% dry matter

reduced the capacity of the equipment to handle the influent waste. For dilute slurries, solids removal efficiencies in order of 45% were attained.

Shutt et al. (1975) conducted extensive studies on the application of a vibrating screen to swine wastewater. The most promising results were obtained with a 0.39 mm opening size screen at an influent flow rate of 67 l/min . When solids were concentrated up to 16.4% on a wet basis, only 0.6% of the inflow was retained on the screen. The filtrate contained 78% of the total solids and 84% of the influent COD, the difference being retained in the separated solids. It was concluded that the vibrating screens should be free of problems for farms where high dilution of manure is practical. These units are very sensitive to flow rate and influent solids concentration. Maintenance to prevent screen blinding is necessary for concentrated wastes. Also there is rapid wearing of the screen due to vibration. The energy required for the reciprocating motion will increase the costs involved.

Overcash et al. (1983) compiled data from various sources for solid separation efficiencies of commercial vibrating and stationary screening devices actually in use on farms. The authors used the percent of total solids as a criterion because the greatest amount of data are available for this parameter. The results for beef, dairy and swine

manures are summarized and illustrated in Figure 2.1. The extent to which these devices can separate the solids from the liquid bulk is based on the critical dimension of the particles in the manure. These dimensions are generally determined by a sieve analysis. Figure 2.2 displays the results of particle size distribution for typical swine manure.

From the comparison of Figures 2.1 and 2.2 , one can observe that commercial units are less efficient in solids removal from swine waste. For example, with a 0.1 mm screen opening size, the stationary and vibrating screen devices removed just about 40% of the solids, and this value is considerably less than that predicted from the sieve analyses, which is in the order of 52% . Overcash et al. (1983) explain that machine characteristics with manure, such as clogging, create conditions which reduce the performance of the screening devices.

Glerum et al. (1971) tested the rotary vacuum filter. It consists of a slow revolving drum of filter material. The vacuum applied in the interior of the drum permits the collection of liquid and the solids are brushed off from the surface. Experiments were performed with pig slurry and a 0.29 mm screen. The solids removal efficieny was 51% and the separated solids presented good stackable qualities and the dry matter content was 21% . The drawback

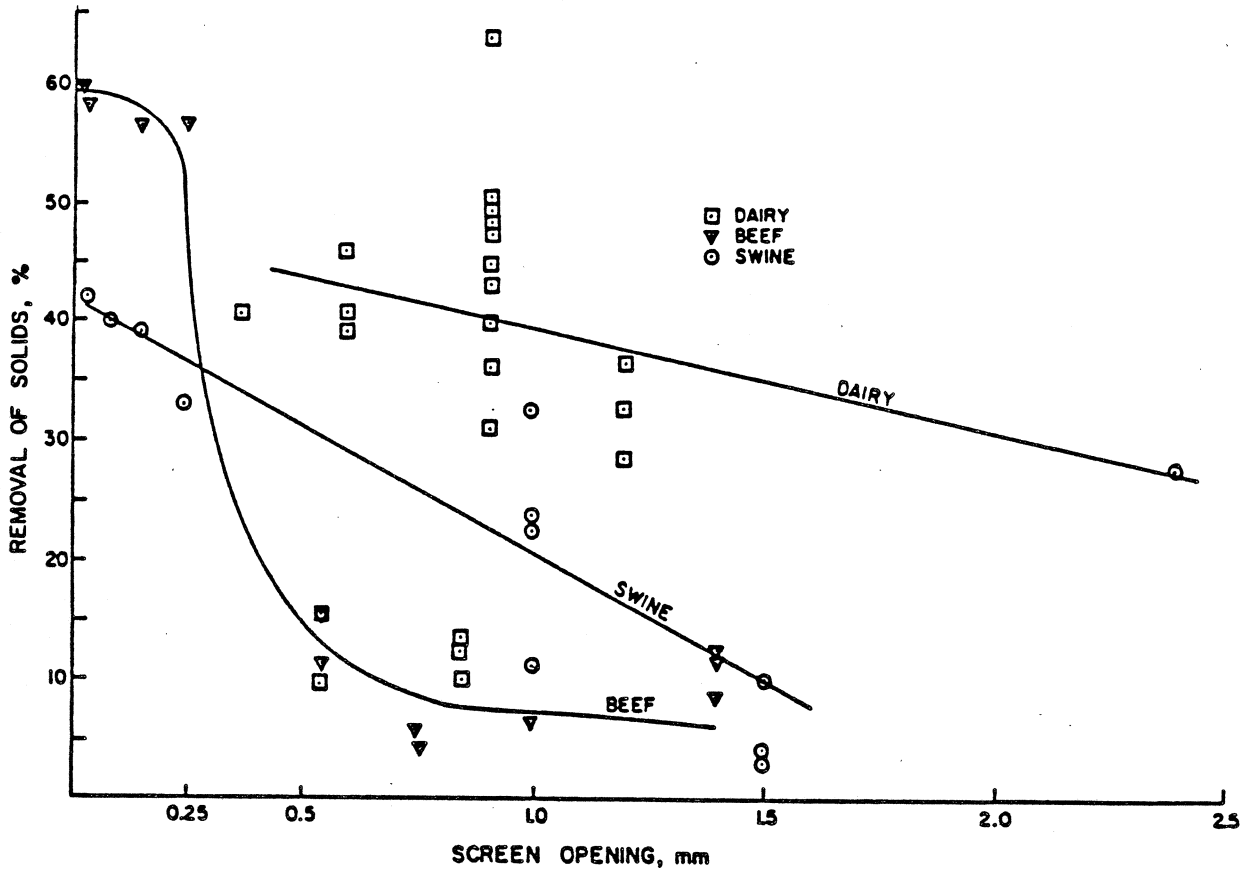


Figure 2.1 Separation efficiency using commercial screen devices
Source: Overcash et al. (1983)

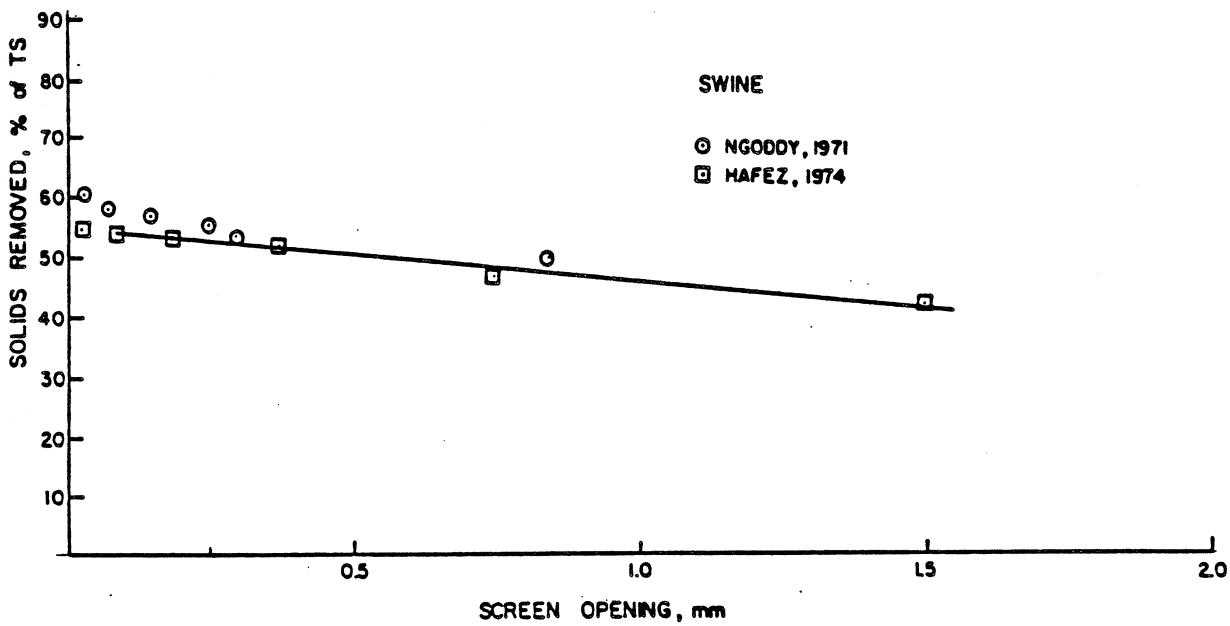


Figure 2.2 Screen separation of swine manure
Source: Overcash et al. (1983)

of this unit was the low hydraulic loading capacity of 0.25 m^3/h, and the 4 hp energy requirement.

Shirly and Butchbaker (1975) developed the rotating Conical Screen Separator. This unit makes use of screening and centrifugal forces. By increasing the diameter of the screened cone, it was possible to achieve effective centrifugal forces with reduced angular velocities, and consequently reduce the energy requirement. For optimum operation, they recommended an influent flow rate between 10-24 $kg/s/m^2$, and a peripheral speed of 51 m/s. The removal efficiency increased as the influent solids concentration and flow rate decreased and the diameter of the cone increased. But when a screen with 1.5 mm opening size was used the screen got plugged and liquid flow ceased.

Pain et al. (1978) tested the rotary screen separator with press rollers. In this unit the liquid drains through the perforations and the wet solids are squeezed by rollers and finally cleared off by a scraper. For pig and cow slurries the solids removal efficiencies were was 50 and 65% respectively. The moisture content of the solids cake ranged between 70 to 80% . This discrepancy was due to the particle size distribution. Cow slurry has a greater fraction of large fibrous particles compared to pig slurry. From this study it was found that an increase of

total solids in the influent slury improved the efficiency of the separation. Major disadvantages of this unit are the energy requirements and maintenance. Also these authors did not indicate the limitations on loading rates which are important operational parameters.

An analysis of the above described screening units shows that the slow biodegradable solids can be effectively removed from the animal wastewater. It is even more desirable to increase this efficiency. Since the absence of solids from the liquid fraction stimulates a faster rate of organics utilization. The soluble substrate is readily assimilated in biological treatment systems.

2.3 Fundamentals of Biological Treatment of Wastewater

The organic matter and nitrogen compounds can be effectively removed through biological processes under a favorable environment for the living microorganisms. The biochemical reactions involved in these processes are very different and require specific conditions which can be, in some instances, incompatible. Therefore, there is a need to understand the key factors affecting the biological activites in the treatment system.

2.3.1 Removal of Organic Matter

The heterotrophic microorganisms utilize organic matter to support both life and growth functions. First, the bacteria use hydrolytic enzymes to convert complex high molecular weight organic substances into simple soluble components, which are diffusable through the cell membrane. The microbial metabolism of the soluble substrate consists of energy yielding oxidation-reduction reactions. Most microorganisms oxidize their food by the enzymatic removal of hydrogen from the molecule. The final hydrogen acceptor is determined by the aerobic or anaerobic nature of the surrounding medium. Literature concerning enzymatic processes of hydrogen transfer and methods of biologically conserving energy released can readily be obtained from Gaudy and Gaudy (1980), and Pelczar et al. (1977).

The anaerobic degradation of organic waste occurs in the absence of free oxygen and it is a two stage process (Hashimoto et al., 1980; Fischer et al., 1979; McCarty, 1964a,b). During the first stage, which is commonly designated as the acid fermentation phase, the complex organic matter is hydrolyzed to products such as triglycerides, fatty acids, amino acids, as well as sugars. The hydrolysis process takes place due to the action of extra cellular enzymes of a heterogenous group of facultative and anaerobic bacteria. Furthermore, these hydrolyzied products

will be subject to fermentation, oxidation and other metabolic processes leading mainly to the formation of volatile acids, alcohol and eventually a new bacterial population.

In the second stage, usually referred as methane fermentation, several species of strictly anaerobic bacteria will convert the end products of the first stage mainly into methane, carbon dioxide, water and new bacterial cells. Although the anaerobic process is presented as being sequential by nature, under optimum operating conditions, both stages take place simultaneously and synchronously in an active, well buffered system (Sievers and Brune, 1978).

Aerobic processes are used effectively in wastewater treatment. In these processes a heterogeneous microbial culture, composed mostly of bacteria, fungi, protozoa and rotifers metabolizes most of the organic matter in the presence of oxygen (Stanier et al., 1979). The resulting end products are carbon dioxide, water and new cells.

Under food-limiting conditions, however, the microorganisms are forced to consume their own cellular mass until they lyse due to enzymatic attack. At this stage, the bacterial cells lose their viability, i.e. the reproduction ability. When lysing predominates, the cell-nutrients are

released into the medium, and become available to the remaining active organisms as food. The general organic aerobic reaction can be summarized as follows:

$$\text{organic matter} + O_2 \text{ ---> } CO_2 + H_2O + \text{non-biodegradable matter} + \text{energy} \qquad (2.1)$$

As pointed out by Wilkinson (1975), the principal significance of oxygen as the external electron acceptor is that the aerobic heterotrophs oxidize the substrate completely and therefore the maximum amount of energy is liberated and becomes available for growth and life-support facilities. Whereas the anaerobic microorganisms produce reduced compounds and the energy yield is much less. As an example, the difference in energy yield from the anaerobic and aerobic metabolism of glucose is 31 and 686 kcal/mole, respectively (Benefield and Randall, 1980). It follows that aerobic treatment has a high rate of synthesis and organic waste removal. Also, the resultant end products do not exert further oxygen demand.

2.3.2 Nitrification Process

In the untreated waste, nitrogen is present principally in the form of organic nitrogen and ammonium.

Other chemical species of interest for nitrogen control in wastewater are nitrite, nitrate and nitrogen gas (Sawyer and McCarty, 1978). Ammonium is the most reduced form of inorganic nitrogen and serves as the starting point for a biological process known as nitrification. A study by Warington (1891) indicated that the oxidation of ammonium ions occurred in two distinct steps, each promoted by a select group of bacteria.

It is generally held that obligate chemo-autotrophic bacteria assume the major role in naturally occuring nitrification. Painter (1977) reported that Nitrosomonas europaea, Nitrobacter winogradskyi and Nitrobacter agilis are mainly responsible for nitrification in wastewater. The genera Nitrosomonas, oxidize ammonia to nitrite which is then further oxidized to nitrate by the genera Nitrobacter. The oxidation reactions can be represented as:

$$2NH_4^+ + 3O_2 \xrightarrow{\text{Nitrosomonas}} 2NO_2^- + 4H^+ + 2H_2O + \text{energy} \quad (2.2)$$

$$2NO_2^- + O_2 \xrightarrow{\text{Nitrobacter}} 2NO_3^- + \text{energy} \quad (2.3)$$

These reactions are thermodynamically favorable, resulting in the release of free energy at rates between 64 and 84 kcal/mole for ammonium oxidation and approximately 17

kcal/mole for nitrite oxidation (Painter, 1970). Both groups of bacteria use the energy liberated in synthesizing cell material from carbon dioxide, carbonates or bicarbonates (Ida and Alexander, 1965).

The cell yield for Nitrosomonas is greater than for Nitrobacter, because the former are able to extract more energy per mole of nitrogen oxidized. The nitrifying bacteria grow very slowly compared to aerobic heterotrophs due to the low energy available through the autotrophic metabolism. Thus, it is very important to provide adequate growth conditions in a mixed culture.

Detailed nutritional requirements are given by Sharma and Ahlert (1977). The list includes inorganic carbon, nitrogen, phosphorus, trace metals, salts and vitamins. There is some controversy about the assimilation of organic compounds. Smith and Hoane (1968) have demonstrated that acetate molecules can permeate the cell wall of Nitrobacter. However, Hooper (1969) observed that inorganic carbon is required for the autotrophic cellular synthesis.

Since the nitrifying organisms are obligate aerobes, adequate dissolved oxygen (DO) concentration is necessary to support the reactions involved. The

theoretical oxygen demand calculated from Equations 2.2 and 2.3 are 3.43 mg O_2. mg^{-1} N for NH_4^+ oxidation and 1.14 mg O_2. mg^{-1} N for NO_2^- oxidation.

The limiting concentration of dissolved oxygen required for nitrification has not been precisely determined, but it is somewhere in the range of 0.3 to 0.5 mg/l, below which nitrification does not occur (Stenstrom and Poduska, 1980; Wild et al., 1971; Johnson and Schroepfer, 1964). Haug and McCarthy (1972) found that a DO level of 60 mg/l presented no constraints on the growth of nitrifiers. In practice the aeration systems are designed to maintain a minimum DO concentration of 2 mg/l. Nevertheless, Prakasam and Loehr (1972), and Heukelekian (1942) have indicated that even with a DO level of 4 to 5 mg/l, the interior of a sludge floc may remain anaerobic due to diffusion limitations.

The pH of the medium has been shown to affect the growth rate of nitrifying bacteria. Various studies have found that the optimum pH lies in the range of 7.0 to 8.5 (Loveless and Painter, 1968; Engel and Alexander, 1958). Research done by Stankewich (1972) and Haug and McCarty (1972) has demonstrated the ability of the nitrifiers to adapt to a lower environmental pH. For a shift in pH from 7.0 to 6.0 approximately, 10 days of acclimatization was required to reestablish the optimal growth rate. Jaworski

et al. (1963) observed active nitrification at pH 5.0 to 5.5 in aerobic digestion of sewage sludge.

As shown in Equation 2.2 , hydrogen ions (H^+) are produced during the ammonium oxidation, and as a result the alkalinity of the system will decrease at a stoichiometric rate of 7.14 mg as $CaCO_3$ per mg of NH_4^+-N (EPA, 1975). A combination of H^+ with bicarbonate alkalinity in the wastewater and subsequent stripping of CO_2 by aeration tends to maintain pH values near neutrality.

Neufeld et al. (1980) and Anthonisen et al. (1976) have investigated the inhibitory effects of un-ionized ammonia (NH_3) and nitrous acid (HNO_2) on the nitrifying bacteria. The formation of these two compounds is pH dependent. Alkaline or acidic medium will stimulate the formation of NH_3 or HNO_2 respectively. The growth of Nitrobacter is inhibited by NH_3 in the range of 0.1 to 1.0 mg/l and HNO_2 between 0.2 and 2.8 mg/l . The Nitrosomonas have a higher tolerance, and they are inhibited by NH_3 in the range of 10 to 150 mg/l. The inhibitive effect is related to an enhanced ability to penetrate the nitrifying organisms, which may make them more toxic than the ion form.

Nitrifying bacteria are subject to inhibition by a variety of substances. Barnes and Bliss (1983), Hockenbury and Grady (1977), and Tomlinson et al. (1966) have reported

many organic compounds that are toxic because they interfere with the electron transport chain. The most significant group of inorganic inhibitors is the heavy metals. At very low concentrations many of them are also essential micronutrients (Painter, 1970). It should be noted that most of these substances are manufacturing industry related compounds, and seldom found in any animal waste at high levels.

Nitrification is profoundly influenced by the temperature of the medium. Focht and Chang (1975) and Wild et al. (1971) have indicated the optimum temperature for nitrifier activity to be in the range of 28 to 36^{o}C , with lower and upper limits of 4 and 50^{o}C respectively. In Quebec, the wastewater temperatures are seldom in the optimum range for nitrifiers. Thus, allowance must be made in a design for the slow bacterial growth rate at winter temperatures.

2.3.3 Denitrification Process

The biological reduction of nitrogenous oxides to gaseous nitrogen is achieved in the presence of a suitable exogenous carbon source (Christensen and Harremoes, 1977). Denitrification is carried out by facultative heterotrophic bacteria belonging to the genera Pseudomonas,

Achromobacter, Bacillus and Micrococcus (Focht and Chang, 1975). Certain autotrophic bacteria are also capable of denitrification by oxidizing inorganic compounds (Payne, 1973).

In a wastewater which has undergone biological nitrification, the concentration of nitrates is expected to be high. Under zero or a very low level of dissolved oxygen, nitrate is the favoured electron acceptor, and dissimilatory reduction takes place. In the assimilatory process, certain bacteria can reduce nitrate primarily to ammonia for subsequent use in the synthesis of cell protein (Nason, 1962).

For the denitrification process, the electron donor is usually an organic molecule. Methanol is the preferred carbon source because it is the least expensive synthetic compound available that can be applied without leaving a residual BOD in the process effluent (Viessman and Hammer, 1985). The overall reaction can be represented as:

$$NO_3^- + 5/6\ CH_3OH \text{ -----> } N_2 + 5/6\ CO_2 + 7/6\ H_2O + OH^- \quad (2.6)$$

The pathway of denitrification beyond the nitrite form has been a subject of controversy. There is strong evidence that nitrous oxide (N_2O) and enzyme-bound nitric oxide (NO) are intermediate products formed during the process (Firestone et al., 1979). The final production of

molecular nitrogen gas from N_2O has been demonstrated in several studies (Knowles, 1982).

During the denitrification reaction, the overall effect is to raise the pH of the solution due to the production of hydroxide ions. The change in pH is dependent upon the buffer capacity of the mixed liquor and the amount of denitrification. Metcalf and Eddy (1979) suggested that 6.5 to 7.5 is the optimum pH range for denitrifying bacteria. Focht and Chang (1975) have indicated that neutral to slightly alkaline pH not only effect faster denitrification rates, but also a more complete reduction to nitrogen gas. The process efficiency may fall by as much as 25% for pH 6 and pH 8 (Moore and Schroeder, 1970).

Several organic compounds have been examined as substrates that can act as hydrogen donors in biological denitrification. The endogenous carbon matter released as a result of auto-oxidation of biomass can be mobilized to meet the needs of denitrifying cells. However, the reaction involved proceeds at quite a low rate (Wuhrmann, 1964).

Raw wastewater is another inexpensive source of organic carbon. This is a rich mixture of colloidal and particulate matter and some soluble exogenous susbstrate. But consideration should be given to the rate of degradation of the insoluble fraction and the availability of the

substrate. Also this source can add unwanted amounts of organic and ammonia nitrogen to the effluent from the treatment facility (EPA, 1975). To assure their removal, a sequence of nitrification-denitrification has to be incorporated in the treatment system (Abufayed and Schroeder, 1986; Christensen, 1975).

When the carbon from the endogenous or the raw waste source is limited, the denitrification reaction can be enhanced by the addition of a supplemental carbon source (Bridle, 1982). Several organic chemicals including sugar and alcohol have been used to achieve denitrification (Timmermans and Van Haute, 1983; McCarty et al., 1969). The use of these compounds will significantly increase the operating cost. On the basis of cost, methanol was seen to be the most favourable carbon source, and its use is well documented.

Essentially, when the medium is oxygen free, the nitrate and nitrite are used as the oxidizing agents. Under aerobic conditions, oxygen is the preferred electron acceptor and aerobic oxidation will prevail. Studies done by Krul and Veeningen (1977) and Loehr (1984) have found an oxygen concentration of 0.2 and 0.5 mg/l to be the critical parameter relevent to efficient denitrification. Loehr et al. (1976) indicated that in an oxidation ditch treating poultry waste, denitrification occurred promptly even though

the DO level was high and nitrification took place simultaneously. This was possible because oxygen diffusion was limited by the floc matrix, which resulted in a very low or zero DO level in the floc interior.

The denitrifiers are active over a wide range of temperatures. The reaction occurs from below 5 to over 60^{o}C, with an optimum value in the range of 35 to 50^{o}C. Payne (1973) notes that possible variation in denitrifying organisms makes it difficult to establish an optimal temperature.

2.4 Biological Treatment Processes for Swine Manure

The biodegradation of animal waste is usually executed using either aerobic or anaerobic biological processes. The following subsection discusses in more detail the advantages and disadvantages of the processes commonly employed in the treatment of swine manure.

2.4.1 Anaerobic Processes

Anaerobic lagoons and digestors are the two most widely known anaerobic processes for manure waste management. The anaerobic lagoon is a very popular system because of its low capital and operating costs, low labour

energy requirements as well as simplicity and ease of operation (Humenik et al., 1980). These units provide partial stabilization of pollutants, and the reduction of total solids, COD and total nitrogen ranges are 50 to 70%, 60 to 90% and 60 to 80% respectively. A better performance can be expected for units properly designed and operating at detention times in excess of 200 days for temperatures above 15^{o}C (Sweeten, 1980; White, 1977). One of the major shortcomings of anaerobic lagoons is the biological degradation of manure and the production of hydrogen sulfide, ammonia and amines which cause acute odour problems (Fulhage, 1980; Nordstedt and Baldwin, 1975).

On the other hand, there is a high interest in the application of anaerobic digestion to animal waste (Sievers and Brune, 1978; Gramms et al., 1971). The major advantage of this process is the production of biogas which can generate energy under controlled conditions. The digester gas contains 60% methane, 38% carbon dioxide and 2% other gases. The potential for electricity generated from biogas is in the order of 2.6 to 6.2 kWh/day for each 1000 kg of liveweight, depending on the species (Smith, 1980; Fischer et al., 1979).

One of the factors which strongly influences the operating ability of anaerobic fermentation process is the ambient temperature. The microbial activity is a temperature

dependent process and the optimum temperature for mesophilic methane production is 35°C . Studies have demonstrated, however, that anaerobic digestors operate quite effectively at temperatures as low as 20°C and as high as 60°C (Chen and Hashimoto, 1980).

Another important factor affecting the anaerobic processes is the accumulation of volatile acids, which could drastically depress the pH and stop the production of methane, unless sufficient alkalinity is provided to maintain the pH value between 6.6 to 7.6 (Hashimoto et al., 1980; McCarty and McKinney, 1961). Furthermore, excessive alkalinity can also influence to some degree the efficiency of the anaerobic system (Jewell and Loehr, 1977). Finally, the accumulation of ammonia, which is generated from the deamination of protein in the waste stream during the digestion process, can be toxic to the anaerobic microorganisms (Kroeker et al., 1979; Robertson et al., 1975).

It is apparent that anaerobic digestion is a complex process and needs a great deal of attention during operation. Moreover, the practicality and feasibility of energy recovery from an anaerobic digestor appear to be questionable for small and medium size livestock operations in cold climates (Agriculture Canada, 1980).

2.4.2 Aerobic Processes

In comparison to the anaerobic processes, it is pertinent to note that there are some advantages and disadvantages associated also with aerobic processes (Loehr, 1984; Adams and Eckenfelder, 1981; EPA, 1979; Ingens and Day, 1966).

The advantages of aerobic processes are:

-The ability to adjust to the various environmental changes due to the presence of a mixed culture of microorganisms. It was found that an aerobic reaction could take place over a wide range of temperatures, pH, and rapidly adjust to shock loads.

-The supernatant withdrawn from the reactor is of higher quality since it contains a low concentration of organic matter and nutrients.

-It requires less operational attention.

-The sludge production is minimal under extended aeration conditions.

The disadvantages of aerobic processes include:

-The inability to produce a useful by-product such as methane gas.

- The higher operating costs associated with the aeration of the mixed liquor.

The most common forms of aerobic treatment which are used in livestock operations are aerobic lagoons and oxidation ditches.

In aerobic lagoons, the oxygen is supplied by: (1) the diffusion of oxygen along the air-wastewater interface, (2) the mixing action of the wind and (3) algal photosynthesis action, which is a function of the turbidity and solids concentration. Thus to ensure a good oxygenation capacity, aerobic lagoons should be shallow in depth and have a very large surface area (Sweeten, 1980; Hart and Turner, 1965). In some cases mechanical aeration is required due to the high oxygen demand. A BOD reduction of 60 to 80% can be expected during a hydraulic retention time of 20 to 60 days at a reasonably warm temperature (Barker et al., 1980; Humenik et al., 1980; Hermanson and Koon, 1972). At very low temperatures the effect of ice formation transforms these lagoons into a storage basins instead of treatment facilities (MWPS, 1983).

The oxidation ditch was first developed in the Netherlands by Pasveer (1954) for treating wastewater

emanating from small communities and industries. Simply stated, the oxidation ditch is a modification of the activated sludge process, and it is charaterized as a completely mixed extended aeration system with a long hydraulic retention time. This process has been frequently used for the treatment of swine waste (Mills and Fenlon, 1976; Day et al., 1971; Jones et al., 1971).

With an inhouse storage and treatment system, the manure falls through slotted floors into an oxidation ditch beneath the floor. This system capitalizes on the fact that there is no need for an extra waste collection system as well as transporation equipment. Since the waste is continuously fed to the oxidation ditch, it minimizes both the temperature losses and the odour spreading problems (Merkel, 1981; Hobson and Robertson, 1977).

Jones et al. (1969) conducted a field study to determine design criteria for oxidation ditches treating swine waste. A hydraulic retention time of 50 days was judged to be the most adequate for attaining the best process performance. The treatment efficiency in these systems can be high, removing 30 to 90% of the nitrogen, up to 90% BOD and 40 to 50% of volatile solids. Sneath (1978) has demonstrated that odours can be consistently reduced when sufficient aeration is practiced.

Owens et al. (1973) and Ingens and Day (1966) have identified the problems that have arisen with oxidation ditches. These are related to the attempts to treat strong wastes of high solids content. Foaming is another nuisance with oxidation ditches (Jones and Patni, 1972). The foaming in an under-house ditch can be so great that it can come up through the slats, cover the pigs and cause death (Day et al., 1971).

At the present time very few animal production facilities utilize this technology because of the high energy costs involved (Sweeten, 1980). Hence, alternative treatment processes have to be investigated so that there is not such a strain on the resources of livestock producers.

2.5 Sequencing Batch Reactor (SBR)

The SBR is a modern version of the fill and draw activated sludge process, first developed by Ardern and Lockett in 1914. The fill-and-draw system provided efficient treatment of municipal raw wastewater. Nevertheless, the concept did not gain popularity, mainly because of the high manual operator attention required and the clogging of air diffusers during the periodic settlement of sludge (Ardern, 1927). There were some scattered efforts made by Hoover (1953) and Porges (1960) to promote the use

of the fill-and-draw method for the treatment of dairy wastes, but the experience was short lived.

During the 1970's, Irvine and coworkers (1971, 1977, 1979) reinstated the fill-and-draw activated sludge process. This resurgence was brought about by the availability of low cost electronic process controllers and improved aeration devices. The fill-and-draw process was further developed, and additional cycles were added to the basic scheme, which expanded the spectrum of treatment capabilities, and the system was designated SBR, for Sequential Batch Reactor.

2.5.1 Technology Description

The SBR system operates in stages which are carried out sequentially in the same tank. These stages are designated according to their functions as fill, react, settle, draw and idle (Irvine et al., 1977). Figure 2.3 schematically illustrates a typical SBR system operating cycle.

In the fill sequence, the reactor is gradually filled over a set period of time to a final predetermined volume. Prior to filling, the tank contains an active and sizeable organism population. Mixing and/or aeration can be

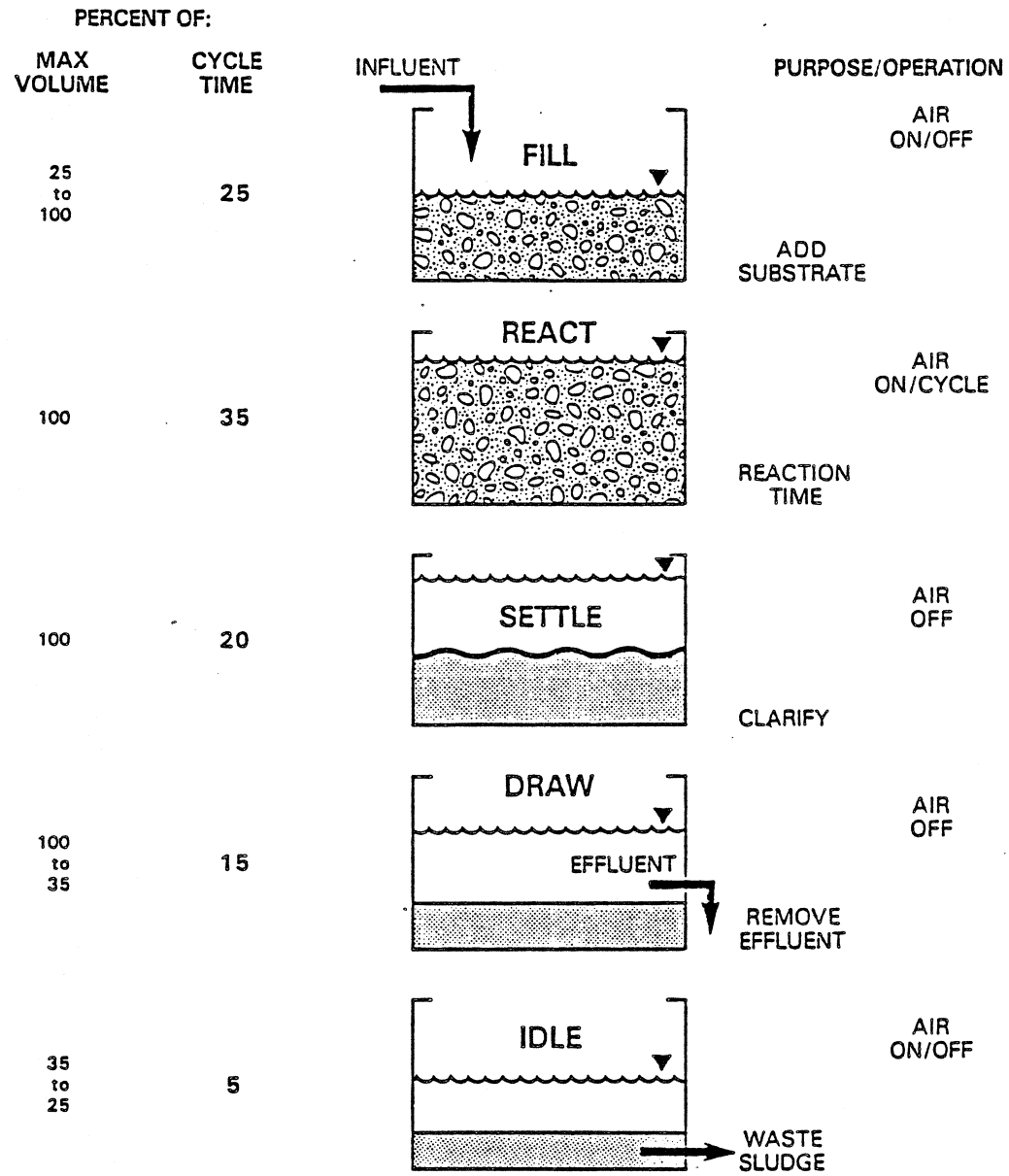


Figure 2.3 Typical SBR operating cycle for municipal wastewater
Source: Irvine et al. (1983)

supplied during this cycle. However, studies by Chiesa and Irvine (1982) have shown that filling under anaerobic conditions provides a better control on the growth of filamentous organisms, which in turn will contribute to better settling characteristics. Fermentative waste degradation by facultative and obligate anaerobic heterotrophs will likely occur with some degree of bacterial synthesis.

Irvine et al. (1983) indicated that the time of fill should be in the order of 25% of the time allotted for the total cycle. The filling time is generally dependent upon the variations in the hydraulic flow rate. As for the volume of raw wastewater to be added during fill, it could be as much as 75% of the reactor capacity, and is determined by performance requirements, the desired loading rate and sludge retention time.

In the react sequence, there is no supply of wastewater, and the reactor operates at a constant volume. Aeration and/or mixing are provided to accomplish the desired reactions. It is an important step during which waste stabilization is expected. The aerobic heterotrophic organisms utilize the organic carbon from the waste to furnish energy and synthesize new cells. The biological conversion of reduced nitrogen to an oxidized state is carried out mainly by the nitrifying species with an

adequate concentration of dissolved oxygen. An anoxic environment during the react sequence can provide suitable conditions for denitrification. According to Dennis and Irvine (1979) typically the react sequence takes up to 35% of the full cycle time, but performance demands might require substantial deviation from this average time. Towards the end of this stage, sludge wasting can be achieved as a means of controlling the sludge age and hence, the biomass concentration.

The settling sequence is conducted without aeration or mixing in order to establish quiescent conditions within the reactor. This way the mixed liquor settles, a sludge blanket is formed in the bottom and a relatively clear supernatant is obtained. The time for settling is usually fixed between 1/2 and 1 h (EPA, 1986). The sludge blanket formed should remain below the withdrawal mechanism and not be raised due to gas formation before the effluent is discharged.

During the draw sequence, the clarified supernantant is discharged. Arora et al. (1985) have reported that floating or adjustable weirs are the most popular decanting mechanisms in current use. Approximately 15% of the cycle time is dedicated to draw the effluent. At the end of this stage the volume of mixed liquor in the reactor is brought to a minimum level.

The idle sequence is used whenever the influent waste flow is not regular. It can also function as a pause in a multi-tank system, by providing time for one reactor to complete its fill 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 (Irvine, 1985). In the idle sequence, the microbial population is maintained in an endogenous phase which can be readily activated by the incoming wastewater during the fill mode.

2.5.2 Evaluation of the SBR

The SBR has its advantages and disadvantages over other types of wastewater treatment processes. In comparison to the conventional continuous flow systems which have been a traditional choice, the main advantages of SBR can be summarized as follows (Manning and Irvine, 1985; Mandt, 1985; Silverstein and Schroeder, 1983; Irvine et al., 1983; Goronszy and Barnes, 1982):

- An SBR tank serves as an equalizing tank during the fill sequence, and therefore can tolerate greater peak flows and/or shock loads of biochemical oxygen demand without degradation of effluent quality;

- Reactions that must be physically separated in continuous-flow systems, like nitrification, denitrification, phosphorus removal and decantation can be accomplished in the same tank by a single sludge biomass under a control operating strategy;

- A potential reduction in the oxygen supply is obtained by providing anoxic periods in the react phase;

- The biomass cannot be washed out by hydraulic surges because it can be held in the reactor as long as necessary;

- Solid-liquid separation occurs under quiescent conditions, and short circuiting is non existent during the settle period;

- Activated sludge recycles are eliminated, because the mixed liquor is always in the reactor;

- Filamentous growth can be controlled easily by varying the operating strategies during fill;

- The SBR is kinetically superior to continuous flow systems, because of its resemblance to an ideal plug-flow reactor. Also the content of ribonucleic acid (RNA) in the SBR is high, which permits the processing of more substrate at a greater rate;

- It is a good application for retrofitting existing plants;

The following disadvantages have been stated by the EPA (1986) concerning SBR operation:

- As the systems get larger, there is an increasing sophistication of the timing units and level sensors used to control process sequences;
- There are difficulties involved in controlling the draw phase in order to minimize the discharge of floating or settled sludge;
- Plugging of aeration equipment during settle, draw and idle periods can be a problem.

2.5.3 Application of SBR

There is an expanding market for the application of the SBR. This system is especially recommended for small communities and industries which generate highly variable load conditions of both hydraulic and biodegradable organic wastes (Goronszy, 1979; Irvine et al., 1979a). Several SBR wastewater treatment plants, ranging from 1900 to 19000 m^3/day in size, are currently operating in Australia, Canada and the United States (EPA, 1986; Barth, 1983; Schmidtke and Topnik, 1983).

Arora et al. (1985) conducted a post-construction evaluation of eight full scale SBR facilities treating domestic wastewater. All plants under consideration met the effluent requirements except one, which operated with unsatisfactory decanter equipment. Nevertheless, a striking difference was observed in the design criteria as well as the operating strategies at these plants. For example, the hydraulic retention time (HRT) varied from 7.6 to 49 hrs, the food to microorganisms ratio (F/M) was in the range of 0.032 to 0.18 per day and the computed sludge age was between 15 and 150 days. The aeration policies during the operating cycle and the time allowed for fill, react, settle and draw were significantly different. Hence, some of the reactors were oversized and power consumption ranged from as low as 0.8 to 22.9 kWh/kg BOD applied. From this study it was concluded that there are no established parameters for the design and operation of a SBR. Consequently, this has been reflected in the costs involved, and in the implementation of the system as a viable option.

In the late 1970's, the Quebec Ministry of Environment authorized the construction of several SBR systems for the treatment of effluents from small and medium size food processing industries, with particular emphasis on animal slaughter operations (Zaloum et al. 1985). Although this technology was recommended because of its many advantages, the SBR in Quebec had very limited success.

Lessard (1984) reported that out of the 23 SBR treatment facilities built in Quebec, 12 were in functioning condition and only 4 of them were performing adequately. Most of the problems encountered with these facilities were attributed to the lack of design parameters, poor understanding of the system's operation, no information regarding the performance of the system at cold temperatures and absence of post construction monitoring.

In the view of this situation, a joint study was conducted by Environment Canada and the Quebec Ministry of Environment to determine the feasibility of the SBR to treat slaughterhouse effluents. For this purpose a pilot scale 138 liter SBR was installed next to a recently constructed full scale SBR, receiving a flow rate of 40 m^3/day, at St-Louis de Gonzague.

Zaloum et al. (1985) and Belanger et al. (1985) have reported the results, covering a period of fourteen months of study, and concerning the above mentioned pilot plant and the full scale SBR operation respectively. The influent raw waste had an average BOD_5 of 1875 mg/l , 3000 mg/l COD, 550 mg/l TSS, 300 mg/l TKN and 16 mg/l total phosphorus. To assure good performance and produce an effluent within the standard guidelines (30/30 g/m^3 BOD/TSS) the following design criteria are recommended: A

HRT between 7 and 16 days and the sludge age from 15 to 40 days. The most stable conditions were attained at 11 days HRT and 30 days biological solids retention time (BSRT). Sludge wasting was done on a daily basis for better control of BSRT. A 24 hour cycle was used (feed with aeration 7 hrs; react with aeration 14 hrs; settle approximately 2.5 hrs and draw 0.5 hrs). Priorities were given to the removal rates of organic carbon and suspended solids, which were in the order of 95 and 90% respectively. The preliminary results have indicated that more than 50% phosphorus reduction and 90% nitrogen removal can be attained in this system. However, temperatures below 5^{o}C greatly affected the process efficiency.

Tam et al. (1986) investigated the effects of temperature on the SBR for the treatment of wastewater from a milking centre. A 6 hour cycle was adopted (fill 10 min, react 3.5 hrs, settle 1.5 hrs, draw 30 min and idle 20 min) The average sludge age was 20 days and the computed HRT was 20 hours. The results obtained indicate that 95% BOD_5, 88% COD, 93% NH_3-N and 95% TSS removal was achieved at 21^{o}C. By increasing the temperature to 29.8^{o}C, the process efficiency did not change significantly. For reactors operating at 3.7 and 10.5^{o}C the average removal of solids and organic matter was quite high, above 86 and 90% respectively. But the ammonia reduction was low and

erratic, since the effluent contained as much as 43% (at 3.7°C) and 56% (at 10.5°C) of the influent ammonia concentration. From this research work it appears that, compared to heterotrophic microorganisms, the activity of nitrifiers was severely affected by low temperatures.

In 1985, Lo et al. carried out experiments with a pilot-scale SBR. The suitability of this process for treating milking parlor wastewater was confirmed. For the 4 and 6 h operating cycles there was no significant difference in quality of the effluent produced. The treatment efficiency in terms of pollutants removal was: 86.5% BOD_5, 78.5% COD, 90.8% TSS and 61.8% TKN. In this research work, a very short fill sequence (40 sec) was introduced, but the authors did not discuss its effects. It is possible that any effects resulting from the organic shock loading were attenuated by the batch features inherent to this treatment system.

Lo and Liao (1987) employed a conventional SBR and a fixed film SBR (FFSBR) for the treatment of dilute dairy manure with a COD concentration between 2000 and 3000 mg/l. Three operating cycles (2, 3 and 4 hrs) and three levels of feeding (1, 2 and 3 l) were tested. For both reactors the treatment efficiency decreased with increased hydraulic loading and shorter cycle time. For the amount of wastewater treated per day (12 l) and percentage of organic removal (81

to 97%), the best operating mode was obtained in a cycle of 4 hrs with a 2 l fill which corresponded to a HRT of 10 hrs The difference in treatment efficiency between the SBR and the FFSBR was more evident at higher organic loading rates. The activity of the attached microorganisms was mainly reponsible for the better performance generally attained in the FFSBR. In this study no efforts were made towards the removal of nitrogen, and the nitrification process was not enhanced. In animal waste management the control of nitrogen forms can be of great interest.

Irvine et al. (1983), Silverstein and Schroeder (1983), Alleman and Irvine (1980a, b) and Goronszy (1979) have demonstrated the feasibility of nitrogen removal in the SBR system through nitrification and denitrification without the addition of a supplemental carbon source. To achieve this, an alternating aerobic / anoxic sequence has to be introduced during the fill and react phases.

Alleman and Irvine (1980a, and b) used the aformentioned concept to remove over 92% of the nitrogen biologically from a synthetic waste being treated in the SBR. A 9.5 hrs cycle was selected with 2 hrs for anaerobic fill, 3 hrs for aerated react, and followed by 3 hrs of anoxic react. During the aerated period the nitrogen forms are oxidized with successive formation of ammonium, nitrite and nitrate. Given the availability of the storage /

endogenous carbon source, the anoxic stage provides for conditions suitable for denitrification. Prior to settling, a brief (15 min) final aeration is necessary to strip the nitrogen gas entrained in floc matrices. It was concluded from this study that the rate of nitrification and denitrification was 0.17 to 0.2 mg NH_4+ -N oxidized/day/mg of mixed liquor suspended solids (MLSS) and 0.18-0.22 mg NO_3- N reduced/day/mg MLSS respectively.

Studies conducted by Palis and Irvine (1985) found that an alternating aerobic/anoxic sequence during the fill period did not seem to affect the activity of nitrifying and denitrifying bacteria. The reactions achieved complete nitrification on a consistent basis and it occurred as soon as air was supplied. During the anoxic operation between 86 and 94% nitrogen removal was observed. However, sludge settleability was poor and it was affected by the aeration policy used. Settling was further deteriorated when an anoxic react sequence was introduced. It is apparent that more research is warranted in this area to better understand the role of aerobic/anoxic operations in the SBR.

2.5.4 Economic Considerations

The EPA (1986) report includes the economic considerations of the SBR system. The construction, and the

annual operating and maintainance costs to handle flow rates from 379 to 18,925 m^3/day were calculated to be between $329,000 and $3,564,000. As the influent flow rate increased, the cost per unit volume for the treated effluent decreased considerably.

A cost comparison between the SBR system and a conventional oxidation ditch and activated sludge system has been presented by Irvine (1985), and Ketchum et al. (1979). A conservative cost estimate approach was utilized because the available information regarding aeration strategies and the state of sludge stabilization was limited. Even under these conditions the cost estimates as of January 1985 show the SBR system to be competitive. For example, the capital and annual operating costs for a 1,893 m^3/day facility are $201,000 and $717,000 more for the oxidation ditch and the activated sludge treatment system, respectively.

Furthermore, substantial energy savings can be made in a SBR by introducing a non-aerated period in the operating cycle without seriously impairing the effluent quality (Alleman and Irvine, 1980b). In aerobic processes it is important to consider the energy required for oxygen supply, since this can be a determining factor for the implementation of the system.

2.6 Summary of the Literature Review

A major problem facing the intensive swine industry is the management of the animal waste produced in massive quantities. According to the legislation of the Environmental Protection Service of Canada and the Provinces, small and large animal producers are required to comply with strict pollution control regulations. Hence there is the need to treat and stabilize the wastes prior to disposal in waterways. The reduction of pollutants can be attained by physical and/or biological treatment.

The physical treatment takes advantage of the density differences and the nature of the solid and liquid phases. Currently several screening devices are available with the solids removal efficiency in the range of 2% to 50%. Literature surveys reveal that the performance of these units is affected by 1) the clogging of the screen due to inadequate cleaning and slime buildup which in turn decreases the hydraulic loading capacity and increases the moisture content in the separated solids, and also 2) the relatively high energy requirements whenever vacuum or roller press features are incorporated to improve the efficiency. Consequently, it would be appropriate to develop a microscreeening unit which could not only overcome the abovementioned shortcomings, but as well augment the efficiency of solids removal, while maintaining constant

hydraulic loading and the ability to handle influent solids concentrations commonly found in animal slurries. This enables the separated solid and liquid fractions to be more easily handled and processed further.

Aerobic and anaerobic biological treatments have been used to alleviate the pollution problems. However, farmers are reluctant to utilize these technologies because of the time required for the system management, high equipment and energy costs, process inhibition at low temperatures and potential toxic components, such as antibiotics passed by the animals and heavy metals.

The sequencing batch reactor (SBR) could be an advantageous alternative. Interest in this technology has now increased rapidly and it is being applied successfully for the treatment of domestic wastewater and effluents from dairy and slaughter operations with a treatment efficiency of more than 90% in the removal of organic matter and nitrogen. In spite of the advantages of SBR referred to in the literature for some specific applications, there is still a lack of information regarding general design concepts and operational parameters. For instance, strong liquid swine waste has not yet been processed in a SBR. The existing full scale operations function in a varied manner, and to date no generalized predictive model for the implementation of the SBR exists for general waste

materials. Plus there is limited information concerning the effects of low temperature on the behaviour of the SBR process. Inevitably more research must be conducted on this treatment system if rational designs are to be made of such systems for pactical applications.

CHAPTER 3 SPECIFIC OBJECTIVES

Hypothetically it is possible to treat liquid swine manure on the farm with an integrated treatment system which incorporates a microscreening unit followed by a sequential batch reactor (SBR). Consequently a substantial reduction in the volume of polluting substances as well as the odour and water pollution impact of the effluent will occur. Thereby the produced concentrated solids require much less storage capacity and are more easily disposed of. The relatively clean water is another product which may be discharged without posing a serious threat to the environment, or can be reused for washing purposes and the like.

The specific objectives of this research were:

1. To develop a continuous belt microscreening unit for solid/liquid separation of swine waste.

2. To evaluate the performance of this unit in terms of solid separation efficiency, and to determine the physical and chemical properties of the influent slurries and those of the resulting solid and liquid separates obtained.

3. To study the effectiveness of using the SBR process for carbon and nitrogen removal from the

screened liquid swine manure.

4. To determine the benefits and effects of anoxic operation as well as the optimal rates of nitrification/denitrification.

5. To develop and verify a mathematical model for the combined removal of organic carbon and nitrogen in a SBR under various process operation conditions.

6. To establish design parameters for the optimal operation of a full scale system based on the theoretical and experimental studies.

CHAPTER 4 PERFORMANCE OF A CONTINUOUS BELT MICROSCREENING UNIT FOR SOLID/LIQUID SEPARATION OF SWINE MANURE

4.1 General Remarks

Many problems in managing livestock slurries are due to difficulties in pumping and handling a heterogenous material with poor flow properties. Separating the coarse fibrous matter from swine manure slurry would improve greatly the handling characteristics of the resulting separates. Evidently this would require two handling systems, one for the solids and another for the liquid fraction. But, it is preferable to a single unmanageable system. According to Overcash et al. (1983), Hegg et al. (1981), and Rorick et al. (1980), primary treatment of animal slurries by solid/liquid separation has a number of advantages:

- The coarse separates can be handled with conventional farm equipment.

- The recovered solids are often a useful byproduct. They can be used to refeed non-lactating animals, as a soil conditioner, for composting, livestock bedding, or even in pyrolysis.

- The reduction of the organic solid load on

subsequent liquid treatment systems will increase their service life and reduce the overall operating costs.

- Removal of organic solids from manure reduces significantly the potential of polluting watercourses.

- The liquid bulk can be sprayed by simple irrigation equipment or reused for flushing waste from barns.

Producers are very interested in the process of dewatering animal slurries. The literature review presents many common mechanical devices for solid/liquid separation based upon size restriction. The efficiency of these units is less than 50%, and they also present some operational problems which have yet to be overcome.

The purpose of this research work was to develop a continuous belt microscreening unit, with a high solids removal efficiency and which could also accept a steady hydraulic loading rate, and have the ability to handle total solids concentrations commonly found in swine waste. The microscreen was to be tested under various operational conditions and the performance evaluated in terms of physical and chemical parameters.

4.2 The Continuous Belt Microscreening Unit

4.2.1 Development of the Unit

The concept of a continuous belt separator evolved from the observation of the fact that substantial amounts of free water can be extracted from liquid swine waste as a result of gravity forces on a screen. In preliminary laboratory tests, by increasing the liquid head, and hence the pressure on the filtering material, it was found that the dewatering process improved. However, the rate of filtration by a stationary screen is quickly limited because the mesh is being continually blocked by solids build-up and the subsequent sealing of the screen within seconds for concentrated waste. Also it was observed that the maximum volume of filtrate was obtained during the first 30 seconds of exposure, for swine manure with 8% solids content, but after that the flow of liquid practically stopped due to the clogging of the screen and the formation of a compact cake of retained solids.

These observations led to the development of an experimental microscreening unit. As shown in Figure 4.1.a, this unit consists of a continuous conveyor belt of woven filter of Swiss polyester monofilament fabric (with a choice of mesh size that is easily exchangeable) which is moved horizontally at an adjustable linear speed by a 560 Watt

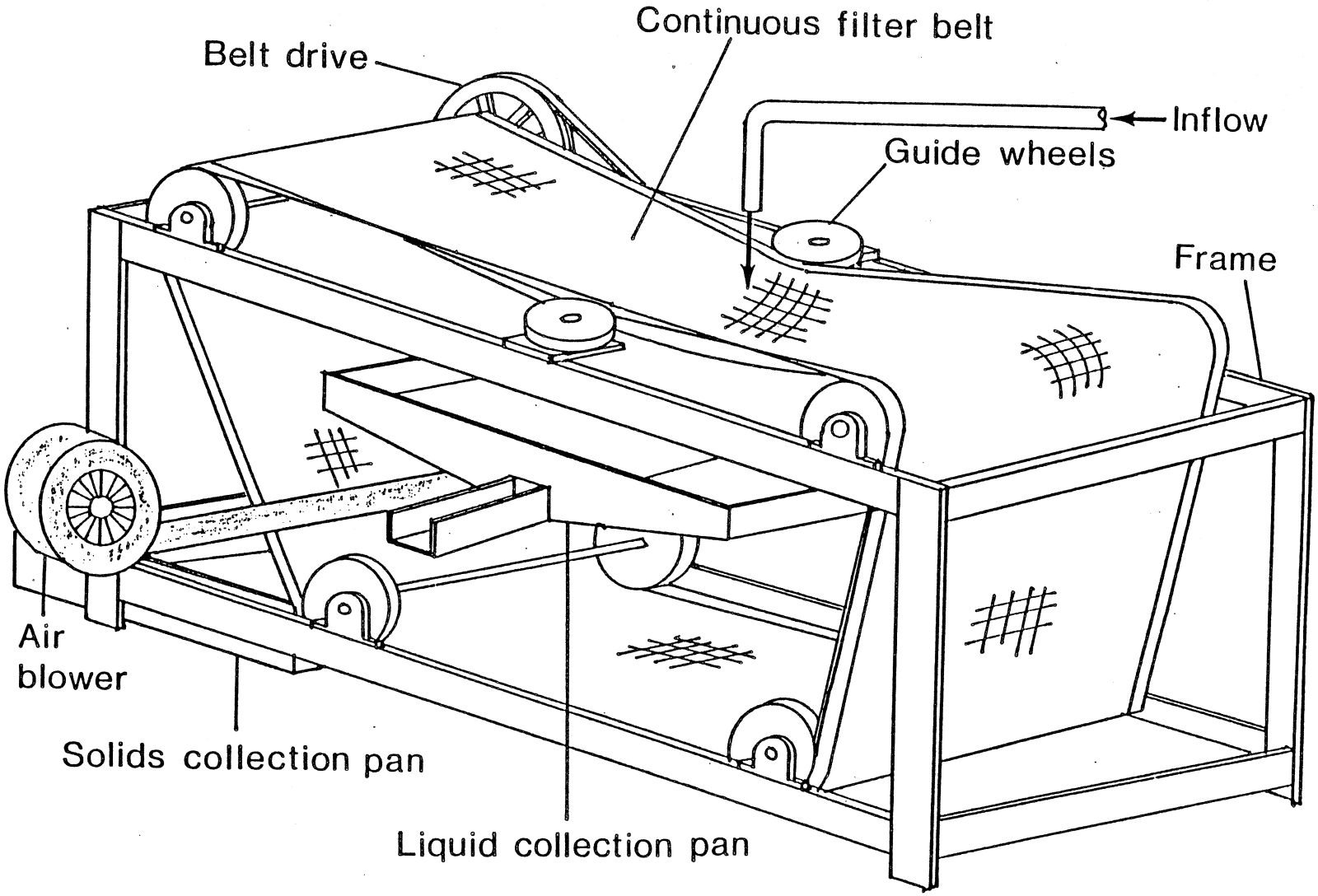


Figure 4.1 Schematic representation of the continuous blet microscreening unit

electric motor. The frame provides support and the pulleys guide the belt in a such a manner that the flat belt with a width of 400 mm is formed into a bag shape at the top, with a width of 200 mm and a maximum depth of about 150 mm.

Liquid waste is continuously discharged by a sump pump onto the belt at the place where the bag starts to form, and the filtrate is collected in a pan under the belt. The solids retained on the screen are removed by two means. Large particles and aggregates fall into a lower pan by themselves as the screen is advanced downwards and the finer particles are blown off by an air knife directed through the bottom of the belt above the solids pan. The air knife consisted of a 3 by 300 mm slot and air was supplied by an 840 Watt electric blower. In this way the filter belt is continuously cleaned and a high rate of manure separation is expected. Figure 4.1.b provides a photograph of the actual microscreening unit under operating conditions. The total cost of the components and fabrication of the single prototype unit was approximately $1500. Models produced in volume, and with a constant speed belt drive motor, would be much cheaper. Subsequently it remained to test the unit under different rates of waste loading, belt speeds and mesh opening size in order to establish its performance envelope.

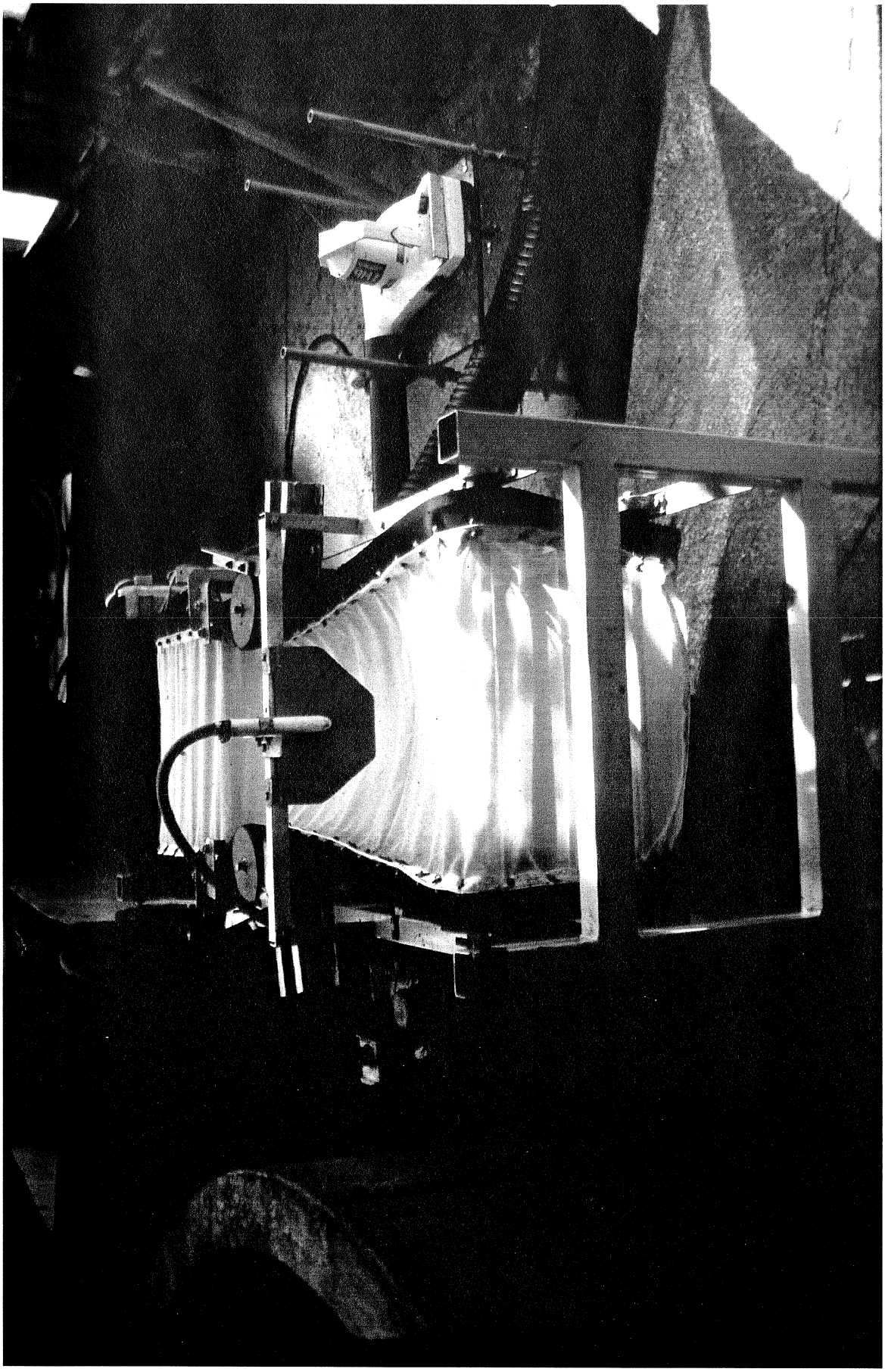


Figure 4.1b) The prototype microscreen

4.2.2 Experimental Procedure

Liquid swine manure from a growing/finishing herd was used for this study. Fresh manure was collected from the barn drain, mixed with water to give final feed materials with various concentrations of total solids and tested the same day. The microscreening unit was operated for a period of three months and between four to six hours per day. The operating variables tested were: influent total solids concentration which ranged from 2 to 8% ; wastewater flow rate was between 10 and 42 l/min; and the linear velocity of the filter belt varied from 1.25 to 5.2 m/min. The combination of the latter two variables resulted in different rates of hydraulic loading of the filter belt, between 0.01 to 0.145 $m^3/min/m^2$ flow rate per belt speed times width.

The filter fabric size opening used was 0.05, 0.1 and 0.2 mm and the temperature during the experimental work ranged between 20 to 25^oC. A minimum of two runs were carried out to test the effects of each of the various levels of the variables considered. Each run involved 300 to 400 l of raw slurry without preliminary separation on a once through basis. During the operation, a complete mixing pattern was obtained for the influent slurry by the action of a high speed impeller type mixer and a bypass return flow from the main hydraulic line.

Samples to be analyzed were collected from the influent slurry, effluent liquid and separated solids during the middle and towards the end of the run. Analyses of raw waste and liquid separate for particle size distribution was performed according to the wet method of sieve analysis adopted from Kemper (1965).

The analyses for total solids (TS), chemical oxygen demand (COD), total Kjeldahl nitrogen (TKN) and total phosphorus (P) were performed using procedures specified in APHA, Standard Methods (1985). To estimate the nutrient value of the screened solids, analyses were conducted according to AOAC (1970) to determine crude proteins (CP), ether extract (EE) and ash. The crude fibre (CF) content was calculated by difference (e.g. %CF = 100 - %CP - %EE - %ash). Wherever it was applicable, analyses were conducted in duplicates and triplicates. The choice of methods used are given in appendix C.

4.3 Results and Discussion

4.3.1 Performance of the Microscreening Unit

Generally, in solid/liquid separation processes, the efficiency is expressed in terms of percentage of solids separated from the influent slurry. It is a well known fact that screens with a smaller mesh size can retain a greater fraction of the particles being supplied to the system. Therefore the microscreening unit was tested with fabric filters which had 0.05, 0.1 and 0.2 mm size openings, and concurrently the permissible hydraulic loading capacity for each of them was established. Table 4.1 summarizes pertinent results regarding the performance of the microscreening unit. Additional results are included in Appendix A.

Table 4.1 PERFORMANCE OF THE MICROSCREENING UNIT AS A FUNCTION OF SCREEN MESH SIZE

Screen Mesh Size (mm)	Maximum Permissible Flow Rate (l/min)	Influent Solids (%DM)	Avg. Solids Removal Efficiency (%)	Dry Matter in Solids Screened (%DM)
0.05	15	7.74	62.4	12.4
0.1	35	8.2	60.5	18.6
0.2	42	7.96	45.5	21.0

All three screens demonstrated a high solids removal efficiency which ranged from 45.5 to 62.4%. The higher efficiency was evidently obtained with the smallest opening size 0.05 mm. However this screen was unable to sustain a flow rate of more than 15 l/min. Under this condition filtration occurred at a slower pace. Consequently, there was a build up of influent wastewater on the screen which restricted the movement of the conveyor belt and prevented the formation of a mat made out of the screened solids, which could act as an additional filtering medium. On average, the separated solids contained 87.6% of moisture which is considerably high, and hence it favoured the occurrence of seepage.

The use of a screen with 0.1 mm opening size permitted an increase of the hydraulic loading from 15 to 35 l/min, i.e. an increase of 2.3 times. This was possible because the influent slurry did not accumulate on the screen. Actually the height of the liquid where the screen formed the bag was about 80 mm. A drawback of the 0.1 mm screen was a slight drop in the efficiency, which was 60.5% compared to the 62.4% obtained with the 0.05 mm screen. On the other hand the separated solids contained a higher percentage of dry matter, the average being 18.6%, and this product was relatively free from seepage problems.

In the case of the 0.2 mm mesh size screen the

hydraulic loading was limited to a maximum of 42 l/min. Above this rate the conveyer belt sagged excessively just from the weight of the incoming wastewater. The efficiency in solids removal dropped abruptly to 45.5%, due to the combined effect of larger openings and the physical action between the influent flow and the screen motion. Hence these conditions were not the most appropriate to trap the finer particles i.e. below 0.2 mm, which was confirmed by the particle size analyses included in Appendix A. The solids retained on the screen were quite dry with a moisture content of 79%, which provides for good stacking properties.

In this study the screened liquid fraction was to be treated further biologically, hence the desired goal was to attain a high efficiency in the removal of solids from the wastewater without substantially reducing the hydraulic loading capacity of the microscreening unit. In this context and based on the previous evaluation, the best performance was judged as the one obtained with the 0.1 mm mesh screen, since it could handle a relatively high flow rate (35 l/min) with a 60.5% of solids removal efficiency. Therefore all the remaining experimental work was conducted with the 0.1 mm mesh screen.

4.3.2 Effect of Slurry Characteristics on the Separated Materials

For the continuous belt microscreening unit equipped with a 0.1 mm screen opening, the range of solids removal efficiency, based on the particle size distribution as a percentage of total solids, was between 47 and 59%, for influent slurry with dry matter contents between 3 to 8%, as indicated in Table 4.2. These results compare favourably with those of existing filtration units as mentioned earlier (Hegg et al., 1981; Rorick et al., 1980; Pain et al., 1978).

TABLE 4.2 PARTICLE SIZE DISTRIBUTION AS A % OF TOTAL SOLIDS

	Solids Remaining in Separated Liquid as % of Original Raw Solids Mass			
	Fresh Raw	Influent Slurry (d.m.%)		
Particle Size (um)	Manure (% of Mass)	3.0	6.0	8.0
<53	52.8	51.7	44.0	40.7
53-75	2.7	0.9	0.5	0.2
75-105	2.5	0.4	0.0	0.0
105-150	1.8	0.0		
150-250	1.3			
250-500	3.1			
500-1180	8.4			
>1180	27.4			
TOTAL	100.0	53.0	44.5	40.9
solid removal efficiency %		47.0	55.5	59.1

Note: These results are an average of 3 trials.

As shown in Table 4.2, the particle size analysis of the separated liquid fraction revealed that some particles smaller than 0.1 mm have also been removed. Should the separation process be based only on the physical size of the filter openings, then as much as 42% of solids would be retained. For swine manure containing 8% dry matter the solid removal efficiency was 59%. Thus the relative efficiency of the process measured in terms of final solids removal, improved by 1.42 times the theoretical minimum. The reason for these high results is probably the fact that as solids from the influent slurry are being discharged onto the belt screen, a thin cake of solids forms almost immediately on the filter belt, and smaller particles are trapped thereon. The thickness of this cake can be varied by changing the belt speed, and it is also a function of input slurry application rate and initial solids content.

In Figure 4.2, the results are shown on the basis of the concentration of solids removed by the screen from the influent raw waste. It also confirms that the efficiency of solids removed increased with the content of dry matter in the raw waste. The results were virtually independent of length of the test or the time at which samples were collected.

A curvilinear relationship exists between slurry dry matter concentration and proportion of solids removed.

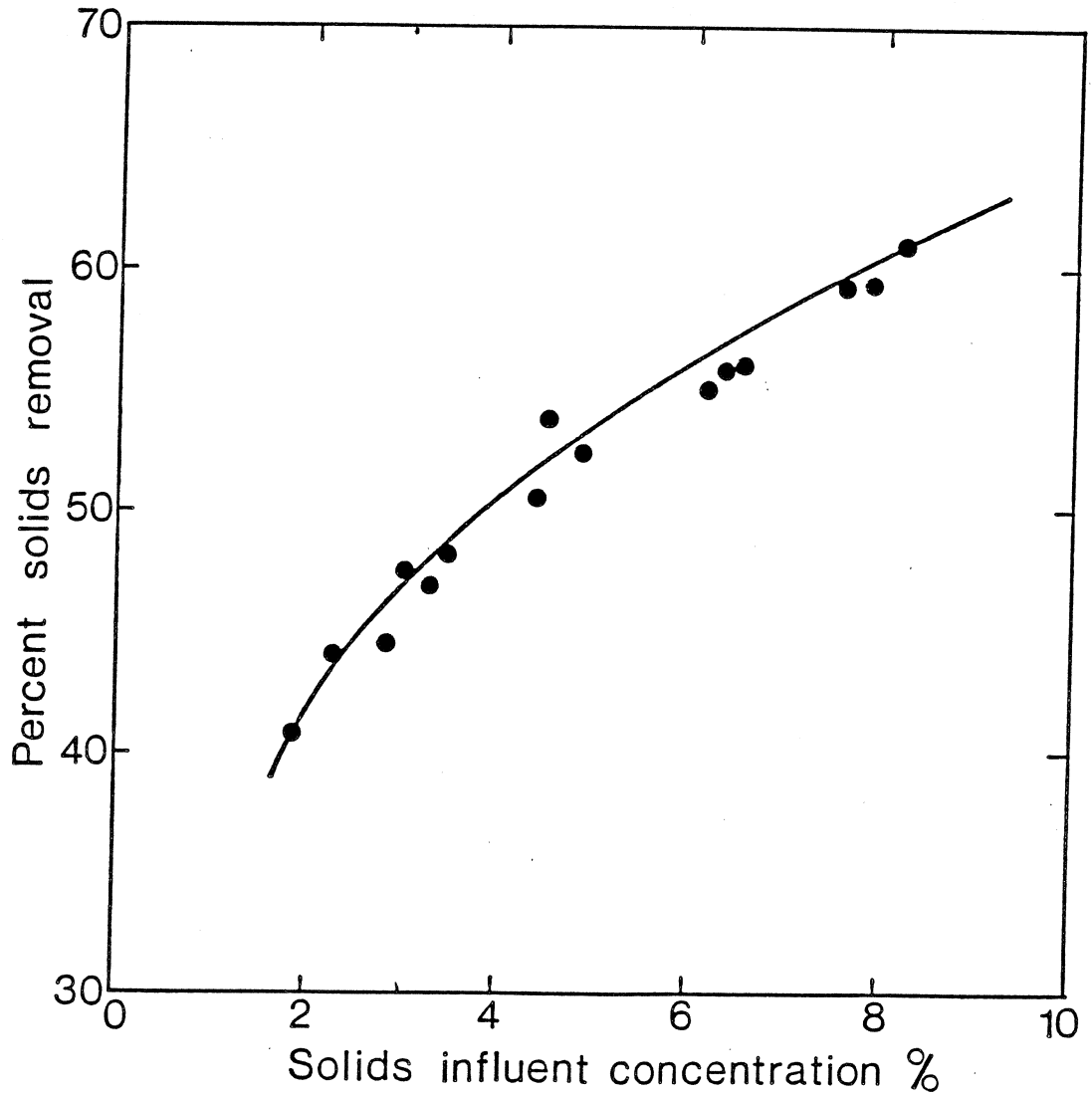


Figure 4.2 The effect of influent solids concentration on the process efficiency

It also appears that the solid removal efficiency should reach a plateau, where increasing dry matter in the slurry would not significantly improve the process. Furthermore, higher solids contents would form a semi-plastic fluid, which would require a different manner of handling. For the experiments in this study, liquid swine waste containing dry matter concentrations in the range of 2 to 8% by weight was tested.

A statistical analysis was performed on both the total solids in the influent slurry and % solids removal efficiency. The General Linear Models (GLM) program was used and the results are shown in Appendix A. The analysis of variance indicated that there is a relationship of solids removal efficiency due to the change in the influent solids concentration, and it is significant at a 0.01 level. Also the least squares method was used to determine the best fitting curve and the respective predictive equation for these data. A coefficient of correlation of 0.95 was found for this equation.

$$Y = 32.91\ X^{0.267} + 2.257 \qquad (4.1)$$

Y = % Solids removal efficiency

X = % Total solids in the influent slurry

As illustrated in Figure 4.3, the dry matter in the screened solids was found to be between 14 and 18.5%,

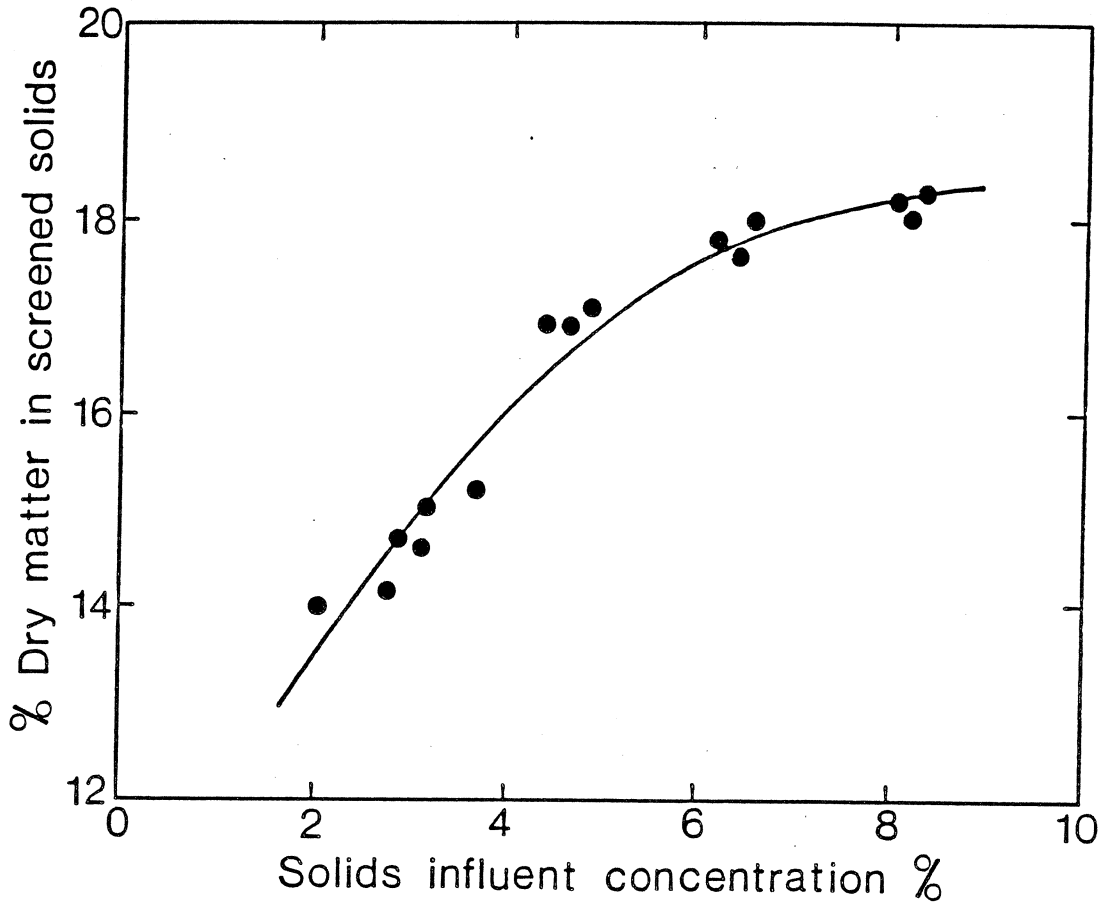


Figure 4.3 The dry matter content in the screened solids as a function of influent slurry

for influent raw manure with 2 to 8% dry matter content. It is readily apparent that there is a direct relationship between solids concentration in the slurry and that in the separated solids. Since there is no pressing stage included in this unit, the only other explanation for the improved dryness of the solids cake would be the self-pressing effect brought by increasing the amount and weight of the slurry solids on the belt.

An analysis of variance was performed on the dry matter in the screened solids and influent slurry by using the GLM procedure and the results obtained are included in Appendix A. This analysis revealed that the solids in the influent slurry had a highly significant effect at 0.01 level, on the reduction of moisture content from the screened solids fraction. By applying the least squares method the best fitting curve and the respective predictive equation with a coefficient of correlation of 0.93 was obtained.

$$S = 6.57 \times^{0.262} + 3.6 \qquad (4.2)$$

S = % Dry matter in the screened solids fraction

Visually it was observed that the solid fraction deposited in the collection pan possessed reasonable solid stacking properties, although some seepage occurred from the screened solids at higher moisture contents. This effect is

likely due to the presence of finer particles in the separated solids which have a greater surface to volume ratio, and better water absorption properties. To further dewater these particles without drying, it would be necessary to apply some type of suction force or roller pressing.

4.3.3 Effect of Hydraulic Loading and Screen Velocity on the Process Efficiency

Figure 4.4 displays the relationship between a wide range of hydraulic loadings and the unit efficiency in terms of solids removal. For input solids loading of 3, 6 and 8%, the average solids removal efficiency was 47, 55 and 59%, respectively. These experiments clearly demonstrated the capacity of the microscreening unit to handle slurries with solids concentration commonly found in swine operations in Quebec.

Within the tested range it was apparent from Figure 4.4 that compared to the influent solids loading, the hydraulic load did not seem to affect the performance of solids removal. The statistical GLM procedure was used for the analysis of hydraulic loading data and the solids removal efficiency at three levels of influent slurries. The results included in Appendix B confirmed that hydraulic loading does not have any significant effect at 0.05 level

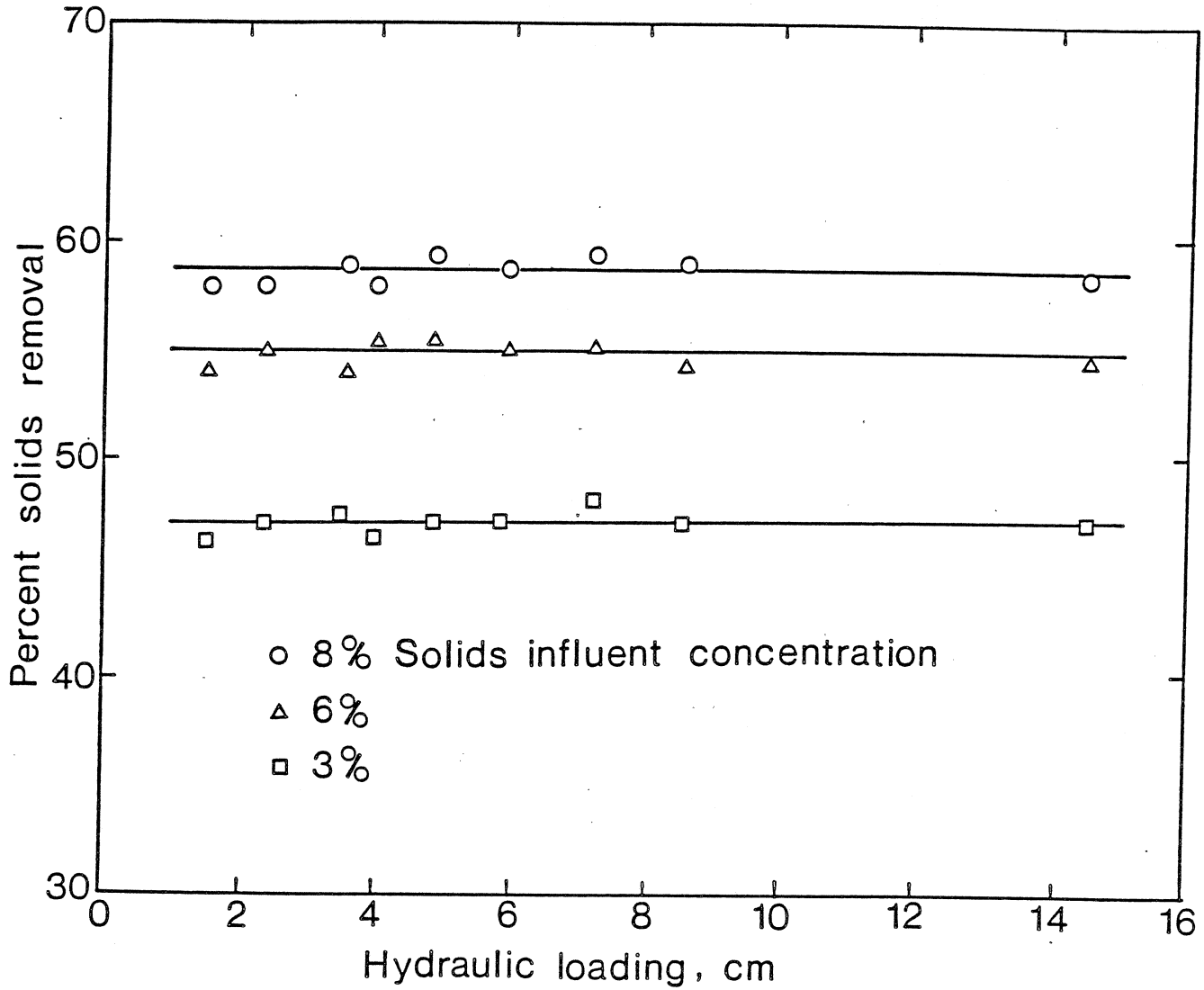


Figure 4.4 Relationship between hydraulic loading and solids removal efficiency

on solids removal efficiency. However, the influent manure concentration had highly significant effect at 0.01 level on the solids removal efficiency.

It was found that the maximum hydraulic loading within safety limits of the microsceening unit was 0.145 m^3/min per m^2/min of slurry flow rate divided by belt speed times width. The physical meaning of these units can be translated into a column of liquid of a known height. Above the refered height, the belt sagged excessively as a result of overloading. Generally, it was possible to overcome this problem by increasing the linear velocity of the belt to at least 1.2 m/min or by decreasing the flow rate applied to this device. Also by implementing minor modifications to the mechanical design of the microscreening unit, it should be possible to increase the specific inflow rate considerably.

The dewatering effectiveness of the equipment was tested by providing linear velocities for screen exposure between 1.2 to 5.2 m/min and supplying influent slurries with 6 and 8% dry matter at a flow rate of 30 l/min. For each combination of variables a set of two tests were conducted and the average results are tabulated in Table 4.3. The moisture content in the separated solids was affected mostly by the solids concentration in the input waste rather than the linear velocity of the screen. However, there was a consistent small increase in solids

moisture level as the belt speed was increased from 1.2 to 5.2 m/min, possibly because the time allowed for draining had decreased accordingly.

TABLE 4.3 DEWATERING EFFECTIVENESS OF THE MICROSCREENING UNIT

Linear Screen Velocity (m/min)	% Moisture Content of Screened Solids	
	Influent Slurry (d.m%)	
	6.0	8.0
1.2	83.9	81.8
2.3	83.5	82.0
3.5	84.2	82.8
4.7	84.5	82.1
5.2	84.4	83.0

4.3.4 Physical and Chemical Properties of the Influent Slurry, the Screened Liquid and the Solid Fraction

The separated liquid fraction can be reused for agricultural purposes as a fertilizer, but it can also be viewed as a source of pollution. Therefore it was important to determine its strength in terms of chemical parameters such as COD, TKN and P. Analyses were made for raw slurries with 4.7 and 8% dry matter and the respective screened liquid effluents. These results were obtained with a constant specific hydraulic loading of 0.145 m^3/min per m^2/min.

Table 4.4 summarizes the measured physical and chemical parameters, these values are averages of three tests. The data obtained indicate that the microscreening unit has a high performance efficiency in the removal of particles, as well as organic carbon, total nitrogen and total phosphorus. These would be of great benefit, for example, if a secondary aerobic treatment was being considered.

TABLE 4.4 PHYSICAL AND CHEMICAL CHARACTERISTICS OF INFLUENT SLURRY AND SCREENED EFFLUENT

Parameters	Influent Slurry	Effluent	% Change in Concentration
Total Solids	47,000	22,000	53.2
(mg/l)	80,000	32,000	60.0
COD (mg/l)	52,300	32,500	39.8
	85,490	52,303	38.9
TKN (mg/l)	3,500	2,380	32.0
	4,170	2,700	35.3
Total P (mg/l)	929	735	20.9
	1,910	1,570	17.8

There is a great interest in recycling dewatered fresh waste solids for refeeding purposes. The inclusion of manure in the animals diet will not only lower the feed cost but also provide a useful means of managing livestock wastes. Among domestic and wild animals, coprophagy is a

common practice. But the health and nutritional aspects are not yet well established. According to McCaskey and Anthony (1979) chemical residues, parasites and pathogenic microorganisms present in the excreta can produce mycotoxins, induce allergic responses and cause infections when fed to animals.

Proximate analyses were used to determine the nutritional value of the screened solids, as specified by Lloyd et al. (1978). Results from four tests showed that input slurry with 4.7 and 8% dry matter did not affect the nutrient content of the resulting solid fraction significantly. Hence these results were averaged and expressed in terms of percentage on a dry matter basis. The dewatered solid contained 3.1% of ether extract which represents the fatty substances. The amount of crude proteins (CP) was 10.2% and it included all of the nitrogenous compounds. The ash content was 11.0% and it is a measure of the mineral content of a feed. The crude fibre (CF) which englobes cellulose, lignin, hemicellulose, sugars, starches and other carbohydrates was determined by difference, i.e. %CF = 100% - %CP - %EE - %ash. Thus the %CF was 75.7%.

An analysis of the dewatered material showed that it is rich in crude fibre, but its proteins and fat values are relatively low. Although more than 55% of the solids

were captured on the screen (see Figure 4.2) there was still a substantial amount of nutrients incorporated in the liquid as can be observed from Table 4.4. Similar trends were found by Jett et al. (1974) when conducting an extensive study on the size distribution and nutritional value of swine manure separates. These researchers concluded that more than 83% of CP, 93% of EE and 87% of ash were contained in the portion of the manures that passed through a 0.250 mm screen. It is quite possible that the screened solids can still be used as a part of animal diet, but definitely it would have to be supplemented with other required ingredients to obtain a well balanced feed.

4.4 Summary

A mechanical continuous belt microscreening unit was developed to ease the problems related to animal waste management. The observations during testing of this unit can be summarized as follows:

- The efficiency of the microsceening unit was in the range of 47 to 60% in terms of total solids removal, over a wide range of influent slurry volume rates and solids concentrations. The unit efficiency improved at higher influent solid concentrations.

- Solids removal efficiency was observed to be independent of hydraulic loading, up to the maximum capacity of the unit, which was 35 l/min for a screen of 0.1 mm mesh opening size.

- The dry matter in the screened solids varied between 14 to 18.5 % and presented reasonable stacking properties.

- The chemical analyses of the influent swine slurry and the liquid effluent showed that as much as 39% COD, 34% TKN and 19% of total P were removed during

the solids separation process.

- The nutritional value of the screened solids was also examined. The feed content was low and it was composed of 3.1% ether extract, 10.2% crude proteins, 11.0% inorganic matter and 75.7% of crude fiber.

- The unit proved to be very reliable after more than 300 hours of operation. There was no observable deterioration of the woven filter material which demonstrated high resilience. Mechanically, this unit operated well throughout the experimental program.

- The cost of a commercially fabricated microscreen unit of the same size would be much less than currently available liquid/solid separator.

CHAPTER 5 INVESTIGATION OF THE APPLICATION OF THE SBR FOR LIQUID SWINE MANURE TREATMENT

5.1 INTRODUCTION

This part of the dissertation centers exclusively on the possibility of biological treatment of liquid swine manure using the sequential batch reactor (SBR) technique. This system was described in the literature as being efficient, easy to operate and economical for the treatment of different types of low strength liquid waste. However, the applicability of the SBR to handle strong wastes is not established. Hence the present study will investigate the following:

1. The quality of the effluent produced when screened liquid swine manure is treated in the SBR, which is operating over a wide range of hydraulic retention time and biological solid retention time. Various physical and chemical parameters are to be used to assess the effectiveness of the treatment process.

2. The behaviour of the mixed liquor in the SBR under a wide spectrum of operating conditions.

3. Nitrogen control through nitrification and denitrification processes.

4. The effect of low temperature on the process performance.

Based on these main goals, a set of recommendations will be presented for the design and operation of the full scale SBR system.

5.2 MATERIALS AND METHODS

5.2.1 Experimental Setup

Four parallel bench scale sequencing batch reactor systems were established in the waste management laboratory in the Department of Agricultural Engineering of McGill University. The reactors were acrylic plastic cylinders with an inner diameter of 140 mm and a height of 400 mm . To minimize evaporation losses, the reactors were covered with acrylic plates with openings for service connections. Figure 5.1 shows the relevant features of the experimental setup.

For each reactor the influent wastewater was pumped at the required rate, and gravity flow was used to withdraw the treated effluent. Compressed air was distributed by air diffusers installed at the bottom of the reactors. To minimize evaporation losses, as well as to remove grease and dirt, the air was previously bubbled through distilled water contained in a humidifier. The air flow was controlled by an air flow meter and regulator valves. The air flow rate was about 1.7 l/min/l of mixed liquor which permitted the maintenance of the dissolved oxygen concentration above 2.5 mg/l, and contributed to the mixing of the liquid content. Additional mixing was provided by a magnetic stirrer.

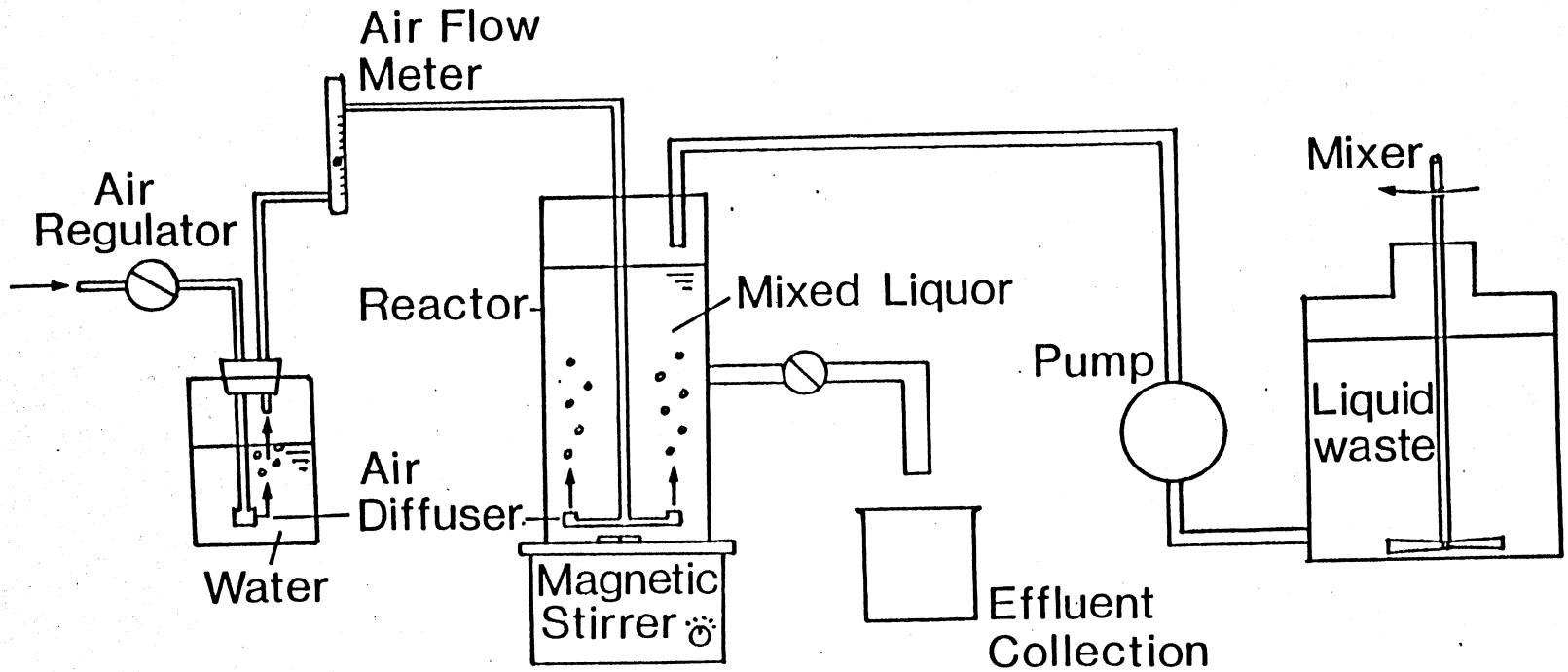


Figure 5.1 Schematic of the SBR Experimental Set-Up

The reactors functionality were checked regularly and the reactors were cleaned daily by resuspending the microbial growth which had accumulated on the inside walls. Once every week the air outlet stones were brushed clean, and tubing and related connections were cleaned by compressive rolling.

5.2.2 Liquid Swine Manure

Throughout this study, swine manure from a growing finishing herd was utilized. Fresh manure was collected from the barn drain and diluted with tap water to obtain feed material with approximately 4% total solids concentration. In operations where hog manure is handled in the liquid form, it is common for dry matter to range from 3 to 5% (Dupont et al., 1984).

Subsequently, the slurry was screened through a 0.1 mm opening size screen. For this purpose the prototype microscreening unit described in Chapter 4 was used. The screened liquid (filtrate) was frozen and stored at -20^{o}C. Prior to the treatment in the SBR system, a required volume of wastewater was thawed and brought to the SBR level of operating temperature.

5.2.3 Experimental Procedure

Initially, the biomass which was used to seed the reactor was obtained from a municipal activated sludge treatment facility at St. Rose, Quebec. For this purpose, a 30 litre sample of mixed liquor from the aeration tank was collected and permitted to settle. The supernatant was removed and the concentrated biomass was equally distributed among the four reactors. Each reactor received 0.5 l of seeding matter and 2.5 l of diluted (1:4) liquid swine manure. Over a period of thirty days the substrate concentration was increased gradually in order to permit the acclimatization of microorganisms to the incoming waste as well as to avoid shock loading.

5.2.3.1 Phase I: Study of HRT and BSRT

The first phase was carried out at room temperature to investigate the effect of hydraulic retention time (HRT) and biological solids retention time (BSRT) on the quality of the effluent and the settleability of the biomass produced during operation. The range of HRT tested was 3, 5, 7 and 9 days. The levels of BSRT were 10, 20 and 30 days. The sludge age was selected by taking into consideration the time necessary for the establishment of the nitrifying population. Also these levels of BSRT are

commonly used for extended aeration systems. The 4 (HRT) x 3 (BSRT) factorial design covered a wide range of the two independent variable settings. Following a change in BSRT, a minimum period of 30 days was allowed for the system acclimatization to the new operating conditions. Each combination of HRT and BSRT was tested over a period of 35 days and samples were taken two times a week. At the end of each trial (HRT/BSRT), track studies covering a period of 24 hours were also carried out.

Preliminary experiments were undertaken with an aerated fill sequence. However, foaming problems became acute and the sequential phasing of the SBR was set according to the following format:

MODE	REACTOR CONDITION	TIME (hrs)
Fill	Add 0.33 to 1.0 liter of waste for HRT from 9 to 3 days. Magnetic stirring on and aeration off.	3.0
React	Maximum operating volume of 3 liters. Aeration and stirring on. At the end of the react period waste 1:10, 1:20 and 1:30 of mixed liquor to control BSRT between 10 to 30 days.	19.0
Settle	Aeration and stirring off. Quiescent mode to allow the settling of solids.	1.0
Draw	Discharge between 0.03 to 0.9 liter of treated effluent.	0.5
Idle	Aeration and stirring off until next cycle begins.	0.5

5.2.3.2 Phase II: Nitrification and Denitrification

This phase was implemented to assess the potential of nitrification and denitrification in the SBR. The reactors were operated at 6 and 9 days HRT and 20 days BSRT, at room temperature. As for the SBR cycling, it was changed to accommodate the anoxic react sequence as follows:

MODE	REACTOR CONDITION	TIME (hrs)
Fill	Add 0.33 to 0.5 liter of waste for HRT of 9 to 6 days. Magnetic stirring on and aeration off.	3.0
React	Maximum operating volume of 3 liters. Aeration and stirring on.	9.0 - 18.0
	Anoxic (air off) and stirring on.	0.0 - 9.0
	Aeration to strip nitrogen gas. Waste 0.15 liter of mixed liquor.	
Settle	Aeration and stirring off. Quiescent mode to allow the settling of solids.	1.0
Draw	Discharge between 0.183 to 0.35 liter of supernatant.	0.5
Idle	Aeration and stirring off. Pause in the cycle.	0.5

After a change in HRT a minimum time of twenty days was allowed for acclimatization purposes. This phase was carried out over a period of five weeks and samples were taken twice a week. At the end of the experimental period, track analyses covering the entire cycle were conducted.

5.2.3.3 Phase III: Operation at Low Temperature

Laboratory experiments were set to determine the effects of low temperatures on the effectiveness of the SBR. The reactor operating with a sequencing schme as described in section 5.2.3.1 and 9/20 (HRT/BSRT days) was tested at 4 and 10^{o}C (+1^{o}C). The temperature was brought down gradually by dropping 2^{o}C every second day. At the desired temperature level, a minimum adaptation period of 15 days was provided before sampling. The duration of each of these experiments was of thirty days, and reactors were sampled twice weekly. In addition, track analyses over a period of 24 hours were conducted at the end of the testing periods.

5.2.3.4 Phase IV: Carbon and Nitrogen Loading Variations

The objective of this phase was to determine the effects of organic carbon and nitrogen loading on the kinetics of the treatment process. The results of this phase are used for model validation in chapter 6. The loading studies were conducted by varying the normal influent waste strength. The dilution factors used in these investigation were 1:4, 1:2, 3:4 and 1:1. A period of 20 days was allowed for the acclimatization of the system. The reactors were tested over a period of five weeks during which eight samples were collected. Also, two track analyses were

conducted during the same period of time. The SBR system was maintained in accordance with the sequence pattern and daily routine as outlined in Section 5.2.3.1.

5.2.4 Sampling and Analyses

For each of the tested operating conditions, samples were taken from: 1) the influent substrate, 2) the corresponding treated effluent and 3) the mixed liquor. During the track analyses, mixed liquor samples were withdrawn at different time intervals from the reactors by a large mouthed pipette. The mixed liquor samples were filtered and the filtrates were analyzed for various physical and chemical parameters. The following is a list of the standard physico-chemical analyses conducted during this research work. Wherever it was applicable, these analyses were conducted in duplicates and triplicates.

1. Total suspended solids (TSS)
2. Volatile suspended solids (VSS)
3. Sludge volume index (SVI)
4. pH
5. Temperature
6. Dissolved oxygen (DO)
7. Oxygen uptake rate (OUR)
8. Chemical oxygen demand (COD)

9. Biological oxygen demand (BOD_5)
10. Total Kjeldahl nitrogen (TKN)
11. Ammonia nitrogen (NH_3 -N)
12. Nitrite-nitrate nitrogen (NO_2- NO_3- N)

The filterable constituents (COD, NH_3-N and NO_2-NO_3-N) from the mixed liquor were determined after passage of the sample through a glass fiber filter. All samples were stored and analyzed according to the procedures outlined in Standard Methods (APHA, 1985). The methods used are listed in Appendix C.

5.3 RESULTS AND DISCUSSION

5.3.1 Summary of the SBR Operating Strategy

The single-batch system, sequencing on a daily cycle, was selected because of the operating simplicity and the ease with which it can accommodate the wastewater flow pattern generated by animal production operations.

The fill sequence was conducted without aeration. This condition permitted the elimination of an excessive foaming problem, which appeared to arise from aerating the high strength organic waste. By not providing air during the fill period, some of the influent substrate can be metabolized under anaerobic conditions. This will lead to a decrease in the organic load within the system during the aerobic biodegradation.

Preliminary experiments carried out with an aerated fill sequence generated large amounts of foam unable to be contained within the reactor. Loehr (1984) and Merkel (1981) previously reported that foaming was objectionable when animal wastes were treated in an oxidation ditch. The use of anti-foaming agents will not only aggravate the treatment costs but it may also cause other undesirable side effects, such as deterioration of the effluent quality.

This experimental study demonstrated that the SBR system with a non-aerated fill sequence can be an effective method to control the foaming nuisance resulting from the aerobic treatment of animal wastewater. Other benefits which can be derived from this strategy include:

1) Energy savings,
2) Substrate removal, and
3) Restricting the growth of aerobic filamentous microorganisms.

Generally, the times allotted for the sequences of react (19 hours) and settle (1 hour) were adequate to accomplish the desired reactions which included: 1) the removal of much organic carbon and nitrogen, and 2) the settling of biological solids. It is important to stress that the process performance was adversely affected by factors such as: 1) short hydraulic retention time, 2) low temperatures and 3) other factors to be discussed below.

For those reactors which displayed good settling, the biomass in the mixed liquor flocculated and settled out rapidly by forming a dense blanket with reasonable shear resistance. This made it possible to decant the effluent by gravity without disturbing the bottom sludge layer. The treated supernatant was withdrawn at the rate of 0.1 to 0.3 l/min during the draw sequence.

The idle stage was reduced to about 30 min, this time being generally used to prepare the feed and check the functioning of the reactors for the next cycle. Since there were no specific functions to be accomplished in this sequence, it could have been totally eliminated from the adopted operational cycle.

5.3.2 Effect of HRT and BSRT on the Performance of SBR

5.3.2.1 Effluent Quality

The performance of the SBR system was evaluated over a range of twelve operating conditions, each specified by one HRT and one BSRT, and from here on designated as HRT/BSRT. For every individual condition, the influent feed and the treated effluent were analyzed periodically and the results will be discussed in the following subsections.

5.3.2.1.1 Suspended Solids (SS)

Figures 5.2 to 5.4 display the average percentage removal of the suspended solids (SS) over a period of 35 days as a function of a) the HRT and b) the BSRT.

At the 3 days HRT, it was evident that the SS removal was relatively poor for all three BSRT examined. The

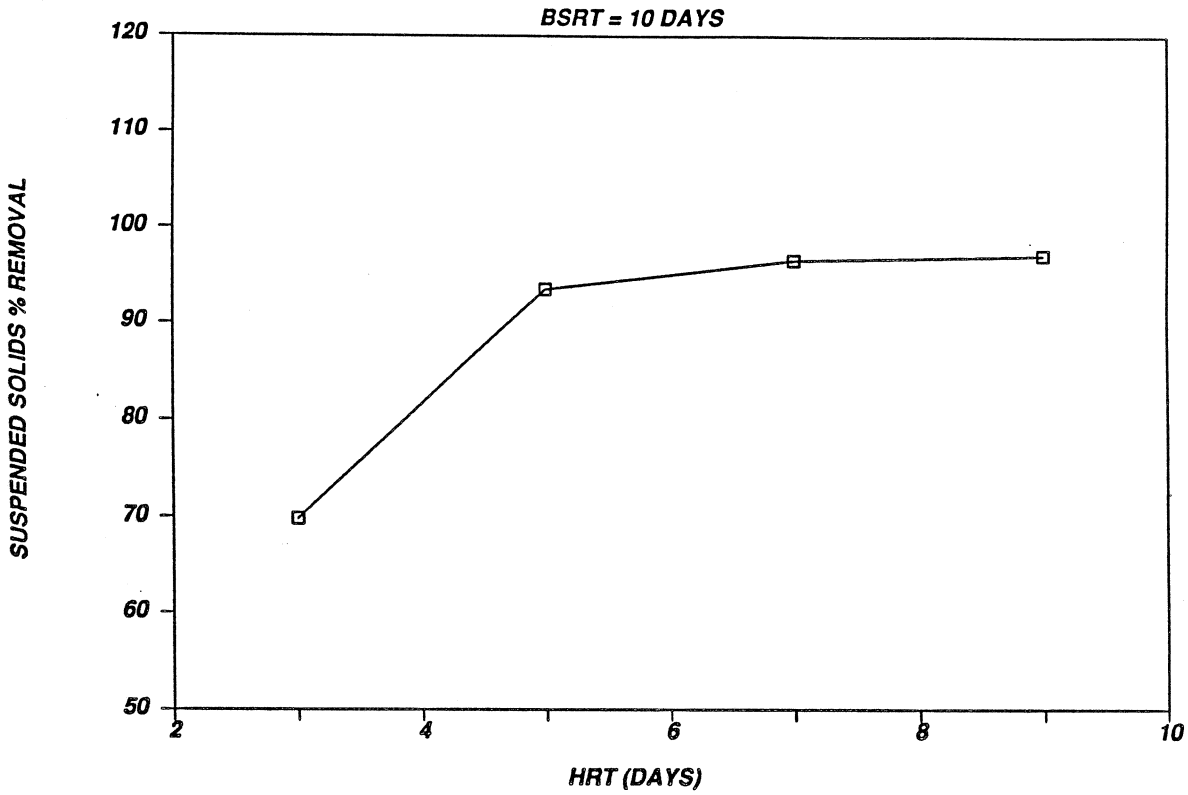


Figure 5.2 SS removal as a function of HRT and 10 days BSRT

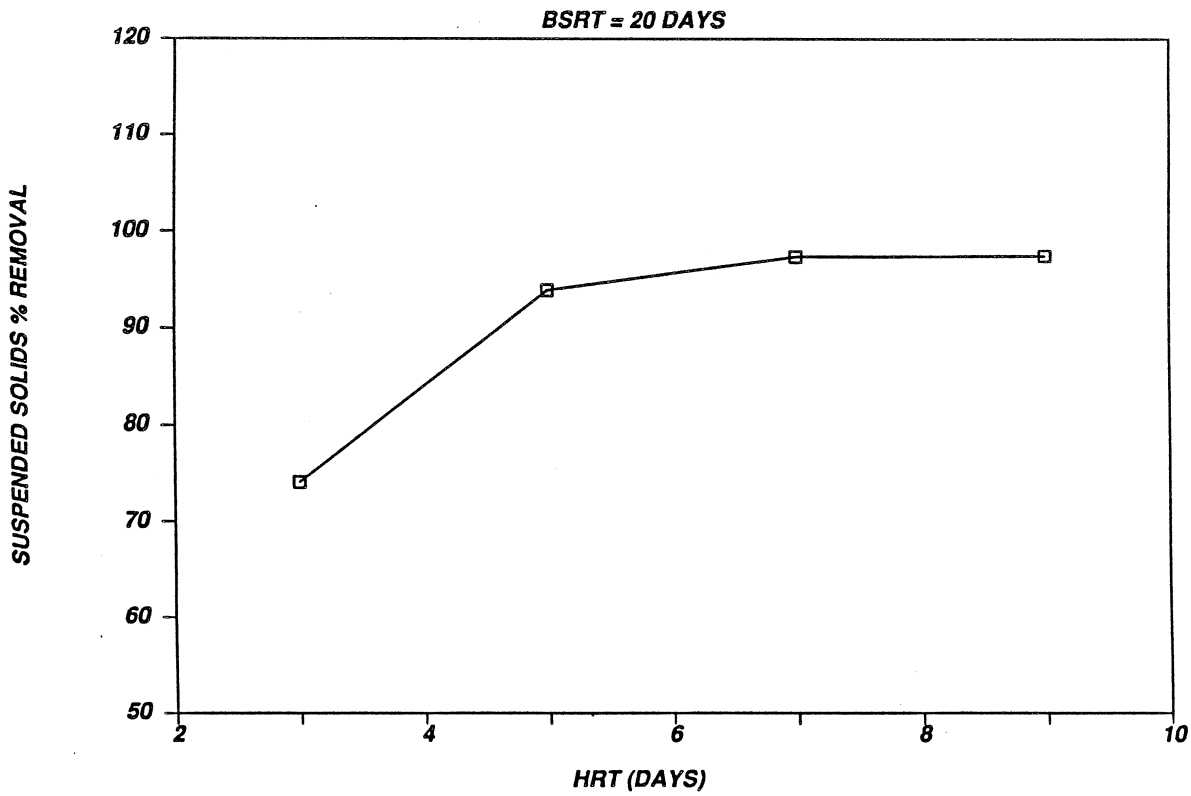


Figure 5.3 SS removal as a function of HRT and 20 days BSRT

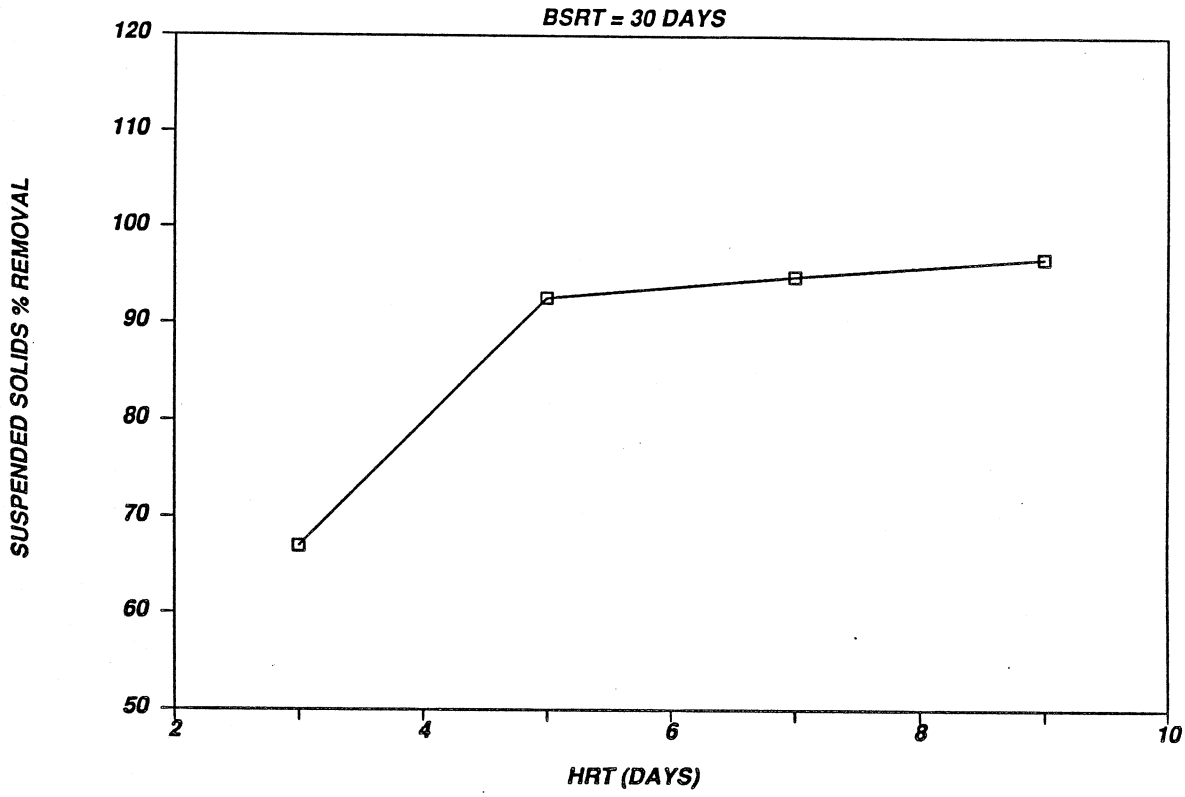


Figure 5.4 SS removal as a function of HRT and 30 days BSRT

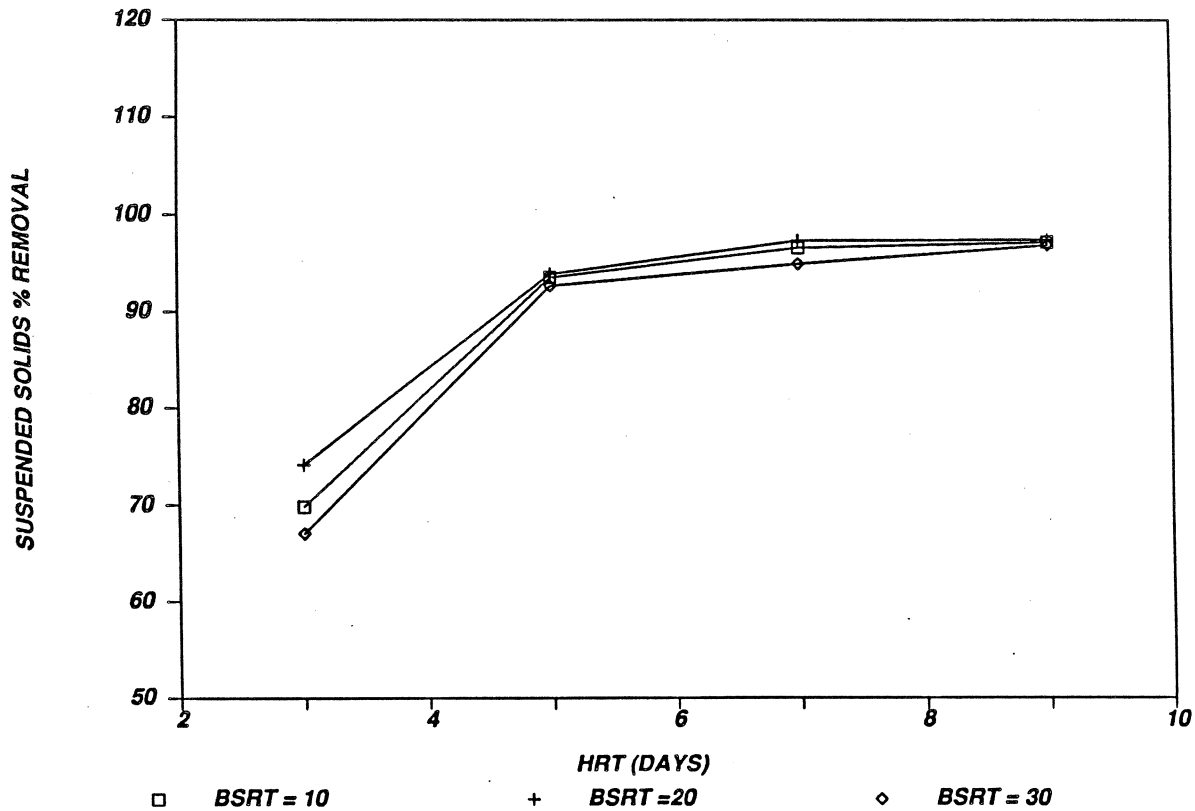


Figure 5.5 Effect of BSRT and HRT on SS % removal

SS percentage removal was found to be in the range of 67 to 75%, with the average value at 70%. In addition, an oscillation of the SS removal behaviour was observed throughout the testing period, this variability being reflected in the large calculated standard deviation. The results obtained demonstrated the immaturity of the treatment process.

As the HRT increased, the removal of suspended solids was more stable and consistently higher, regardless of the BSRT level being tested. For conditions 5 days HRT/10, 20 and 30 days BSRT; 7 days HRT/10, 20 and 30 days BSRT; and 9 days HRT/10, 20, and 30 days BSRT, the average removal of SS was 92.6 to 93.8%, 94.8 to 97.3% and 96.8 to 97.4%, respectively. As shown in Figure 5.5 the reactor operating at 30 days BSRT displayed a distinct trend showing the effluent to be slightly of inferior quality compared to the effluents from the reactor operating at 10 and 20 days BSRT. This obsevation could be attributed to the high concentration of microorganisms present in the latter reactors and their physiological state. At longer sludge ages, and under food limiting conditions, more living cells will lyse and release soluble and non-soluble material to the medium, and thus deteriorate the effluent quality.

Based on the previous observations, one can see that regardless of the biological solids retention time,

prolonging the HRT from 7 to 9 days did not change the SS removal efficiency to any appreciable extent, and hence it will not justify the larger reactor volume that would be required in the latter case. On the other hand, it is worthwhile to mention that a HRT period of 7 days and a BSRT of 20 days appears to be the most promising combination for the removal of suspended solids (removal rates were observed to be up to 97.3%).

The organic fraction in the effluent suspended solids was determined by volatilizing these residues at 600^{o}C. The volatile solids analysis is commonly used to measure the biological stability of the wastewater. A high ratio of volatile suspended solids (VSS) to total suspended solids indicates that the residue contains a large percentage of biodegradable organic matter and less of inorganic or inert material.

In this study, the VSS/SS was determined for all the tested conditions and the data are included in Appendix B. The average VSS/SS ranged from 76% to 82%. The lower values were obtained for reactors operating at 7 and 9 days HRT/10, 20 and 30 days BSRT, which means these treatments were more effective in reducing the organic load. Comparatively, reactors operating at 3 and 5 HRT/10, 20 and 30 days BSRT released more organic matter into the effluent,

which indicates a less efficient treatment. However, the performance of the SBR system should not be judged only on the basis of VSS reduction. Other parameters such as COD and NH_3 should also be investigated.

5.3.2.1.2 Chemical Oxygen Demand (COD)

The chronological effluent data for COD are shown in Figures 5.6 to 5.8, which display the reduction of the influent organic carbon. During the 35 days of the sampling period, it was evident that an erratic behaviour occurred in the reactors operating at 3 days HRT, regardless of the BSRT examined. It is believed that high organic substrate loading and a maladjusted microbial population were mainly responsible for the poor quality effluent. The percentage of COD removal ranged from a low of 70% to a high of 88%, with 80% being the average value. The effluent quality also deteriorated because some biomass was present in the effluent, due to poor settling of the mixed liquor.

The effluent quality improved substantially at 5 days HRT/10, 20 and 30 days BSRT. Under these conditions, the reactors were able to cope better with the strength of the influent wastewater. The COD removal ranged from 92 to 95% with the tendency to stabilize at 94%.

Figure 5.9 illustrates the effect of BSRT on the

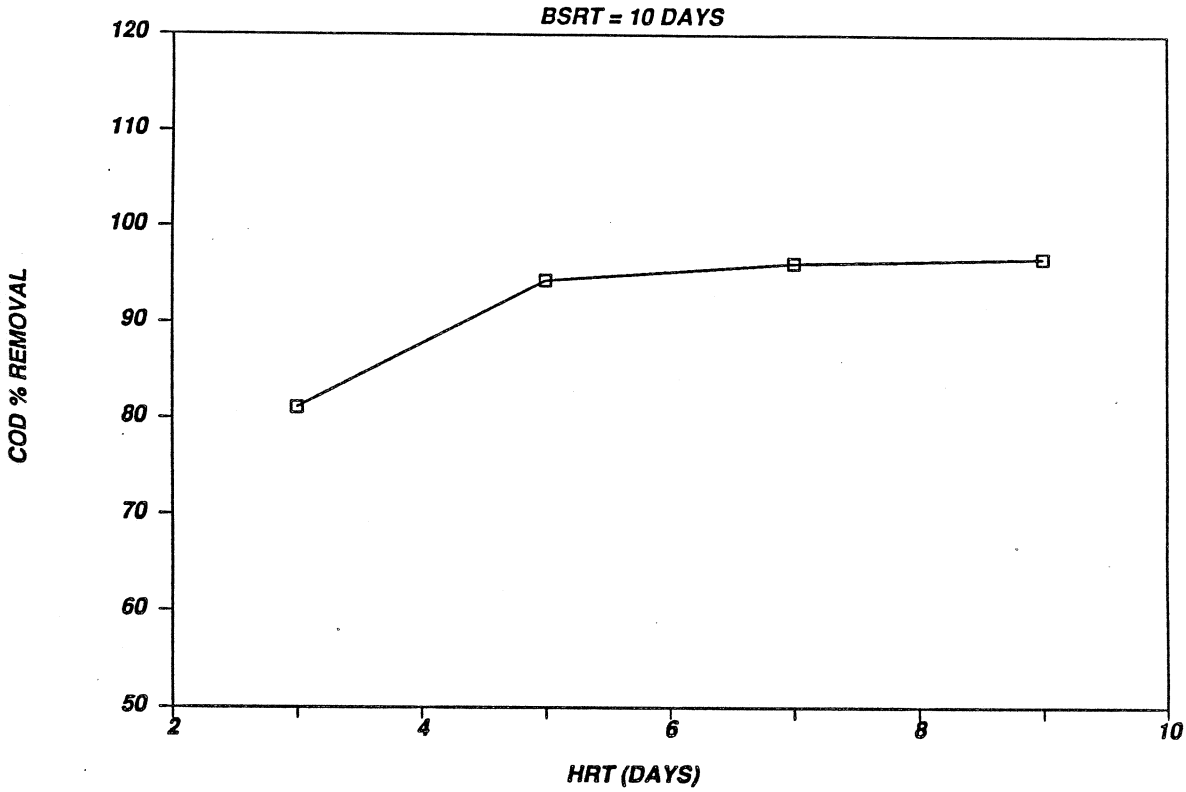


Figure 5.6 COD removal as a function of HRT and 10 days BSRT

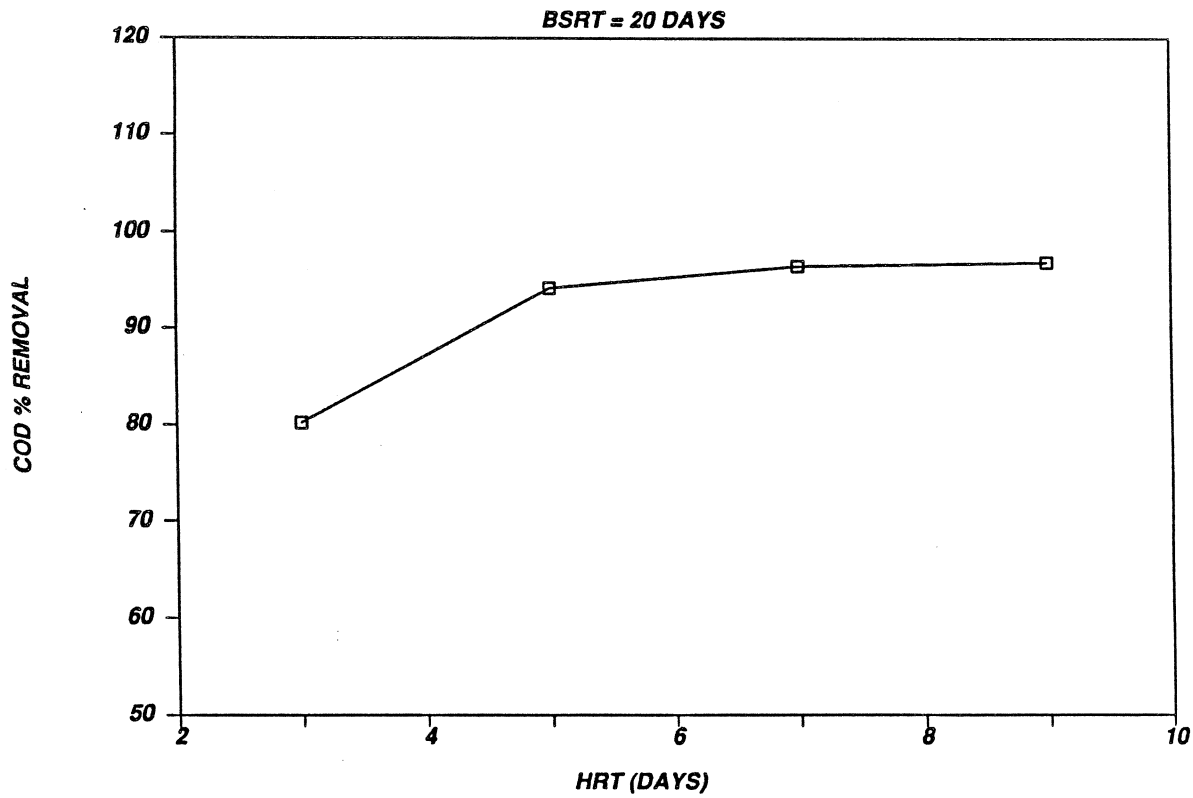


Figure 5.7 COD removal as a function of HRT and 20 days BSRT

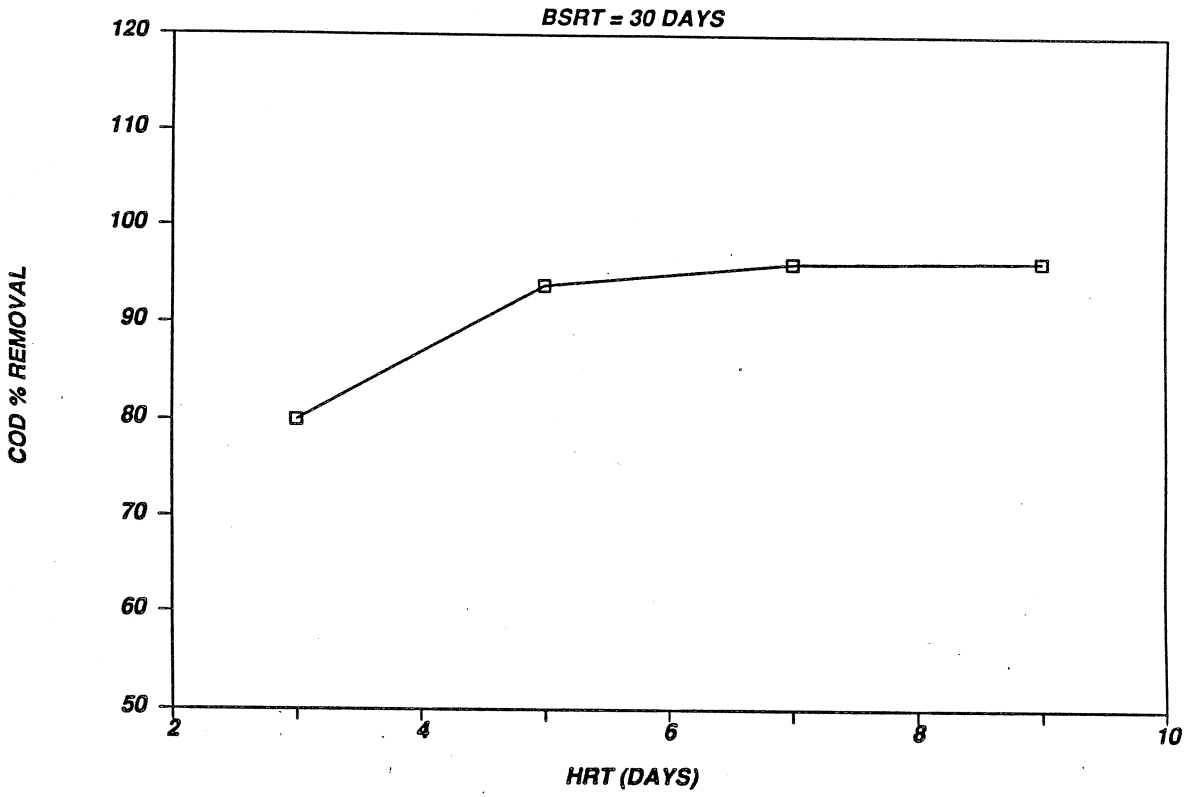


Figure 5.8 COD removal as a function of HRT and 30 days BSRT

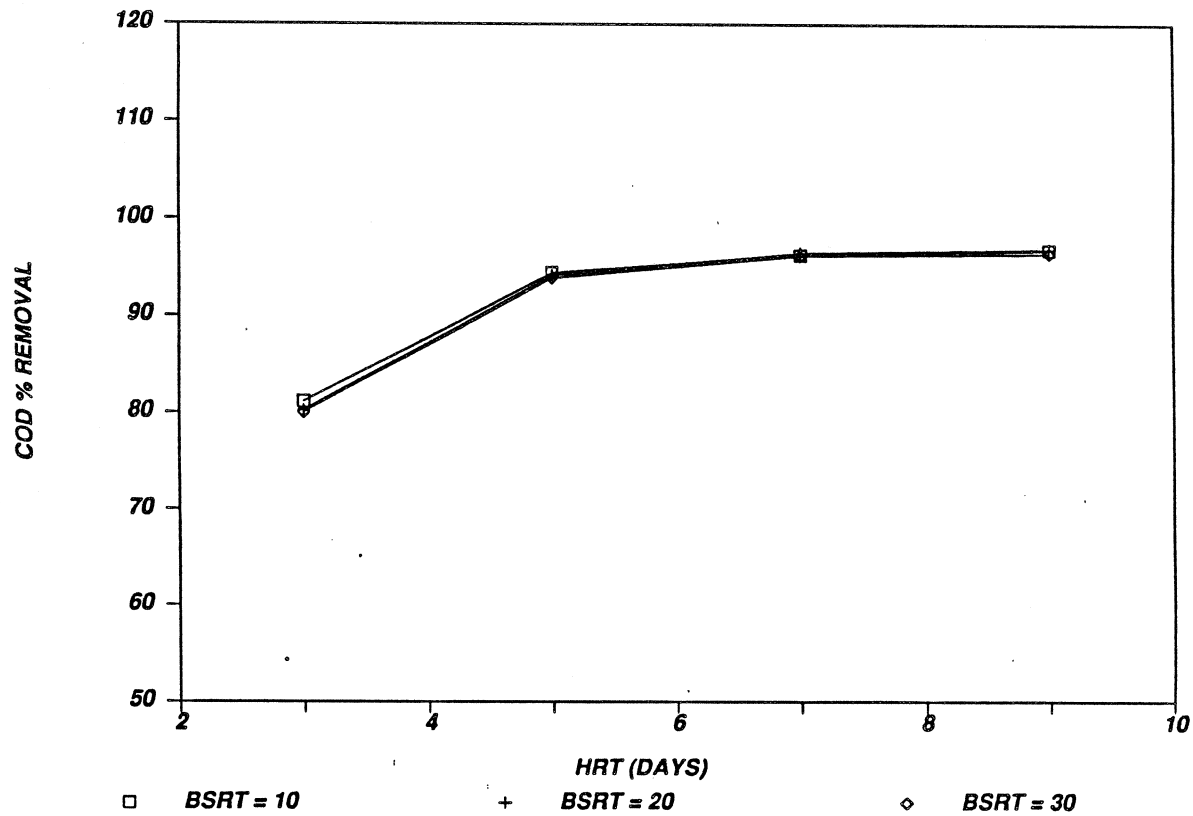


Figure 5.9 Effect of BSRT and HRT on COD % removal

process performance in terms of COD reduction. For all the tested levels of HRT, the change in BSRT between 10 to 30 days did not appear to have much influence on the amount of organic matter present in the effluent. Biological solids retention time is closely related to microorganisms specific growth rate, and it is a valuable operating parameter in a continuous system, such as activated sludge. However, in the SBR system the periodic batch sequence provides conditions for microbial growth to occur until it is limited by the availability of substrate. Therefore BSRT loses its importance in the SBR process, except for maintaining an adequate level of biomass in the reactor. It is also a way to remove inert materials accumulating in the treatment system, which otherwise would reduce the effective capacity of the reactor. Zaloum et al. (1985) reported also that optimum BSRT was in the range of 15 to 40 days, and outside these limits operation problems occurred and the effluent quality was low. However, their results were not fully conclusive and more research needs to be carried out on this aspect of the problem.

Almost a straight line type of behaviour was observed for those reactors operating at 7 and 9 days HRT/10, 20 and 30 days BSRT, reflecting the ability and success of SBR in reducing the organic carbon load. For the studied conditions, the COD removal achieved a value of 97% (on average). Since there was no significant difference in

the effluent quality for 7 and 9 days HRT, it would be economically more advantageous to choose the shorter 7 days HRT. It is important to emphasize the fact that although high efficiency was attained in the removal of COD, the effluent still contained a considerable amount of organic carbon. However, most of this matter would not be in a readily biodegradable state, and thus it should not pose as a serious threat to the environment.

Actually, the ratios between BOD_5 and COD (calculated in Appendix B), for the influent screened liquid manure and the treated effluent, were 0.62 and 0.38, respectively. This is a clear indication that most of the biodegradable matter in the raw waste was metabolized during the biological treatment, and hence the resulting BOD_5/COD ratio was low. This also confirms that the effluent contained a large fraction of nonbiodegradable organic matter.

The existing standards set by the Environmental Protection Service of Canada for discharging effluents in watercourses are 30 mg BOD_5/l and 30 mg SS/l. But these values are seldom attainable when primary physical and biological processes are applied for treatment of wastewater from livestock operations. Therefore, the treated effluents are generally applied to the land, in which case the

criteria for limiting the organic matter, nitrogen and other nutrients will depend on the type of soil and the crops produced thereon.

5.3.2.1.3 Nitrogen Forms: Ammonia Nitrogen and Nitrite - Nitrate Nitrogen

From the present study it was evident that 3 days HRT was not adequate for the stabilization of NH_3 - N. As it can be seen from Figures 5.10 to 5.12, for reactors operating at 3 days HRT/10, 20, 30 days BSRT, the average removal of NH_3 - N was 34%, 71% and 66%, respectively. The NH_3 -N in the effluent ranged from 227 to 910 mg/l (Appendix B). The unreacted ammonia could be ultimately hazardous if disposed of with no regard to the environment.

The reactor operating at 3 days HRT/10 days BSRT condition had the poorest performance of all the combinations tested (see Figure 5.13). The high values of NH_3 -N were presumably due to the low activity of the nitrifiers. These bacteria, which are known to be slow growers (Barnes and Bliss, 1983), appeared to be unable to flourish in the stressful medium created by a short HRT, the concomitant high nitrogen loading (average influent 1178 mg/l NH_3 -N) and the short BSRT. These observations were confirmed by the values of NO_2 and NO_3-N measured in the effluent, with

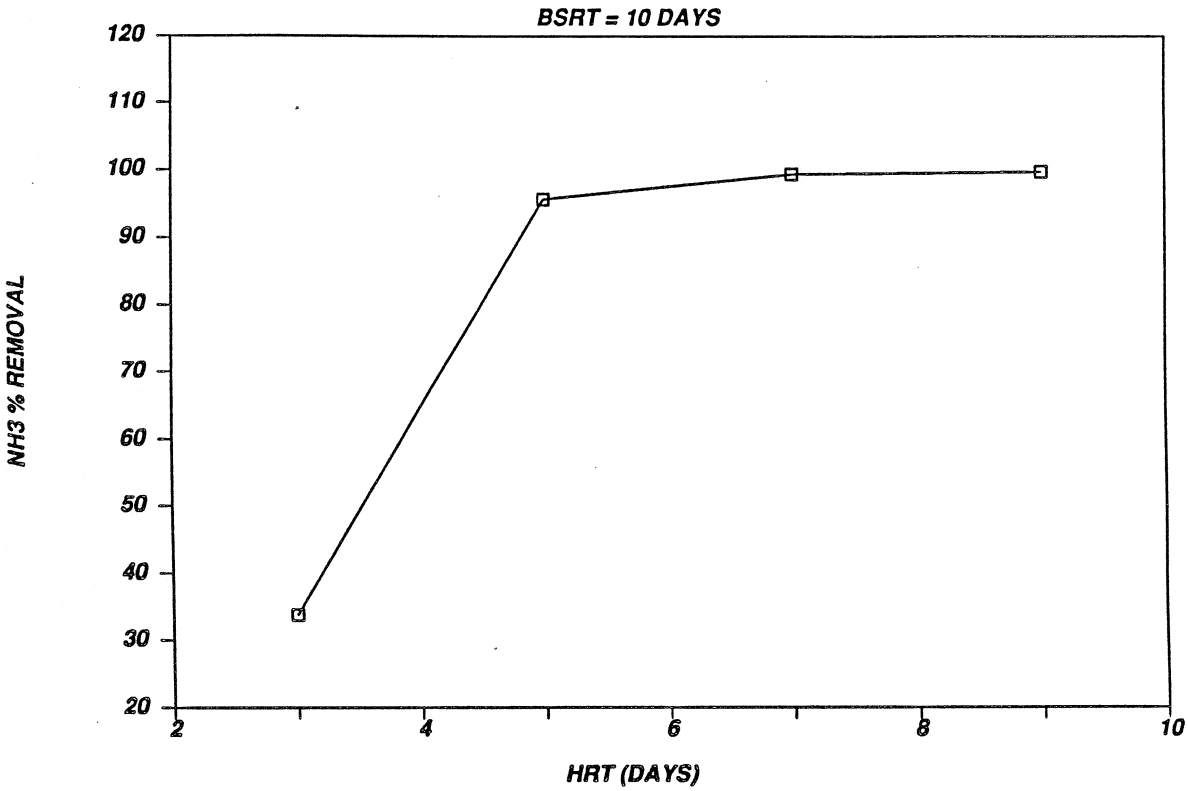
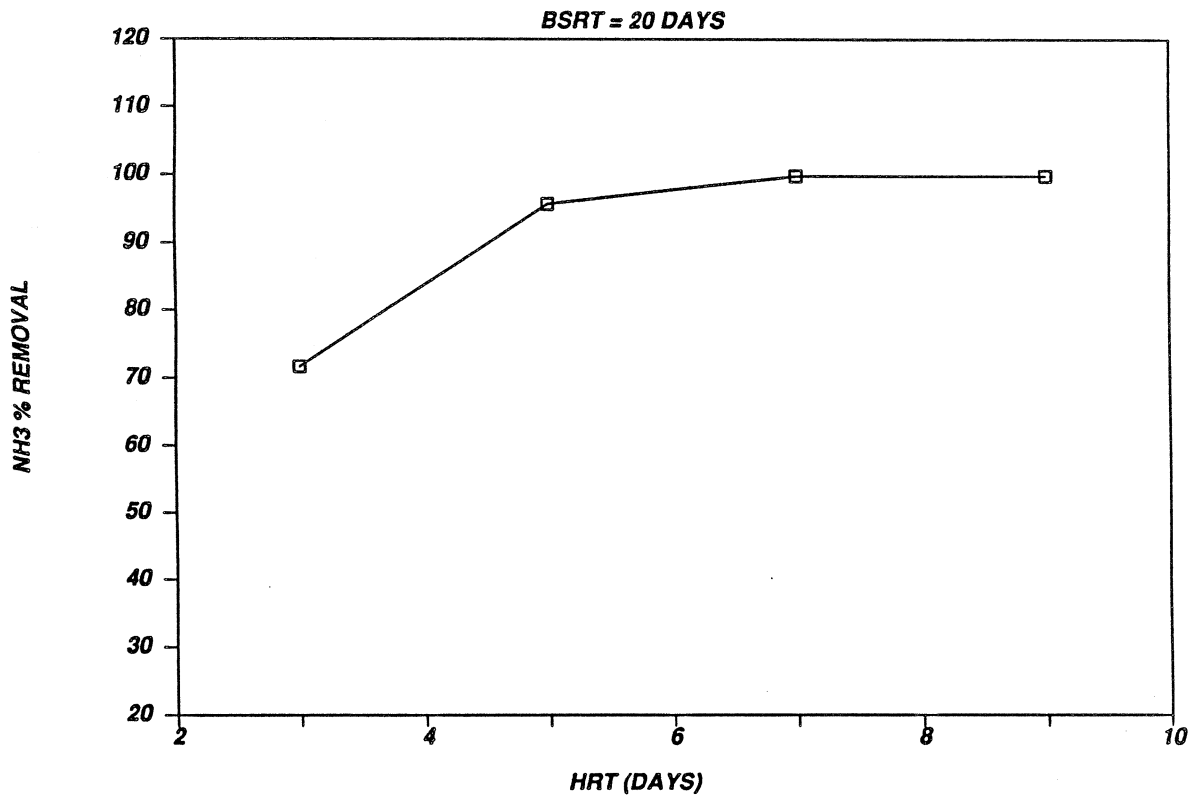


Figure 5.10 NH_3-N removal as a function of HRT and 10 days BSRT



Fgiure 5.11 NH_3-N removal as a function of HRT and 20 days BSRT

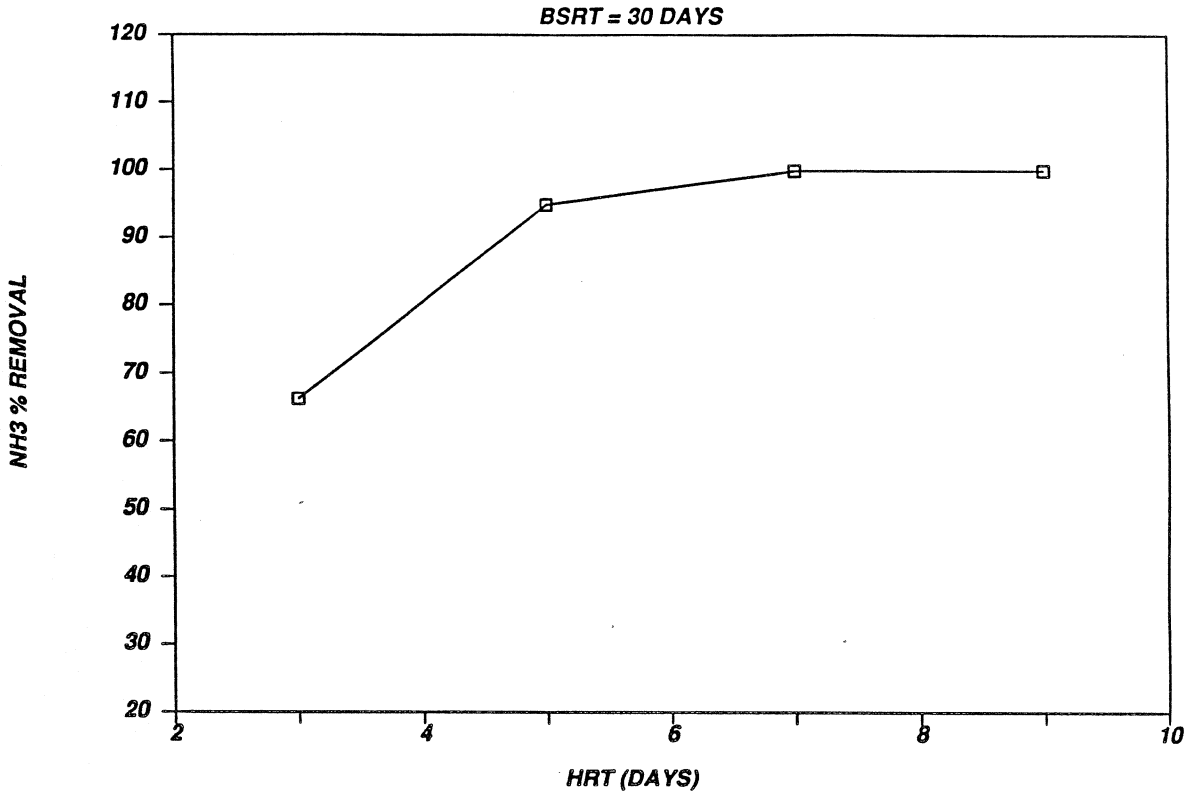


Figure 5.12 NH_3-N removal as a function of HRT and 30 days BSRT

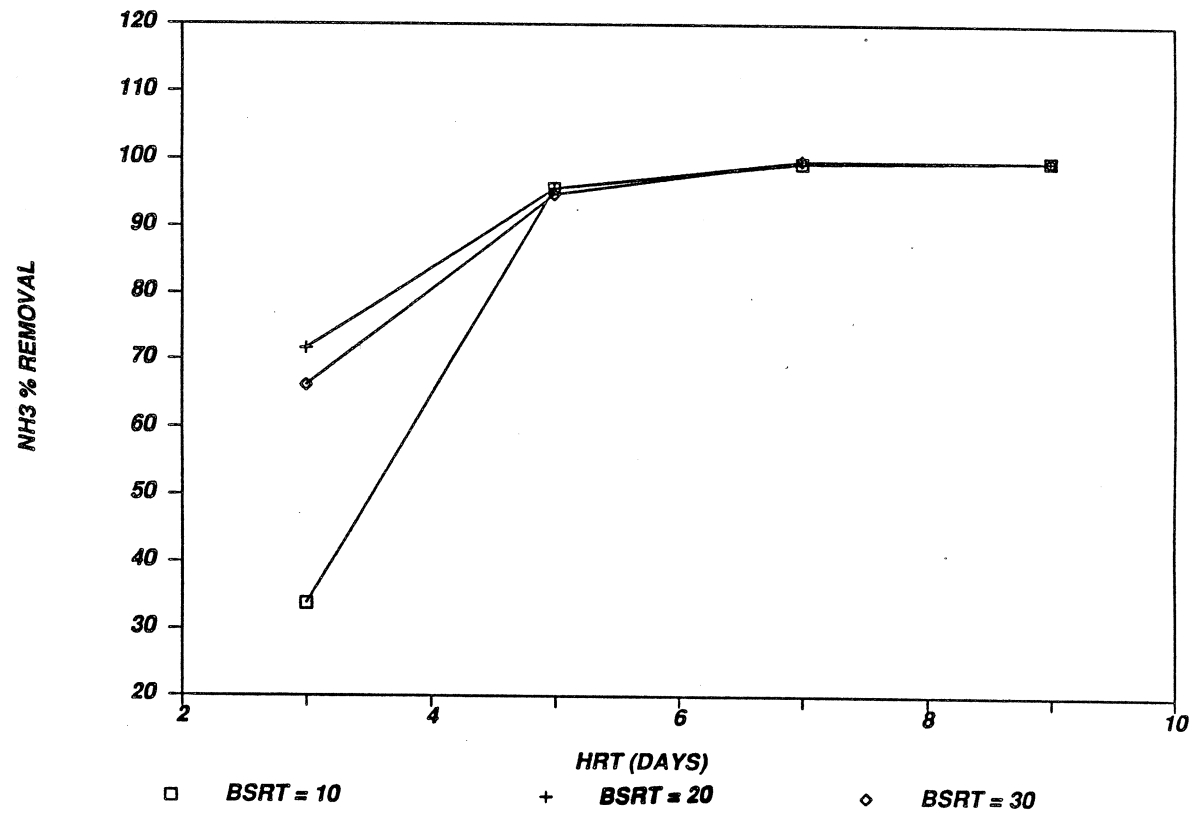


Figure 5.13 Effect of BSRT and HRT on NH_3-N % removal

an average of 12.14 mg /l (see Figure 5.14) indicating that most of the nitrogen remained in the non-oxidized form.

On the other hand, the nitrification process improved remarkably for those reactors operating at 3 days HRT/20 and 30 days BSRT conditions. As shown in Figures 5.11, 5.12, 5.15 and 5.16 the effluent produced contained in average 347 and 408 mg/l NH_3 - N, and NO_2 - NO_3 - N was 311 and 302 mg/l. From Figure 5.17 it is apparent that longer BSRT affected positively the nitrification process, and this is even more evident at short HRT. Possibly the 30 days BSRT favoured the establishment of the slow growing nitrifiers and thus, it enhanced the nitrification process. But, this improvement by itself was not sufficient to produce a fully nitrified effluent. Cearly, the nitrogen loading applied to these reactors was much too high, and possibly it affected the activity of the nitrifyers adversely.

Over 95% of ammonia nitrogen was removed from the influent treated in the reactors operating at 5, 7 and 9 days HRT and 10, 20 and 30 days BSRT. From these studies, it was evident that the hydraulic retention time (HRT) is the major determining factor in the process efficiency. As a matter of fact, for reactors operating between 3 and 9 days HRT and a constant 10 days BSRT, the efficiency in NH_3 - N reduction varied from a minimum of 33% to a maximum of 99.8%. Figure 5.13 shows that the independent variable BSRT

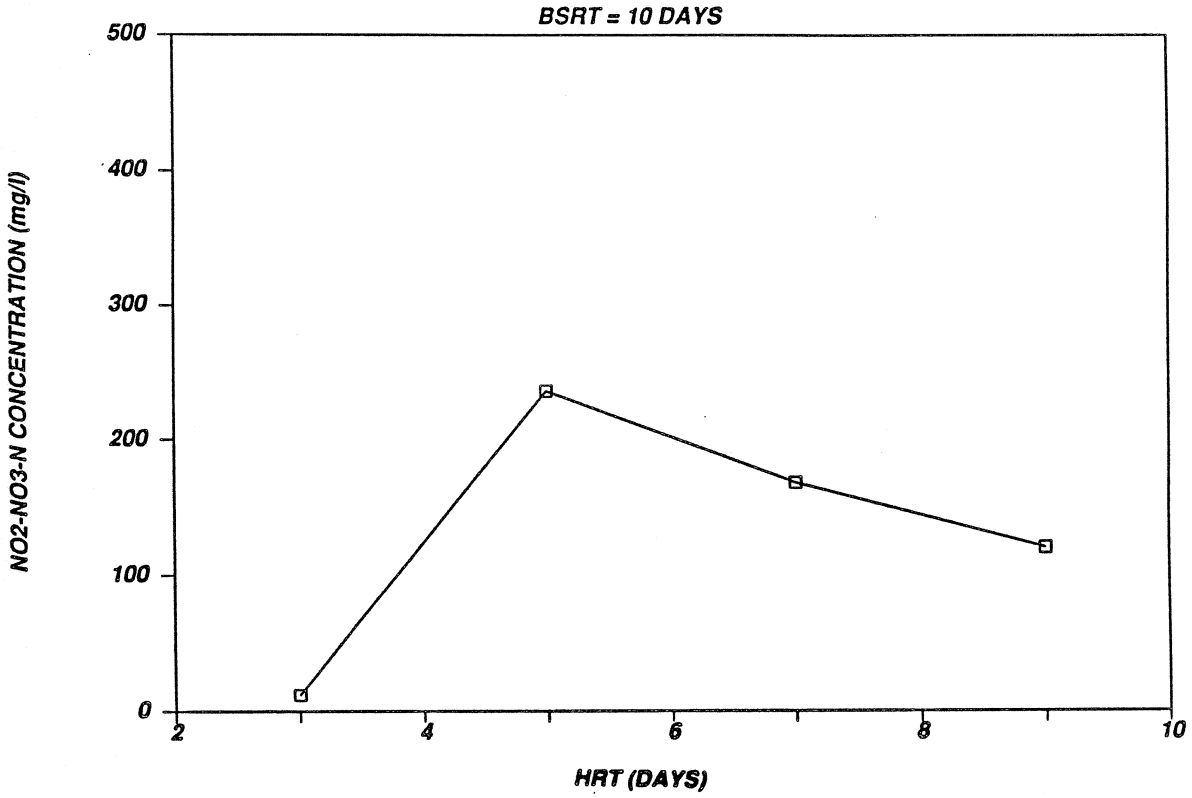


Figure 5.14 Average NO_x in the reactors operating from 3 to 9 HRT/10 BSRT (days)

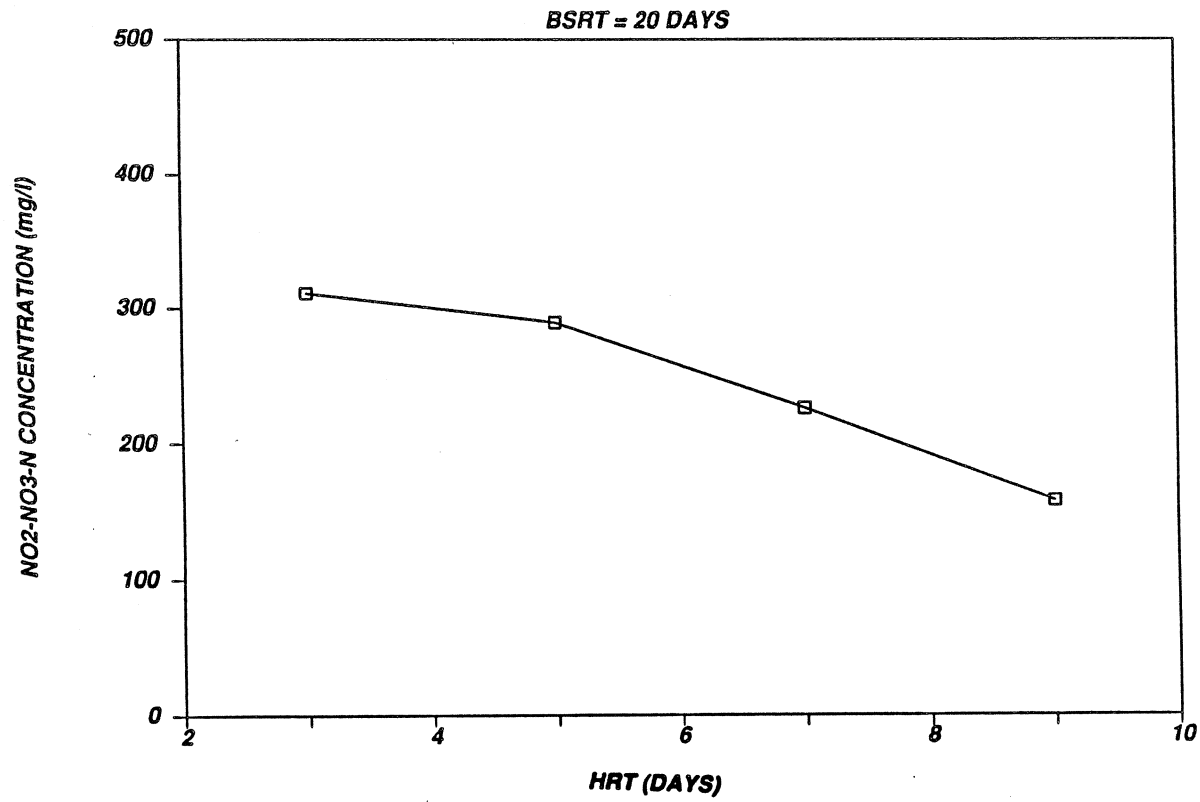


Figure 5.15 Average NO_x in the reactors operating from 3 to 9 HRT/20 BSRT (days)

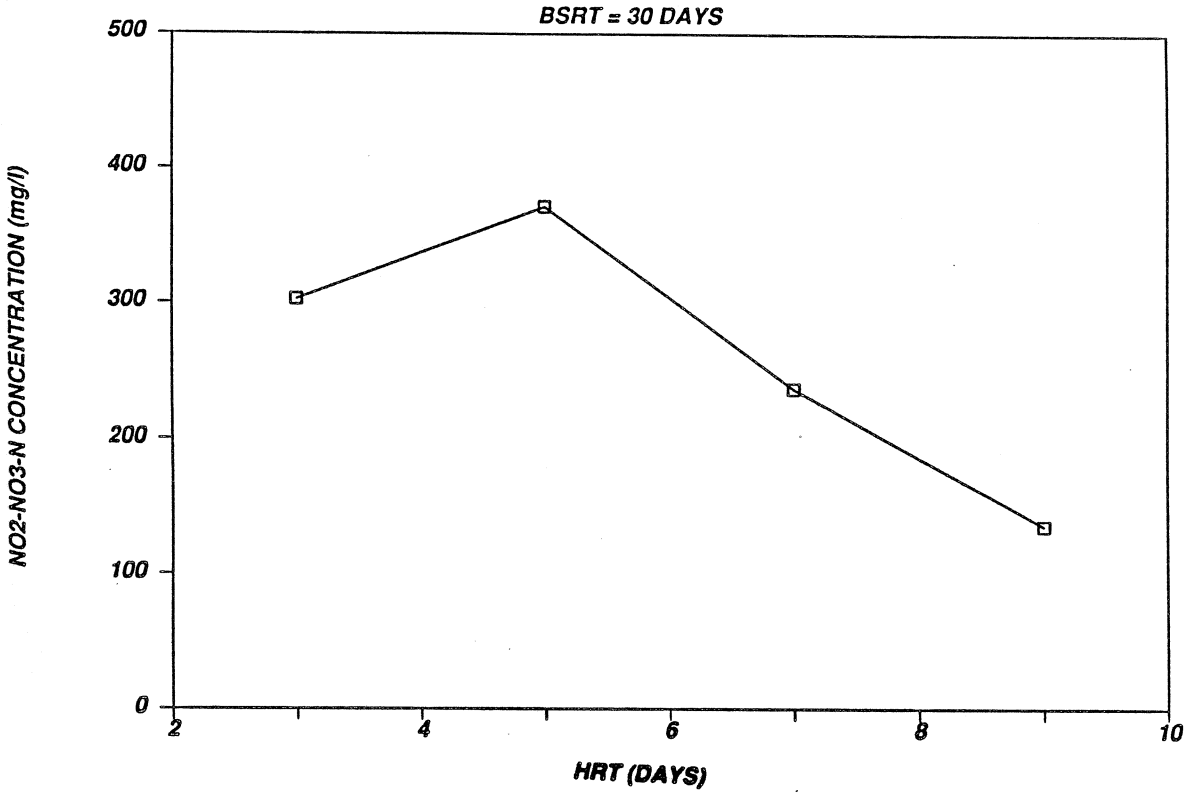


Figure 5.16 Average NO_x in the reactors operating from 3 to 9 HRT/30 BSRT (days)

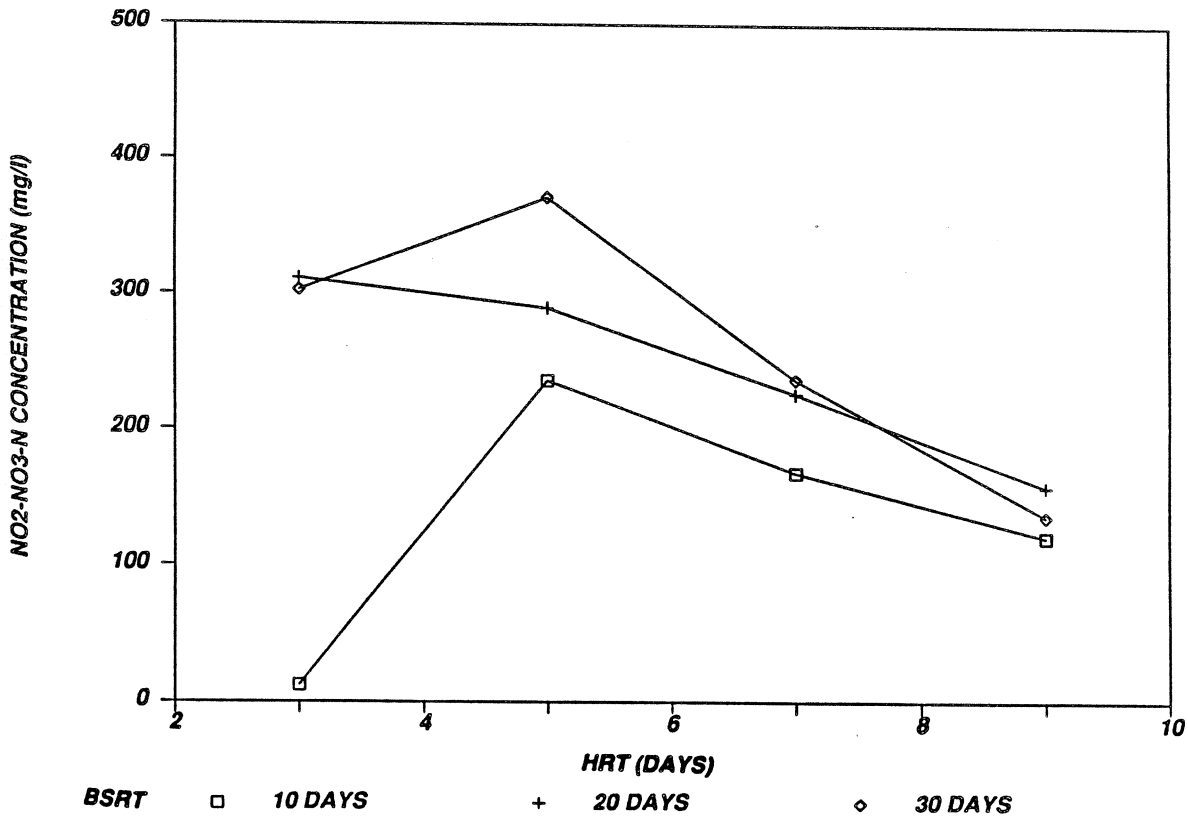


Figure 5.17 Effect of HRT and BSRT on the formation of oxidized nitrogen

has a negligible effect on the quality of the effluent when the HRT ranges from 5 to 9 days. By increasing the HRT, the substrate loading within the system decreases and therefore less stress is exerted on the microorganisms involved in the treatment process. The biological oxidation of nitrogen was carried out in all nine reactors and a highly nitrified effluent [NO_2 -NO_3 -N] concentration ranging from 120 to 371 mg/l was produced as shown in Figures 5.14 to 5.16. Especially for reactors operating at 7 and 9 days HRT the residual NH_3 -N was on average about 2.5 mg/l, this is an indication that almost total conversion of ammonia to oxidized nitrogen forms had occurred, which was one of the desirable goals of the present study.

5.3.2.2 Odour Stability

Untreated swine manure can readily develop offensive odours because of the anaerobic decomposition of organic matter and release of a complex mixture of volatile gases. Researchers have identified more than forty odourous compounds in manure gases. But amines and sulfides are considered to be the major contibutors to odour formation (Welsh et al., 1976; Merkel et al., 1969). An effective way of controlling odour nuisance is by aerobic biological conversion of waste to stable end products, such as carbon dioxide, water and inert material.

The SBR treatment system under investigation demonstrated a high capacity for odour control. Indeed the offensive smell of raw manure gradually disappeared during the fill sequence and a mild earthy odour took over. During the aerated react sequence, it was possible for sulfides to be oxidized to sulfates, and amines were oxidized to ammonia and eventually to nitrate. Thus the supernatant produced had an inoffensive odour, except for reactors operating at 3 days HRT which gave off acute smells of ammonia. This was a consensus reached by 7 people who were asked about the degree of odour offensiveness. It was obvious that the residual odour was a function of time that the waste remained in the reactor, and 3 days HRT was not sufficient to stabilize the odours produced. It was also interesting to note that for all other reactors involved the unaerated fill, settle and idle did not allow odour to regenerate, possibly due to the presence of nitrates. These act as a reservoir of oxygen that several types of heterotrophic bacteria can use for oxidizing organic matter, and thus restricting the fermentation process.

5.3.2.3 pH

In biological treatment systems it is essential to monitor the pH, because it affects the growth of microorganisms. According to Gaudy and Gaudy (1980),

bacteria cannot tolerate a pH level below 4.0 or above 9.5. The low pH causes denaturation of key enzyme proteins and at a high pH the hydroxyl ions exert toxic effects. A pH between 6.5 and 8.5 is an optimum for activated sludge.

As shown in Table 5.1, for the SBR the pH of the mixed liquor just before settling was between 5.31 and 8.6, which is still within the tolerable range for the microbial activity.

Table 5.1 AVERAGE MIXED LIQUOR pH DATA FOR PHASE I

HRT (days)	BSRT (days)		
	10	20	30
3	8.60	7.63	7.31
5	5.37	5.31	6.56
7	6.83	6.88	6.95
9	7.12	6.82	7.05

The pH of the influent slurry was around 7. Yet for reactors operating at 3 days HRT and 10, 20 or 30 days BSRT, the pH rose up to 8.6. This was probably due to the enzymatic reaction of organic nitrogen, containing amines and amino acids, with oxygen to form ammonia/ammonium and other compounds. Since ammonia is a slightly alkaline

inorganic mineral (pH 7.5-8.0) its concentration will naturally increase the pH of the liquid medium. In the remaining reactors, the pH dropped considerably, with the minimum detected being 5.31. These results are closely related to the nitrification process and the production of hydrogen ions, which induced acid conditions to prevail in the reactors,as noted by Loehr (1984).

5.3.2.4 Oxygen Uptake Rate

Typical behaviour of the oxygen uptake rate (OUR) is shown in Figures 5.18 and 5.19 for phase I studies. These results were obtained from track analyses conducted at the end of the testing period for each reactor. Remaining data are tabulated in Appendix B.

The oxygen consumption rate by the microorganisms is a measure of biological activity of the mixed liquor in a treatment system. From the curves presented in Figures 5.18 and 5.19 the OUR varied considerably during the react sequence. The maximum respiration rates in the reactors, operating under HRT/BSRT conditions 3/20, 7/10, 7/20 and 7/30, were 240.0, 159.2, 136.5 and 121.2 mg O_2/l - hr, respectively.

Throughout the aeration time, reactor 3/20 demons-

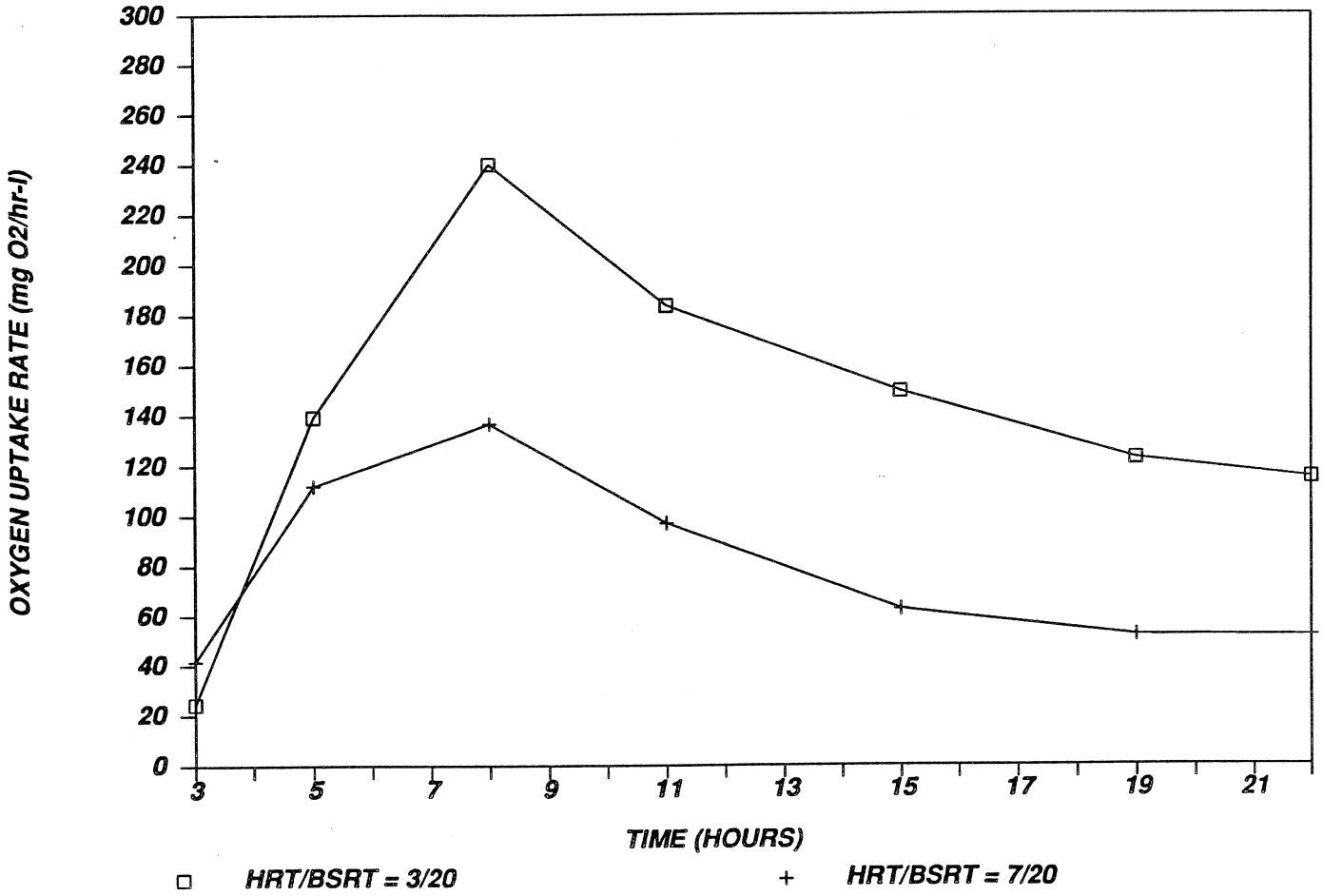


Figure 5.18 OUR vs. time for reactors 3/20 and 7/20 (HRT/BSRT)

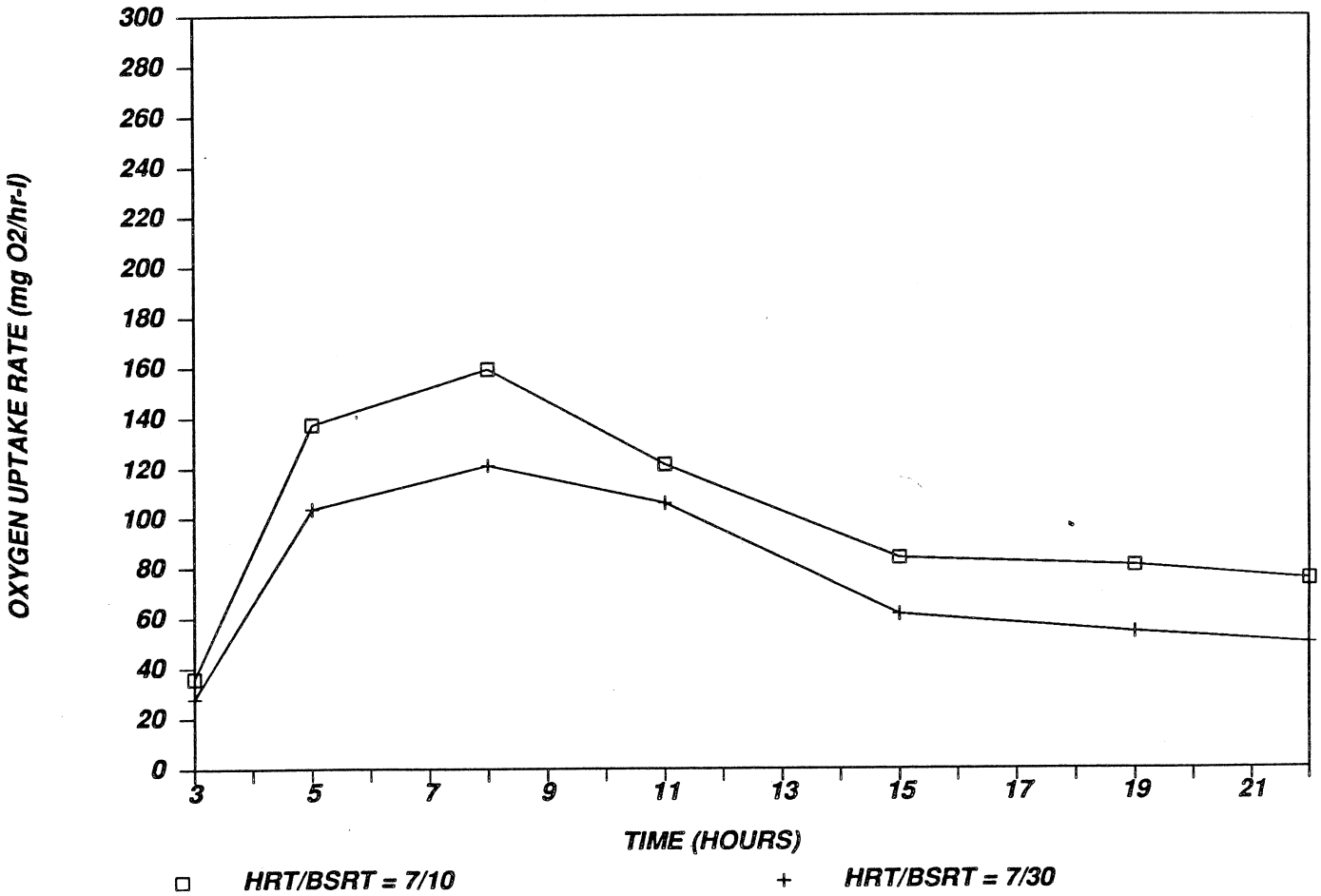


Figure 5.19 OUR vs. time for reactors 7/10 and 7/30 (HRT/BSRT)

trated a high OUR in relation to reactor 7/20. Since the only difference between the two reactors was the volume of wastewater supplied daily, these results lead to the belief that substrate loading and consequent microbial growth had much to do with the OUR observations.

At the end of the react sequence, reactor 3/20, which received 2.3 times more organic load than reactor 7/20, displayed an OUR of 114.8 mg O_2/ l-hr. This is an indication that high microbial activity was still taking place and nutrients were being oxidized in the former reactor. At the same time, reactor 7/20 showed a consumption of only 46 mg O_2/l- hr, which reflects a low rate of metabolic oxidation because of the substrate paucity. As a matter of fact, the OUR rate in the latter reactor decreased considerably after the eleventh hour of the SBR cycle, which suggests that most of the readily available nutrients were already utilized by the microbes, and eventually the influent wastewater was stabilized.

The OUR pattern of reactors 7/10 and 7/30 was similar to the one described for reactor 7/20. The exception was that reactor 7/30 consistently displayed a lower OUR compared to the other two reactors. This behaviour was linked to the extended BSRT (or sludge age) and consequent endogenous respiration. Under these circumstances the microbes lose viability to reproduce, and maintenance energy

requirements provided by oxidative reactions are much lower than necessary for growth functions.

To complement these studies, frequent measurements of mixed liquor dissolved oxygen (DO) were also taken towards the end of the react sequences. The results revealed that the average DO was lowest (2.5-3.2 mg O_2/l) in the reactor operating at 3 days HRT, thus confirming a high rate of oxygen utilization due to microbial synthesis. The DO rose up to 7.1 mg O_2/l in the reactors operating at 9 days HRT, which is a clear demonstration of a decline in oxygen consumption as the pollutants from the wastewater were stabilized.

5.3.2.5 Settling Characteristics of Sludge

In biological reactors such as the SBR, the biomass is separated periodically from the liquid phase by quiescent settling within the reactor. This requires the heterogeneous microbial population to be able to flocculate, that is to form a floc matrix of relatively increasing bulk density as the total biological mass settles down, and the supernatant is developed in turn. However, if the settling process fails, then the withdrawal of the clear supernatant, as well as the efficiency of the biological treatment, will be seriously affected.

In the course of the present investigation, the settling characteristics of the mixed liquor as a valuable design parameter of the SBR was closely monitored. The evaluation of the settling was conducted in the form of periodically measuring:

1. the mixed liquor suspended solids (MLSS) concentration at the end of the react stage,
2. the sludge volume after 30 min of settling in a 1.0 liter graduated cylinder,
3. the height of the sludge blanket in the reactor.

Figure 5.20 displays the relationship between HRT and the volume occupied by the biological solids for different BSRT values. It was evident that the volume of the biological solids underwent a steady increase as the BSRT progressively increased from 10 to 30 days. This observation was consistent in most of the cases examined, regardless of the hydraulic retention time applied to the reactor.

For example, at an organic loading corresponding to 5 days HRT, the sludge volume varied from 580 to 757 ml as the corresponding BSRT changed from 10 to 30 days. This deterioration was possibly brought about by the physiological state of the microorganisms present in the reactor. This observation was strongly supported by a previous investigation by Benefield and Randall (1980), who

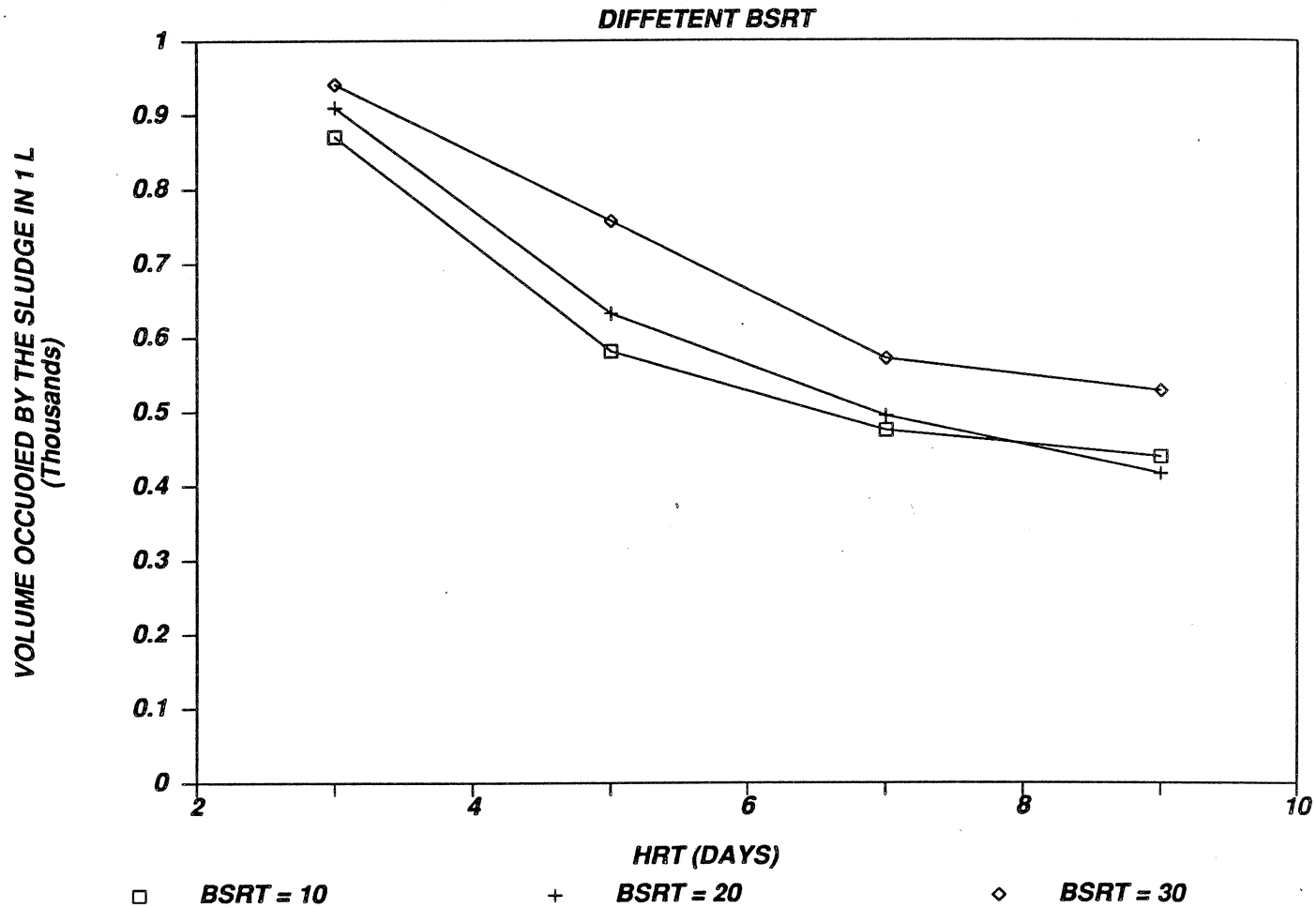


Figure 5.20 Relationship between HRT and the volume occupied by sludge for various BSRT

found that the larger the BSRT, the lower is the ability of the microbes to flocculate. On the other hand, a longer BSRT enhanced the activity of the nitrifying bacteria, which are known for their poor flocculation nature. Similarly, a fairly low BSRT would eventually jeopardize the occurrence of nitrification, and possibly deteriorate the settling process as a result of the ecosystem immaturity.

Figure 5.21 depicts the same information as a function of BSRT which is a valuable engineering design parameter of the SBR on the settling characteristics of the mixed liquor in the reaction. For the same BSRT and various HRT quantities, the quality of the settling, expressed by a smaller volume occupied by the settled sludge, has the tendency to improve with the successive increase in HRT. For the purposes of initial design estimates from these bench scale tests, both Figures 5.20 and 5.21 contain the same information, and either can be used depending on which is found to be more convenient.

The following is a numerical example to illustrate the above described behaviour. For the conditions of HRT/BSRT equal to 3/20 and 9/20 days, the settled sludge volumes were 910 and 452 ml, respectively (i.e. about a 50% improvement), which could be considered as a valuable achievement in a full scale SBR operation. One should also bear in mind that increasing the hydraulic retention time

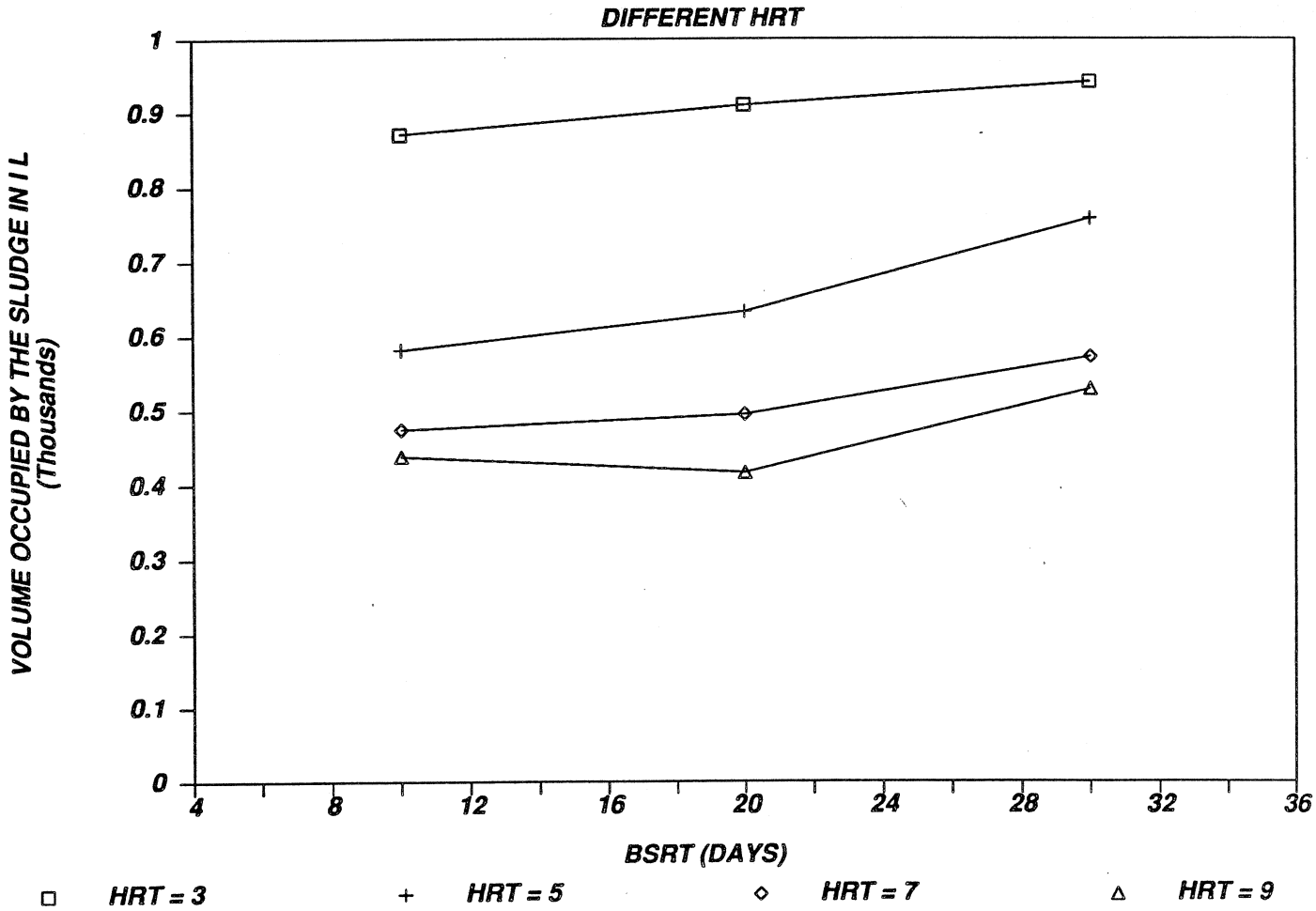


Figure 5.21 Relationship between BSRT and sludge volume for various HRT

causes a respective increase in the volume of the reactor for a fixed inflow rate, which may not be the most economic design. A compromise must be made between the best operating conditions and process economics.

The poor settling observed at a low HRT (3 days) could be attributed to adverse environmental conditions created by: 1) insufficient aeration, 2) the presence of toxic substances, 3) the substrate loading rate, and 4) the resultant accelerated growth rate of microorganisms. From the visual observations during the settling period, the 3 days HRT reactor revealed poor flocculation properties of the mixed liquor, which in fact never aggregated or settled satisfactorily. On the contrary, in the reactors operating at higher HRT of 5, 7 and 9 days the bioflocculation process was quite evident. This performance most likely occurred due to: (1) adequate substrate loading (the daily COD load applied to the reactors ranged between 3.3 to 6.0 gm/l) and (2) the growth of a mixed culture with good settling properties. Sludge bulking is frequently caused by excessive growth of filamentous microorganisms. These species can compete effectively with other aerobic bacterial growth in a medium rich in carbohydrates and low in dissolved oxygen, because of the their large surface area per unit of mass.

It is a common practice in the activated sludge treatment process and its various modifications to express the sludge settling efficiency in terms of the sludge volume

index (SVI). It is the volume in milliliters occupied by 1 g of a suspension after 30 min of settling. Metcalf and Eddy (1979) have stated that for low strength waste (e.g. municipal waste) an SVI below 120 is an indicator of good settling, while for values above this limit poor sludge settling can be expected.

In the present study the SVI was found to be in the narrow range of 28.7 to 44.1 with no conclusive trend. The SVI results are summarized in Table 5.2, and based on these values one could infer that the sludge settling is excellent in all of the reactors examined. However the volume occupied by the sludge was much too high in some of the reactors (e.g. for the 3 days HRT in Table 5.2) which in turn indicates the inability of the SVI to illustrate perfectly the settleability conditions observed.

The low values of SVI obtained in the present study were mainly due to the effect of high microbial concentrations in the mixed liquor, which ranged from 12,850 to 22,415 mg/l (see Table 5.2). Therefore, it is strongly recommended that the percentage of the volume occupied by the settled sludge in reference to the total reactor volume should be reported, as well as the concentration of suspended solids in the mixed liquor. This practice would be particularly beneficial when describing the treatment of concentrated wastewater.

Table 5.2 SUMMARY OF SLUDGE SETTLEABILITY FOR PHASE I

Parameter	BSRT (Days)	HRT (Days)			
		3	5	7	9
Sludge Volume 1 liter Cylinder (ml)	10	871	581	474	438
	20	910	632	494	451
	30	941	757	570	527
MLSS (mg/l)	10	19745	16050	14465	12850
	20	20705	18467	17169	15295
	30	22415	19360	18240	16080
SVI (mg/l)	10	44.1	36.2	32.8	34.1
	20	43.9	34.2	28.7	29.5
	30	41.9	39.1	31.2	32.8
* Reactor (H Sludge/ H Total)	10	0.840	0.519	0.416	0.385
	20	0.875	0.577	0.439	0.398
	30	0.913	0.710	0.530	0.486

* Fraction of reactor volume occupied by the sludge blanket.

It is important to mention that throughout the experimental period the height of the sludge blanket in the reactor after 30 min. of settling was consistently lower compared to the height in the one liter graduated cylinder used in the standard SVI test. This difference was between 3 to 14% and it can attributed to the surface area to volume

ratio, which is greater in the cylinder than in the reactor. The aspect ratio of the experimental vessel must be known in order to avoid an incorrect interpretation for application to a full scale SBR, where a 10% difference could lead to a change in the design of the safety factor.

In summary, emphasis must be placed on both engineering parameters, BSRT and HRT, as they affect the sludge settleability. Based on the findings of this experimental work, a BSRT of 10 to 20 days, and a HRT between 7 to 9 days would produce a sludge with good settling performance. Furthermore, it is worthwhile to observe that BSRT had considerably more effect on the volume of settled sludge than it did on either COD or SS (see Figures 5.5 and 5.9).

5.3.3 Study of Nitrification/Denitrification

This phase of the study was designed to investigate the fate of the nitrogen species in the SBR system. In order to achieve nitrogen removal, variations in the reactor cycling were introduced to encourage the exogenous and endogenous carbon denitrification. For this effect two non-aerated periods were implemented, one being of three hours during the fill sequence and the other ranging from zero to nine hours incorporated towards the end of the react sequence. Within this context, four operating modes were examined, the ratios of nitrification/ denitrification periods for the fill plus react sequence comprised of 19/3, 16/6, 14/8 and 10/12 hours. Furthermore, each of these operating modes was tested for reactors functioning at 6/20 and 9/20 (HRT/BSRT).

5.3.3.1 Effect of Aerobic and Anoxic Operation the Performance of SBR

Tables 5.3 and 5.4, summarize the average results of the various physical and chemical parameters used for evaluation of process performance in terms of the quality of effluent produced. Additional information concerning this section is inserted in Appendix B.

Table 5.3 AVERAGE INFLUENT/EFFLUENT AND MIXED LIQUOR CHARACTERISTICS FOR SBR OPERATING AT 9/20 (HRT/BSRT, DAYS) AND VARIOUS NITRIFICATION/ DENITRIFICATION PERIODS

Parameter	Influent	Effluent			
	Waste	Nitrification/Denitrification (hrs)			
		19/3	16/6	14/8	10/12
NH_3-N (mg/l)	1265	2.2	2.4	6.1	11.5
NO_2-NO_3-N (mg/l)	6.2	182.1	150.4	102.9	74.5
Inorganic N (% Removal)	---	85.5	88.0	91.4	93.2
TKN (mg/l)	2580	185	200	250	275
COD (mg/l)	31175	845	815	830	1250
TSS (mg/l)	10690	283	307	296	350
Parameter		Mixed Liquor			
		Nitrification/Denitrification (hrs)			
		19/3	16/6	14/8	10/12
MLSS (mg/l)	---	15750	15165	14080	13435
Sludge Vol. 1.0 liter Cylinder (ml)	---	475	510	635	800

Note: Denitrification time includes fill and part of react sequence.

Table 5.4 AVERAGE INFLUENT/EFFLUENT AND MIXED LIQUOR CHARACTERISTICS FOR SBR OPERATING AT 6/20 (HRT/BSRT, DAYS) AND VARIOUS NITRIFICATION/ DENRIFICATION PERIODS

Parameter	Influent	Effluent			
	Waste	Nitrification/Denitrification (hrs)			
		19/3	16/6	14/8	10/12
NH_3-N (mg/l)	1195	5.5	9.3	14.8	217.0
NO_2-NO_3-N (mg/l)	5.8	230.8	195.7	136.5	9.7
Inorganic N (% Removal)	---	80.3	83.0	87.0	---
TKN (mg/l)	2410	270	305	355	460
COD (mg/l)	30680	1105	1075	1120	1595
TSS (mg/l)	11852	326	360	427	539

Parameter	Mixed Liquor			
	Nitrification/Denitrification (hrs)			
	19/3	16/6	14/8	10/12
MLSS (mg/l)	18635	18150	17300	16565
Sludge Vol. 1.0 liter Cylinder (ml)	500	625	740	930

The reactors operating at condition 9/20 (HRT/BSRT) and four levels of non-aerated periods demonstrated a high capacity for ammonia nitrogen removal, process efficiency was above 99.0%. As the non-aeration time increased from 3 to 12 hours the NH_3 - N in the effluent ranged from 2.2 to 11.5 mg/l. The difference in the residual ammonia resulted from the environmental changes occurring within these reactors. Extended anaerobic periods may have caused the death of some bacterial species, and eventually the cell nutrients were released to the medium. Also, under these conditions the nitrifiers activity was suppressed and thus, further nitrogen oxidation could not take place.

A similar trend was displayed by the reactors operating at 6/20 (HRT/BSRT). Although these systems received a daily load of about 600 mg of ammonia nitrogen, the residual amount of NH_3 - N in the treated wastewater was between 5.5 to 14.8 mg/l, for reactors with denitrification periods ranging from 3 to 8 hours, respectively.

An exceptional behaviour was shown by the reactor operating under condition 6/20 with a 12/10 hours of denitrification/aeration. The average NH_3 - N in the treated effluent was 217.0 mg/l, and the nitrite plus nitrate nitrogen concentration was 9.7 mg/l. These results clearly indicate that nitrification was suppressed in this system. The absence of dissolved oxygen for a long period of time,

coupled with high total organic loading applied daily to the reactor, which was in average 15340 mg of COD, certainly created inhibitory conditions to the growth of nitrifying organisms in the highly competitive environment.

The sensitivity of nitrifyers to low dissolved oxygen concentration can prevent the accomplishment of nitrification process in a heavily loaded system, mainly because of: 1) high oxygen demand exerted by the predominantly heterotrophic population and 2) deficient oxygen transfer within the sludge matrix. Coincidentally, the average pH of the mixed liquor was above 8.0 and thus about 5% of the ammonia present in the reactor would be in the un-ionized state. This quantity was most probably well above the minimum amount responsible for inhibition of the nitrifying species. Loehr (1984) has reported free ammonia concentrations that begin to inhibit nitrosomonas are in the range of 10-150 mg/l, and those that begin to inhibit nitrobacter are in the range of 0.1-1.0 mg/l.

The concentration of nitrites plus nitrates differed substantially within the treated liquid manure. For reactors operating at 6 and 9 days HRT/20 days BSRT, the oxidized nitrogen compounds decreased from 230.8 to 136.5 mg/l, and 182.1 to 74.5 mg/l, when the non aerated period was augmented from 3 to 8 hours, and 3 to 12 hours, respectively. The observed loss of nitrite-nitrate-nitrogen

was a consequence of the denitrification reaction, in which case these compounds are used as terminal electron acceptors and reduced to elemental nitrogen gas when the molecular oxygen is not available in the treatment system. The dissimilatory reduction involved is referred to as an anoxic process to distinguish it from other anaerobic processes. The anoxic reduction of NO_2-NO_3-N was noticeably enhanced by increasing the periods of zero dissolved oxygen in the operating cycle.

During the course of the present study, the SBR system demonstrated a high capacity for removal of inorganic nitrogen, defined here as ammonia plus oxidized nitrogen. As indicated in Tables 5.3 and 5.4, the actual removal of inorganic nitrogen was between 80.3% to 93.2%. These results compare fairly well with the 86% obtained by Palis and Irvine (1985) and the 90% reduction achieved by Irvine et al. (1983) in the treatment of domestic wastewater.

The measured effluent TKN was found to be relatively low in those reactors where nitrification/ denitrification was taking place. Compared to average influent TKN concentrations of 2410 and 2580 mg/l, the removal efficiency ranged from 81% to 93%. The analyses of the results do not show a significant difference between reactors operating with 3 and 6 hour anoxic periods. Nevertheless, a consistent trend was visible when anoxic

operation increased from 3 to 12 hours. The total nitrogen concentration in the effluent was highest for the latter reactors. Based on these results, it is evident that most of the remaining nitrogen was in the organic form, since the ammonia component in the effluent was quite low.

The removal of organic nitrogen was influenced by the anoxic conditions in the reactors. Increasing the non-aerated period from 20 to 58% must have limited the oxidation of organic nitrogen, and its subsequent utilization. In addition, it could be that the residual protinaceous matter was slow to biodegrade. Possibly under the present operating conditions, the microorganisms were unable to develop the appropriate enzymatic mechanism which would permit the decomposition of high molecular weight organic compounds to their constituents, which could then integrate in the microbial metabolic pathway.

After successful establishment of nitrification/ denitrification, the SBR also demonstrated a good capacity for the removal of organic carbon. All reactors consistently maintained a quality effluent with above 95% COD removal. These results compare very well with the ones obtained in Section 5.3.2.

Nevertheless, the COD removal efficiency obtained in the present experimental work is high relative to the

80% achieved by Lo and Liao (1987) in the treatment of dairy manure. Certainly, the waste characteristics had a bearing on the treatment efficiency. Compared to swine, the dairy manure has a larger fraction of fiber, comprising of complex compounds such as lignin and cellulose, which decomposes at a very slow rate, and hence can contribute to the low process performance.

There was no significant difference in terms of organic carbon removal between reactors operating with 3, 6 and 8 hours anoxic period. Presumably, carbon oxidation was accomplished within the first few hours of the aerobic react sequence. For reactors operating at 6 and 9 days HRT with 12 hours anoxic, the COD released in the effluent was 1595 and 1250 mg/l, respectively. The low level of substrate utilization was the result of elongating the non-aeration time. Consequently, this shifted the microbial population largely towards facultative and anaerobic species. This point of view was supported by the fact that a gray color sludge was produced in both reactors, rather than a brown sludge which is commonly displayed by aerobic treatment system. The anaerobiosis in these reactors also contributed to exudation of foul odour.

It is a general practice to evaluate the performance of biological treatment processes in terms of suspended solids (SS) concentration in the effluent. The

test involved is inexpensive, easy to perform and expresses adequately the magnitude of pollution impact. As shown in Table 5.3, the average effluent SS varied from 283 to 350 mg/l. This again attests to the high performance of the SBR in the removal of suspended solids. Anoxic operation between 3 to 12 hours did not seem to affect seriously the solids removal. Nevertheless, the effluent was of an inferior quality with SS concentration increasing by as much as 24% when the anoxic operation was augmented to 12 hours. This fact also coincided with relatively poor floc settleability occurring in the reactor with 12 hours anoxic.

Typically, for reactors operating at the 6/20 condition and with 3 to 12 hours anoxic, the amount of suspended solids in the effluent ranged from 326 to 539 mg/l. It is obvious that the short hydraulic retention time combined with the effect of anoxic operation were responsible for the deterioration of the effluent. The amount of suspended solids in the effluent increased from 15 to 54%, when compared to the reactors operating at 9 days HRT. The differences were more significant within those reactors operating with 8 and 12 hours anoxic. This observation could have been due to: 1) the environmental conditions created, 2) the type and the number of microorganisms capable of biodegrading the available substrate, and 3) the ability of microorganisms to flocculate and eventually settle.

The anoxic operation had a profound impact on the settling process in the SBR. It was visually noticed that during the experimental work the flocculation and consequent settling improved drastically as the anoxic period decreased from 12 to 3 hours. This improvement was in the order of 77 and 85%, for those reactors operating at 6 and 9 days HRT, respectively.

In general, the reactors operating with an anoxic period of 3 and 6 hours displayed rapid flocculation as soon as the aeration and mixing was off. The flocs were relatively big, chocolate brown in color, and with a zone settling velocity between 1.2 and 1.5 m/hr. The sludge blanket formed was compact and it settled well below the effluent discharge port. As for the overlying supernatant it was devoid of pin floc type turbidity. Settling of mixed liquor was noticeably of a poorer quality in the reactors that operated with 8 and 12 hours anoxic. The sludge produced was of grayish color, fluffy and occupied a large volume of the reactor to the extent that it jeopardized the removal of the treated effluent and consequently, this affected adversely the efficiency of the SBR treatment system. Similar findings were also reported by Palis and Irvine (1985), where domestic wastewater was treated in an SBR that operated with an alternating aerobic /anoxic fill and a totally anoxic react sequence. The calculated SVI for

the later reactor was above 200, which is a very clear indication of a bad settling performance.

5.3.3.2 Track Studies on Ammonia, Nitrite-Nitrate (NO_x), Total Nitrogen and pH

In order to gain a better understanding of the SBR phase operation, track studies were conducted at the end of the experimental phase. This timing was chosen to ensure that ecological equilibrium was prevalent in all reactors being investigated. A track study entails detailed physical and chemical analyses throughout the duration of one cycle (i.e. including fill, react, settle, draw and idle).

The results from the track analyses, for reactors operating at 9/20 with 3 to 12 hours of anoxic operation, are presented in Figures 5.22 to 5.25. Track data for the set of reactors operating at 6/20, are provided in Appendix B.

The nitrogen species displayed a typical behaviour in all the reactors that were investigated. As shown in Figure 5.22, the concentration of ammonia nitrogen increased during the fill sequence from a minimum of 2.2 mg/l at the beginning of the cycle to a maximum of 170.2 mg/l at the end of the fill sequence, and subsequently during the aerated

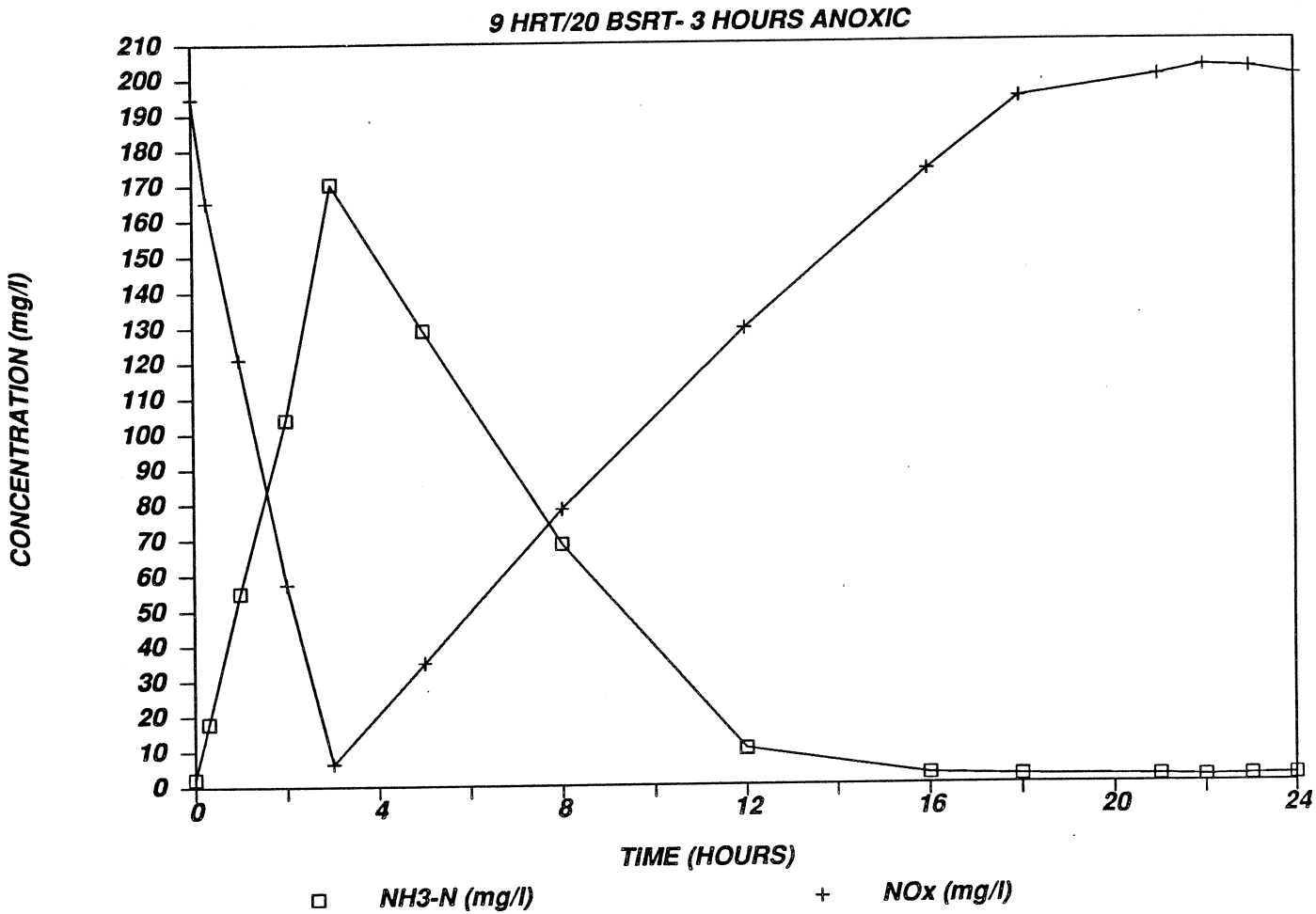


Figure 5.22 Track analysis of ammonia and nitrite-nitrate nitrogen for reactor with 3 hrs anoxic operation

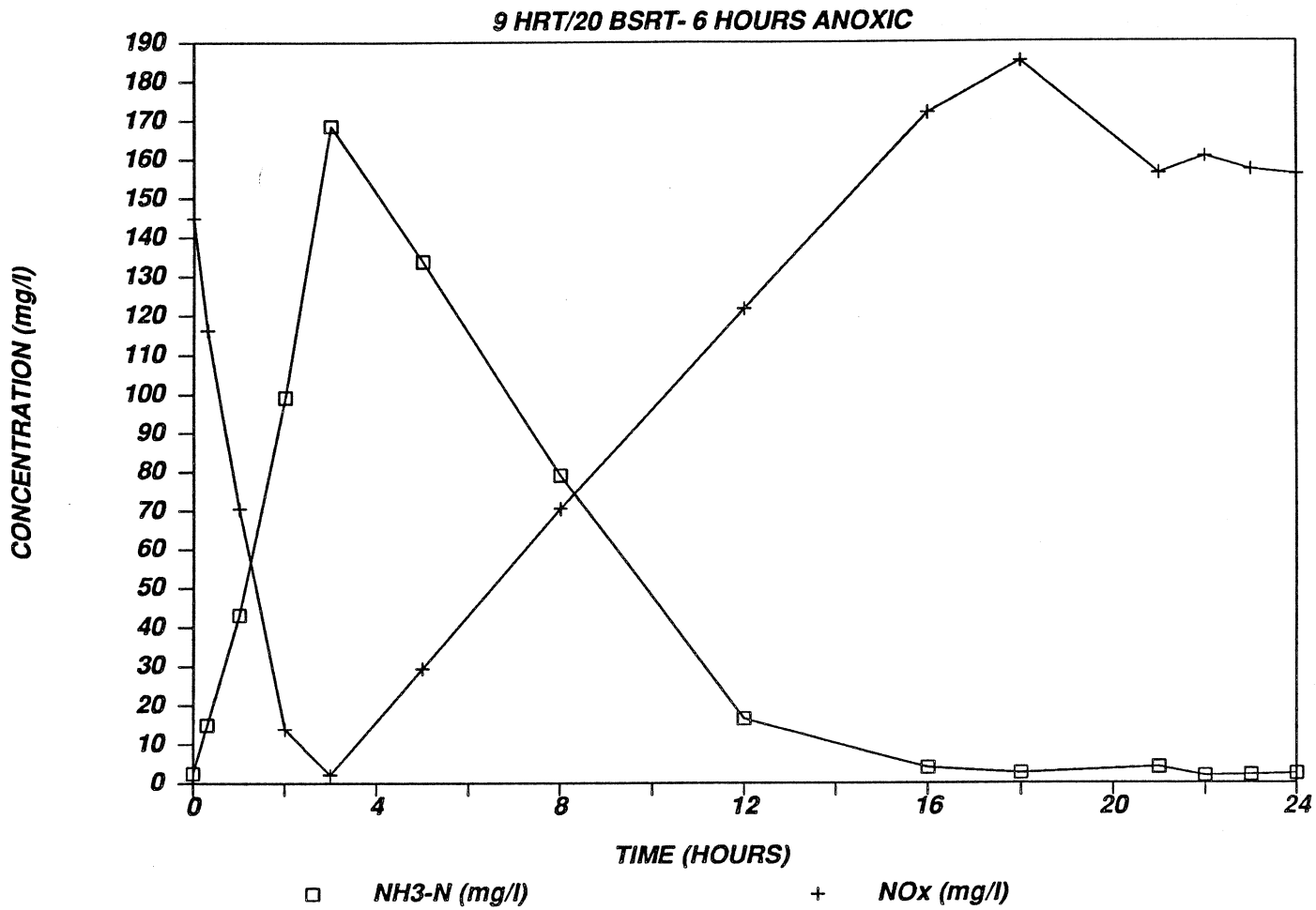


Figure 5.23 Track analysis of ammonia and nitrite-nitrate nitrogen for reactor with 6 hrs anoxic operation

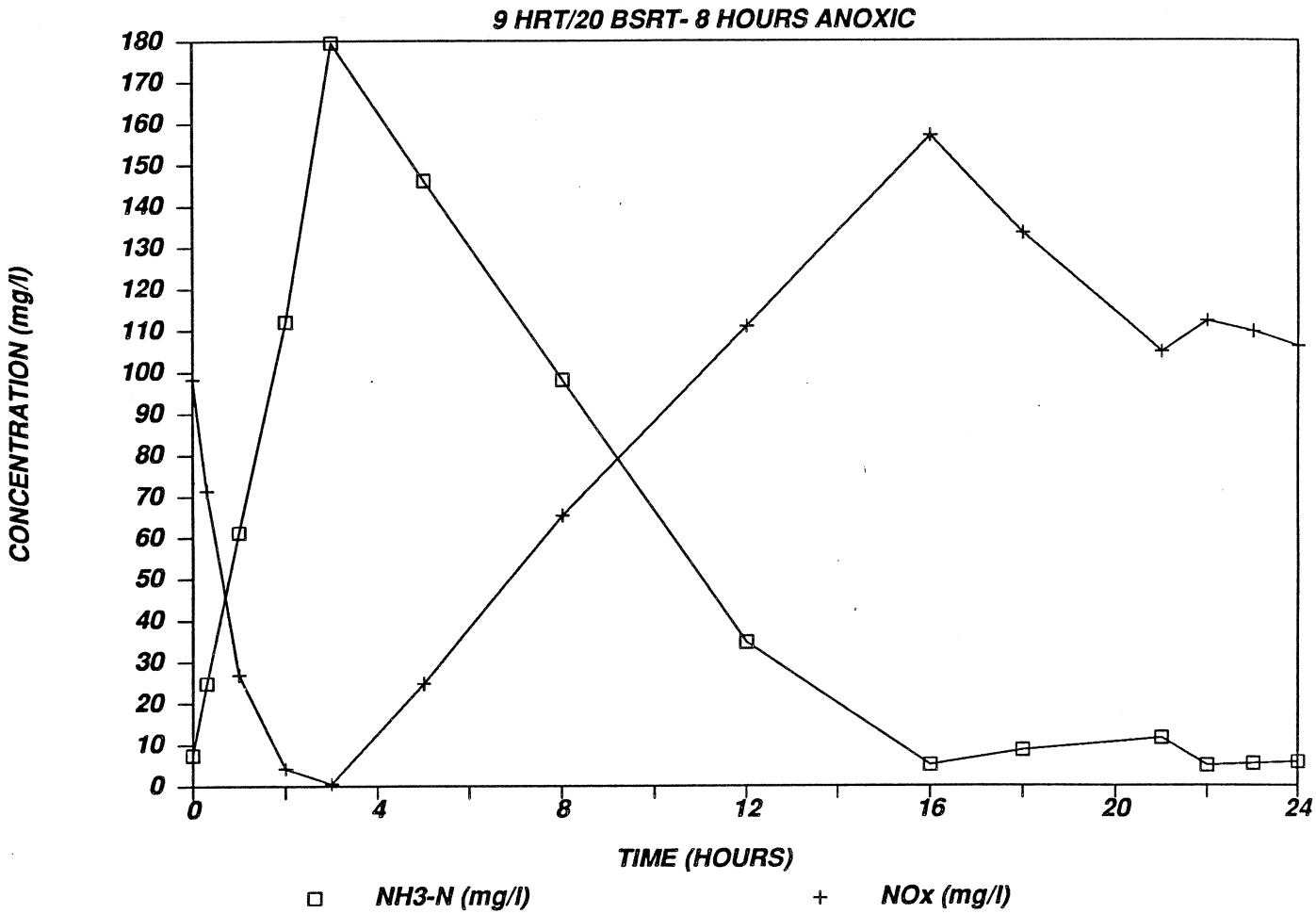


Figure 5.24 Track analysis of ammonia and nitrite-nitrate nitrogen for reactor with 8 hrs anoxic operation

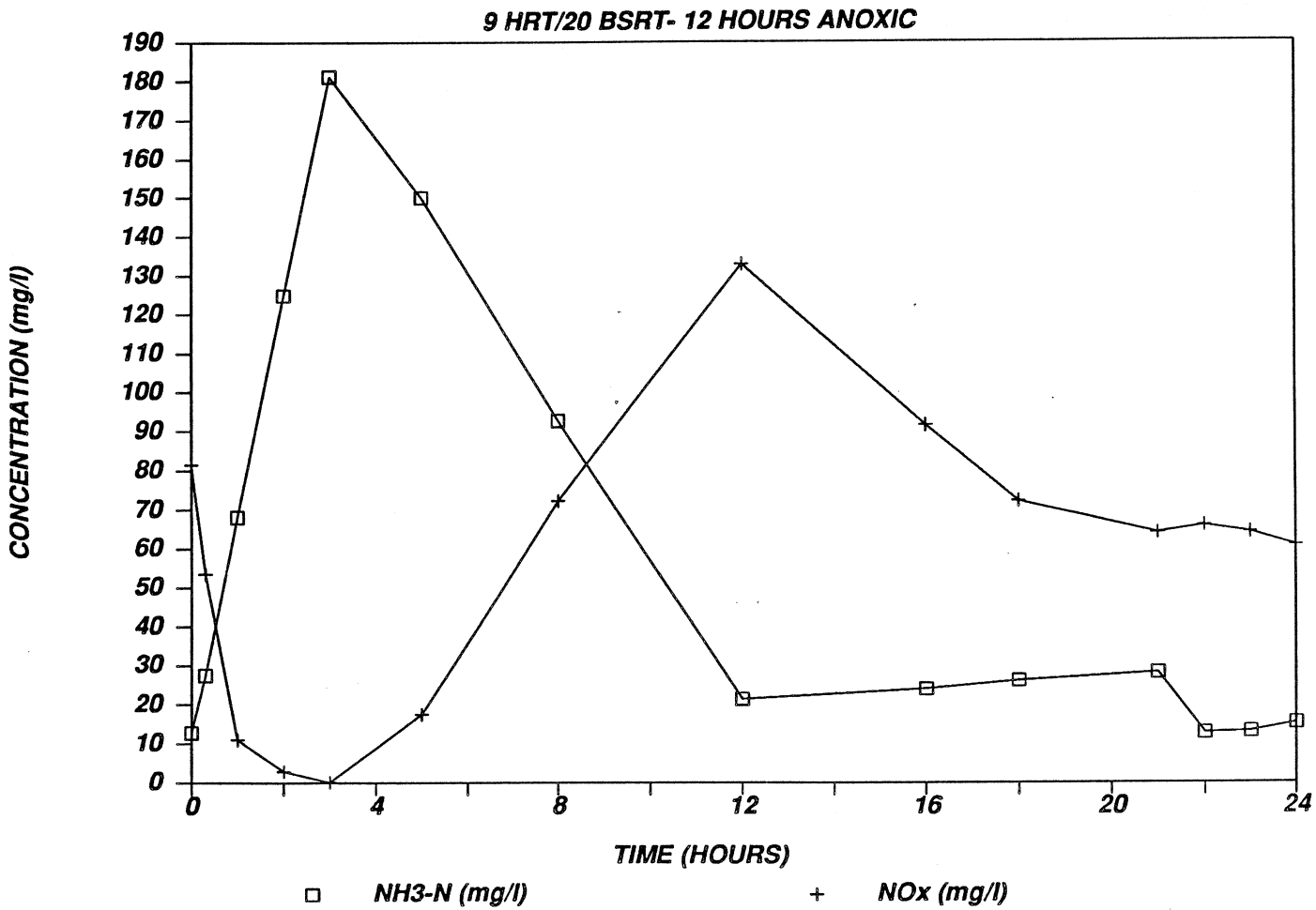


Figure 5.25 Track analysis of ammonia and nitrite-nitrate nitrogen for reactor with 12 hrs anoxic operation

react phase it dropped sharply to less than 2 mg/l.

The observed increase in ammonia concentration was caused by the strength of the influent liquid manure, which was 1205 mg/l NH_3-N/l. Furthermore, there is a strong possibility that ammonium ions were released during the metabolism of organic nitrogen either through fermentative deamination or oxidative degradation permitted by oxygen transfer across the air wastewater interface. Simultaneously, removal of ammonia could have also taken place to some extent, through bacterial assimilation or by volatilization. However, these losses of ammonia did not have any drastic effect on the accumulation of this compound in the reactor during the fill period.

With the supply of the air in the react sequence, ammonia started to decrease rapidly, and most of it was consumed in less than twelve hours. After that its concentration was almost constant until the end of the cycle. The consumption of ammonia nitrogen was mainly due to two reasons: a) the chemoautotrophic bacteria of the genera Nitrosomonas oxidized ammonium to drive the energy required for their growth and maintenance, and b) ammonium possibly provided the nitrogen element, which is an essential building block for cell synthesis. Presumably, the conditions were ideal for the growth of microorganisms,

since an aerobic environment rich in carbonaceous matter as well as other nutrients should prevail, at least during the first hours of the react sequence.

In Figure 5.22 the profile displayed by the oxidized nitrogen forms is sharply in contrast with the one observed for ammonia. At the start of the cycle the mixed liquor detained in the reactor exhibited a high concentration of NO_2-NO_3-N (190 mg/l). Subsequently, during the three hours fill, a steep drop in oxidized nitrogen occurred and it attained the minimum value of 6.2 mg/l. With the advent of the aerated react sequence, the nitrite-nitrate nitrogen concentration started to rise very rapidly and it reached a high level of 195 mg/l in about 18 hours, after which it stabilized with an average value of 200 mg/l.

The steep drop in NO_2-NO_3-N concentration, at a rate of 62.7 mg/hr, during the fill period was mainly due to the occurrence of the denitrification process. In the absence of dissolved oxygen the denitrifying bacteria were able to thrive in an anoxic environment rich in oxidized nitrogen forms and organic carbon which was being supplied by the influent wastewater. These microorganisms biologically reduced nitrite plus nitrates to nitrogen gas while metabolizing the organic substrate. Thus, only residual amounts of oxidized nitrogen were detected at the end of the fill sequence. Gases being released from the

mixed liquor during the fill period were also perceived.

It should be noted that dilution had some influence on the decrease of NO_2-NO_3-N concentration, but its contribution was considered to be minimal. Although the influent had only 6 mg/l of oxidized nitrogen compared to the resident mixed liquor which had 195 mg/l, the dilution ratio was of 1:9 and therefore it contributed merely 11% of the decrease of oxidized nitrogen.

The concentration of NO_2-NO_3-N began to increase immediately following the aeration of mixed liquor. Oxidized nitrogen was produced at a rate of 12.5 mg/hr, and nitrification was essentially completed in about 15 hours, after that a constant plateau in the order of 200 mg/l of oxidized nitrogen was attained. It was very interesting to note that the nitrifying bacteria were well established in the SBR treatment system and the absence of dissolved oxygen during settle, idle and fill, a total of 5 hours, did not have any serious inhibitory effect on their activity.

Most of the oxidized nitrogen encountered in the reactor was in the nitrate form and there were only trace amounts of nitrite. Based on the results obtained it is obvious that complete nitrification occurred and neither un-ionized ammonia or nitrous acid (HNO_2) caused inhibitory

reactions to both the nitrosomonas and the nitrobacter organisms. The concentration of these compounds is pH dependent, since the concentrations of HNO_2 will increase with low pH and NH_3 increases with pH in the basic range. It is conceivable that this treatment system had an adequate buffering capacity to maintain a favorable pH for the nitrifying population.

For reactors operating with an anoxic period of 6 to 12 hrs, the inorganic nitrogen profiles are shown in Figures 5.23 to 5.25, and they are all quite similar in appearance to Figure 5.22. The concentration of ammonia nitrogen increased during the fill sequence, although there were good chances for some of it to volatilize since the pH in these two reactors was in the alkaline range (see Figure 5.29). As for the oxidized nitrogen concentration a sharp decrease was observed, this behaviour is due to the same reasons as previously explained for the reactor with 3 hrs of anoxic operation.

However, in the react sequence the oxidized nitrogen forms followed two distinct paths. With the aeration of the mixed liquor the nitrification process resurged and it proceeded at a relatively fast rate. The nitrite plus nitrate concentration increased from a residual value of 2 mg/l to a maximum of 185 mg/l in 15 hrs, for the

reactor with 6 hrs of anoxic operation. Based on the formation of oxidized nitrogen the average rate of nitrification was 12.3 mg NH_4-N oxidized / hr. For the reactors with 8 and 12 hrs of anoxic operation, the rates of nitrification were 12 and 14.7 mg NH_4-N oxidized / hr, respectively. It is evident from Figures 5.22 to 5.25 that nitrification occurred at a much faster rate in the first 12 hrs of the react sequence, and subsequently it slowed down due to limited amounts of readily available ammonium ions.

Once the biological nitrification was nearly completed, it was advantageous to cut the aeration supply, because facultative heterotrophic bacteria can utilize inorganic ions like nitrate, phosphate and sulfate as the oxidizing agent. In the present case, oxidized nitrogen would be a suitable electron acceptor due to its abundance. With this in mind, three denitrification periods of 3, 5 and 9 hrs were tested towards the end of the react sequence. The results indicate that the highest drop in oxidized nitrogen was 69 mg/l, and it was observed in the reactor with 9 hrs of anoxic react. For reactors with 3 and 5 hrs of anoxic react the NO_2-NO_3-N decrease was 29 and 52 mg/l, respectively.

The rates at which denitrification occurred in the above-mentioned reactors were in a short range of 7.6 to 10.4 mg NO_2-NO_3-N / hr. These values are low compared to

the rates obtained during the fill sequence, which were in the range of 27.2 to 62.5 mg NO_2-NO_3-N / hr. These results could be explained on the basis of the availability of organic carbon.

In the fill sequence, emphasis is placed on the availability of influent substrate carbon to drive the denitrification reaction and oxidized nitrogen would be the limiting factor for this process to continue. On the other hand, denitrification during react was limited by the availability of readily biodegradable organic carbon. It can be observed from COD profiles included in Appendix B, most of the soluble organic carbon was utilized within the first hours of the react sequence. Thus as a source of biodegradable organic carbon the denitrifying bacteria would have to rely on the release of this material as a result of endogenous activity or through hydrolysis of large organic molecules. It is needless to say that, due to the paucity of easily biodegradable organic carbon, denitrification in the react sequence did not come to completion and therefore considerable amounts of oxidized nitrogen were released in the treated effluent.

As expected, the ammonia nitrogen concentration decreased rapidly during the aerated react sequence due to nitrification and through bacterial assimilation. The subsequent anoxic environment in the reactors prevented

further oxidation of ammonium ions, and on the contrary an increase of 1.5 to 6.7 mg/l in NH_3 - N was consistently observed during this period. This buildup of reduced nitrogen was more accentuated in the reactors with 8 and 12 hours anoxic, and it reflected either an assimilatory reductose mechanism in excess of anabolic needs or the release of constitutive cellular nitrogen from endogenous activity. Although the final aeration step, just before the settling process, did permit some oxidation of ammonia to occur, a nominal discharge of this compound in the effluent was inevitable.

The pattern followed by TKN is depicted in Figures 5.26 and 5.27 and is very similar to the one displayed by ammonia nitrogen. Except that TKN concentrations determined were much higher, because of the organic nitrogen fraction, which was about 0.55 in the influent liquid manure.

For a reactor operating with 3 hours anoxic, TKN raised up to 490 mg/l at the end of the fill as a result of the strength of the influent waste coupled with a low rate of consumption taking place in the reactor. There is a strong possibility that complex organic nitrogen colloids were hydrolyzed by extracellular enzymes produced by the microorganisms indigenous to the process, and hence increased the amount of soluble organic nitrogen. In the aerobic react stage, the TKN started to decrease progressively until

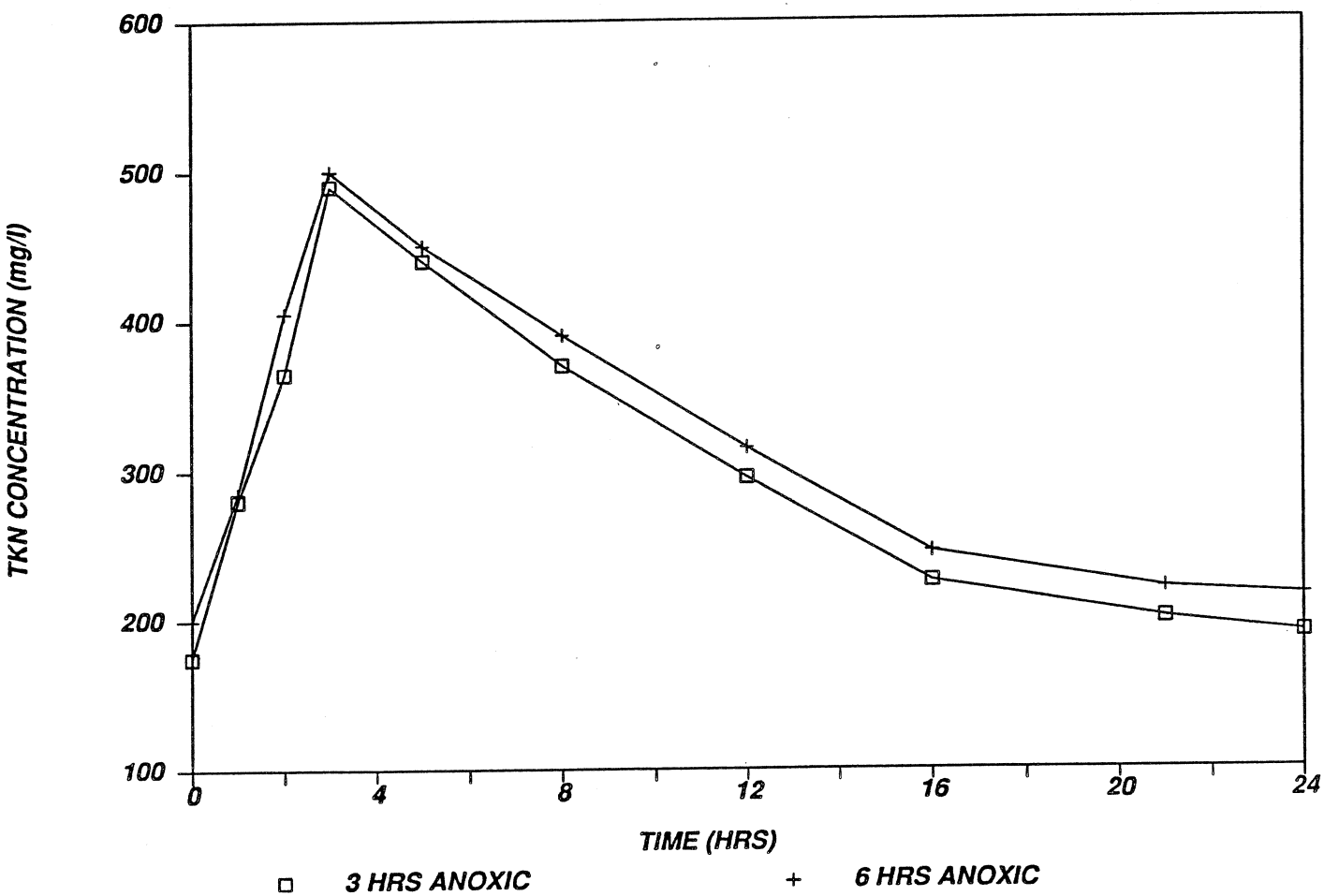


Figure 5.26 TKN track data for reactors operating with 3 and 6 hrs anoxic and 9/20 (HRT/BSRT days)

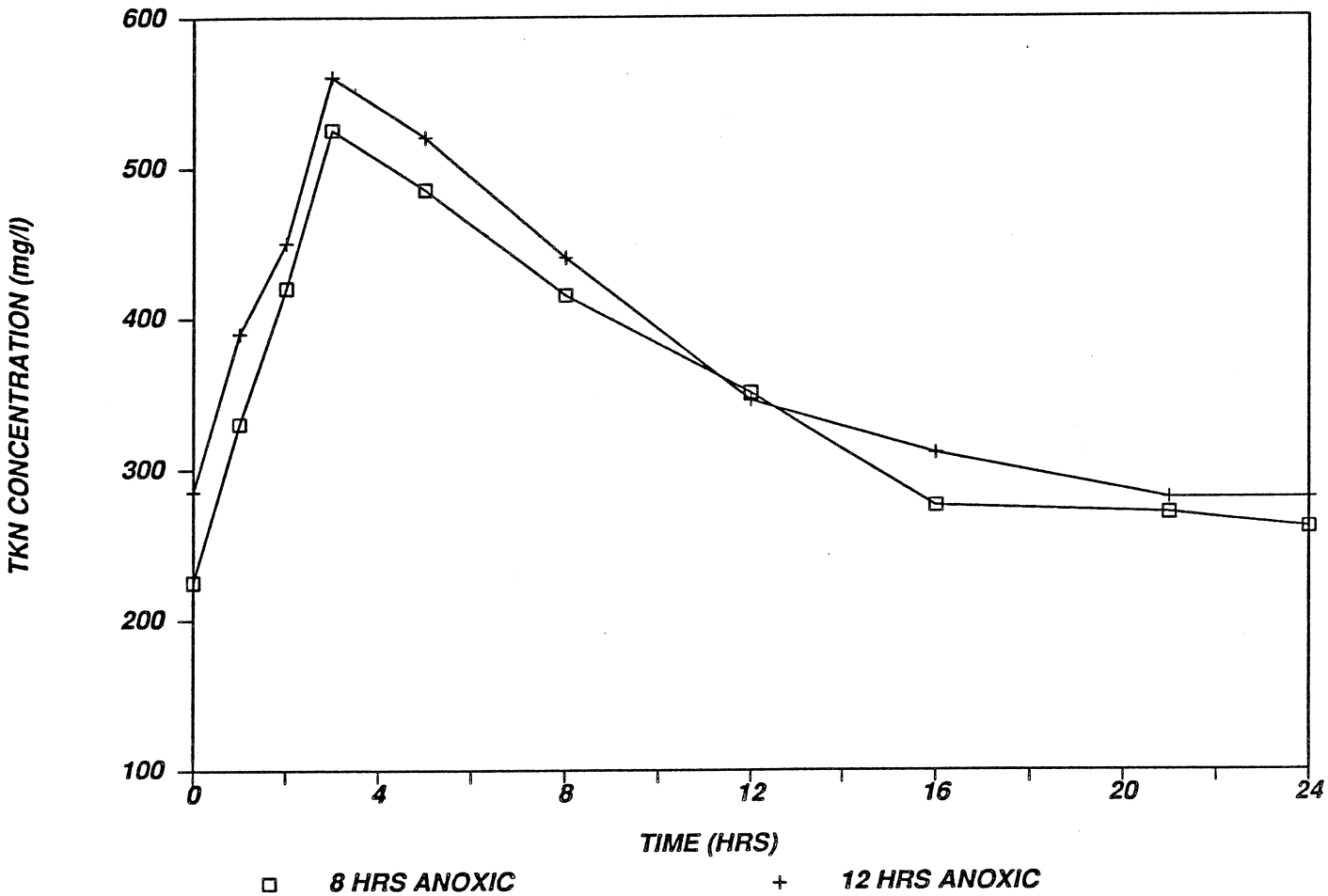


Figure 5.27 TKN track data for reactors operating with 8 and 12 hrs anoix and 9/20 (HRT/BSRT, days)

it reached a low value of 200 mg/l. It is apparent from Figure 5.26 that the biodegradation of the remaining nitrogen would occur at a much slower rate possibly due to its complex organic structure.

Again the behaviour of TKN for the reactor with 6 hrs anoxic was nearly identical to the one obtained for the reactor with 3 hours anoxic. But, reactors operating with 8 and 12 hours anoxic were less efficient in the biodegradation of TKN. It is possible that the type of microbial population established in these reactors (mostly facultative and anaerobic bacteria) were less successful in breaking down organic nitrogen compounds to their elementary constituents such as ammonia, which is then metabolized by the microorganisms. Therefore reactors with 8 and 12 hours anoxic contained as much as 260 and 280 mg TKN/l, respectively, at the end of the cycle.

Figure 5.28 shows a typical track analysis of the pH parameter for reactors operating with an anoxic period of 3 and 6 hours. At the start of the fill sequence the pH of the mixed liquor was quite neutral. However in both reactors the pH began to rise and reached values of 7.63 and 7.79 in about 5 hours. Since the incoming waste had a neutral pH, this rise can be mainly related to the release of hydroxide ions from the denitrification reaction during fill and possibly due to ammonia generation through deamination of orga-

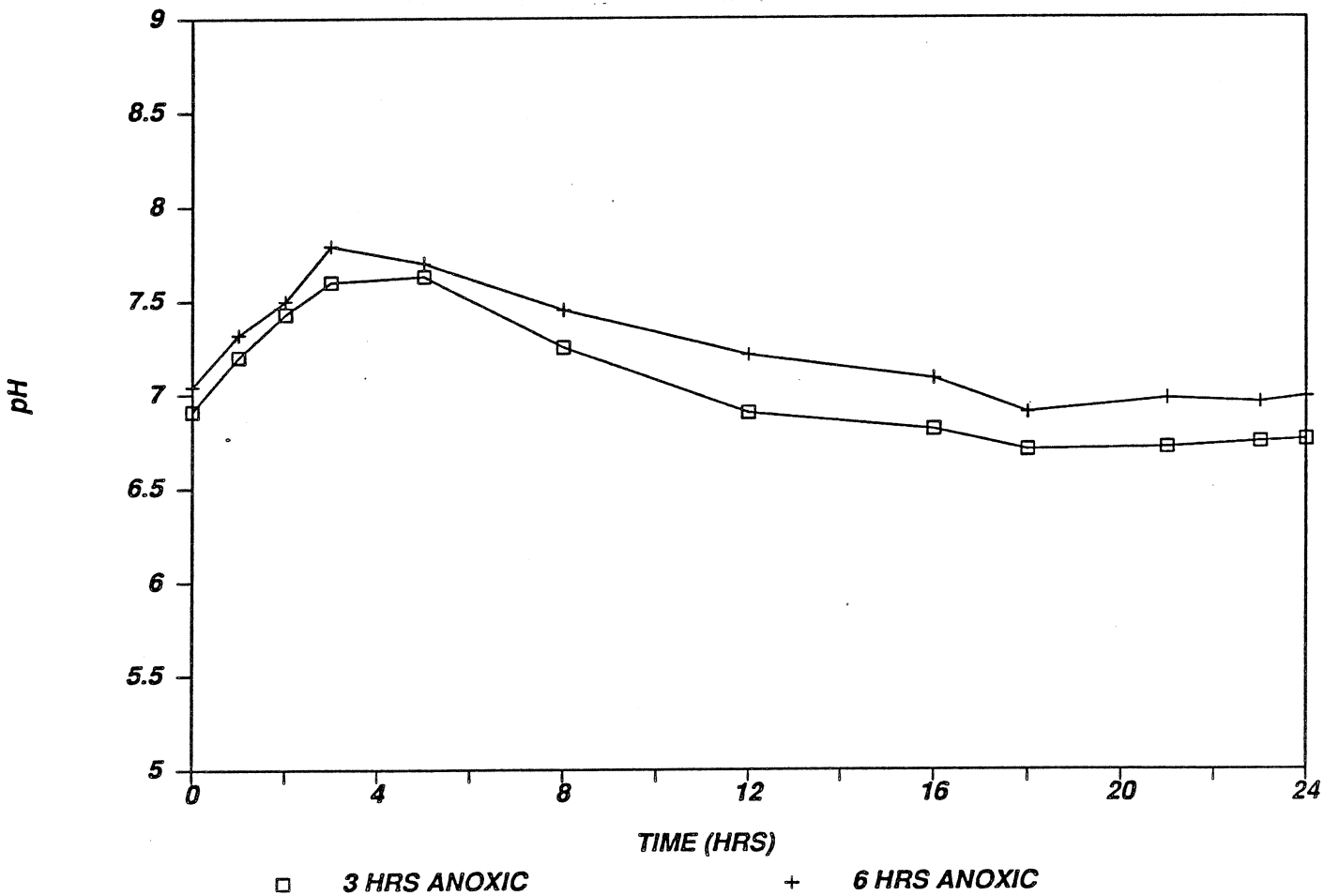


Figure 5.28 pH track data for reactors operating with 3 and 6 hrs anoxic and 9/20 (HRT/BSRT, days)

nic nitrogen. They may have been some other compounds that contributed to the alkalinity of the system, but these were not monitored. With the aeration of the mixed liquor in the react sequence the pH dropped by as much as 0.93 units, and it was due to the production of hydrogen ions during the oxidation of ammonium to nitrite, therefore favouring acidic conditions to prevail in the digestors.

The pH values determined for reactors with 8 and 12 hours of anoxic operation are plotted in Figure 5.29. It is relevant to note that throughout the cycle the pH in these reactors was in the basic range. In general the pH ascended during the anoxic operation, and the maximum increase observed was of 0.7 units. During the aerated react sequence the pH dropped by as little as 0.5 units. Obviously, within the anoxic operation the alkalinity of the system increased and hence, its capacity to accept protons released during nitrification was also augmented.

5.3.4 The Performance of SBR at Low Temperatures

In order to assess the effect of low temperature on the the treatment of liquid swine manure in the SBR, the reactors were operated at 5, 10 and 21^{o}C. Table 5.5 summarizes the average results attained for the various physical and chemical parameters used to ascertain the pro-

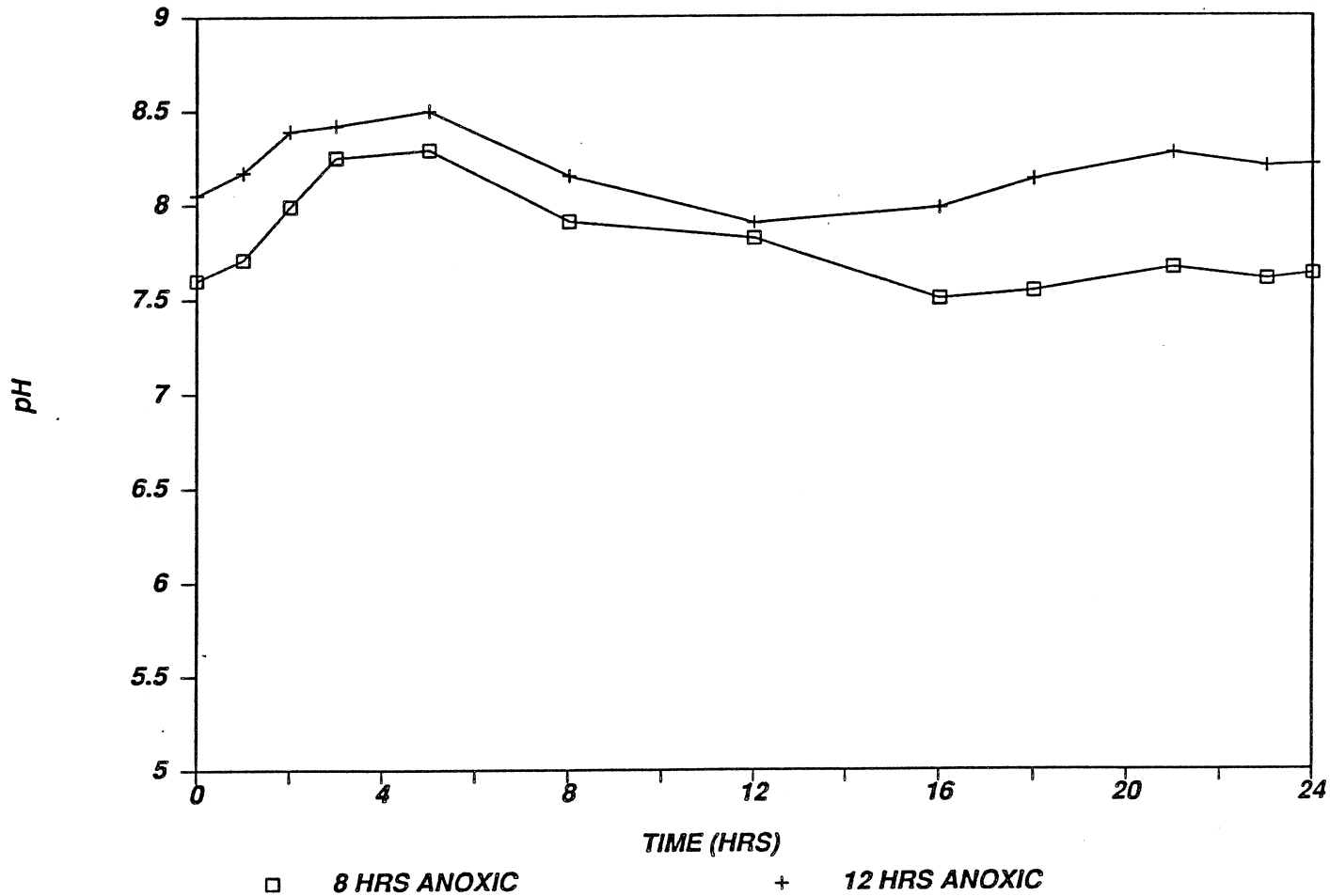


Figure 5.29 pH track data for reactors operating with 8 and 12 hrs anoxic and 9/20 (HRT/BSRT, days)

Table 5.5 RESULTS FROM INFLUENT/EFFLUENT AND MIXED LIQUOR ANALYSES FOR SBR OPEATING AT 9/20 (HRT/BSRT) AND TEMPERATURES OF 5, 10 AND 21° C

Parameter	5°C		10°C		21°C	
	Raw	Treated Waste	Raw	Treated Waste	Raw	Treated Waste
NH_3-N (mg/l)	1055	638	1126	34	1265	2
NO_2-NO_3-N (mg/l)	4	5	5	96	6	182
COD (mg/l)	30220	2050	30268	986	31175	845
SS (mg/l)	10485	751	10055	390	10690	283

Parameter	Mixed Liquor		
	5°C	10°C	21°C
MLSS (mg/l)	14925	14160	15750
Sludge Vol. 1.0 l cylinder (ml)	876	591	475
pH	8.5	7.3	6.6
DO (mg/l)	7.1	6.8	6.4

cess effectiveness at these temperatures.

From the analyses of the data it can be seen that the ammonia concentration in the treated effluent varied significantly between the three reactors. The NH_3-N reduction was 39.5%, 97% and 99.8% for reactors operating at 5, 10 and 21°C, respectively. It is evident that the oxidation of ammonia was adversely affected by temperatures below 10°C. As a matter of fact the average NO_2-NO_3-N concentration for the reactor operating at 5° was only 5 mg/l, while the average ammonia nitrogen was 638 mg/l. The performance of the reactor operating at 10°C was quite satisfactory, and a distinct improvement in the nitrification process was achieved. The effluent contained on average 96 mg/l of oxidized nitrogen, but still the amount of reduced nitrogen (33.6 mg/l) released was relatively high. Comparatively, the reactor operating at 21°C displayed the best results, due to the high degree of nitrification. Only residual amounts of ammonia (2 mg/l) were detected in the treated effluent.

The difference in the results obtained for these three reactors can be explained as a consequence of slow microbial activity at low temperatures and inhibitory reaction of nitrifiers to free ammonia. The optimum temperature for nitrifiers growth is in the range of 28 to

36°C (Focht and Chang, 1975). Probably below 10°C these mesophilic bacteria were supressed in an inhospitable environment, and thus they may have spored or functioned at a very slow rate consequently restricting the nitrification process. Furthermore to aggravate the problem, the pH in the 5°C reactor was consistently above 8, and hence contributed to the presence of un-ionized ammonia which could had inhibited the growth of nitrosomonads and nitrobacters.

According to Loehr (1984), there is conflicting information regarding nitrification at low temperatures. For conventional activated sludge treating domestic waste some studies have indicated that nitrification did not develop below 10°C, other researchers were able to accomplish nitrification at temperatures as low as 1°C with a submerged filter. Tam et al. (1986) conducted research on the treatment of wastewater from a milking center using a SBR, however they did not find a conclusive trend relating low temperatures and nitrification. At 3.7°C and 10.5°C the effluent produced contained 43 and 56% of the influent NH_3-N and oxidized nitrogen, respectively. It is difficult to compare the results from the previous studies, since process configuration and operation conditions such as HRT and BSRT varied as well as the characteristics of the waste treated. Based on the findings of the present experimental work, there is definitely a trend showing that nitrification was severely affected when the temperature

decreased below 10°C. This problem could be overcome by decreasing the nitrogen loading in the system, and for that purpose the HRT would have to be increased and hence the size of the reactor would be greater. Another alternative is to insulate the reactor and take advantage of the exothermic nature of aerobic biodegradation.

Temperatures between 5 to 21°C had little effect on the stabilization of organic matter, the COD removal ranged from 93 to 97%. Nevertheless, it was found that for temperatures less than 10°, an oscillation of COD in the treated effluent was observed during the testing period, this variability being reflected in the large calculated standard deviation. This behaviour could possibly be related to microbial population dynamics shifting from the mesophilic bacteria to psychrophilic bacteria. Although the type and number of microorganisms present in the reactors were not determined, the environmental conditions such as temperature, pH and dissolved oxygen were at a favorable level for the establishment of psychrophiles. These microorganisms grow slowly and utilize substrate at a lower rate, thus affecting the degree of the treatment efficiency of the SBR in terms of COD stabilization.

A noticeble fact was the persistent turbidity observed in the treated supernatant from the reactor operating at 5°C, and it was reflected in the high concen-

tration of SS (751 mg/l) discharged in the effluent. These occurrences could be related to the poor settling in this reactor, since on average the sludge occupied 876 ml of volume in the 1.0 liter graduate cylinder test. As indicated in table 5.5, there was a great improvement in the performance of the reactors functioning at 10 and 20°C. On average the settled sludge occupied 591 and 475 ml and the SS in the treated effluent were 390 and 283 mg/l, respectively. From the analysis of the results obtained for temperatures between 5 to 21°C, it is not a recommended procedure to operate the SBR below 10° C, for liquid swine manure treatment.

5.4 Summary

The results from the present experimental work show that liquid swine manure can be effectively treated in a sequential batch reactor operating under appropriate conditions. The SBR functions on the basis of cyclic variations and thus it permits considerable flexibility in the temporal coordination of reactor phases, to provide the requisite conditions for the combined carbon and nitrogen removal. While an assortment of potentially suitable operating modes could be devised, the format chosen included an anoxic fill that encouraged denitrification with exogenous carbon forms, followed by an aerobic react sequence for oxidation of organic and inorganic contaminants. An anoxic operation in the final segment of react permitted endogenous denitrification. Finally, the settling of the biomass in the reactor allowed the drawing of the treated supernatant.

The following observations are made based on the experimental results:

- The performance of SBR was evaluated at 21^{o} C over a range of 3 to 9 days hydraulic retention time and 10 to 30 days of biological solids retention time. The HRT was seen to be the determining factor in the process

efficiency, in terms of organic carbon and nitrogen treatment. In the tested range, the BSRT showed to have a negligible effect on the quality of the effluent, especially when the HRT ranged from 5 to 9 days. The function of BSRT was mainly to maintain an adequate level of biomass in the reactor.

- Analyses of the influent waste and the treated effluent revealed that for the SBR operating at 5, 7 and 9 days HRT/10, 20 and 30 days BSRT, the average removal of SS and COD ranged between 93 to 97% and 94 to 97%, respectively. It was also observed that prolonging the HRT from 7 to 9 days did not significantly change the process efficiency, and hence it would not justify the larger reactor volume that would be required in the latter case.

- The reduction in NH_3 - N concentration was above 99% for the SBR operating at 7 and 9 days HRT/ 10, 20 and 30 days BSRT, and the residual NH_3-N discharged in the effluent was 2.5 mg/l on the average. Ammonia removal dropped to 95% as the HRT decreased to 5 days. High levels of unreacted ammonia, ranging from 227 to 910 mg/l, were released in the effluents from the reactors operating at 3 days HRT.

- In general, the nitrification process was well established in the SBR. The average oxidized nitrogen concentration, which was mainly in the nitrate form, ranged from 120 to 371 mg/l for reactors operating at 5, 7 and 9 days HRT/ 10, 20 and 30 days BSRT. Due to the release of hydrogen ions from the nitrification reaction the pH was depressed in these reactors. It was found that reactor 3/10 (HRT/BSRT) had the poorest performance, the NO_2-NO_3-N concentration being only 12 mg/l on the average. In this case the combination of nitrogen loading and high pH must have led to the formation of free ammonia, which inhibited the activity of the nitrifiers.

- Nitrification/denitrification processes were conducted in the same reactor by including alternatively aerobic and anoxic periods in the SBR cycle. An anoxic fill stage, rich in organic carbon, favoured denitrification, and as a result a steep drop in NOx occurred. With the aeration supply during react, oxidation of ammonia that had accumulated during fill occurred without any inhibitory reaction from the nitrifiers. Within 15 hours nitrification was accomplished, and a second anoxic period was conducive to endogenous denitrification, but it occurred at a slow rate due to limited ammounts of readily biodegradable organic

carbon. The removal of inorganic nitrogen ranged from 86 to 93% for reactors operating with 3 to 12 hours anoxic and 9/20 (HRT/BSRT). Based on the overall performance it is not recommended to operate the SBR under more than 8 hours of anoxic conditions.

- The performance of the SBR was adversely affected by temperatures below 10°C. In the reactor operating at 5°C and 9/20 (HRT/BSRT), the low temperature plus the high substrate loading totally suppressed the activity of the nitrifyers and produced an unacceptable effluent. A remarkable improvement in the treatment process was attained when the temperature was raised to 10°C, the combined removal of NH_3-N and COD being 97 and 96%, respectively.

- The settling characteristics of the mixed liquor in the SBR was affected by HRT and BSRT. The volume occupied by the sludge showed a decreasing trend as the HRT was increased from 3 to 9 and BSRT was lowered from 30 to 10 days. Based on these studies, a BSRT of 10 to 20 days and a HRT between 7 to 9 days can produce a sludge with good settling performance. It was observed that operating the SBR with an anoxic period greater than 8 hours and temperatures below 10°C jeopardize the mixed liquor settling process.

- The SBR demonstrated a high capacity for odour control. Indeed the offensive smell of raw manure gradually disappeared during the fill sequence, and a mild earthy odour prevailed during the rest of the cycle in the reactors operating between 5 to 9 days HRT. However, a mixed foul and ammonia smell was exuded by the reactors operating with a short HRT, such as 3 days or those with a prolonged anoxic period in the order of 12 hours.

- The anoxic fill operation proved to be an effective means of controlling the excessive foaming problem that commonly occurs when strong waste like swine manure is digested aerobically.

- Substantial cost savings can be made on aeration supply as a result of operating the SBR under anoxic conditions.

CHAPTER 6 MATHEMATICAL SIMULATION OF SBR OPERATION FOR THE TREATMENT OF LIQUID SWINE MANURE

6.1 Introduction

This chapter deals with the modeling and prediction of SBR performance in the treatment of concentrated organic wastes. The SBR system is a time dependent process which has proved to be very efficient in the treatment and reduction of the organic carbon and nitrogen load in the manure, when operated under appropriate conditions. The mathematical model describing the SBR system requires data on stoichiometric and kinetic coefficients for waste removal, microbial growth, oxygen supply and settling rates, which are derived from experiments.

The mathematical model considered in this study addresses two aspects only, namely waste removal (organic carbon and nitrogen) and the growth of microorganisms. Two periods, fill and react, are modeled using the mass balance approach. It is pertinent to note that most of the microbial activity, organic reduction and nitrification-denitrification processes take place in these two periods, and 85% of the operation time is consumed therein.

The mass balance approach is considered in this study for mathematically modeling the SBR system because of

its simplicity and the ability to describe better the different operation conditions. Basically this approach follows the continuity equation and the first law of thermodynamics.

The objective of this chapter is to examine the ability of the mass-balance models to describe the SBR system and provide the necessary operation design parameters.

6.2 Formulation of the Mathematical Model

In order to develop the model which will provide information on the concentration of substrate and microorganism during fill and the react periods of a sequencing batch system, stoichiometric relationships and the corresponding reaction kinetics must be defined first.

6.2.1 Fill Period

In this period, the reactor is partly filled gradually over a set period of time to a final, predetermined volume. Fermentative waste degradation by obligate and facultative anaerobic heterotrophs will likely occur in the fill sequence, with some degree of bacterial

synthesis. For carbohydrate biodegradation, the final end products consist of organic acids and alcohols. Proteinaceous matter subjected to anaerobic fermentation will release ammonium ions.

6.2.2 React Period

The react period is the principal step of the cycle during which it is expected that waste will be stabilized, and it may account for 40% to 85% of the cycle time. In this stage, provisions can be made to provide both an aerobic and an anoxic environment at different times. During the aerated period, aerobic heterotrophic organisms utilize the organic carbon from the waste to synthesize new cells and to furnish energy for synthesis. The overall reaction is:

$$\underset{\text{compounds}}{\text{organic carbon}} + O_2 \text{ -----> } \underset{\text{(new cells)}}{C_5H_7O_2N} + CO_2 + H_2O \qquad (6.1)$$

The rate of microbial metabolism is limited by the availability of nutrients and environmental factors such as temperature, pH and toxic matter. Under food-limiting conditions, an endogenous process occurs and microorganisms

lyse and release nutrients that are subsequently used by the living microorganisms.

The biological conversion of reduced nitrogen to an oxidized state is carried out mainly by autotrophic species of Nitrosomonas and Nitrobacter genera, and to a lesser extent by certain heterotrophic bacteria. The overall reaction is:

$$NH_4^+ + 2O_2 \text{ -----> } NO_3^- + 2H^+ + H_2O + \text{energy} \qquad (6.2)$$

The anoxic react sequence will provide suitable conditions for denitrification. This is a process by which nitrite nitrogen is reduced to nitrous oxides and nitrogen gas in the absence of dissolved oxygen. The reaction involved is:

$$NO_3^- \text{ ------> <----- } NO_2^- \text{ ----> <----- } N_2O \text{ -----> <----- } N_2 \qquad (6.3)$$

In the context of bacterial activity, denitrification serves not only as a means to obtain reduction of nitrogen, but also as a coupled reaction for the utilization of the incoming organic carbon substrate and the one resulting from the endogenous respiration.

6.2.3 Kinetics

The design of a biological wastewater treatment process requires the knowledge of the relationship between microbial growth and substrate utilization. Previous studies have found that when all the requirements for growth are met, microorganisms will follow the normal exponential growth pattern. However, it is commonly the case that substrate concentration decreases, and consequently a declining growth phase will follow the initial growth period. Both these cases are considered in Monod's model (1949). From experimental studies, it was observed that bacterial growth rate was a function of organism concentration, and of the limiting nutrient concentration, and it can be expressed quantitatively as:

$$\frac{dX}{dt} = UX \quad , \quad \text{where} \quad U = U_{max} \frac{S}{K_s + S} \qquad (6.4)$$

where: dX/dt = biomass growth rate (mass.volume^{-1}.time^{-1})

U = specific growth rate (time^{-1})

X = biomass concentration (mass.volume^{-1})

U_{max} = maximum value of U at saturation concentration of growth limiting substrate (time^{-1})

S = growth limiting substrate (mass.volume^{-1})

K_s = saturation constant equal to substrate concentration at which $U = U_{max}/2$ (mass.volume.$^{-1}$)

The specific increase in biomass which results from the utilization of a unit quantity of substrate is defined as growth yield, y, as follows:

$$y = \frac{dX}{dS} \tag{6.5}$$

The substrate utilization rate is proportional to the concentration of biomass present:

$$\frac{dS}{dt} = qX \qquad \text{and} \qquad q = K_{max} \frac{S}{K_s + S} \tag{6.6}$$

where: $\frac{dS}{dt}$ = rate of substrate utilization (mass.volume^{-1}. time^{-1})

q = specific substrate removal rate (time^{-1})

It is possible to develop a relationship between specific substrate utilization rate, specific growth rate and growth yield which is expressed as:

$$q = \frac{dS/dt}{X} = \frac{(dX/dt)/X}{(dX/dS)} = \frac{U}{y} \qquad (6.7)$$

Other factors such as lysing of cells, death and predation must be considered. These factors are lumped together, and the rate at which biomass is lost to endogenous respiration is proportional to the biomass present:

$$\left(\frac{dX}{dt}\right)_e = - K_d X \qquad (6.8)$$

where: dX/dt = rate of biomass reduction due to endogenous respiration (mass.volume^{-1}.time^{-1})

K_d = microbial decay coefficient (time $^{-1}$)

An equation that describes the relationship between net growth rate of microorganisms and the rate of substrate utilization can be expressed as:

$$\frac{dX}{dt} = y \frac{dS}{dt} - K_d X \qquad (6.9)$$

6.3 Mass Balance Equation

Models for biological waste water treatment processes can be developed by applying material balances and the abovementioned kinetic relationships. It is pertinent to note that models that have been proposed to describe the activated sludge processes are based on steady-state conditions within the treatment system (e.g. Lawrence and McCarty, 1970).

In the SBR process, steady-state conditions cannot be maintained, and therefore a dynamic model will have to be developed in order to simulate the removal of organic carbon and the nitrification process. The general material balance which is applied to the reactor is given by:

rate of mass accumulation	=	rate of mass flow into reactor	-	rate of mass flow out of reactor	+ -	reaction rate of utilization

(6.10)

6.3.1 Fill Period

During the fill period, the substrate concentration in the reactor may be expressed by the

following mass balance relationship:

$$\frac{d}{dt}(VS) = QS_o \pm VKSX \qquad (6.11)$$

where: V = reactor volume

S = substrate concentration in the reactor (mass. volume^{-1})

Q = influent flow rate (volume.time^{-1})

S_o = influent substrate concentration (mass.volume^{-1})

K = specific substrate removal rate (mass.volume^{-1}. time^{-1})

X = microorganism concentration (mass.volume^{-1})

A positive sign denotes a source and a negative sign denotes a sink. In the case of organic carbon, the sign should be negative to express the decrease in the organic carbon content due to the microbial activity. In that case equation 6.12 can be written in the following format:

$$\frac{d}{dt}(VS) = QS_o - VKXS \qquad (6.12)$$

Where the first term on the right-hand side of the Equation 6.12 refers to the substrate inflow to the reactor and the second term refers to the microbial consumption of the organic content.

The term on the left hand-side of equation 6.12 represents the change of the substrate concentration during the fill period, and it is an important parameter in the design of the SBR system. Equation 6.12 can be used to design the SBR system fill sequence using the following input data:

- S_o and S are the initial and final substrate concentration
- X the microorganism concentration
- K describes the reaction rate of substrate utilization and can be expressed by the Monod equation as:

$$K = K_{max} \frac{S}{K_S + S}$$

- dS/dt is the function describing the change in substrate concentration with respect to time. dS/dt can be obtained by curve fitting techniques using the experimental data obtained in Chapter 5. This function changes from one type of substrate to another, i.e. different functions express the change of organic carbon and nitrogen substrate.
- Q is the rate of flow of the substrate to the reactor.

Having used these input data in Equation 6.12, the volume of the reactor required for obtaining the best SBR performance can be determined. To simplify equation 6.12 for solution, the following assumptions are made.

- Although the volume varies during the fill sequence, it does not change a great deal (10% total for 10 days HRT, 12.5% for an 8 days HRT and so on). Thus it may be considered constant as a first approximation.

- The concentration of microorganisms, X, does not change significantly during the fill phase and thus a new variable $K_1 = KX$ can be used for the specific consumption rate of organic substrate. The results of mixed liquor suspended solids support this assumption as shown in Appendix B.

Applying these two assumptions to equation 6.12 leads to the following simplifications and solution.

$$\frac{d(VS)}{dt} = V\frac{dS}{dt} = QS_0 - K_1VS \qquad (6.13)$$

$$\frac{dS}{dt} = \frac{QS_0}{V} - K_1S \qquad (6.14)$$

$$\int_{S_1}^{S} \frac{dS}{\left(\frac{QS_0}{V} - K_1 S\right)} = \int_0^t dt$$

$$ln\left[\frac{\frac{QS_0}{V} - K_1 S}{\frac{QS_0}{V} - K_1 S_1}\right] = -K_1 t$$

$$\frac{S_0 Q}{V} - K_1 S = \left(\frac{S_0 Q}{V} - K_1 S_1\right)\left(e^{-K_1 t}\right)$$

$$S = \frac{S_0 Q}{K_1 V}\left(1 - e^{-K_1 t}\right) + S_1 e^{-K_1 t}$$

$$\boxed{S = \frac{S_0 Q}{K_1 V} + \left(S_1 - \frac{S_0 Q}{K_1 V}\right) e^{-K_1 t}} \qquad (6.15)$$

The general mass balance for microorganisms can be written as:

$$\frac{d(VX)}{dt} = \pm V U_1 X \qquad (6.16)$$

Equation (6.16) is based on the assumption that inflow does not contain a significant quantity of active microorganisms.

where: X = biomass concentration in the reactor (mass. volume^{-1})

U_1 = specific growth rate (time^{-1})

Also U_1 can be expressed as:

$$U_1 = \frac{y}{X}\left(\frac{dS}{dt}\right) - K_d \qquad (6.17)$$

where : $$\frac{dS}{dt} = K_{max}\frac{S}{K_S + S}X \qquad (6.18)$$

hence : $$U_1 = \frac{y \cdot K_{max} \cdot S}{K_S + S} - K_d \qquad (6.19)$$

The positive or negative sign denotes an increase or a decrease in microorganism concentration and this depends on the type of microbial species present in the reactor and the phase of operation as well.

6.3.2 React Period

In the react period, the flow rate is equal to zero and volume is constant. Thus the mass balance for substrate and microorganisms reduces to:

$$\frac{dS}{dt} = \pm K'XS \qquad (6.20)$$

and $$\frac{dX}{dt} = \pm U_2 X \qquad (6.21)$$

where: K′= specific substrate removal rate during the react sequence (mass.volume^{-1}.time $^{-1}$)

U_2 = specific microbial growth rate during react (time^{-1})

During the react period the solution of these Equations (6.20 and 6.21) will provide information necessary to predict, in similar situations, the expected change in substrate and the microorganism concentrations. These parameters are of great importance in the assessment of the SBR design.

If it is assumed that the change in microorganism population is not a large fraction (see Appendix B), then a simplification which allows a solution to the equation is as follows:

$$K_2 = K'X \tag{6.22}$$

$$\frac{dS}{K_2 S} = dt$$

$$\boxed{S = S_2 e^{-K_2 t} + S_p} \tag{6.23}$$

where: S_2 = initial substrate concentration during the react sequence

t = time beginning with the start of the react phase

S_p = relatively unbiodegradable substrate concentration which is not utilized by microorganism in the time scale of the SBR cycle.

It is important to point out that environmental factors such as temperature, toxicity effect, etc. can be accounted for by adjusting the specific rates of reactions.

Table 6.1 summarizes the differential equations which will be used to simulate the carbon removal and nitrification performance of a completely mixed sequential batch reactor under dynamic conditions. The experimental results will be used to verify the form of the theoretical model and to calculate the reactor rate variables.

6.4 Evaluation of Model Parameters

In this section, the determination of K_1, K_2 and S_p will be conducted using equation 6.15 and 6.23 derived previously in section 6.3. The input parameters required to determine K_1, K_2 and S_p are:

1. The initial substrate concentration during fill period (S_1) in $mg.l^{-1}$,
2. The initial substrate concentration during react period (S_2) in $mg.l^{-1}$,

TABLE 6.1 SUMMARY OF THE MATHEMATICAL MODEL

Fill Period

Substrate Removal

$$(\text{COD} ; \text{NH}_3 ; \text{NO}_x) \qquad S = \frac{S_0 Q}{K_1 V} + \left(S_1 - \frac{S_0 Q}{K_1 V} \right) e^{-K_1 t} \tag{6.15}$$

Microorganisms Growth

$$\frac{d(VX)}{dt} = V U_1 X \tag{6.16}$$

React Period

Substrate Removal

$$(\text{COD} ; \text{NH}_3 ; \text{NO}_x) \qquad S = S_2 e^{-K_2 t} + S_p \tag{6.23}$$

Microorganisms Growth

$$\frac{dX}{dt} = U_2 X$$

3. The influent substrate concentration (S_o) in $mg.l^{-1}$,
4. The volume of the reactor (V) in l,
5. The wastewater inflow rate (Q) in $l.h^{-1}$,
6. The time at which any of the parameters should be calculated (t) in h.

A worksheet was designed using the Lotus Computer Program to simplify the determination of the unknown parameters (K_1, K_2 and S_p) and incorporating the input above mentioned values. Copies of these worksheets are presented in Appendix B.

Two main concerns arose during the evaluation of K_1 and K_2. The first was whether K_1 and K_2 change considerably during both fill and react periods, or can they be considered as constant values according to the theoretical model. The second was whether the back-calculated concentrations (S_1 and S_2), using the estimated K_1 and K_2 fit the experimental data. In addition, it was hoped that K_1 and K_2 could be found to be practically constant for the full range of HRT and BSRT and influent concentrations which were used in the experimental program. If this were found to be true, than it would lend considerable validity to the theoretical models (equations 6.15 and 6.23) for use in the design of the SBR process parameters for a wide range of practical applications. Thus, it was important to evaluate the statistical properties of

both K_1 and K_2, as well as the best fit for the back-calculated concentrations.

The part of the K_1 and K_2 evaluations that involved the best fit to test data was relatively straightforward in concept. The objective of this procedure was to obtain the estimated K_1 and K_2 coefficients that gave the best fit between the observed data and model calculations.

Four sets of calculations were performed to determine K_1 and K_2 during: 1) the biodegradation of organic matter measured as COD, 2) the rate of NH_3 utilization, 3) the formation and utilization of oxidized nitrogen and 4) microorganism growth reflected by MLVSS. K_1 and K_2 were calculated for conditions: (1) different influent concentration at the same HRT and BSRT (three parameters were examined: COD, NH_3 and MLVSS), and (2) the same influent concentration and different HRT and BSRT. HRTs were 3, 5, 7 and 9 days, while BSRTs were 10, 20 and 30 days (three parameters were examined: COD, NH_3 and NO_x). The predictions of back-calculated concentrations were attempted for the same series, and HRT and BSRT effects were investigated.

6.4.1. Comparison between the Form of Experimental and Predicted Result Changes with Time

6.4.1.1 Change in Initial Concentration

In this subsection the effect of changing the influent concentration of COD, NH_3 and MLVSS on the evaluation of K_1 and K_2 were investigated.

Figures 6.1 to 6.4 display both experimentally measured and best fit predicted COD values at different dilutions of the screened liquid manure and designated from here on as series 1:4, 2:4, 3:4 and 4:4. HRT and BSRT were selected to be 8 and 20 days, respectively. The best fit values of coefficients K_1 and K_2 in the model equations 6.15 and 6.23 were chosen by using the total of the experimental values at all levels of HRT and these same K_1 and K_2 values were then used in the model for all conditions of concentrations and BSRT. It was observed that:

1. The overall configuration of both measured and predicted curves is very close. The regression analysis coefficients r^2 were found to be 0.947, 0.959, 0.847 and 0.963 for Series 1:4, 2:4, 3:4 and 4:4, respectively.

2. The predicted values in the fill period were generally higher than the measured values except for Series 4:4. The regression coefficient of this part of the curve

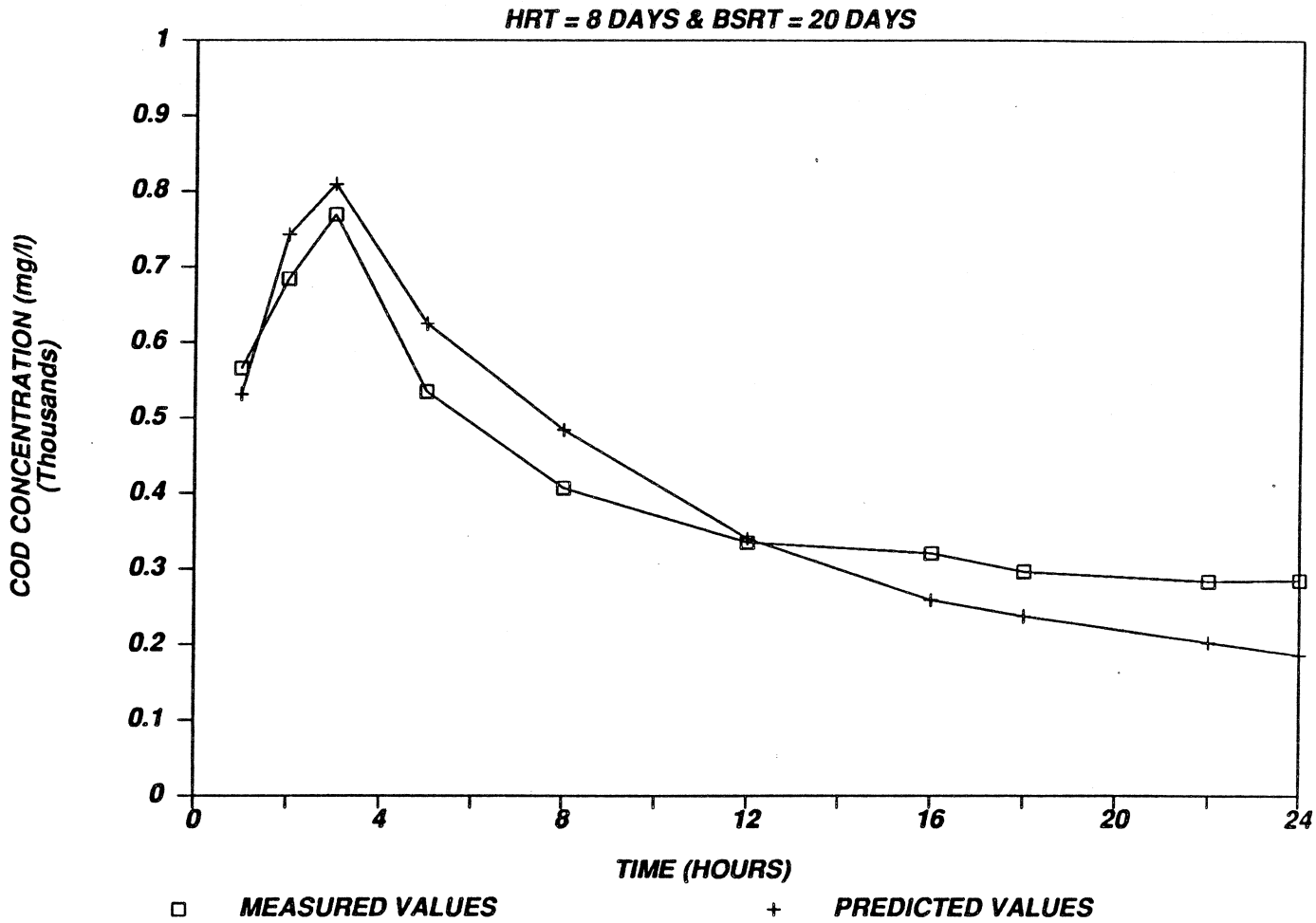


Figure 6.1 Measured and Predicted COD Concentration Profiles (1:4)

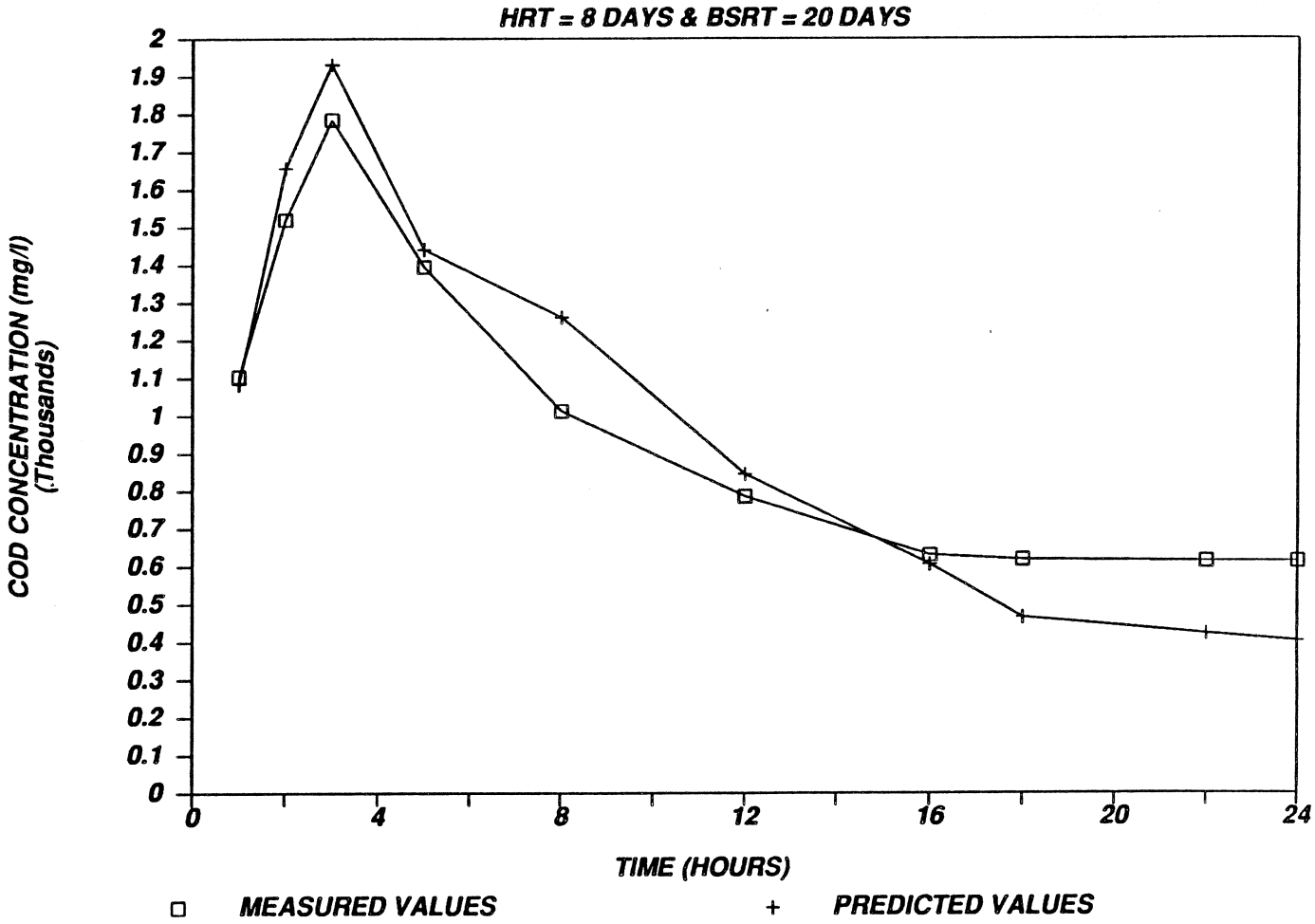


Figure 6.2 Measured and Predicted COD Concentration Profiles (2:4)

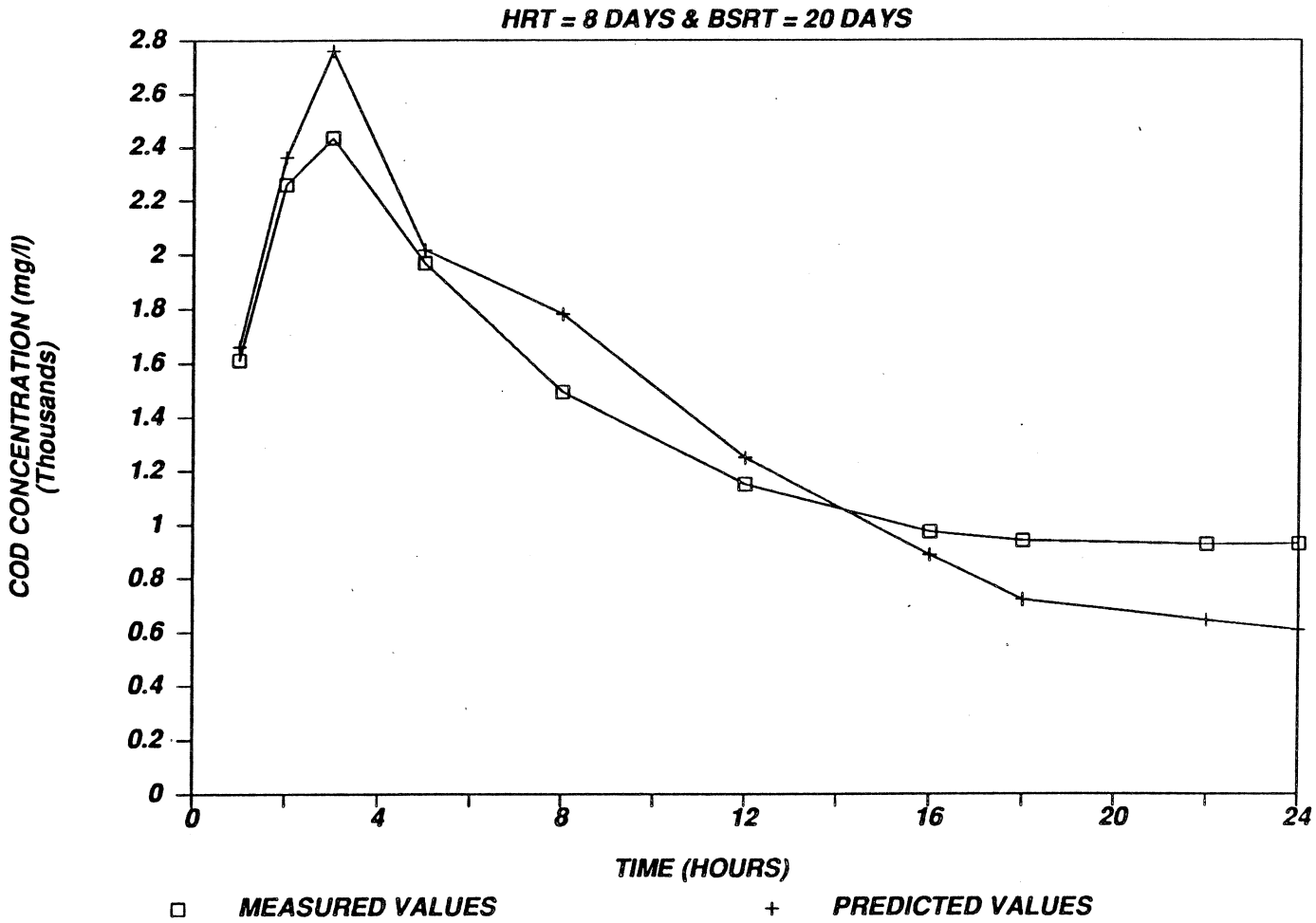


Figure 6.3 Measured and Predicted COD Concentration Profiles (3:4)

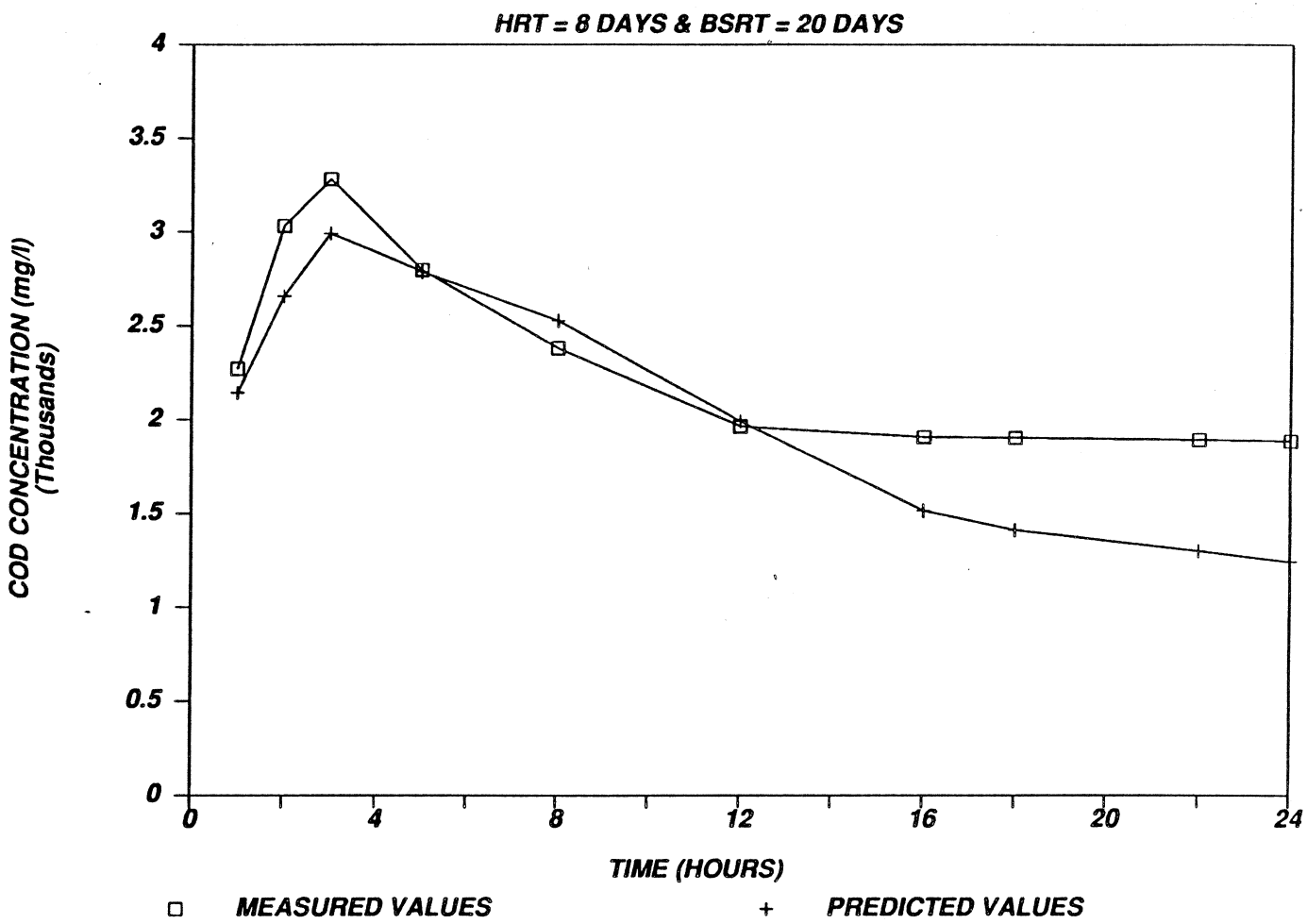


Figure 6.4 Measured and Predicted COD Concentration Profiles (4:4)

(fill period) was 0.95 on the average.

3. In the react period, the mathematical model over-predicted the COD values from 3 to 12 hours, however, it under-predicted the COD values in the rest of the cycle, which suggests the possibility of having two different reaction rates (K_2) in the react period. The first one should take into consideration the active reaction period where the available nutrient is mostly in the soluble biodegradable state and microorganisms are very active, and the second takes into account the substantial decrease in the growth limiting nutrients. This hypothesis was investigated, and the typical results are presented in Figures 6.5 and 6.6, for Series COD - HRT 8/BSRT 20 days both 1:4 and 4:4 (other results for Series COD - HRT 8/BSRT 20 days 2:4 and 3:4 influent concentration are presented in Appendix B). It was noticed that using a variable K_2 did not improve the properties of the fit a great deal. Table 6.2 summarizes the results of the regression analysis coefficients r^2. For practical purposes, a single value of K_2 is simpler to use and gives results which are just about as accurate as using two.

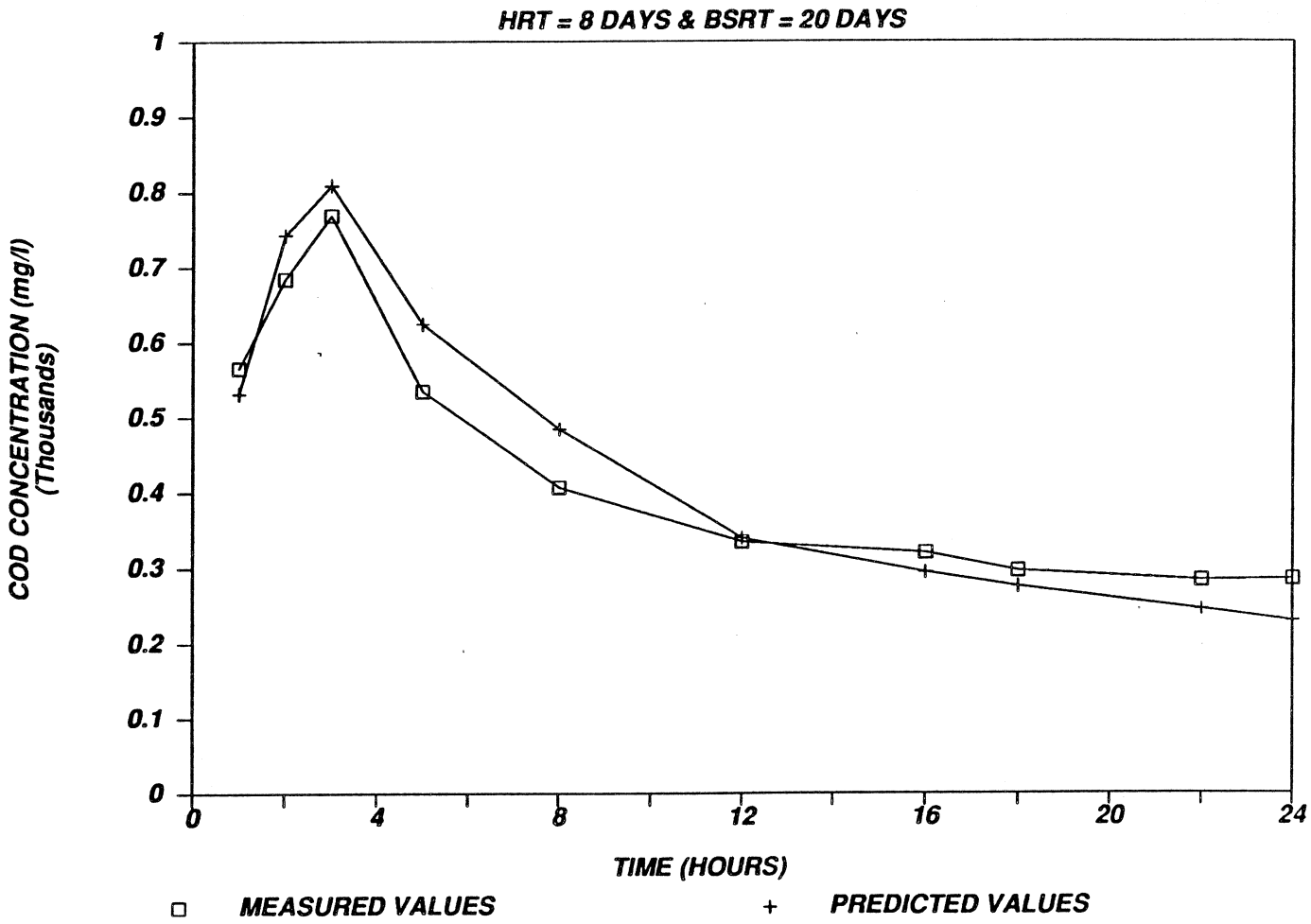


Figure 6.5 COD Concentration Profiles Based on Variable K_2 (1:4)

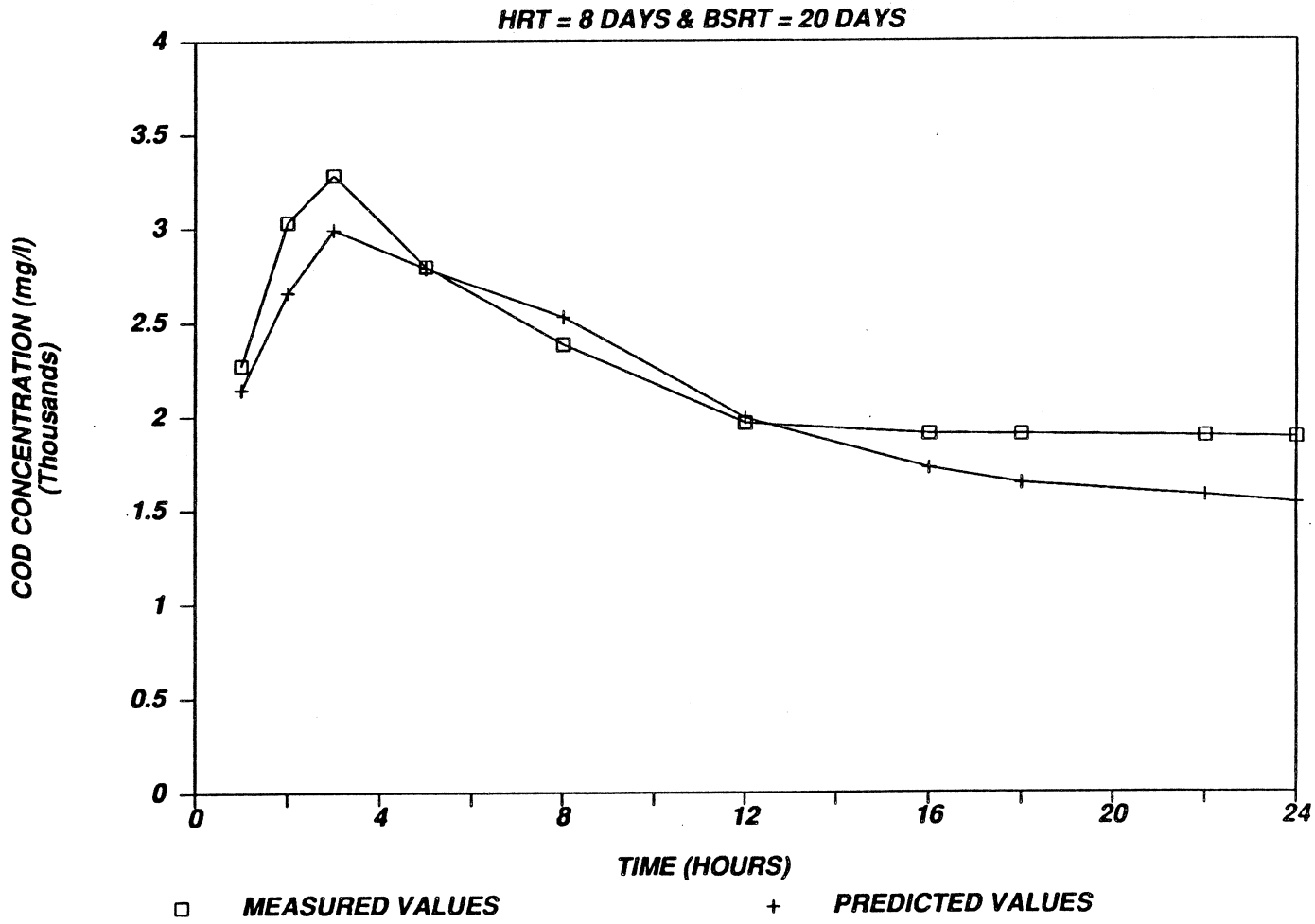


Figure 6.6 COD Concentration Profiles Based on Variable K_2 (4:4)

TABLE 6.2 REGRESSION ANALYSIS COEFFICIENT r^2 FOR COD - HRT 8/BSRT 20 DAYS

Series	r^2	
	K_2 fixed	K_2 variable
COD 1:4	0.947	0.959
COD 2:4	0.959	0.966
COD 3:4	0.847	0.971
COD 4:4	0.963	0.984

In the case of NH_3, the predicted values were higher than the experimentally determined data, especially at 2 and 3 hours (fill period). However, the fit improved significantly during the react period. The regression coefficients r^2 were 0.961, 0.977, 0.952 and 0.949 for Series NH_3 - HRT 8/BSRT 20 days and influent concentration 1:4, 2:4, 3:4 and 4:4, respectively. Figures 6.7 to 6.10 illustrate the NH_3 behavior for both measured and predicted values.

The predictions of MLVSS were basically unsuccessful due primarily to the relatively small variations of the parameter itself. MLVSS did not vary more than +_ 10% during the fill and react periods. The fits of model predictions were very poor as illustrated in Figures 6.11 and 6.12. (Series MLVSS HRT 8/BSRT 20 days and influent concentration 1:4 and 4:4). Nevertheless, it was found that by averaging the MLVSS values in the two periods of fill and

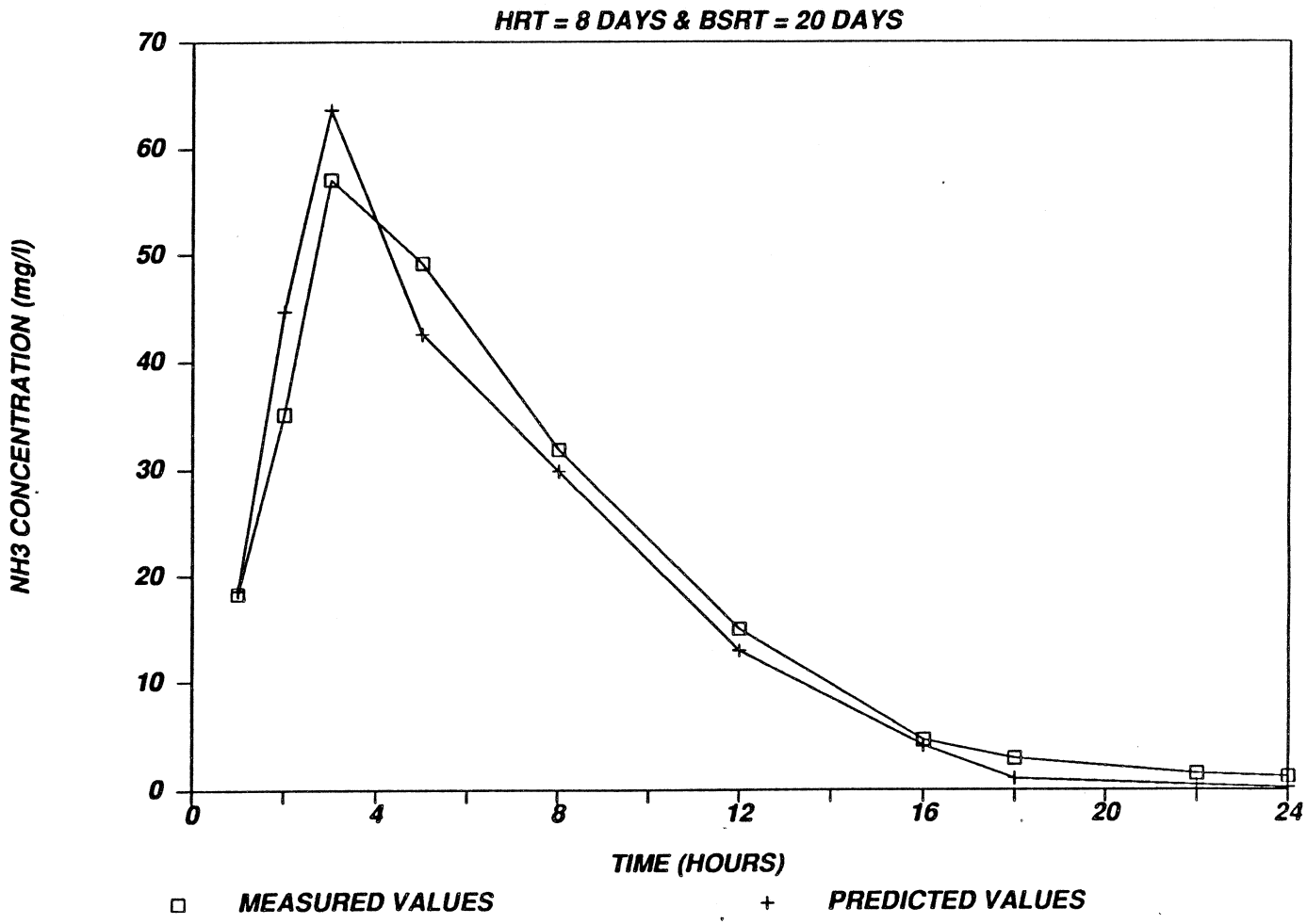


Figure 6.7 Measured and Predicted NH_3-N Concentration Profiles (1:4)

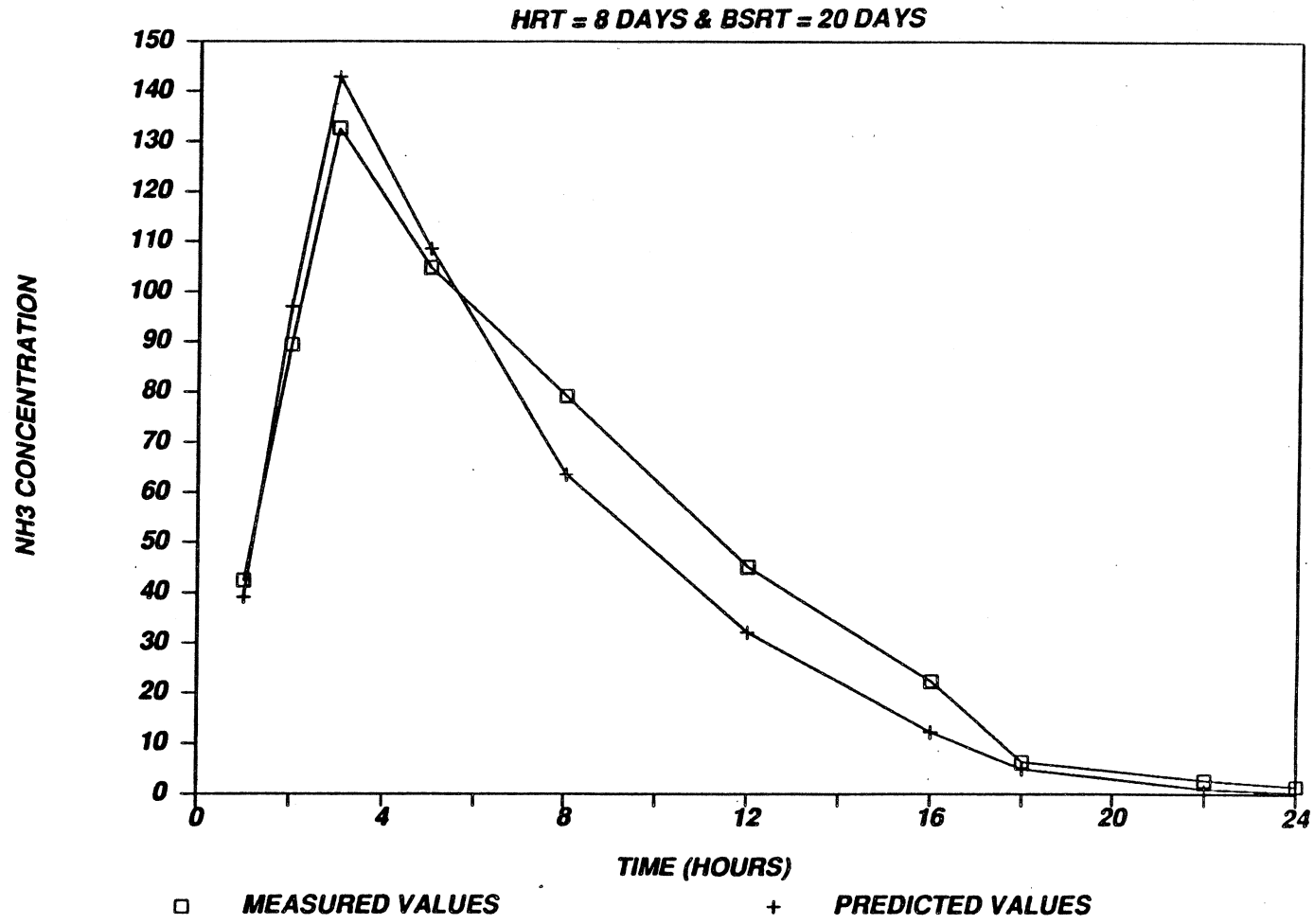


Figure 6.8 Measured and Predicted NH_3-N Concentration Profiles (2:4)

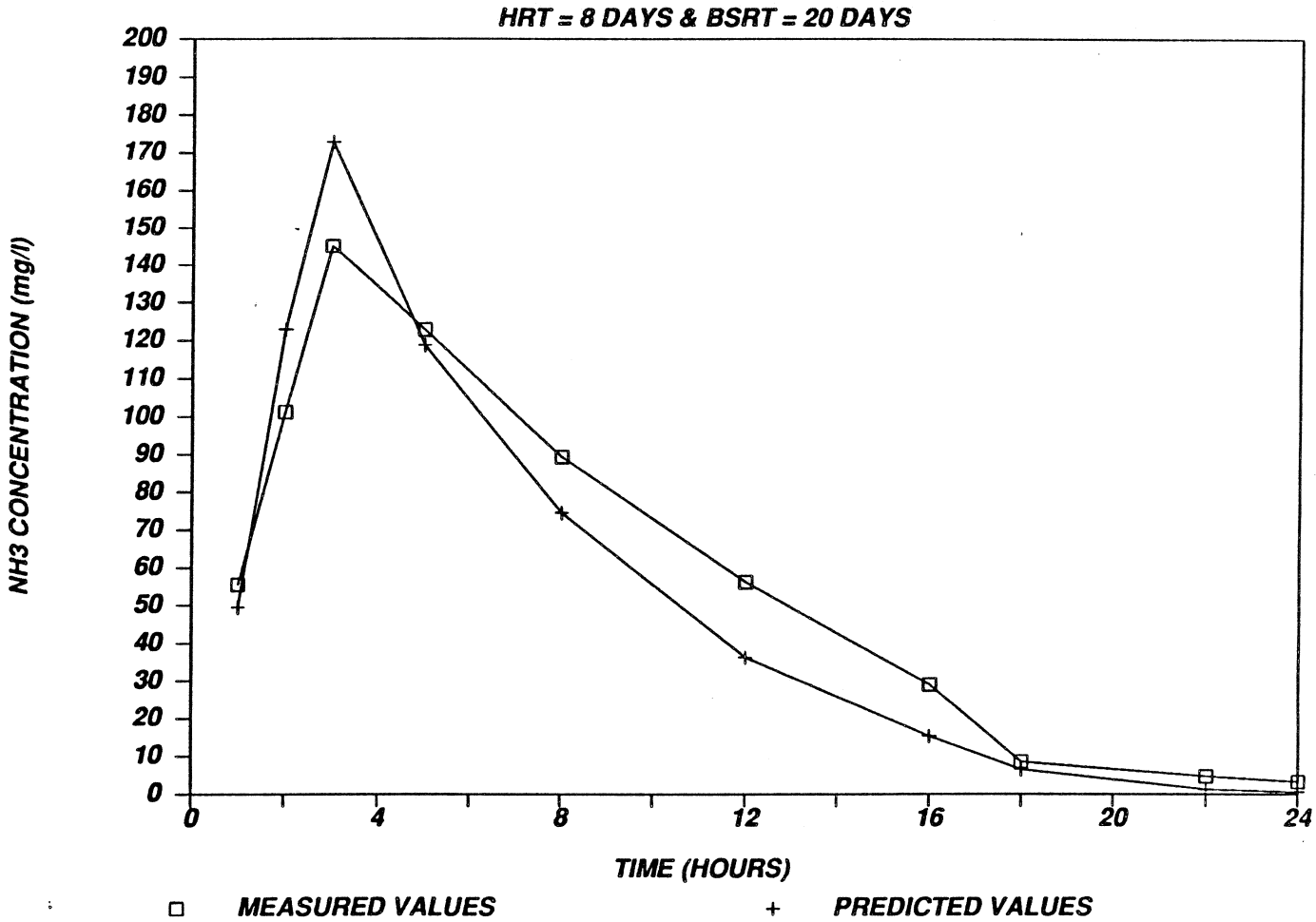


Figure 6.9 Measured and Predicted NH_3-N Concentration Profiles (3:4)

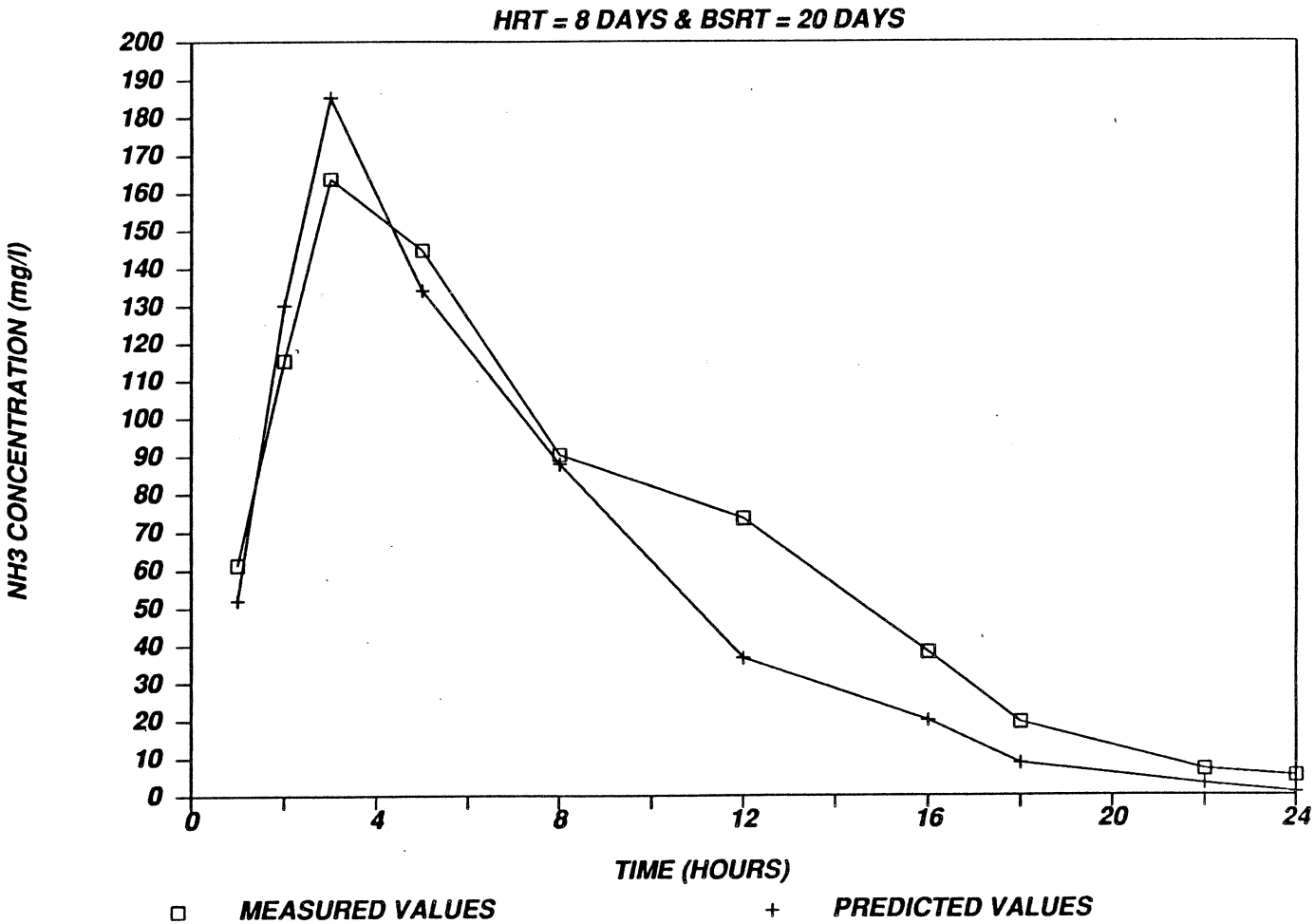


Figure 6.10 Measured and Predicted NH_3-N Concentration Profiles (4:4)

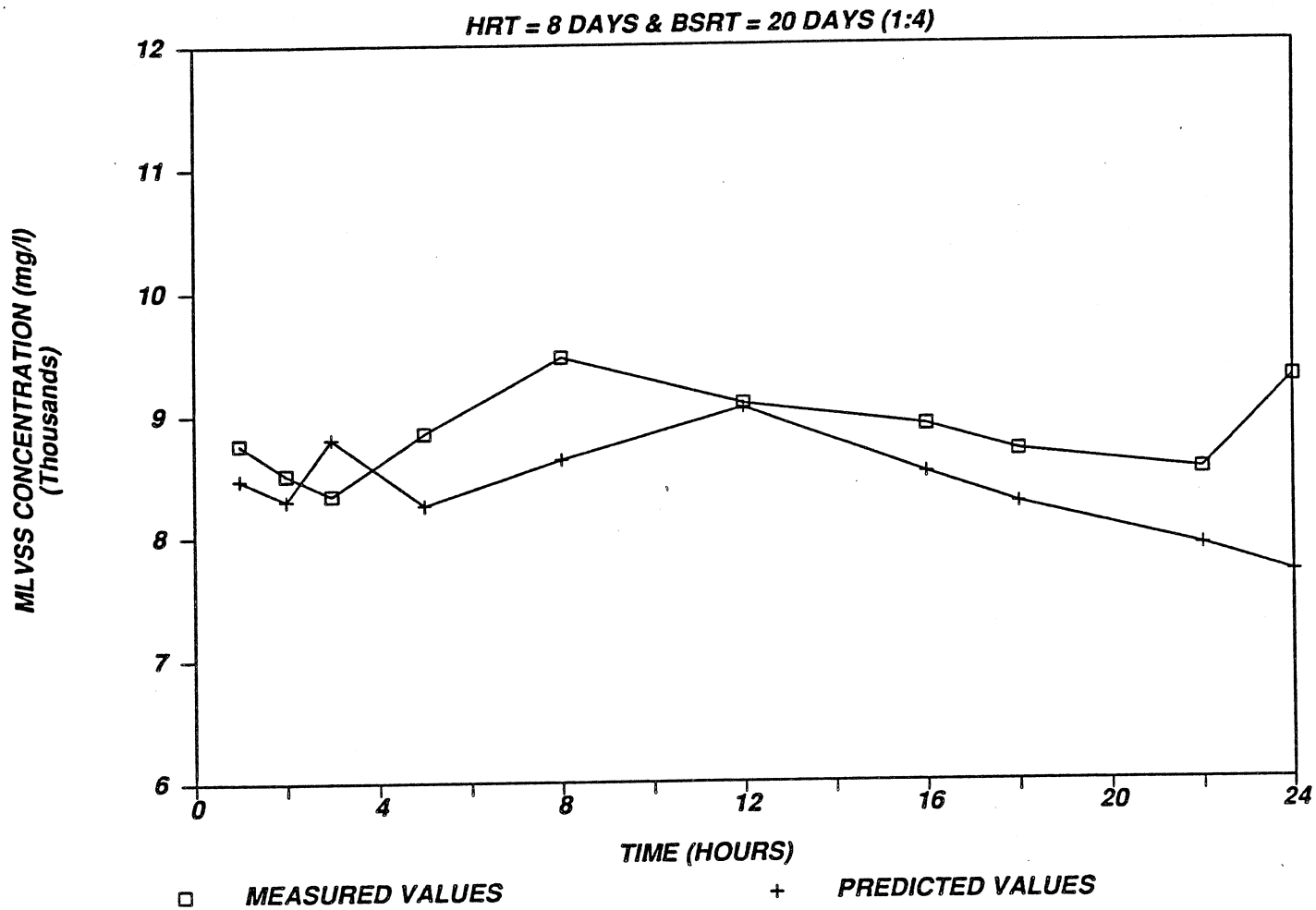


Figure 6.11 Measured and predicted MLVSS concentration profiles (1:4)

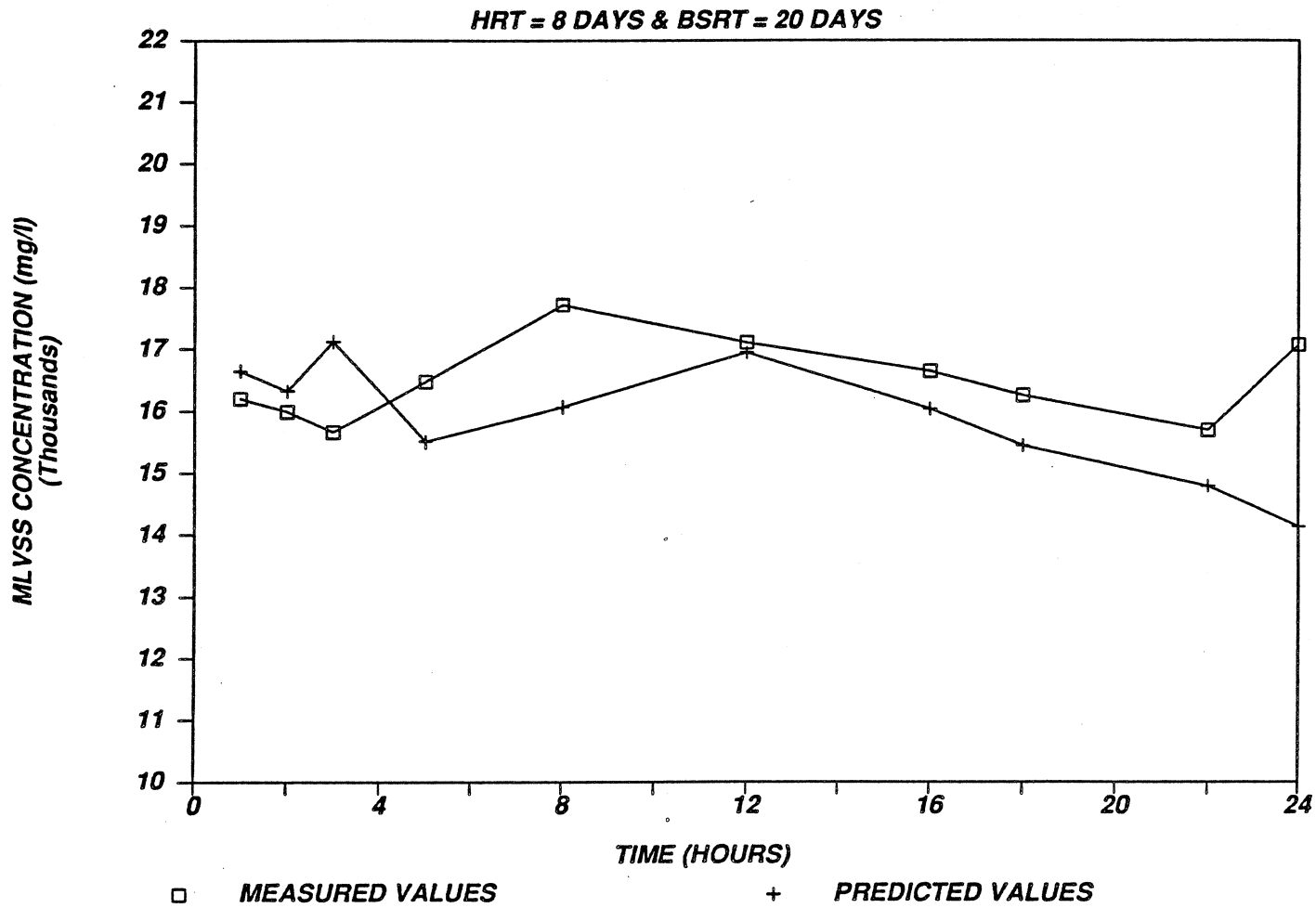


Figure 6.12 Measured and predicted MLVSS concentration profiles (4:4)

react, the MLVSS predictions slightly improved, and typical results from these observations are presented in Figure 6.13. (Results of other series are presented in Appendix B.)

Table 6.3 summarizes the results of the reaction rate coefficients (K_1 and K_2) for COD, NH_3 and MLVSS. It was observed that the same values of K_1 and K_2 could be used for all test conditions, and they did not have to be changed due to concentration changes. This observation was not unexpected because changing the concentration should not affect the rates of reaction.

TABLE 6.3 REACTION RATES K_1 AND K_2 FOR COD, NH_3 AND MLVSS (HRT 8/BSRT 20 Days)

Series	K_1 (h^{-1})	K_2 (h^{-1})
COD 1:4	0.4	0.02
COD 2:4	0.4	0.02
COD 3:4	0.4	0.02
COD 4:4	0.4	0.02
NH_3 1:4	0.2	0.1
NH_3 2:4	0.2	0.1
NH_3 3:4	0.2	0.1
NH_3 4:4	0.2	0.1
MLVSS 1:4	0.05	0.005
MLVSS 2:4	0.05	0.005
MLVSS 3:4	0.05	0.005
MLVSS 4:4	0.05	0.005

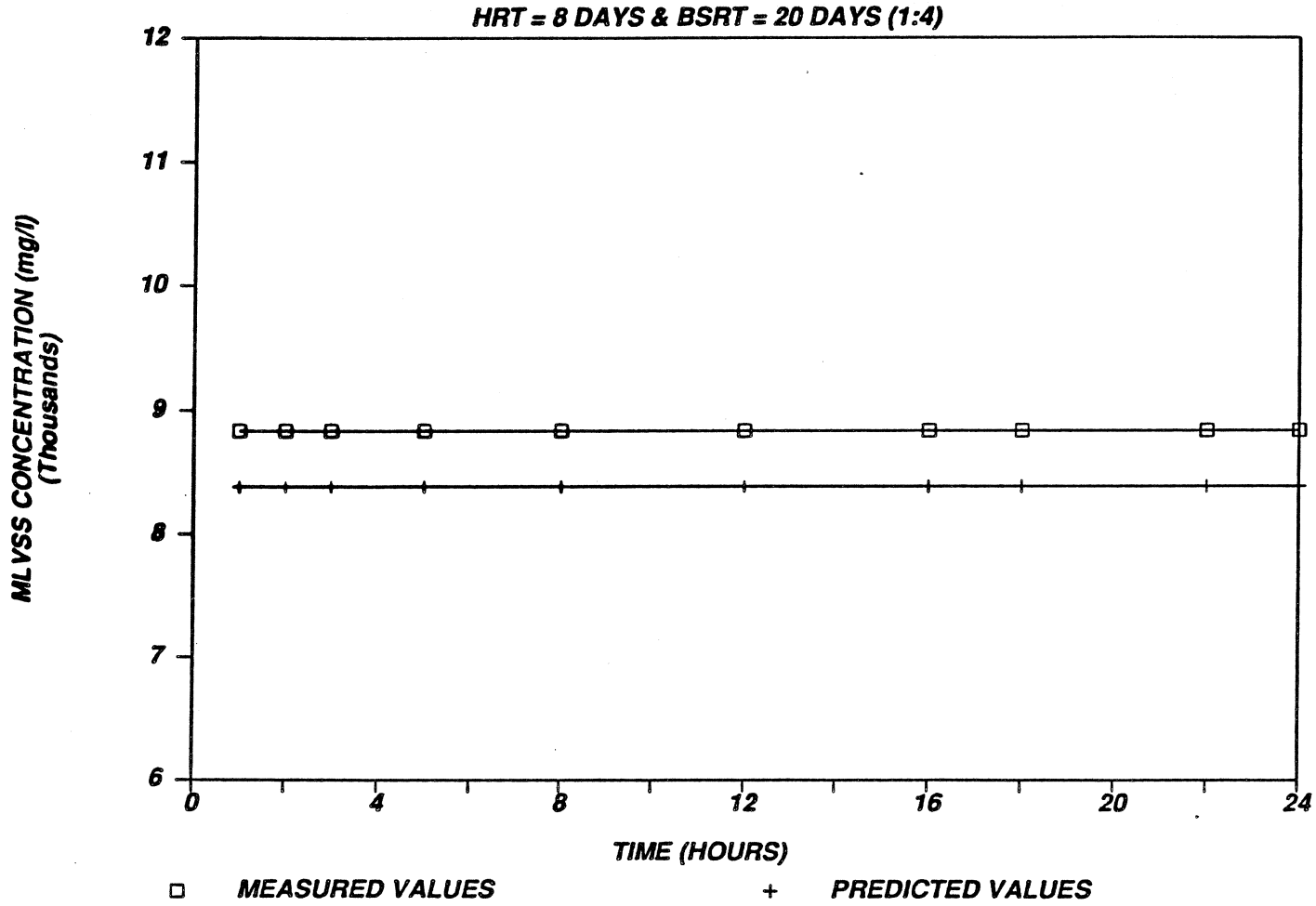


Figure 6.13 Measured and predicted average MLVSS concentration profiles (1:4)

The fact that the same values of K_1 and K_2 were found to give the best fit of average experimental measurements to the model, the successful prediction of behaviour in time is encouraging as far as the validity of the model is concerned. Had K_1 and K_2 been quite different for different influent concentrations, the applicability of the model, as summmarized by equations 6.15 and 6.23, would have been seriously suspect.

6.4.1.2 Change in HRT and Constant BSRT

The effects of changing the hydraulic retention time (HRT) on the variations of K_1 and K_2 were investigated for three different parameters: COD, NH_3 and NO_x. The HRTs were 3, 5, 7 and 9 days, while the selected BSRT was 20 days.

Figures 6.14 and 6.15 display the COD measured and predicted profiles at HRT 3 and 7 days, respectively, using the same values of K_1 and K_2 as were used in Figures 6.1 - 6.4. (Other results are presented in Appendix B.) The regression analysis showed that the overall r^2 coefficients ranged from 0.913 to 0.966 (Table 6.4). It was observed that:

1. During the fill period, especially in the first two hours of fill, the predicted values are higher than the

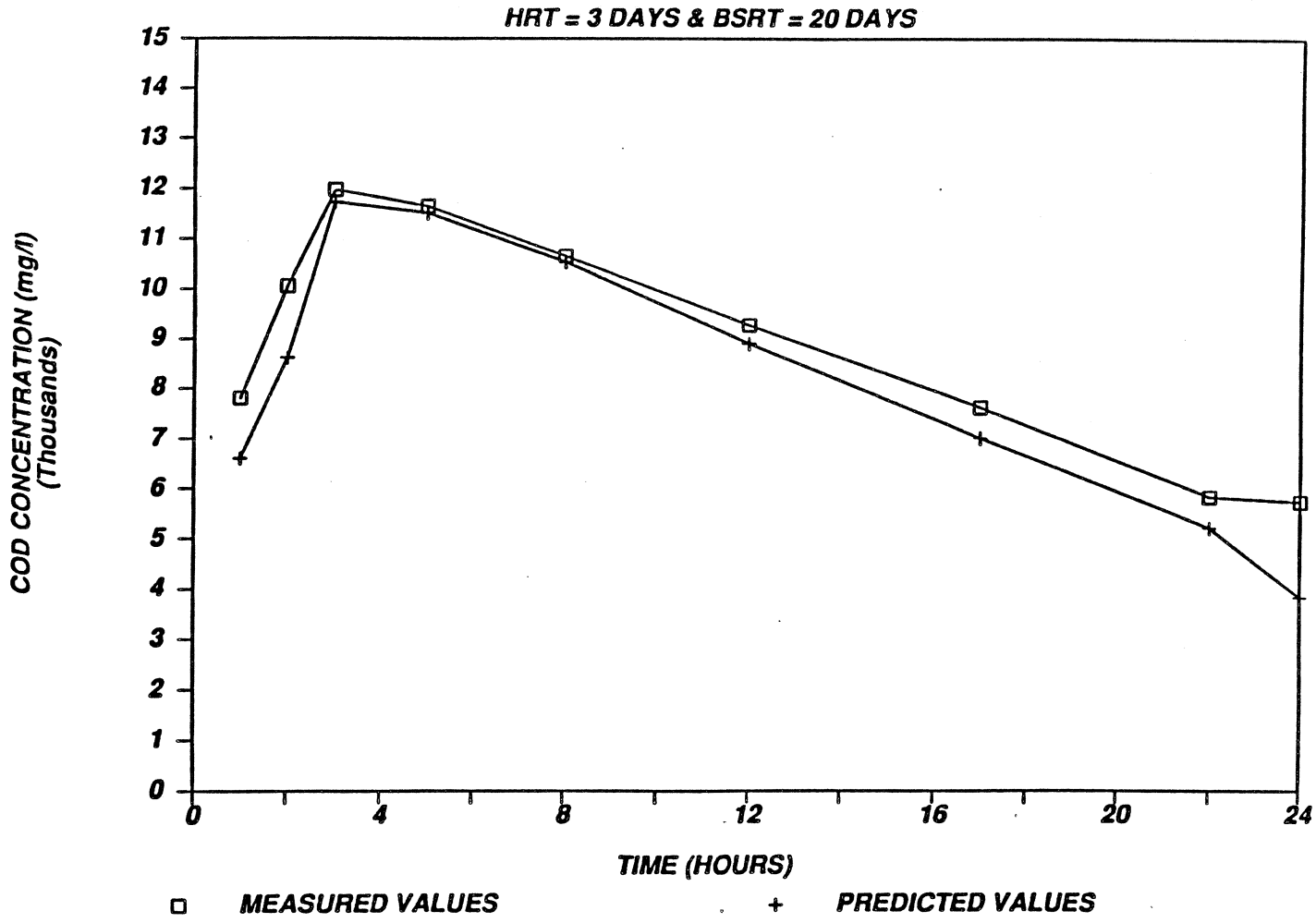


Figure 6.14 Measured and predicted COD concentration profiles for reactor operating at 3/20 (HRT/BSRT)

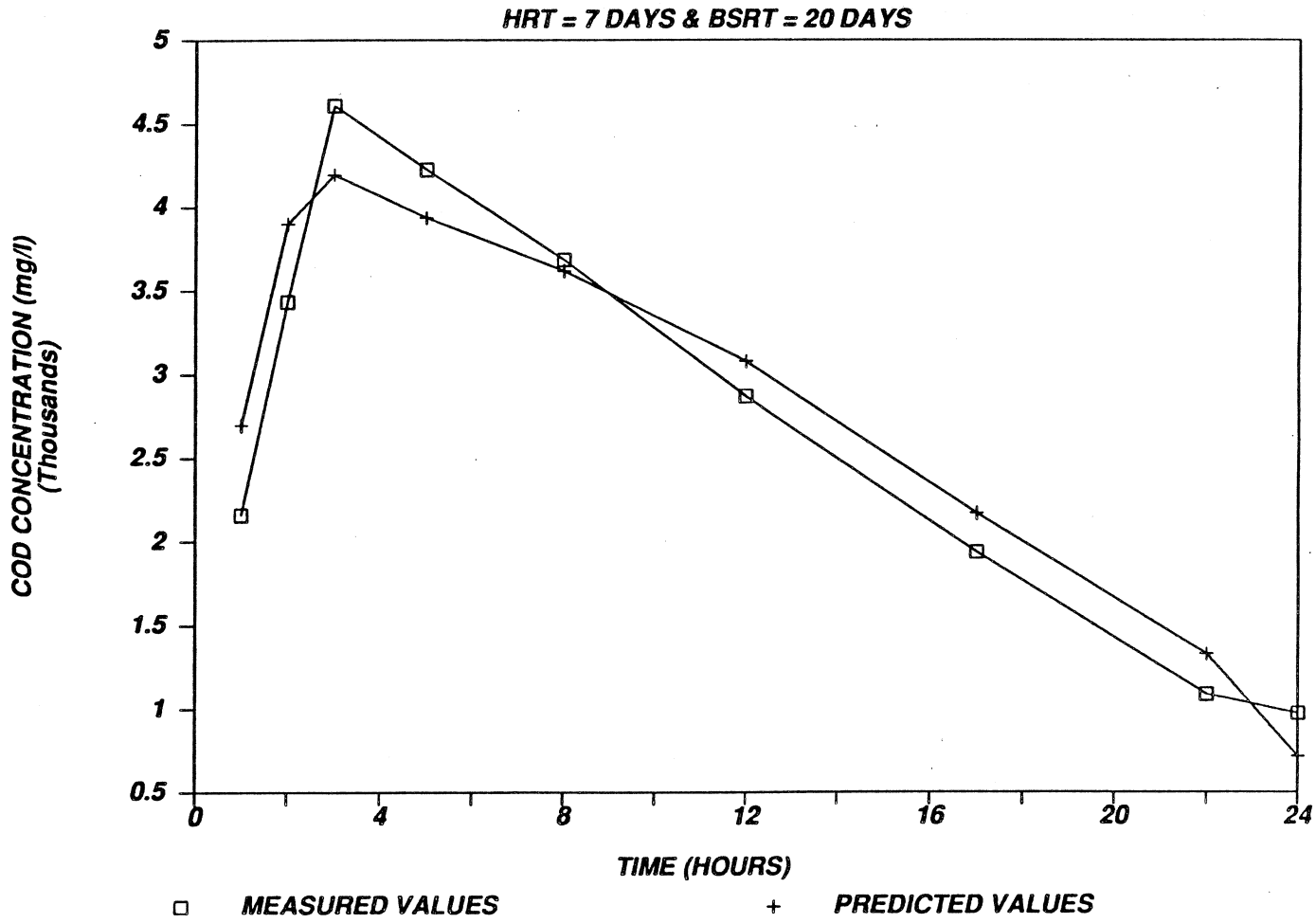


Figure 6.15 Measured and Predicted COD Concentration Profiles for Reactor Operating at 7/20 (HRT/BSRT)

TABLE 6.4 REACTION RATES K_1 AND K_2 FOR COD, NH_3 AND NOx AS WELL AS THE REGRESSION COEFFICIENTS r^2 (HRT = 3, 5, 7 AND 9 DAYS AND BSRT = 20 DAYS)

Series	K1	K2	r^2
COD 3/20	0.4	0.02	0.966
COD 5/20	0.4	0.02	0.942
COD 7/20	0.4	0.02	0.959
COD 9/20	0.4	0.02	0.913
NH_3 3/20	0.2	0.1	0.969
NH_3 5/20	0.2	0.1	0.955
NH_3 7/20	0.2	0.1	0.933
NH_3 9/20	0.2	0.1	0.930
NO_x 3/20	0.2	0.01	0.984
NO_x 5/20	0.2	0.01	0.982
NO_x 7/20	0.2	0.01	0.889
NO_x 9/20	0.2	0.01	0.921

measured ones. However, at the end of the fill the prediction is closer to the measured values. This basically indicates that the estimated K_1 is close to the actual one towards the end of the fill period, and this observation can be attributed to the combined effect of fresh incoming organic waste and the presence of oxidized nitrogen in the reactor itself, which enhanced the activity of the denitrifying bacteria that were under restricted environmental conditions during the settle, draw and idle periods in the previous cycle.

2. The predicted COD during the react period closely followed the time change pattern of measured values, with no definite trend of divergence. In case of the HRT of 3 days, the predicted COD values were lower than the measured values, where in the other cases (HRT 5, 7 and 9 days) the predicted COD values were slightly higher than the measured ones. As noticed in the previous Section 6.4.1.1, the predictions deviated more towards the end of the cycle. It is worthwhile to mention that towards the end of the react period, the remaining organics in the reactor are basically nonbiodegradable, so that the model predicts a higher substrate removal than actually occurred at the end of this period.

3- The reaction rates K_1 and K_2 for COD values at HRT 3, 5, 7 and 9 days; BSRT 20 days, are summarized in Table 6.4. It was observed that K_1 and K_2 values could be kept

constant despite the change in the HRT from 3 to 9 days.

Examination of the NH_3 predictions at different HRTs and BSRTs revealed that:

1- The regression analysis coefficients r^2 were in the range of 0.930 to 0.969.

2- During the fill period the model over-predicts NH_3 (especialy at 2 hour fill period), and this to some degree affected the statistical characteristics of the curve fitting.

3- The model followed measured results more consistently during the react period.

Figures 6.16 and 6.17 typically show the NH_3 measured and predicted values, and more cases are displayed in Appendix B.

The reaction rates during the fill and react periods for NOx are shown in Table 6.4 and typical results of the measured and predicted NOx values are illustrated in Figures 6.18 and 6.19 (other results are included in Appendix B). The regression analysis coefficients r^2 were in the range 0.889 to 0.984, which can be considered a good fit. It was observed that the predicted values are relatively close to the measured ones in the period between

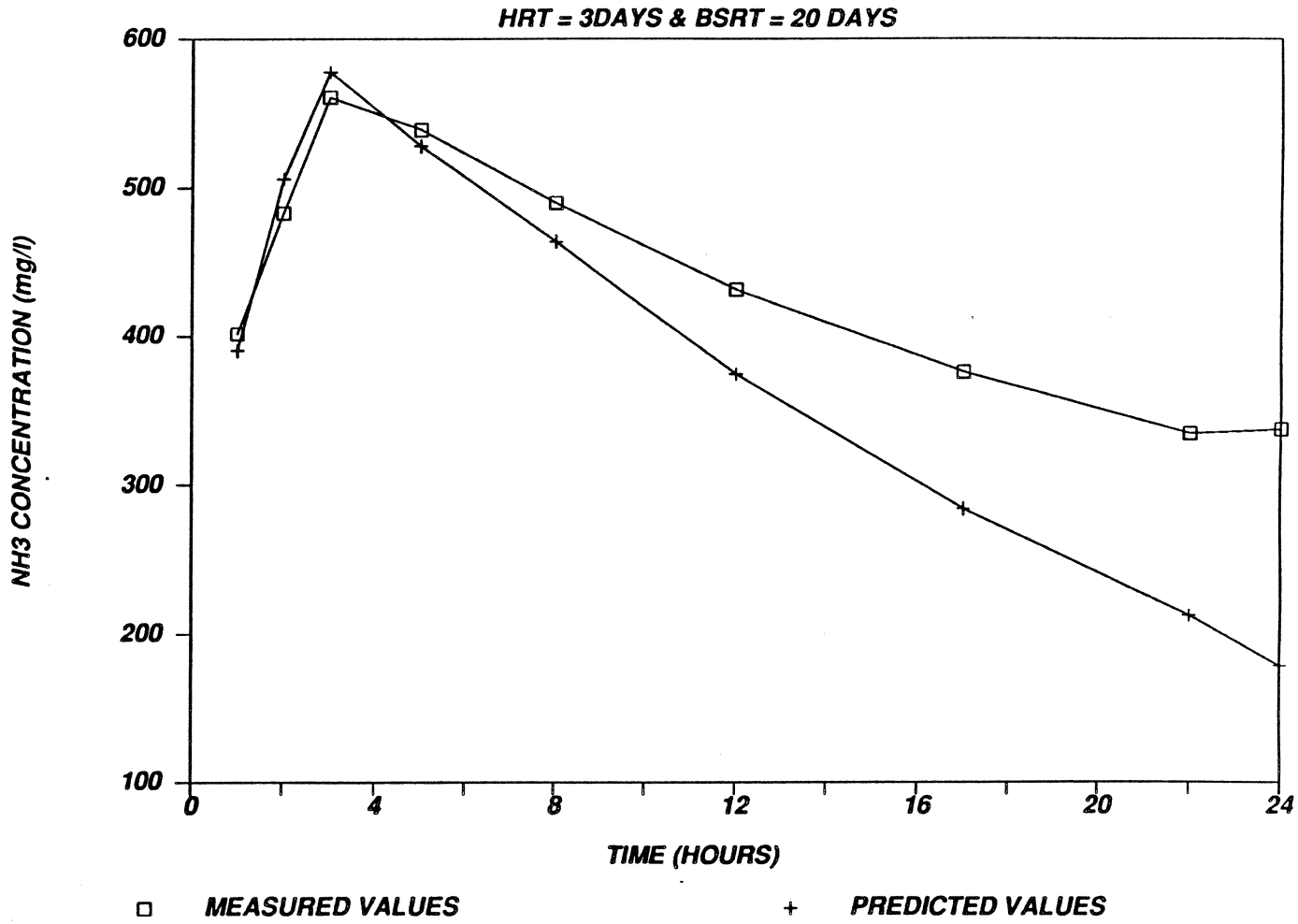


Figure 6.16 Measured and Predicted NH_3-N Concentration Profiles for Reactor Operating at 3/20 (HRT/BSRT)

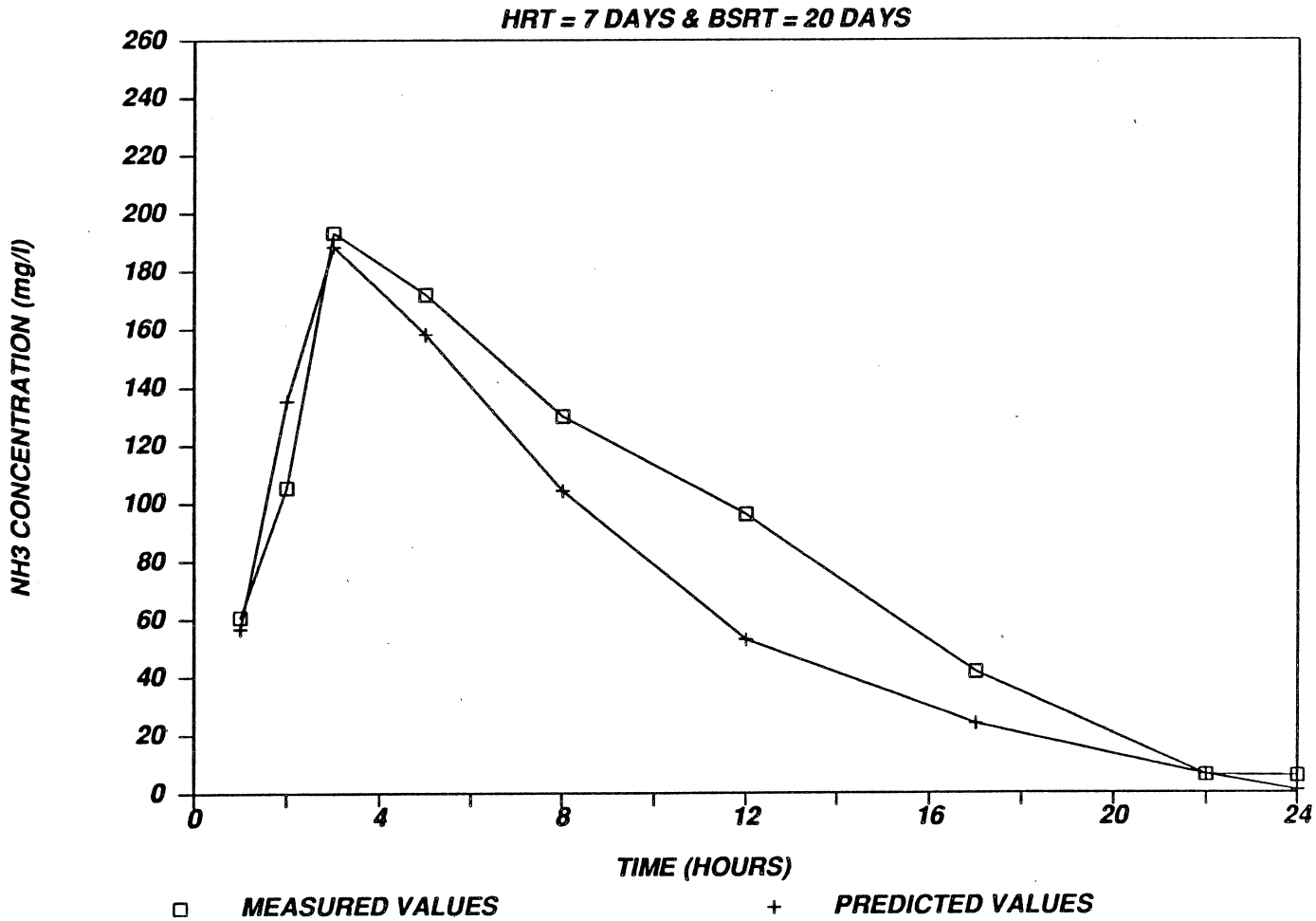


Figure 6.17 Measured and Predicted NH_3-N Concentration Profiles for Reactor Operating at 7/20 (HRT/BSRT)

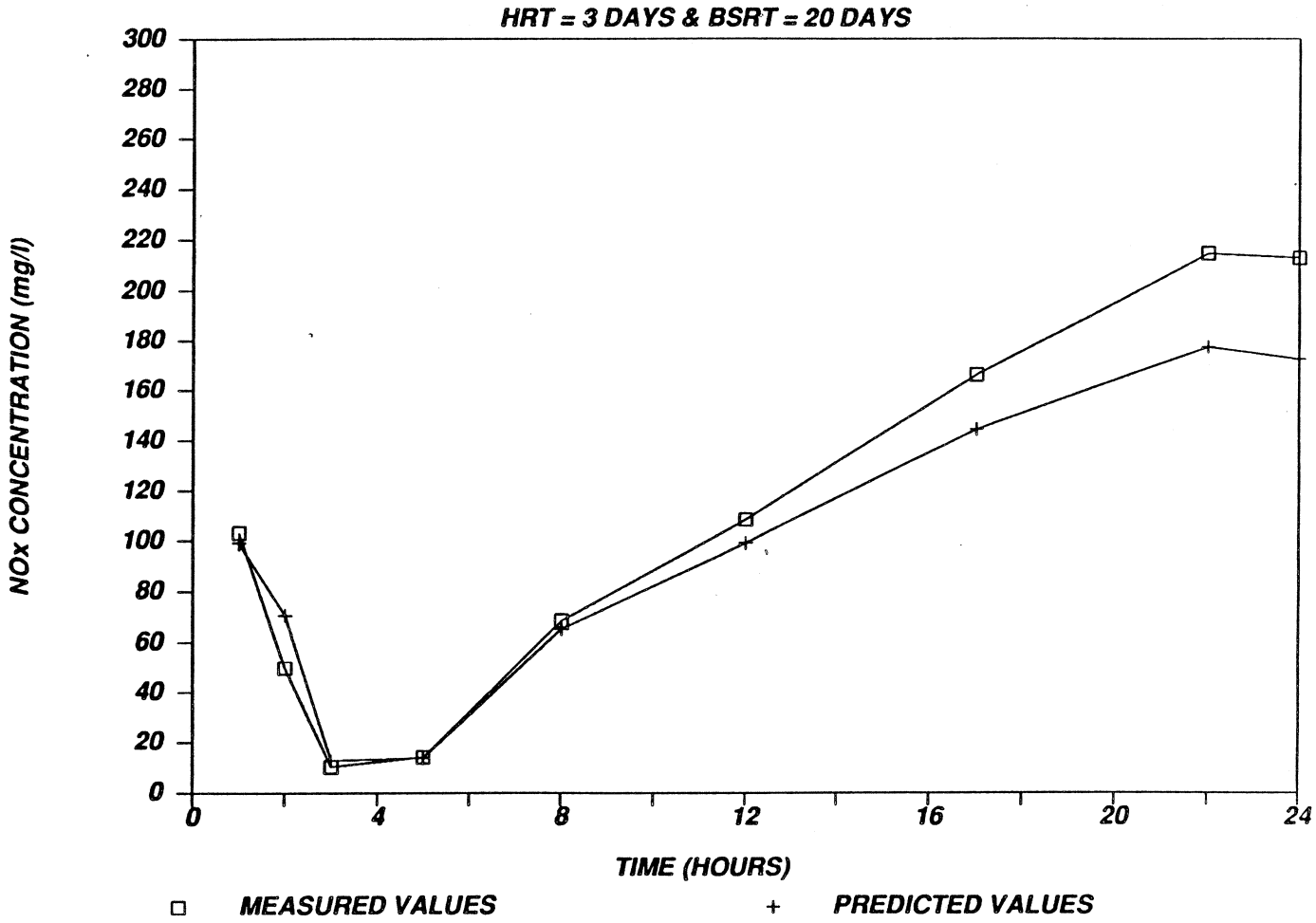


Figure 6.18 Measured and Predicted NOx Concentration Profiles for Reactor Operating at 3/20 (HRT/BSRT)

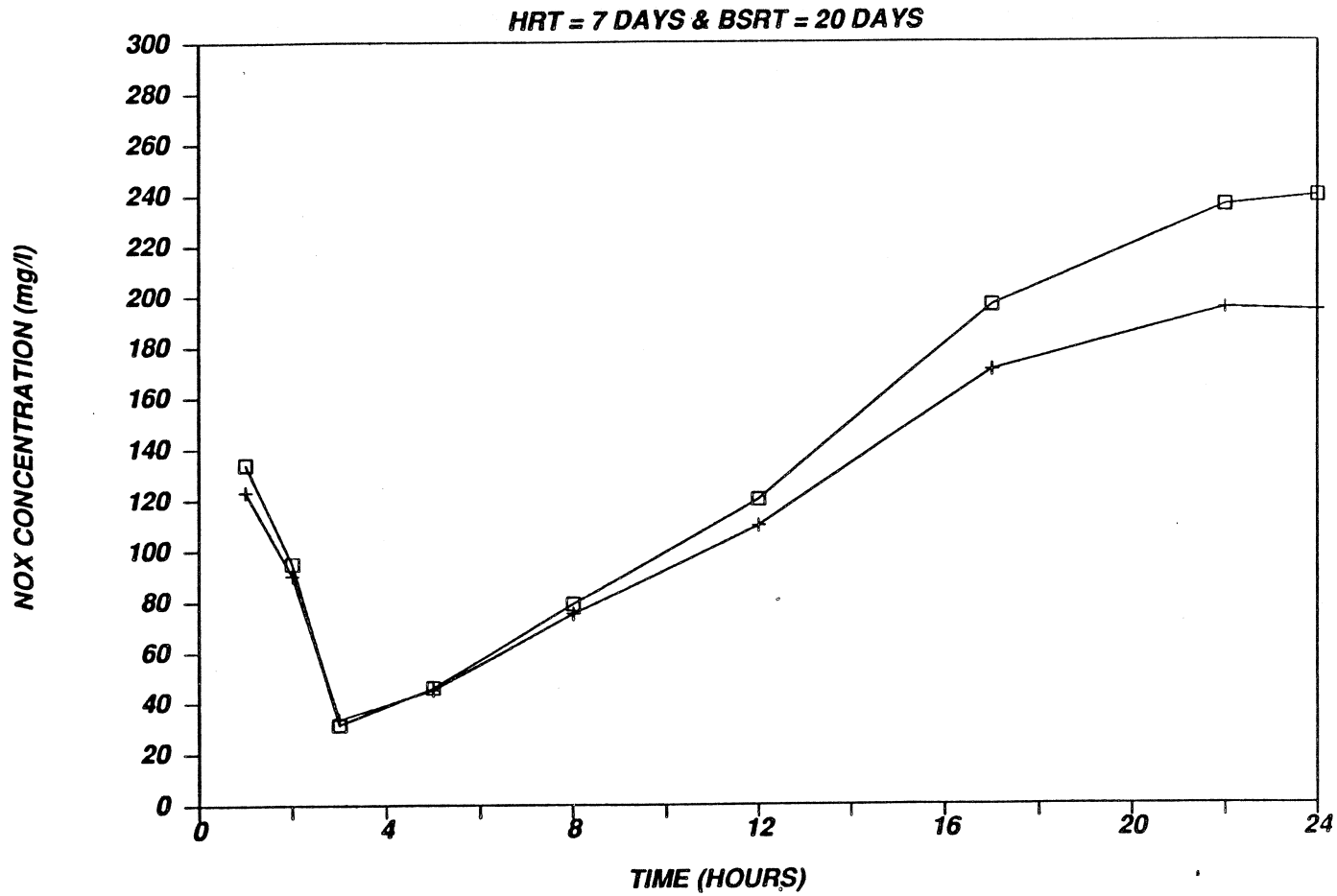


Figure 6.19 Measured and Predicted NOx Concentration Profiles for Reactor Operating at 7/20 (HRT/BSRT)

3 and 12 hours. However, they were to some extent less accurate in the other periods.

Finally, the effect of BSRT change on the estimation of: (1) K_1 and K_2 and (2) the prediction of the substrate concentration was investigated. However, by comparing figures 6.19 and 6.20, which were generated with the same values of K_1 and K_2, it can be concluded that the BSRT change did not affect both abovementioned coefficients.

In summary, it can be concluded that the mathematical model derived in this chapter is efficient and reliable based on the following facts:

1. The regression analysis coefficient r^2 were on the average 0.95, which is considered to be a reasonable fit.

2. The predictions were consistent with different operation conditions, and succeeded in providing good predictions of the substrate concentration over time.

3. The same coefficients K_1 and K_2, characterizing the theoretical specific biological rates of substrate consumption during time in the mass balance dynamic model, could be used for different HRT settings from 3 to 9 days and BSRT values of 10 to 30 days. This is the important criterion for judging the validity of the

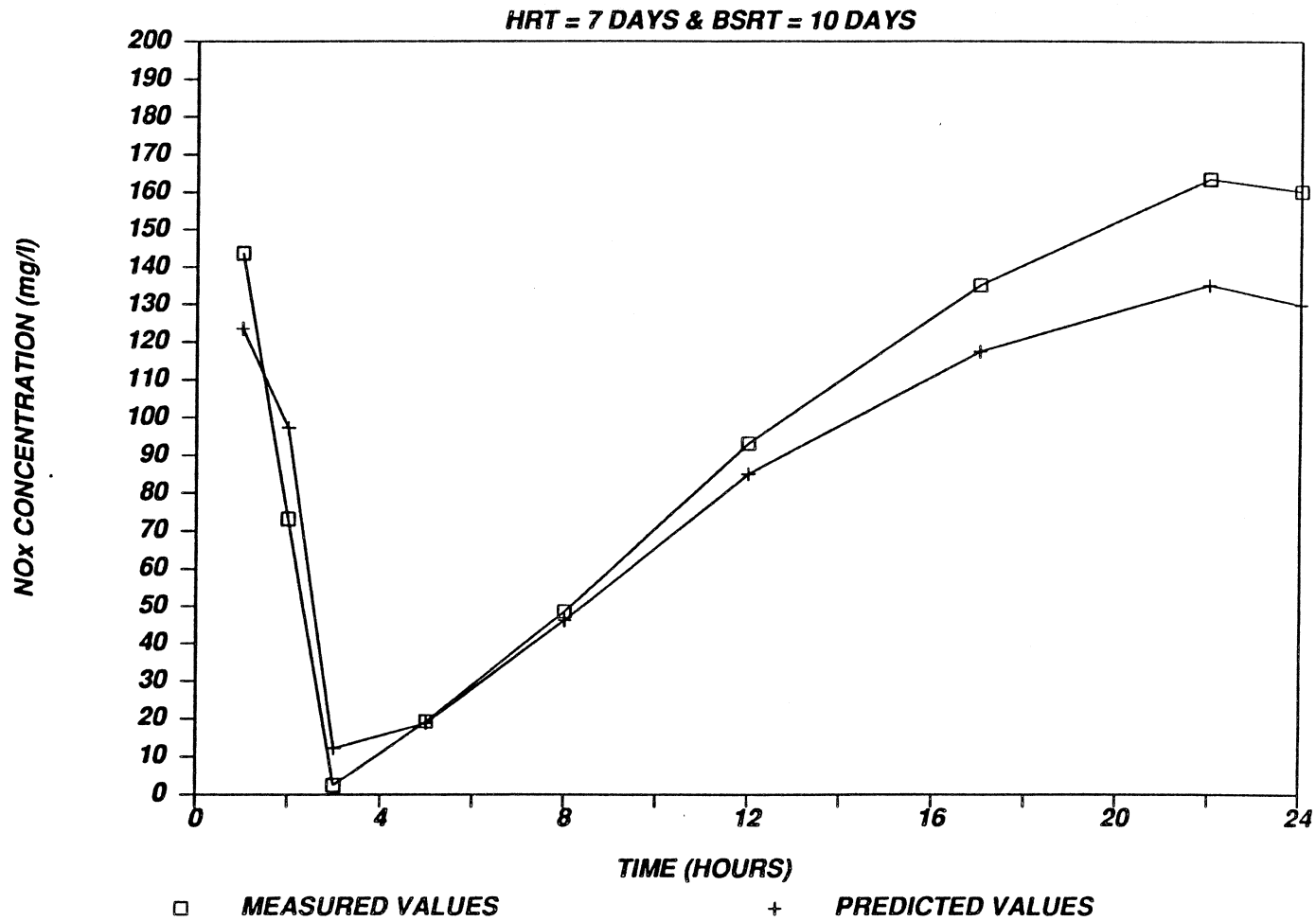


Figure 6.20 Measured and Predicted NOx Concentration Profiles for Reactor Operating at 7/10 (HRT/BSRT)

theoretical models (Equations 6.15 and 6.23).

6.4.2 Using the Model in the Design of the SBR System

The point of interest in making predictions of substrate concentration under several operation conditions in the SBR system obviously centers around the rates of reactions, K_1 and K_2, in the fill and react sequences for different substrate concentrations and HRT values. Once these two values are determined, it is possible to determine the reactor volume required for treating certain wastes to a desired level of digestion. Knowing the initial substrate concentration, and both K_1 and K_2, the mathematical models described in Equations 6.15 and 6.23 can be used to map the variation in the concentration during the times allocated for both fill and react periods. The other three periods, settle, draw and idle, have been ignored here because of their minimal contribution to the stabilization of pollutants in the SBR. As mentioned previously these phases represent only 8.3% of the total time of the SBR cycle.

To better illustrate the ability of the model to predict substrate variations, a numerical example is given below. Consider a waste which has the following characteristics (after microscreening):

COD = 32,000 mg/l

NH_3 = 1,400 mg/l

NOx = 10 mg/l

SS = 1,600 mg/l

The time allocated for the fill period = 3 hours

The time allocated for the react period = 19 hours

The time for settle, draw and idle = 2 hours

The HRT = 7 days and BSRT = 20 days

The reaction rates during fill and react periods are given in Table 6.5, assuming that they will be the same in a full-sized reactor. The predicted values of COD, NH_3 and NO_x are 860, 2.0 and 260 mg/l, respectively as shown in Figures 6.21 to 6.23.

TABLE 6.5 K_1 and K_2 USED IN THE NUMERICAL EXAMPLE

Parameter	K_1	K_2
COD	0.4	0.02
NH_3	0.2	0.1
NO_x	0.2	0.01

In this example a discharge of 10 m^3/day is considered, based on the conditions that a typical medium size farm with 1000 growing finishing hogs will produce on average 10 liters of liquid manure per animal per day.

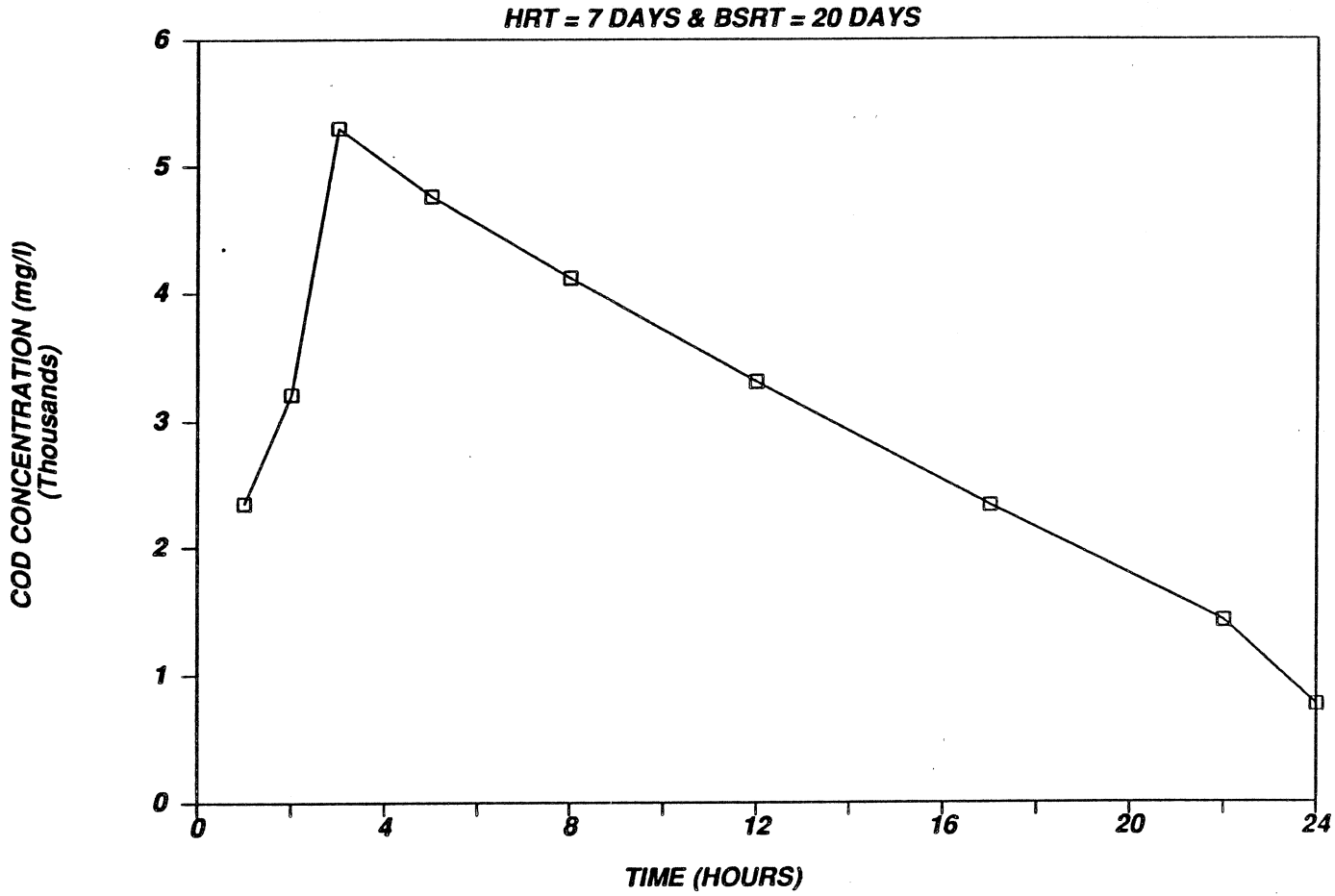


Figure 6.21 Predicted COD Values

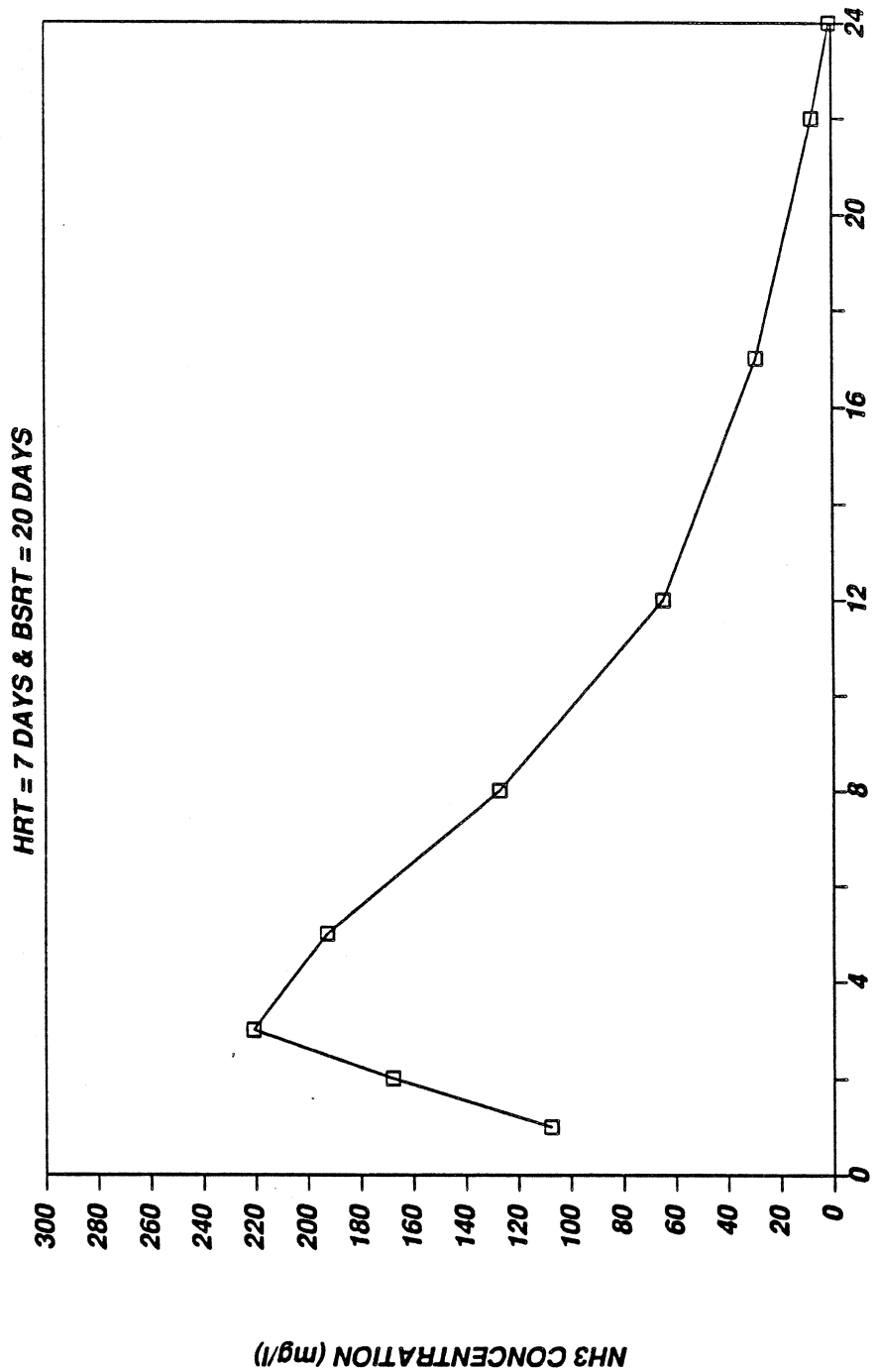


Figure 6.22 Predicted NH_3-N Values

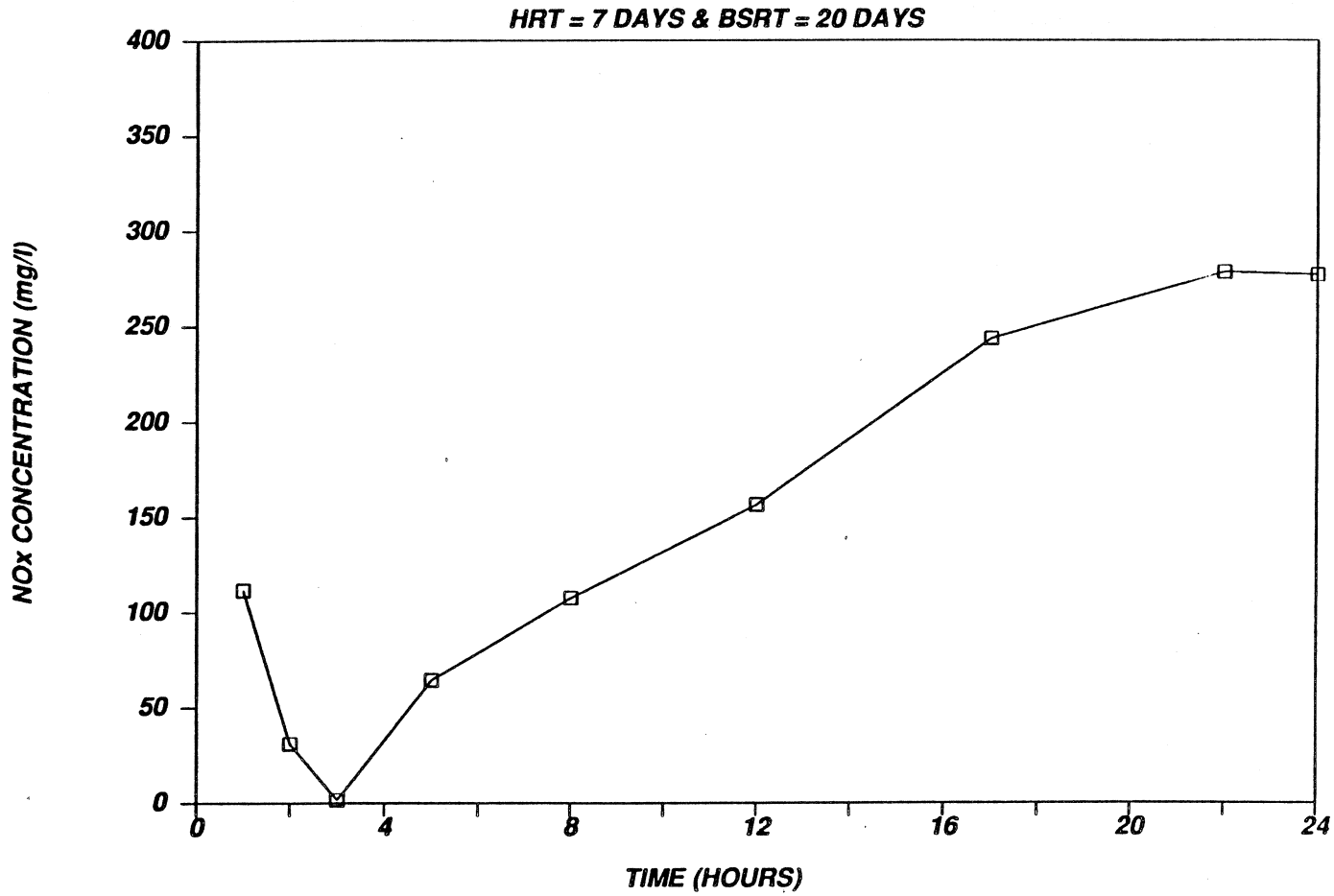


Figure 6.23 Predicted NOx Values

Based on these predicted results presented in Figures 6.21 to 6.23, the final effluent quality and the volume required for the sequential batch reactor in the full scale operation can be determined. The removal percentage of organic and nitrogenous compounds can be calculated from the prediction of the effluent quality determined from the mathematical model presented in equations 6.15 and 6.23. However, these removal percentage results might be unsatisfactory, and do not agree with the regulatory agencies' effluent criteria, which is site specific for land application. In this case the SBR designer can pre-set the effluent quality to be achieved (e.g. 99% removal) and select the number of SBR treatment in series to reach this goal.

Basically, the size of the treatment facility required depends on:

1. The rate at which organic and nitrogenous compounds reactions and conversions occur.
2. The influent discharge.
3. The waste retention time of 7 to 9 days.

Having mentioned the above conditions, the volume will be simply the product of the influent discharge (Q) times the hydraulic retention time (HRT). An addition of 20% volume increase will also be required as a factor of safety to accommodate the possible overflow.

6.5 Summary

A mathematical model based on the dynamic mass balance approach was developed in this chapter. It is a simple reliable model capable of predicting the substrate concentration during both the fill and react periods. The model can be used in the case of a full-scale SBR design to determine the effluent quality. The following are the steps which can be taken to use the mathematical model expressed in Equations 6.15 and 6.23 in the design and operation of the SBR system:

1- Determine the influent concentration for the following parameters: COD, NH_3 and MLVSS.

2- Run a bench scale SBR system to determine the rates of reaction K_1 and K_2.

3- Determine the optimum operation conditions, i.e. HRT, BSRT and anoxic conditions, to give the best effluent results.

4- Substitute the initial concentration values and the rates of reaction into the mathematical model to determine the projected values of the substrate concentration over time during the SBR cycle.

5- Determine the final substrate concentration of the different parameters.

6- Compare these values with desired quantities, or standard values for discharging effluent into surface water or for disposal on land. Based on this comparison, one can select the optimum HRT and BSRT which will give the desired effluent quality.

7- If the results are unsatisfactory, then a higher degree of treatment is required, thus the suggested alternative can be two SBR treatments in series. In this specific case, it is important to investigate the influent nutrient balance in the second treatment step.

CHAPTER 7 CONCLUSIONS, CONTRIBUTIONS TO KNOWLEDGE AND SUGGESTIONS FOR FUTURE STUDIES

7.1 Conclusions and Contributions to Knowledge

In recent years, there has been a considerable interest in the treatment of livestock waste, to abate agricultural pollution. In this context, research was conducted on the treatment of liquid swine manure in an integrated system, incorporating microscreening followed by a sequential batch reactor (SBR). The two unit operations proved to be very efficient in the removal of potential pollutants from swine waste, and yield a high quality effluent with stable end products.

The present study will contribute to extend the knowledge in the field of swine waste management. The experimental results obtained during this research provide quantitative information regarding various parameters and operating conditions pertinent to the use of SBR and the microscreen which was developed as a part of this project. Furthermore, the conclusions derived from these experimental findings may have significant implications for other types of agricultural wastes. It should be noted that the application of the SBR was limited up to now to the treatment of low strength wastes. This study can serve to provide

a better perception of the SBR behaviour when treating strong wastes like manure under aerobic and anoxic conditions.

This treatment system can be easily implemented on a farm. It is cost effective, due to the nature of the SBR process itself, and substancial savings can be made in the operating costs, by including aerated and nonaerated periods in the SBR cycle. Also the microscreening unit can be produced at a relatively low cost. Livestock producers can operate this flexible treatment system in a fashion to produce an effluent which is compatible with the various farm conditions and also comply with the existing regulations for disposal of animal waste on the land.

The following conclusions and contributions to knowledge are presented on the basis of experimental findings:

1. The continuous belt microscreening device developed in this research work has a high hydraulic loading capacity, and it can accept influent slurries containing a wide range of dry matter. The efficiency of this unit ranged from 47 to 60%, in terms of solid/liquid separation, which is relatively high compared to the efficiencies of other commercial units in the same category. The screened solids fraction can be used as

a part of the animal diet or as a compost. The screened liquid was free from substantial amounts of organic matter, nitrogen and phosphorus.

2. The SBR, functioning under appropriate operating conditions, can be a suitable new alternative for treatment of strong liquid swine waste. Such a facility would greatly reduce the potential for polluting the environment when discharging the effluents produced. The best overall performance for SBR was achieved for reactors operating at room temperature, with a hydraulic retention time of 7 to 9 days and a biological solids retention time of 20 days. In these reactors on average the removal of SS, COD and NH_3-N was 97, 97 and 99^+%, respectively. Also the sludge settling process was very efficient in these reactors and yielded a clear supernatant free from offensive odours. The efficiency of the SBR was adversely affected by 1) short HRT, i.e. less than 5 days; 2) temperatures below 10^oC; and 3) extended anoxic operation, greater than 8 hrs.

3. The nitrification process was fully achieved in the abovementioned SBR systems. In the react sequence under aerobic conditons, nitrification came to completion in about 15 hrs, and the oxidized nitrogen was mainly in the nitrates form. The nitrifiers did

not appear to be inhibited by the absence of dissolved oxygen for several hours during the cycle, nor by the substrate carbon and nitrogen loading applied to these reactors.

4. Denitrification occurred promptly during anoxic fill. The heterotrophic bacteria actively reduced oxidized nitrogen in an environment that was rich in exogenous carbon. However, the denitrification process was carried out at a much slower space during react due to limited amounts of biodegradable organic carbon. As a result of nitrification/denitrification it was possible to remove between 86 to 93% of the influent inorganic nitrogen applied to the SBR system.

5. Other advantages associated with the SBR anoxic operation are: 1) cost savings due to reduce air supply, and 2) an anoxic fill can effectively control the problem of excessive formation of foam.

6. A mathematical model based on the dynamic mass balance approach was developed in this study. It is a reliable model capable of predicting the COD, NH_3-N and NO_2-NO_3-N concentrations during both the fill and react periods of the SBR. The model hopefully can be used in the case of a full-scale SBR treatment system design to predict the effluent quality. The input data

required for the predictions are described, and the predictions of the model for the results of tests conducted in the laboratory show good agreement between the values predicted and the values measured. The same basic specific reaction rate coefficients could be used succesfully in the model over a wide range of waste dilutions, hydraulic retention times and biological solids retention times.

7.2 Suggestions for Future Studies

The following topics are suggested for future research:

1. Evaluate the effect of luxury phosphorus uptake by the microorganisms on the performance of the SBR treatment system.

2. Gain a better understanding on the interaction of the different type of microorganisms present in the SBR, under various process configurations. This knowledge would assist in the manipulation of these species to enhance the treatment efficiency.

3. The mathematical model predictions should be tested for other types of waste being treated in a SBR by expanding on the experimental part of the study.

4. Before generalized design procedures are formulated for the application of the sequential batch reactor to livestock wastes, it is advisible to investigate a full scale SBR system under different initial and boundary conditions, in order to confirm the design parameters and operating conditions at full scale.

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APPENDIX A

TABLE A.1 Evaluation of the Microscreening Unit as a Function of Screen Mesh Size and Flow Rate

Screen Mesh Size [η m]	Flow Rate [l/min]	Influent Solids (%DM)		Solids Removal Efficiency (%)		Screened Solids (%DM)	
		Trial 1	Trial 2	Trial 1	Trial 2	Trial 1	Trial 2
	10	7.64	8.2	49	47.5	22.5	21.7
	15	7.64	8.2	40.2	40.8	22	20.5
200	25	8.45	8.71	49.5	47.2	20.9	20.1
	35	8.45	8.71	46.1	48	18.7	21.3
	42	7.8	8.13	44.7	46.3	21.4	20.6
	10	7.92	8.35	57.1	56	19.5	19.2
	15	7.92	8.35	60.6	59.5	18.1	18.5
100	25	8.14	8.27	62	58.4	18	18.3
	35	8.14	8.27	59.8	61.2	17.8	19.4
	42						
	10	7.59	7.9	62.4	60.9	13.5	13.4
50	15	7.59	7.9	61.8	63	12.7	12.1
	25						

TABLE A.2 Particle Size Distribution Analyses of Influent Manure with 8% DM and the Screened Liquid

Particle Size [η m]	Influent Manure (% mass)	% Solids in liquid fraction	
		mesh 200 ηm	screen 50 ηm
<53	51.85	50.7	38.1
53-75	2.6	2.3	
75-105	2.49	2	
105-150	1.31	1.1	
150-250	1.58	0.7	
250-500	3.4		
500-1180	9.25		
>1180	27.53		
TOTAL	100	56.8	38.1
% Solid Removal		43.2	61.9

TABLE A.3 EFFECT OF INFLUENT SLURRY ON SOLIDS REMOVAL AND SCREENED SOLIDS
General Linear Models Procedure

Replicate	Influent Slurry [%d.m.]	Solids Removal Efficiency	Screened Solids [%d.m.]
1	2.0	41.0	14.0
2	2.4	43.7	14.2
3	2.7	44.5	14.8
1	3.0	47.6	14.6
2	3.5	48.0	15.2
3	3.2	47.2	15.0
1	4.8	52.5	17.0
2	4.4	50.5	16.8
3	4.6	53.9	16.8
1	6.3	55.5	17.6
2	6.7	56.7	17.9
3	6.5	56.9	17.4
1	7.9	58.2	18.2
2	8.0	59.5	18.0
3	8.2	60.5	18.5

Number of Observations in Data Set = 15

DEPENDENT VARIABLE: Solids Removal Efficiency

SOURCE	DF	SS	MS	F-Calc	Prob
Corrected Total	14	534.9773			
Model	3	511.8995	170.6331	81.33	0.0001
Replicate	2	6.7573		1.61	0.2435
Influent Slurry	1	505.1422		240.78	0.0001
Error	11	23.0777	2.0979		

R-SQUARE 0.95
C.V. 2.79

Regression Analysis

Solids Removal Efficiency = a* influent slurry ^b + c

Solution:
a = 32.9094
b = .2669
c = 2.2569
Maximum error is 2.1823

DEPENDENT VARIABLE: Screened Solids

SOURCE	DF	SS	MS	F-Calc	Prob
Corrected Total	14	534.9773			
Model	3	32.7888	10.9296	46.40	0.0001
Replicate	2	0.1240	0.26		0.7733
Influent Slurry	1	32.6648	138.67		0.0001
Error	11	2.5911	0.2355		

R-SQUARE 0.92
C.V. 2.94

Regression Analysis

Screened Solids = a* influent slurry ^b +c

Solution:
a = 8.5708
b = .2625
c = 3.6001
Maximum error is .5539

TABLE A.4 EFFECT OF HYDRAULIC LOADING ON THE PROCESS EFFICIENCY
General Linear Models Procedure

Hydraulic Loading	Solid Removal Efficiency (%)					
	Influent Slurry (%DM)					
	3		6		8	
	Trial 1	Trial 2	Trial 1	Trial 2	Trial 1	Trial 2
0.017	44.6	46.2	53.8	53.5	57.2	58.8
0.022	46.9	45	53.4	57	57.9	57.4
0.033	48.5	46.1	50.7	56.7	60	58.5
0.038	46	46.8	54.3	57.1	55.9	60.7
0.048	44	50	59.2	52.6	57.5	61.5
0.057	47.1	47.3	57.1	53.9	55.6	62
0.072	46.5	50.7	55	54.8	60.3	58.3
0.084	49.4	46	53.2	55.3	59	60.2
0.145	44.9	50.2	52.7	54.5	61.4	57.1

Number of Observations in Data Set = 54

DEPENDENT VARIABLE: Solids Removal Efficiency

SOURCE	DF	SS	MS	F-Calc	Prob
Corrected Total	53	1502.8387			
Model	4	1315.2100	328.8025	85.87	0.0001
Trial	1	12.6150		3.29	0.0756
Influent Slurry	2	1299.4781		169.68	0.0001
Hydraulic Load	1	3.1168		0.81	0.3714
Error	49	187.6286			

R-SQUARE 0.87
C.V. 3.65

APPENDIX B

TABLE B1. CHRONOLOGICAL INFLUENT/EFFLUENT DATA FOR PHASE I

SUSPENDED SOLIDS

TIME (days)	INFLUENT SS (mg/l)	EFFLUENT SS (mg/l) HRT/BSRT (days) 3/10	5/10	7/10	9/10	INFLUENT: SS (mg/l)	EFFLUENT SS (mg/l) HRT/BSRT (days) 3/20	5/20	7/20	9/20	INFLUENT (mg/l)	EFFLUENT SS (mg/l) HRT/BSRT 3/30	5/30	7/30	9/30
1	8475	1240	750	340	244	10160	3190	714	305	295	9400	3055	670	470	340
4	9025	1935	604	452	260	10400	2608	625	280	240	10250	3781	805	529	275
8	8700	1500	469	235	250	9735	1625	390	198	174	9715	2075	460	608	219
12	7430	1710	385	265	210	10030	3012	730	326	305	10380	4119	595	650	366
16	8050	3725	565	244	205	9850	1925	595	175	195	9845	2908	830	415	380
20	10155	2185	540	285	285	9240	2750	537	250	269	9675	2888	790	480	299
24	9650	3400	495	240	290	11075	3365	885	376	335	8395	2320	618	352	285
28	7145	2915	570	325	240	9350	1820	593	272	250	10530	4300	850	495	390
32	8520	3240	760	270	245	10620	3185	619	295	285	9550	3267	645	410	245
35	9160	3575	485	310	270	9965	2495	500	240	260	7950	2790	783	507	280
AVERAGE	8631	2605	562	296	249	10042	2597	618	271	260	9569	3150	704	491	307
ST.DEV.		925	118	65	28		623	136	59	49		729	126	90	58
%REMOVAL		69.8	93.4	96.5	97.1		74.1	93.8	97.2	97.4		67.0	92.6	94.8	96.7

VOLATILE SUSPENDED SOLIDS

TIME (days)	INFLUENT (mg/l)	EFFLUENT VSS (mg/l) HRT/BSRT (days) 3/10	5/10	7/10	9/10	INFLUENT: (mg/l)	EFFLUENT VSS (mg/l) HRT/BSRT (days) 3/20	5/20	7/20	9/20	INFLUENT (mg/l)	EFFLUENT VSS (mg/l) HRT/BSRT 3/30	5/30	7/30	9/30
12	6293	1457	331	214	160	8521	2530	582	251	230	8657	3400	503	501	267
16	6738	3138	452	190	158	8085	1611	490	138	146	8165	2390	685	316	294
20	8342	2322	439	219	228	7619	2170	436	200	208	7890	2271	634	370	225
24	7845	2804	407	186	219	8997	2629	720	261	250	6839	1931	483	265	224
28	6073	2369	453	255	190	7720	1506	469	215	193	8775	3615	650	382	300
32	7082	2690	621	208	185	8825	2580	510	232	217	7773	2655	531	325	190
35	7330	2816	400	243	206	8243	2105	412	190	205	6562	2286	626	361	215
AVERAGE	7110	2514	443	216	192	8287	2217	517	212	207	7809	2650	587	360	245
ST.DEV.	755	503	82	24	25	487	425	94	39	30	782	579	73	68	39
%REMOVAL		65	93	96	97		73	93	98	98		66	92	95	97
VSS/SS	0.83	0.82	0.82	0.78	0.77	0.82	0.81	0.81	0.76	0.76	0.82	0.82	0.80	0.76	0.76

TABLE B1. CHRONOLOGICAL INFLUENT/EFFLUENT DATA FOR PHASE I (CONT.)

CHEMICAL OXYGEN DEMAND

TIME (days)	INFLUENT COD (mg/l)	EFFLUENT COD (mg/l) HRT/BSRT (days) 3/10	5/10	7/10	9/10	INFLUENT COD (mg/l)	EFFLUENT VSST COD1) HRT/BSRT (days) 3/20	5/20	7/20	9/20	INFLUENT (mg/l)	EFFLUENT VSST COD1) HRT/BSRT BSRT (days) 3/30	5/30	7/30	9/30
1	27554	4650	1983	1128	862	30240	5200	2200	1206	970	29765	4735	1570	1225	1042
4	25875	3853	1552	851	776	28320	3650	1638	805	825	30250	5960	2318	1240	1200
8	26979	4019	2070	1018	920	30480	5800	1997	1200	800	27500	3010	1425	1045	960
12	24840	3220	1098	828	817	25900	5250	1480	948	850	32000	9305	2110	1500	1350
16	26375	3864	1340	914	851	28325	6580	1562	1135	1015	28105	4540	1250	820	785
20	27830	6100	1817	1254	1040	31622	7425	1800	1237	1075	29610	5165	1649	900	865
24	27036	4876	1157	1086	750	26300	5075	1630	900	928	28475	4816	1650	935	950
28	27122	4681	1576	1093	856	28100	5830	1525	985	850	31850	8145	2285	1385	1226
32	28118	7992	1309	989	948	28445	5705	1650	938	945	29250	7450	2014	1193	1205
35	27542	7618	1150	1052	905	28820	6250	1385	850	830	28428	5920	1930	1105	1080
AVERAGE	26927	5087	1505	1021	872	28655	5676	1686	1020	908	29523	5904	1820	1134	1066
ST.DEV.		1630	354	131	85		1006	248	159	92		1894	365	216	178
%REMOVAL		81.1	94.4	96.2	96.7		80.1	94.1	96.4	96.8		80.0	93.8	96.1	96.3

AMMONIA NITROGEN

TIME (days)	INFLUENT NH N (mg/l)	EFFLUENT COD (mg/l) HRT/BSRT (days) 3/10	5/10	7/10	9/10	INFLUENT NH N (mg/l)	EFFLUENT NH N (mg/l) HRT/BSRT (days) 3/20	5/20	7/20	9/20	INFLUENT (mg/l)	EFFLUENT NH N (mg/l) HRT/BSRT BSRT (days) 3/30	5/30	7/30	9/30
1	1260	890	35	7.8	2.4	1156	335	54	2.5	2.3	1062	392	64	2.7	2.8
4	1105	735	74	15.2	2.8	1138	418	60	2.1	1.8	1240	425	75	3.5	3.7
8	1216	815	59	11	2.1	1066	320	31	2.4	2.5	1105	315	51	2.5	2.1
12	987	630	65	3.5	2.7	1289	390	65	2.8	2.8	1320	590	82	3.8	3.7
16	1150	726	38	6.2	2.6	1260	365	70	2.1	1.9	995	280	47	2	2.2
20	1261	910	46	9.1	2.3	1335	482	76	4.1	3.2	1245	415	55	3.5	3.4
24	1185	654	37	4.7	2.8	1200	340	45	2.2	2.4	1260	370	43	2.9	2.5
28	1156	710	52	5.5	2.4	1184	280	29	2.5	2.6	1380	528	80	3.4	3.4
32	1254	862	55	6	2.8	1170	315	38	2	1.8	1215	350	59	2.8	2.8
35	1210	875	48	6.4	2.7	1195	227	52	2.3	2.1	1240	412	73	3	2.6
AVERAGE	1178	780	50	7	2	1199	347	52	2	2	1206	407	62	3	2
ST.DEV.		102	12	3			71	16	0	0		93	14	0	0
%REMOVAL		33.7	95.7	99.4	99.8		71.0	95.7	99.8	99.8		66.2	94.8	99.8	99.8

TABLE B1. CHRONOLOGICAL INFLUENT/EFFLUENT DATA FOR PHASE I (CONT.)

NITRITE-NITRATE-NITROGEN (mg/l)

TIME	INFLUENT NOX	EFFLUENT NOX (mg/l) HRT/BSRT (days)				INFLUENT NOX	EFFLUENT NOX (mg/l) HRT/BSRT (days)				INFLUENT NOX	EFFLUENT NOX (mg/l) HRT/BSRT BSRT (days)			
(days)	(mg/l)	3/10	5/10	7/10	9/10	(mg/l)	3/20	5/20	7/20	9/20	(mg/l)	3/30	5/30	7/30	9/30
1	6	6.5	280	165	123	5	375	276	226	186	4.6	310	394	255	180
4	9	18.5	310	195	98	8.8	215	300	250	145	5	358	440	180	112
8	7.3	9.3	235	174	168	6.3	361	318	258	157	9.8	177	360	243	137
12	6.1	6	170	145	95	6	280	289	219	125	6.2	405	285	245	134
16	6.4	15.4	218	150	125	9.4	370	265	195	190	4.5	220	350	230	145
20	7	8.5	280	165	116	6.2	214	243	173	165	6	380	395	165	142
24	6.1	10.8	195	156	127	4.5	337	275	204	112	5	300	450	250	109
28	6.5	5.2	227	175	105	7.1	394	350	285	173	9.5	375	260	312	120
32	4.9	25	205	180	120	6.8	260	292	237	162	5.7	290	405	260	135
35	5.5	16.2	238	172	125	10.5	305	286	212	159	6.1	210	375	225	140
AVERAGE	6	12	235	167	120	7.06	311	289	225	157	6.24	302	371	236	135
ST.DEV.		6	43	14	20		66	29	32	24		79	61	41	20
%FORMATION		87	3538	2486	1754		4306	3999	3099	2129		4747	5851	3690	2069

TABLE B2. MIXED LIQUOR DATA FOR PHASE I

SLUDGE VOLUME IN 1 LITER GRADUATE CYLINDER

TIME	SLUDGE VOL. (ml) HRT/BSRT (days)				SLUDGE VOL. (ml) HRT/BSRT (days)				SLUDGE VOL. (ml) HRT/BSRT (days)			
(days)	3/10	5/10	7/10	9/10	3/20	5/20	7/20	9/20	3/30	5/30	7/30	9/30
8	882	570	510	470	915	650	530	450	950	725	635	490
12	934	625	485	520	970	680	604	427	942	860	570	570
16	960	585	450	405	934	597	450	386	975	750	625	520
20	875	600	470	370	910	544	475	470	912	690	540	525
24	869	580	525	490	870	738	420	418	928	745	480	425
28	813	560	490	425	825	615	518	526	937	705	495	590
32	795	550	464	415	906	600	490	475	940	765	640	614
35	840	575	400	410	950	630	465	460	948	815	580	485
AVERAGE	871	580	474	438	910	631	494	451	941	756	570	527
ST.DEV.	56	23	38	50	45	58	56	42	18	56	61	62

TABLE B2. MIXED LIQUOR DATA FOR PHASE I (cont.)

MIXED LIQUOR SUSPENDED SOLIDS (mg/l)

TIME	HRT/BSRT (days)											
(days)	3/10	5/10	7/10	9/10	3/10	5/20	7/20	9/20	3/30	5/30	7/30	9/30
8	19150	16494	15240	12970	22380	19225	17950	12825	23730	21610	19425	19030
12	18955	18020	12190	10080	19510	18110	15167	16190	21950	19585	21080	16515
16	17300	13150	16750	15675	21715	20115	19635	18300	19180	17520	16315	14170
20	22480	14925	11260	11480	23380	17390	15810	15715	24825	18130	18160	17295
24	18120	18480	16125	14170	17950	15575	16240	13280	22170	17510	15920	15605
28	21620	17426	13490	12200	22585	21385	20160	16535	20590	19645	18465	13910
32	19590	14395	16250	15635	20630	19200	17415	13810	24135	21815	20120	15325
35	20745	15510	14415	10590	17640	16735	14975	15705	22740	19065	16435	16790
AVG.	19745	16050	14465	12850	20705	18467	17169	15295	22415	19360	18240	16080
ST.DEV.	1644	1753	1876	2018	2057	1767	1845	1735	1756	1566	1782	1582

MIXED LIQUOR VOLATILE SUSPENDED SOLIDS (mg/l)

TIME	HRT/BSRT (days)											
(days)	3/10	5/10	7/10	9/10	3/10	5/20	7/20	9/20	3/30	5/30	7/30	9/30
8	16270	13970	12850	11360	18790	15180	12790	9105	20775	18540	15250	13790
12	15545	14795	10015	7958	17150	15665	11530	12210	19185	17565	18910	11720
16	15660	11180	13445	13590	18370	14895	13646	14457	15580	15540	15255	11985
20	17540	11650	8787	10230	19690	14350	14070	10970	23205	14695	15470	15510
24	14855	14580	12740	10645	15480	11095	14235	9375	21160	13480	14216	10925
28	17940	14725	10250	8780	20265	19030	17850	12000	17590	14215	13110	8750
32	15635	11580	14135	12640	18360	17472	15610	10025	22025	20070	15150	13025
35	16390	12740	12685	8315	14270	13250	12950	11620	19435	16295	14285	12360
AVG.	16229	13152	11863	10439	17784	15117	14085	11220	19869	16300	15205	12258
ST.DEV.	981	1443	1788	1908	1936	2277	1810	1643	2305	2130	1582	1870

TABLE B2. MIXED LIQUOR DATA FOR PHASE I (CONT.)

FRACTION OF THE REACTOR VOLUME OCCUPIED BY THE SLUDGE BLANKET

TIME	SLUDGE VOL. (ml) HRT/BSRT (days)				SLUDGE VOL. (ml) HRT/BSRT (days)				SLUDGE VOL. (ml) HRT/BSRT (days)			
(days)	3/10	5/10	7/10	9/10	3/20	5/20	7/20	9/20	3/30	5/30	7/30	9/30
16	.905	.535	.420	.370	.910	.570	.425	.365	.935	.725	.550	.475
20	.890	.515	.445	.350	.890	.540	.430	.410	.915	.675	.525	.490
24	.810	.490	.440	.420	.850	.620	.415	.385	.920	.690	.480	.400
28	.790	.530	.405	.395	.835	.610	.470	.445	.895	.695	.480	.510
32	.780	.480	.400	.395	.880	.550	.455	.395	.890	.735	.600	.585
35	.865	.565	.385	.380	.890	.575	.440	.390	.925	.740	.545	.455
AVERAGE	.840	.519	.416	.385	.875	.577	.439	.398	.913	.710	.530	.486
ST.DEV.	.37	.23	.18	.17	.38	.25	.19	.17	.40	.31	.23	.21

pH OF THE MIXED LIQUOR

TIME	HRT/BSRT (days)				HRT/BSRT (days)				HRT/BSRT (days)			
(days)	3/10	5/10	7/10	9/10	3/20	5/20	7/20	9/20	3/30	5/30	7/30	9/30
8	8.58	5.42	6.90	7.29	7.69	5.26	6.41	7.24	7.84	6.15	7.20	7.19
12	8.59	5.18	7.15	7.34	7.30	4.90	7.25	6.22	7.25	6.73	7.03	6.92
16	8.71	6.25	7.42	6.58	8.15	5.12	6.93	7.01	6.92	7.19	6.65	6.84
20	8.50	5.49	6.80	6.73	7.82	4.75	7.05	6.80	7.10	6.42	6.82	7.15
24	8.62	5.17	6.59	7.65	7.49	5.34	7.15	6.72	7.34	7.37	7.05	7.71
28	8.69	5.62	7.23	7.16	6.95	6.02	6.40	7.13	7.50	6.23	7.39	6.50
32	8.55	4.73	6.34	6.95	7.53	5.71	6.83	6.54	6.81	5.91	7.06	6.96
35	8.59	5.11	6.22	7.25	8.10	5.40	7.01	6.90	7.76	6.48	6.40	7.13
AVERAGE	8.60	5.37	6.83	7.12	7.63	5.31	6.88	6.82	7.31	6.56	6.95	7.05

DISSOLVED OXYGEN (mg O /l)

TIME	HRT/BSRT (days)				HRT/BSRT (days)				HRT/BSRT (days)			
(days)	3/10	5/10	7/10	9/10	3/20	5/20	7/20	9/20	3/30	5/30	7/30	9/30
8	2.3	3.7	3.9	6.2	1.9	3.1	5.8	6.9	2.7	3.5	5.4	7.4
12	2.2	4.5	4.3	6.0	2.3	4.7	4.2	6.4	2.5	4.2	5.1	6.9
16	1.7	4.0	4.1	6.9	4.0	3.5	5.1	7.0	3.0	3.7	6.3	7.0
20	2.5	3.9	5.0	5.7	2.2	3.8	6.0	6.8	4.5	3.5	6.0	7.1
24	3.4	3.8	3.5	6.4	2.5	3.4	4.3	5.5	3.8	4.5	5.5	6.5
28	2.8	5.0	5.0	7.0	2.8	4.1	4.9	7.3	3.0	4.8	4.7	7.5
32	1.9	3.6	4.4	6.2	3.2	5.2	5.5	6.1	3.6	5.0	5.0	7.8
35	3.1	2.9	5.2	6.6	3.5	4.9	5.3	6.8	2.5	4.4	5.9	6.8
AVERAGE	2.5	3.9	4.4	6.4	2.8	4.0	5.1	6.6	3.2	4.2	5.5	7.1

TABLE B2. MIXED LIQUOR DATA FOR PHASE I (CONT.)

TRACK FOR MIXED OXYGEN UPTAKE RATE (mg O /hr.l)

TIME	SLUDGE VOL. (ml) HRT/BSRT (days)				SLUDGE VOL. (ml) HRT/BSRT (days)				SLUDGE VOL. (ml) HRT/BSRT (days)			
(days)	3/10	5/10	7/10	9/10	3/20	5/20	7/20	9/20	3/30	5/30	7/30	9/30
3.15	16.9	15.6	35.9	24.0	24.5	38.1	41.9	36.5	17.5	19.6	28.0	21.5
5.0	145.7	128.0	137.4	107.5	139.3	114.6	112.0	89.2	122.8	112.5	103.6	74.9
8.0	173.5	160.5	159.2	142.6	240.0	135.0	136.5	114.0	165.0	140.0	121.0	135.0
11.0	219.1	179.0	121.4	138.5	183.5	142.9	97.0	105.5	174.5	137.2	105.9	96.1
15.0	237.0	155.2	83.6	104.7	149.2	136.5	62.4	89.0	159.2	115.0	61.2	88.5
19.0	251.5	130.7	80.2	95.2	122.5	94.2	51.8	71.4	113.5	81.4	53.7	68.2
22.0	228.4	104.9	75.0	81.5	114.8	87.5	51.2	65.9	108.1	63.5	49.5	47.5

TABLE B3. RATIO BETWEEN BOD AND COD

	INFLUENT WASTE			EFFLUENT FROM REACTOR 7/20 (HRT/BSRT)			EFFLUENT FROM REACTOR 9/20 (HRT/BSRT)		
	BOD mg/l	COD mg/l	BOD /COD	BOD mg/l	COD mg/l	BOD /COD	BOD mg/l	COD mg/l	BOD /COD
	20715	31622	.65	545	1237	.44	430	1075	.40
	17065	28445	.60	330	938	.35	340	945	.36
	17540	30240	.58	455	1206	.38	360	970	.37
	16575	25900	.64	350	948	.37	330	850	.39
AVG.	17974	29052	.617	420	1082	.385	365	960	.38

TABLE B4. CHRONOLOGICAL INFLUENT/EFFLUENT DATA FOR PHASE II

TOTAL KJELDAHL NITROGEN

TIME	INFLUENT	EFFLUENT TKN (mg/l)				INFLUENT	EFFLUENT TKN (mg/l)			
(days)	TKN (mg/l)	REACTOR 6/20 (HRT/BSRT):ANOXIC HRS				TKN (mg/l)	REACTOR 9/20 (HRT/BSRT):ANOXIC HRS			
		3	6	8	12		3	6	8	12
12	2375	215	275	340	420	2530	150	155	200	265
16	2510	300	310	360	435	2805	195	215	315	305
20	2560	290	315	415	525	2245	135	155	175	210
24	2155	185	245	250	345	2775	240	270	295	325
28	2615	355	365	450	570	2440	180	205	230	215
32	2405	265	295	280	550	2375	145	170	210	250
35	2250	280	330	390	375	2890	250	230	325	355
AVG.	2410	270	305	355	460	2580	185	200	250	275
ST.DEV.	154	52	36	66	82	227	43	40	56	51

TABLE B4. CHRONOLOGICAL INFLUENT/EFFLUENT DATA FOR PHASE II (cont.)

AMMONIA NITROGEN

TIME (days)	INFLUENT NH -N(mg/l)	EFFLUENT NH -N (mg/l) REACTOR 6/20 (HRT/BSRT):ANOXIC HRS				INFLUENT NH -N(mg/l)	EFFLUENT NH -N (mg/l) REACTOR 9/20 (HRT/BSRT):ANOXIC HRS			
		3	6	8	12		3	6	8	12
1	1275	7.1	11.5	16.5	162.0	1239	2.5	2.8	5.5	8.6
4	1370	6.2	8.4	17.1	180.7	1454	2.0	2.5	4.7	16.7
8	1240	3.8	9.8	11.6	231.5	1126	2.2	1.7	6.5	14.2
12	1430	7.9	12.0	19.0	197.0	1085	2.3	2.2	8.5	10.9
16	1190	4.3	7.8	15.5	285.0	1390	1.4	2.0	6.4	14.5
20	1165	5.2	4.6	12.3	169.3	1400	1.9	2.5	5.0	8.4
24	1210	6.0	10.5	17.0	247.5	1192	2.7	2.5	10.2	13.1
28	1015	3.7	8.0	10.0	269.8	1264	2.3	2.8	3.8	7.3
32	970	4.2	11.3	15.2	195.4	1325	2.6	2.3	6.1	11.5
35	1085	6.5	9.1	13.8	231.8	1175	2.1	2.2	4.3	9.8
AVG.	1195	5.5	9.3	14.8	217.0	1265	2.2	2.4	6.1	11.5
ST.DEV.	138	1.4	2.1	2.6	40.3	118	0.4	0.3	1.9	2.9
%REMOVAL		99.5	99.2	98.7	81.8		99.8	99.8	99.5	99.0

NITRITE-NITRATE-NITROGEN

TIME (days)	INFLUENT NH -N(mg/l)	EFFLUENT NH -N (mg/l) REACTOR 6/20 (HRT/BSRT):ANOXIC HRS				INFLUENT NH -N(mg/l)	EFFLUENT NH -N (mg/l) REACTOR 9/20 (HRT/BSRT):ANOXIC HRS			
		3	6	8	12		3	6	8	12
1	7.5	189.6	164.1	116.2	92.6	4.6	243.2	197.9	144.0	13.5
4	4.8	173.9	140.8	95.6	101.5	3.9	197.6	156.2	116.5	10.8
8	7.5	210.7	174.5	90.8	83.9	6.0	237.8	206.6	158.1	6.2
12	8.0	214.0	155.1	107.3	50.9	5.8	216.1	221.5	109.5	10.5
16	4.1	203.8	131.9	122.5	47.5	5.7	245.5	164.1	136.2	7.9
20	7.3	170.2	144.0	103.9	81.6	8.3	238.0	172.6	154.5	9.5
24	6.1	146.8	170.3	86.5	59.4	7.1	242.6	218.1	140.5	7.3
28	5.0	175.0	135.0	94.2	62.5	5.2	229.3	230.9	128.4	9.0
32	5.7	151.4	149.4	108.5	93.0	4.9	226.4	205.4	145.5	10.7
35	6.0	185.7	138.5	103.5	75.1	6.5	231.6	183.5	130.8	11.6
AVG.	6.2	182.1	150.4	102.9	74.8	5.8	230.8	195.7	136.4	9.7
ST.DEV.	1.3	22.0	14	10.7	17.9	1.0	14.0	24.2	14.7	2.2

TABLE B4. CHRONOLOGICAL INFLUENT/EFFLUENT DATA FOR PHASE II (cont.)

SUSPENDED SOLIDS

TIME (days)	INFLUENT SS (mg/l)	EFFLUENT SS (mg/l) REACTOR 6/20 (HRT/BSRT):ANOXIC HRS				INFLUENT SS (mg/l)	EFFLUENT SS (mg/l) REACTOR 9/20 (HRT/BSRT):ANOXIC HRS			
		3	6	8	12		3	6	8	12
1	10510	270	323	394	628	12530	327	285	415	322
4	11615	225	465	537	490	10235	219	280	332	418
8	13985	380	288	491	513	11740	292	301	229	309
12	14730	435	340	338	445	11515	312	245	340	365
16	12445	278	382	468	551	10875	275	365	252	380
20	13420	410	456	257	685	7880	260	250	391	496
24	9710	317	275	450	460	9365	241	378	194	258
28	9905	283	381	312	525	9850	273	405	230	316
32	10165	349	372	593	636	12295	346	268	275	284
35	12035	308	321	431	457	10620	284	293	302	347
AVG.	11852	326	360	427	539	10690	283	307	296	350
ST.DEV.	1696	63	60	99	80	1355	37	53	69	66
%REMOVAL		97	97	96	95		97	97	97	96

CHEMICAL OXYGEN DEMAND

TIME (days)	INFLUENT COD(mg/l)	EFFLUENT COD (mg/l) REACTOR 9/20 (HRT/BSRT):ANOXIC HRS				INFLUENT COD(mg/l)	EFFLUENT COD (mg/l) REACTOR 6/20 (HRT/BSRT):ANOXIC HRS			
		3	6	8	12		3	6	8	12
1	31100	820	1160	855	1490	29460	940	1105	1130	1715
4	30500	745	685	710	1200	30285	1020	880	810	1825
8	29840	860	750	585	670	33750	1175	1050	1045	1330
12	31530	1040	855	1105	1170	28495	965	1115	1190	1675
16	32100	920	670	8[illegible]	1315	31810	1155	970	1200	1650
20	33150	1100	735	1170	1850	34525	1430	1250	1240	1895
24	28225	780	1100	620	1215	28540	875	815	860	1450
28	31195	715	710	950	940	30010	1215	1410	1350	1575
32	31500	655	675	725	1360	29185	970	1135	1215	1225
35	32610	820	810	765	1295	30750	1305	1020	1160	1610
AVG.	31175	845	815	830	1250	30680	1105	1075	1120	1595
ST.DEV.	1341	133	177	184	296	1979	170	164	161	199
%REMOVAL		97	97	97	96		96	97	96	95

TABLE B6. INFLUENT/EFFLUENT AND MIXED LIQUOR DATA FOR PHASE III

REACTOR OPERATING AT 10 DEG. C

TIME (days)	COD (mg/l) Inf.	COD (mg/l) Eff.	SS (mg/l) Inf.	SS (mg/l) Eff.	NH -N (mg/l) Inf.	NH -N (mg/l) Eff.	NO (mg/l) Inf.	NO (mg/l) Eff.	MLSS (mg/l)	Sludge Vol. cylinder (ml)	pH	DO (mg/l)
1	29045	915	10340	354	940	25.6	4.2	103	12542	540	7.3	7.1
4	29310	760	9275	261	1165	60.9	4.6	68	11850	515	8.1	6.6
8	31070	895	11740	430	1155	43.5	5.1	92	14329	590	7.5	5.9
12	30850	1070	10800	377	1320	17.2	7.5	124	16150	420	6.5	6.8
16	30690	1146	10650	526	1280	21.5	6.0	109	15295	455	6.9	7.5
20	27500	928	8470	295	1075	19.0	4.1	116	16880	535	6.7	6.5
24	32515	1025	11215	502	932	32.7	3.8	71	12740	485	7.5	6.7
30	31160	1150	7950	380	1140	48.5	5.6	85	13495	610	7.8	7.3
AVG.	30268	986	10055	390	1126	33.6	5.1	96	14160	519	7.3	6.8
ST.DEV.	1461	127	1261	86	132	14.9	1.5	19	1699	60	0.43	.66

REACTOR OPERATING AT 5 DEG. C.

TIME (days)	COD (mg/l) Inf.	COD (mg/l) Eff.	SS (mg/l) Inf.	SS (mg/l) Eff.	NH -N (mg/l) Inf.	NH -N (mg/l) Eff.	NO (mg/l) Inf.	NO (mg/l) Eff.	MLSS (mg/l)	Sludge Vol. cylinder (ml)	pH	DO (mg/l)
1	29500	2364	9875	1027	920	465	2.8	3.5	15225	740	8.0	7.5
4	30280	1987	10505	902	1015	547	2.5	4.1	14330	825	8.4	6.9
8	31640	1260	11660	850	1132	735	6.1	7.0	12905	910	8.6	7.1
12	32075	2570	12045	660	1175	722	4.9	5.5	16500	960	8.5	7.8
16	26950	1165	8150	435	860	490	1.6	2.9	13935	885	9.2	7.3
20	28140	1485	9535	330	946	615	5.2	8.0	15010	905	8.3	6.4
24	33125	2845	11390	850	1238	830	7.1	6.1	16120	920	8.9	7.0
30	30050	2730	10720	955	1156	705	3.5	4.0	15375	865	8.6	6.9
AVG.	30220	2050	10485	751	1055	638	4.2	4.7	14925	876	8.5	7.1
ST.DEV.	2051	632	1195	236	129	122	1.8	1.6	1097	64	0.34	0.4

TABLE B7. TRACK ANALYSES FOR PHASE III

TIME	REACTOR AT 5° C			REACTOR at 10° C		
	NH_3-N (mg/l)	NO_x (mg/l)	COD (mg/l)	NH_3-N (mg/l)	NO_x (mg/l)	COD (mg/l)
0.0	642	3.1	1970	30	82	870
1.0	683	2.5	3015	68	56	1915
2.0	710	0.9	3865	107	34	3005
3.0	735	1.1	4990	145	7	3810
5.0	721	1.0	4735	130	19	3545
8.0	690	1.2	4310	114	31	3120
12.0	645	2.7	3880	89	48	2465
17.0	628	2.8	3325	65	70	1715
22.0	604	4.3	2650	42	96	1065
24.0	595	4.0	2640	39	91	1025

TABLE B8. MIXED LIQUOR VOLATILE SUSPENDED SOLIDS

TIME	INFLUENT CONCENTRATION				HRT/BSRT (days)		
hrs.	1:4	2:4	3:4	4:4	7/10	7/20	7/30
0.0	8524	11176	13812	15557	11164	13236	14672
1.0	8247	10764	13552	15249	-	-	-
2.0	8013	10345	13287	15052	10328	12600	14396
3.0	7852	10438	13258	14744	10700	12688	14136
5.0	8326	10611	13924	15500	11248	12900	14280
8.0	8908	11000	14328	16672	11300	13184	14476
12.0	8546	11424	14030	16094	11432	13528	14496
16.0	8376	11138	13672	15656	11612	13580	14700
18.0	8175	10784	13289	15288	11856	13792	14812
22.0	8020	10486	12964	14760	11836	13804	14832
24.0	8728	11056	13940	16052	12008	13996	15172

TABLE B9. WORKSHEET FOR PHASE IVE REACTOR (1:4)

CALCULATION OF RATE OF REACTION FOR COD REMOVAL

t	Q	So	V	SoQ/V	S	K1	A/K1
1	0.125	9500	2.81	422.5978	566	0.4	845.1957
2	0.125	9500	2.81	422.5978	684	0.4	845.1957
3	0.125	9500	2.81	422.5978	769	0.4	845.1957
5					535		
8					407		
12					335		
16					320		
18					296		
22					283		
24					285		

S1	S1-A/K1	K1t	EXP-K1t	B*C	A/K1+D	S2	K2
328	-517.195	0.5	0.606530	-313.695	531.5006		
566	-279.195	1	0.367879	-102.710	742.4853		
684	-161.195	1.5	0.223130	-35.9676	809.2281		
769						650	0.02
535						535	0.02
407						407	0.02
335						335	0.02
320						320	0.02
296						296	0.02
283						283	0.02

t-3	K2*t-3	EXP-K2*T	S1*EXP	So-FILL	Sp-REACT	Sp	S
				531.5006		531.5006	556
				742.4853		742.4853	684
				809.2281		809.2281	769
2	0.04	0.967089	624.5131		624.5131	624.5131	535
5	0.1	0.904837	484.0880		484.0880	484.0880	407
9	0.18	0.835270	339.9549		339.9549	339.9549	335
13	0.26	0.771051	258.3022		258.3022	258.3022	320
15	0.3	0.740818	237.0618		237.0618	237.0618	296
19	0.38	0.683861	202.4229		202.4229	202.4229	283
21	0.42	0.657046	185.9442		185.9442	185.9442	285

TABLE B10. WORKSHEET FOR PHASE IV REACTOR (1:4)

CALCULATION OF RATE OR REACTION FOR NH_3 REMOVAL

t	Q	So	V	SoQ/V	S	K1	A/K1
1	0.125	442	2.81	19.66192	18.33	0.2	98.30960
2	0.125	442	2.81	19.66192	35.1	0.2	98.30960
3	0.125	442	2.81	19.66192	57.04	0.2	98.30960
5					49.2		
8					31.87		
12					15		
16					4.62		
18					2.9		
22					1.5		
24					1.2		

S1	S1-A/K1	K1t	EXP-K1t	B*C	A/K1+D	S2	K2
0.97	-97.3396	0.2	0.818730	-79.6949	18.61467		
18.33	-79.9796	0.4	0.670320	-53.6119	44.69767		
35.1	-63.2096	0.6	0.548811	-34.6901	63.61943		
57.04						52	0.1
49.2						49.2	0.1
31.87						31.87	0.1
15						15	0.1
4.62						4.62	0.1
2.9						2.9	0.1
1.5						1.5	0.1

t-3	K2*t-3	EXP-K2*T	S2*EXP	Sp-FILL	Sp-REACT	Sp	S
				18.61467		18.61467	18.33
				44.69767		44.69767	35.1
				63.61943		63.61943	57.04
2	0.2	0.818730	42.57399		42.57399	42.57399	49.2
5	0.5	0.606530	29.84130		29.84130	29.84130	31.87
9	0.9	0.406569	12.95737		12.95737	12.95737	15
13	1.3	0.272531	4.087976		4.087976	4.087976	4.62
15	1.5	0.223130	1.030861		1.030861	1.030861	2.9
19	1.9	0.149568	0.433748		0.433748	0.433748	1.5
21	2.1	0.122456	0.183684		0.183684	0.183684	1.2

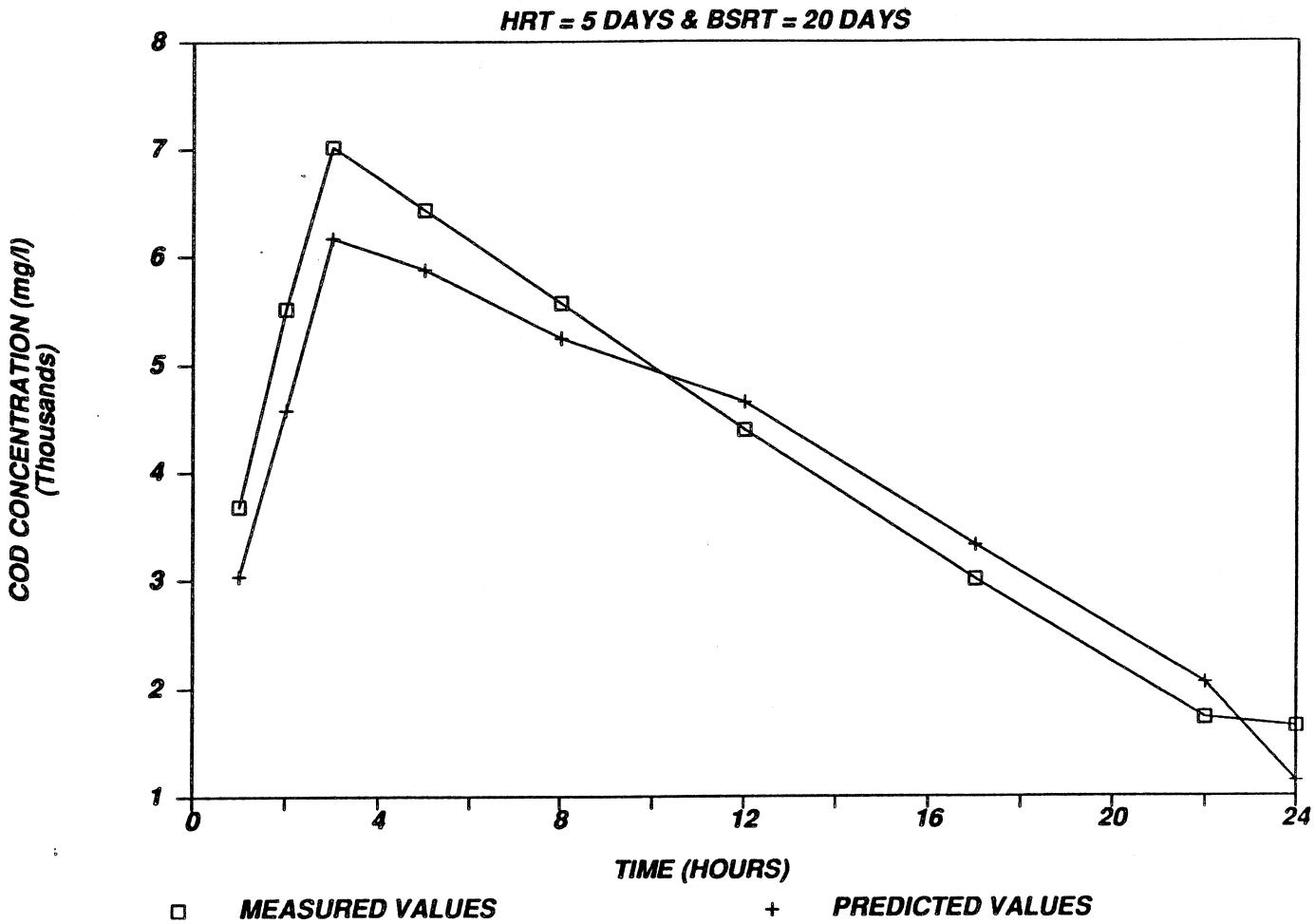


Figure B.1 Measured and Predicted COD Concentration Profiles for Reactor Operating at 5/20 (HRT/BSRT)

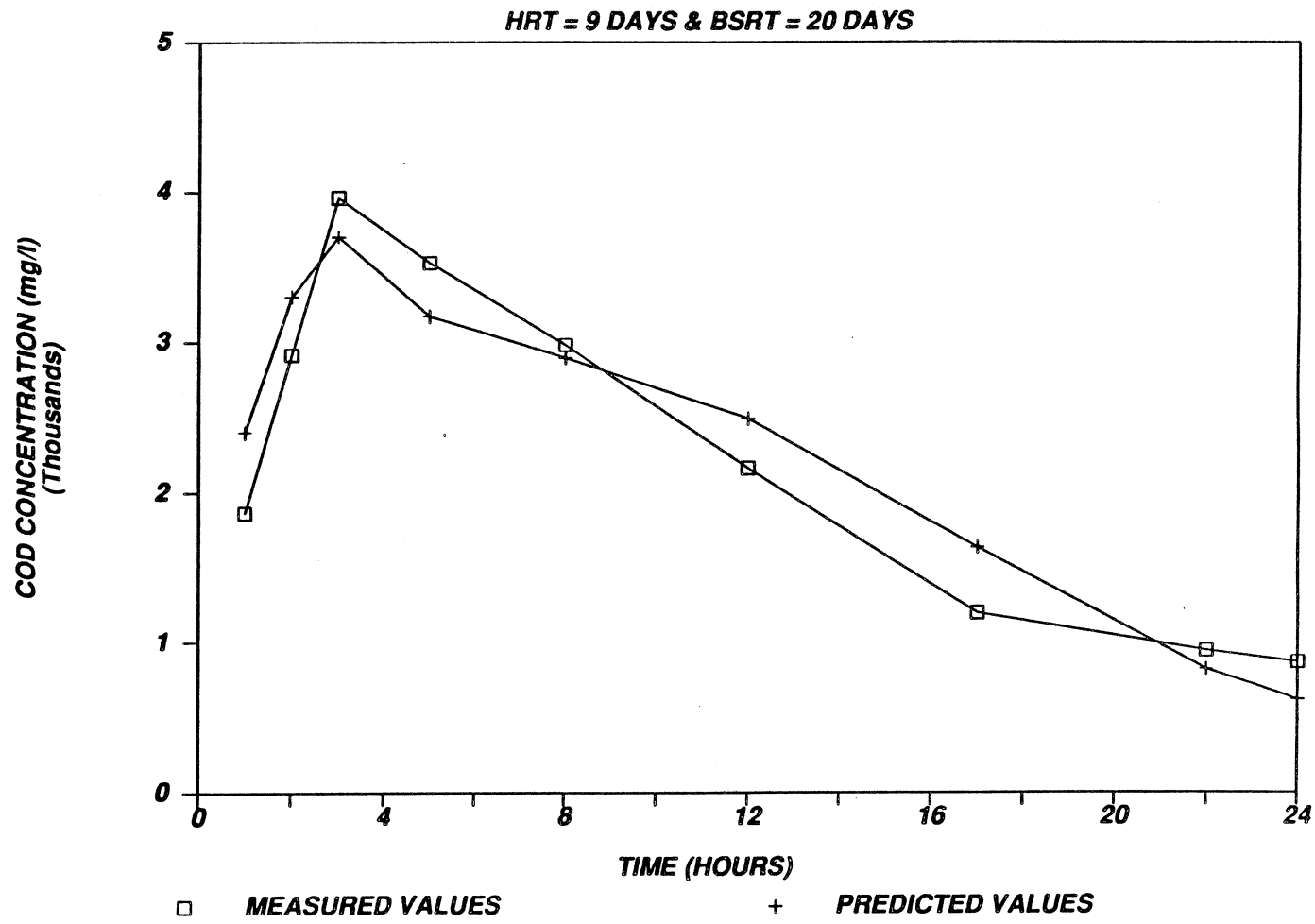


Figure B.2 Measured and Predicted COD Concentration Profiles for Reactor Operating at 9/20 (HRT/BSRT)

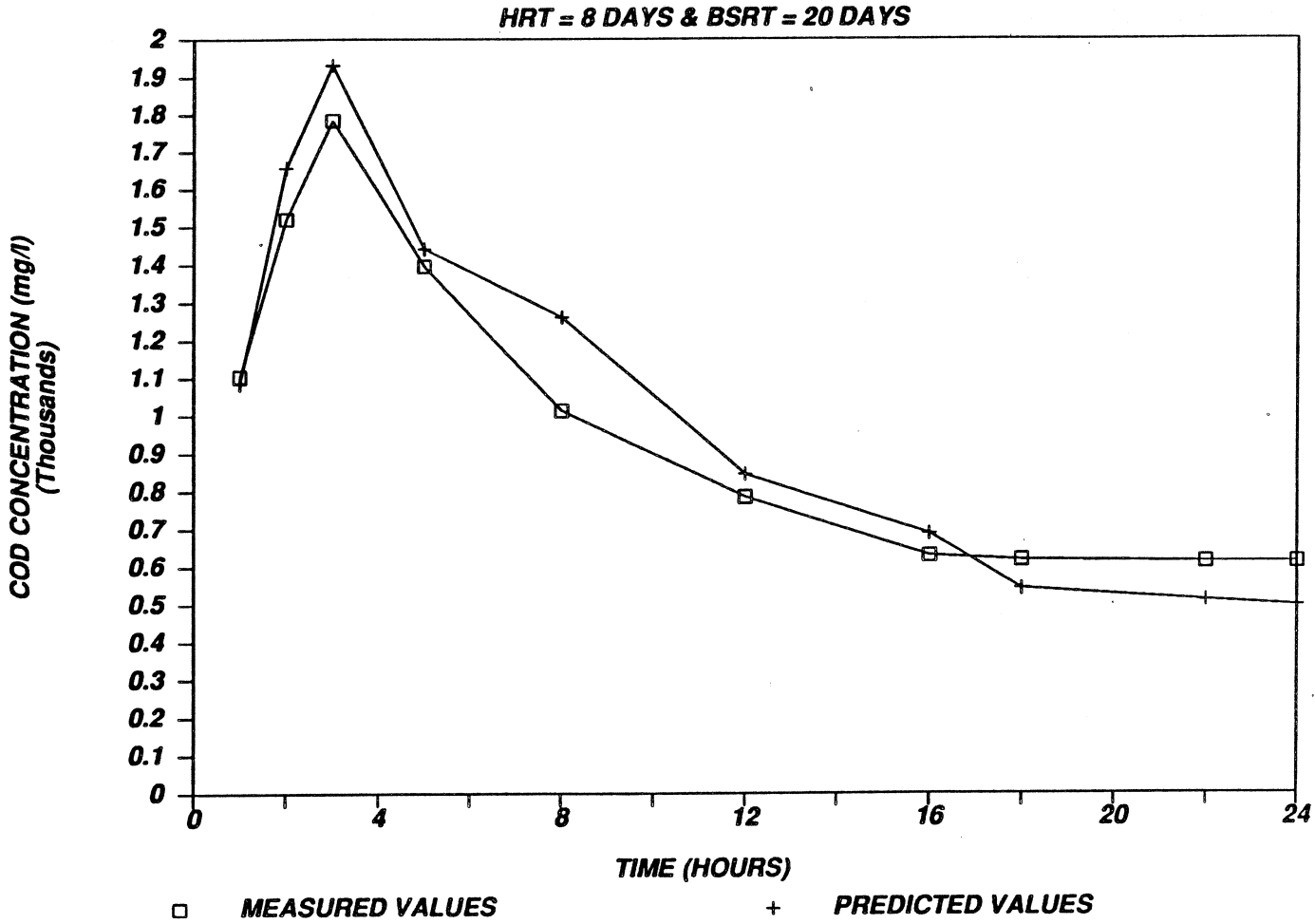
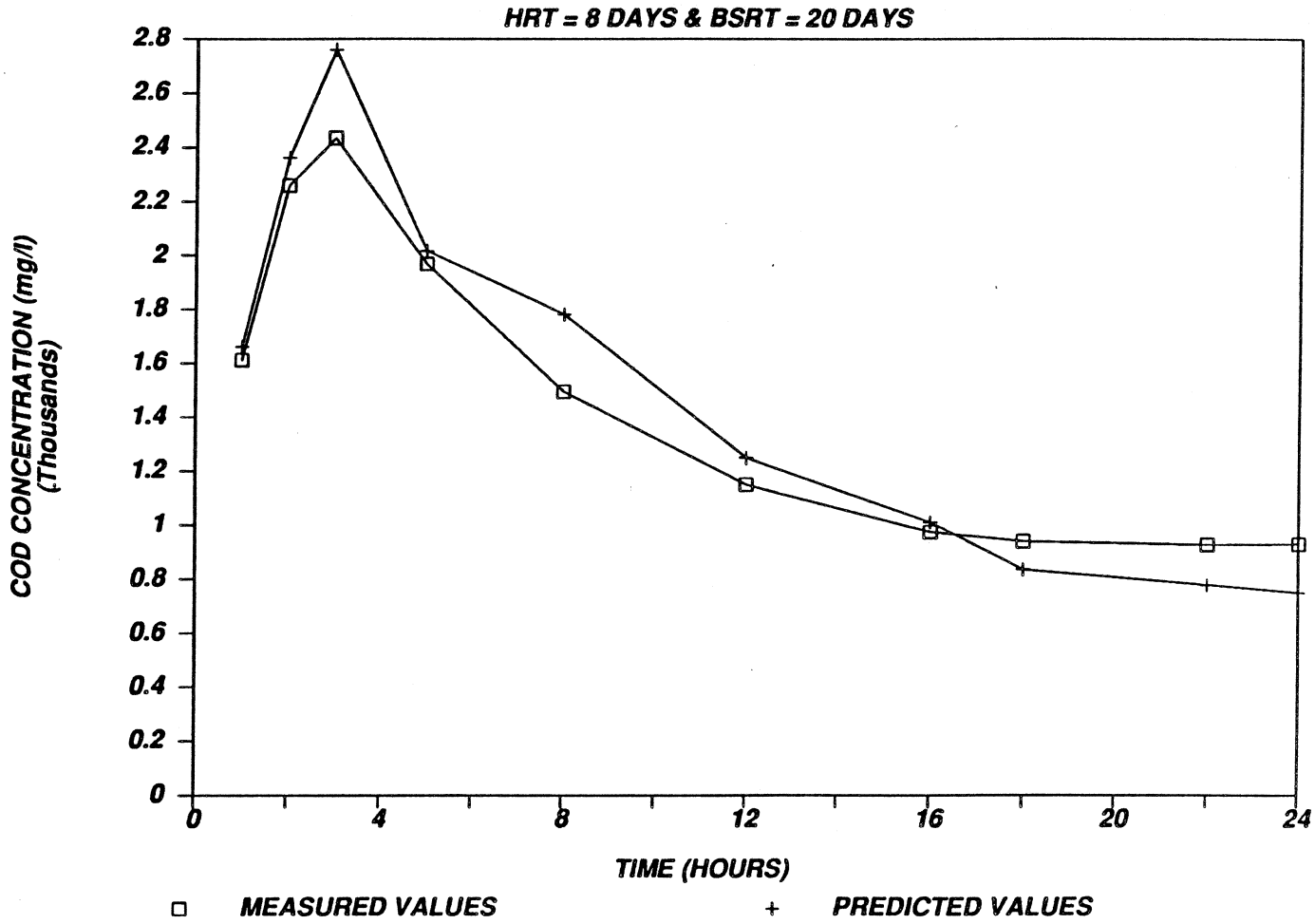


Figure B.3 COD Concentration Profiles Based on Variable K_2 (2:4)



B.4 COD Concentration Profiles Based On Variable K_2 (3:4)

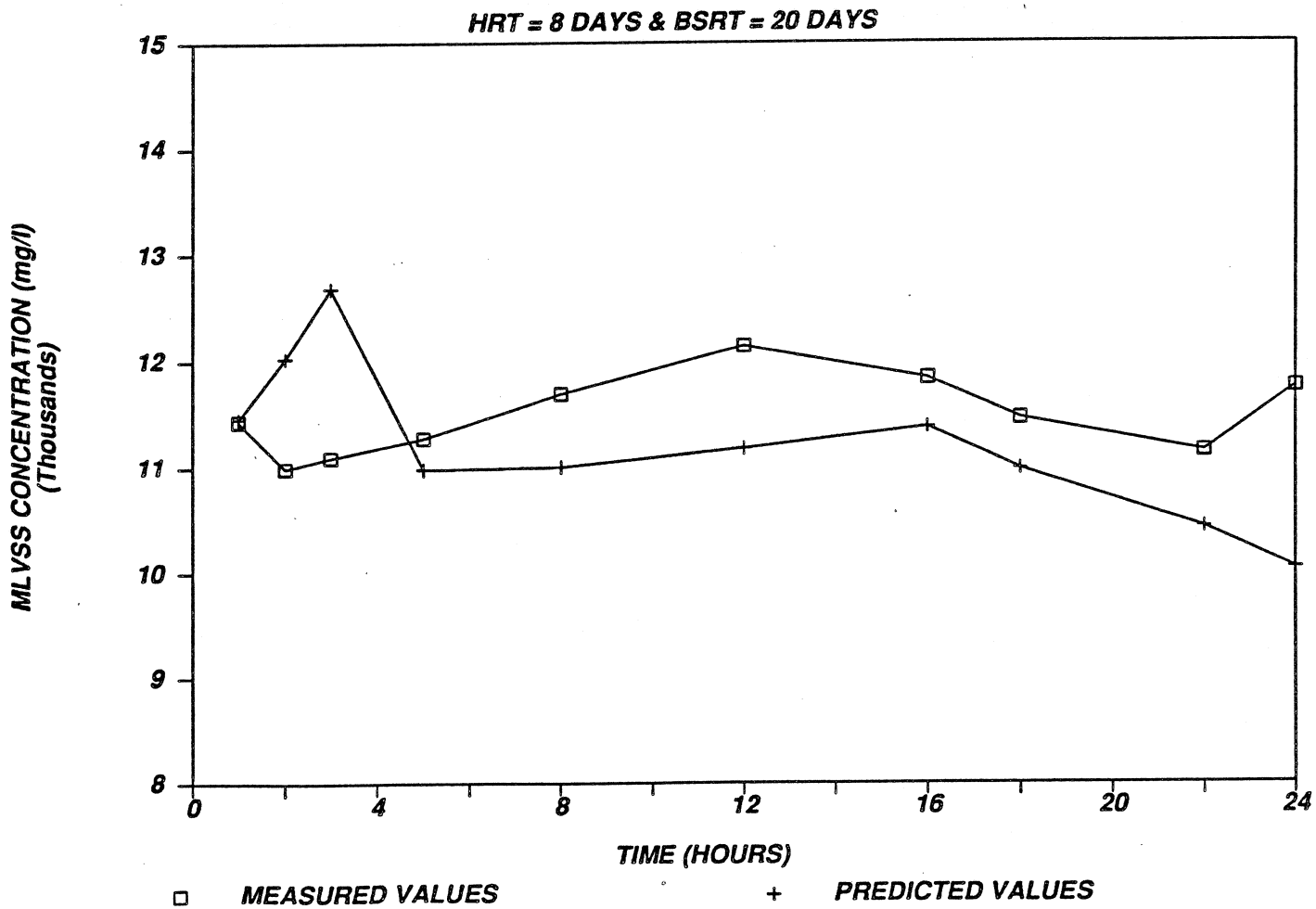


Figure B.5 Measured and predicted MLVSS concentration profiles (2:4)

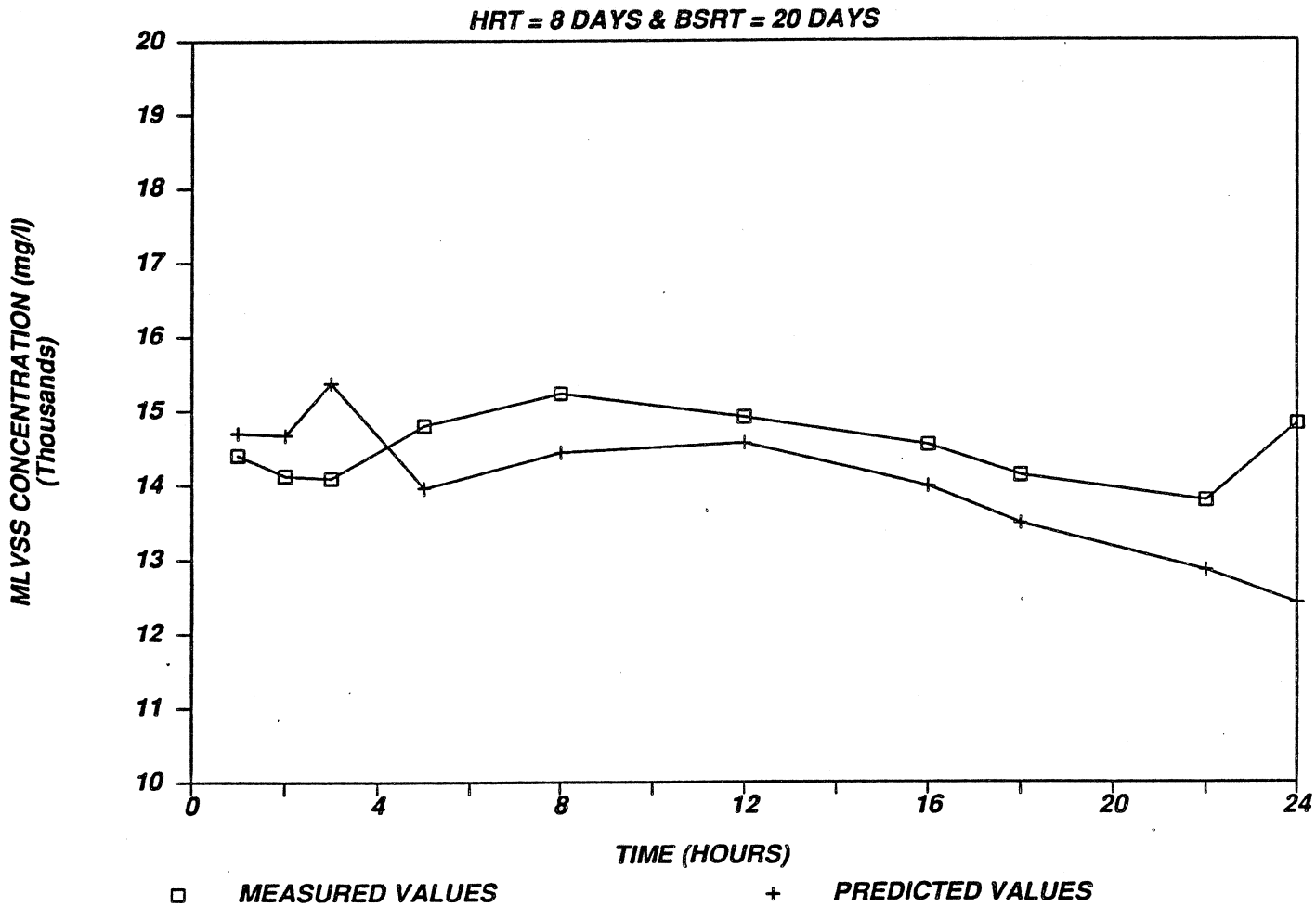


Figure B.6 Measured and predicted MLVSS concentration profiles (3:4)

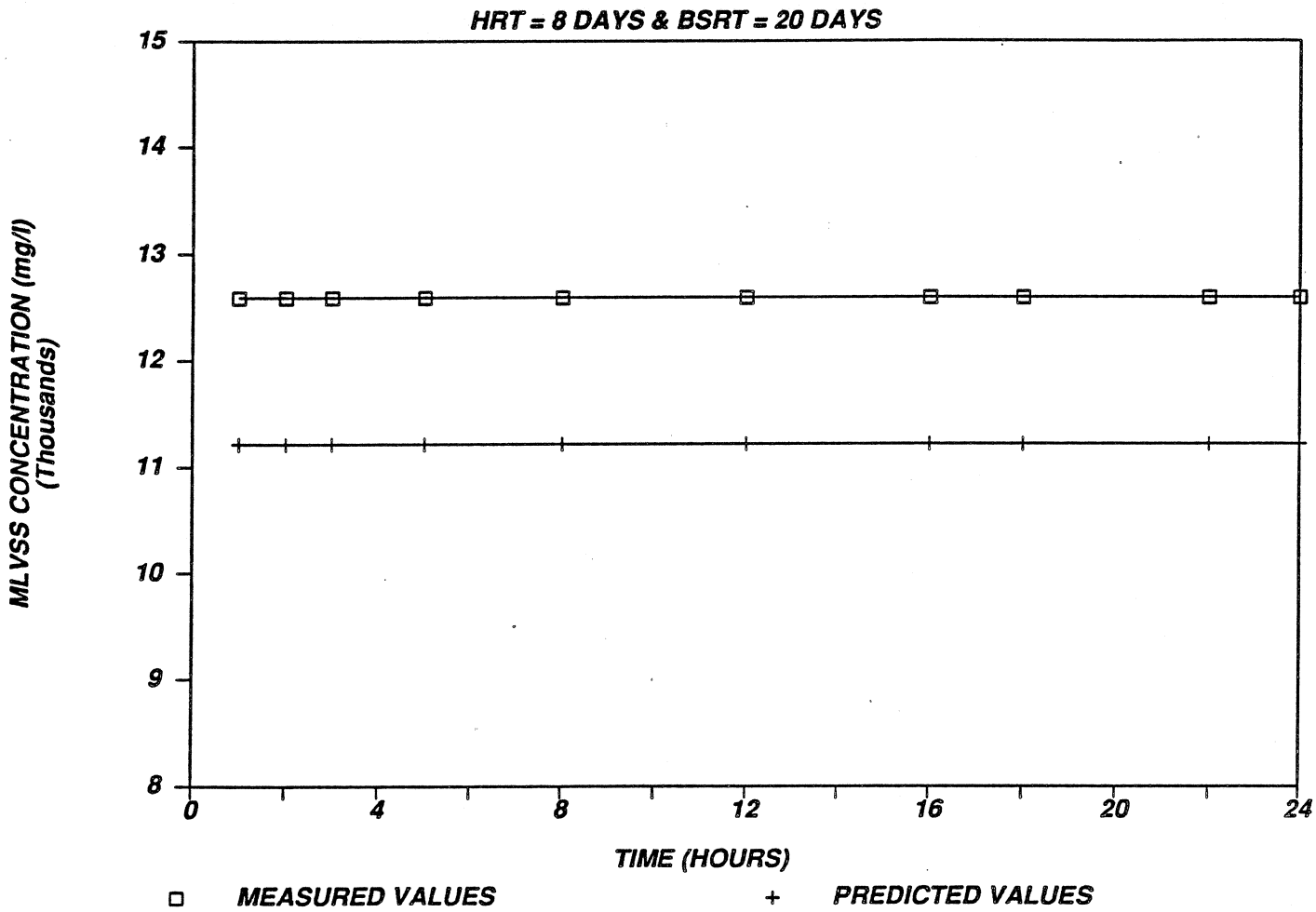


Figure B.7 Measured and predicted average MLVSS concentration profiles (2:4)

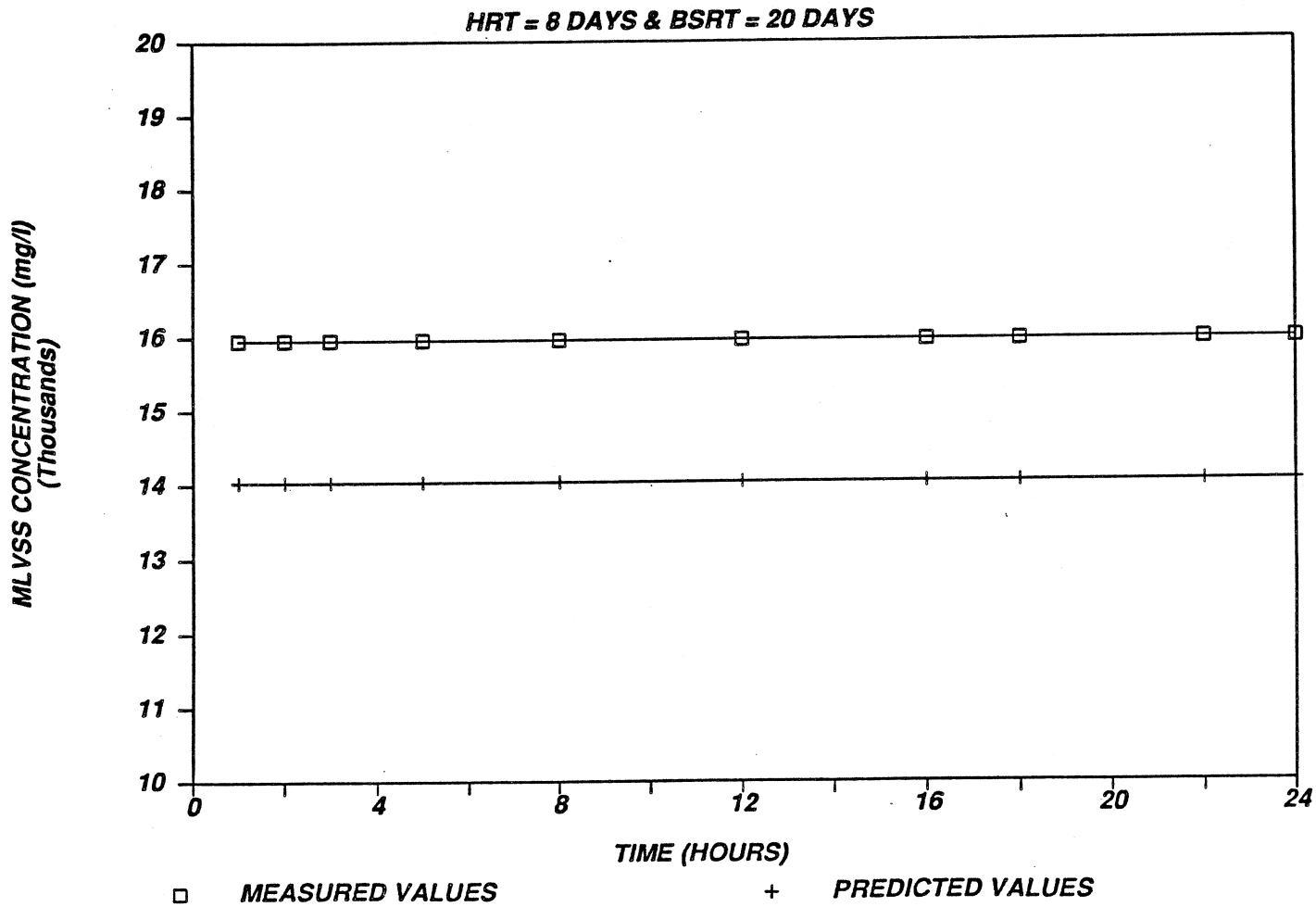


Figure B.8 Measured and predicted average MLVSS concentration profiles (3:4)

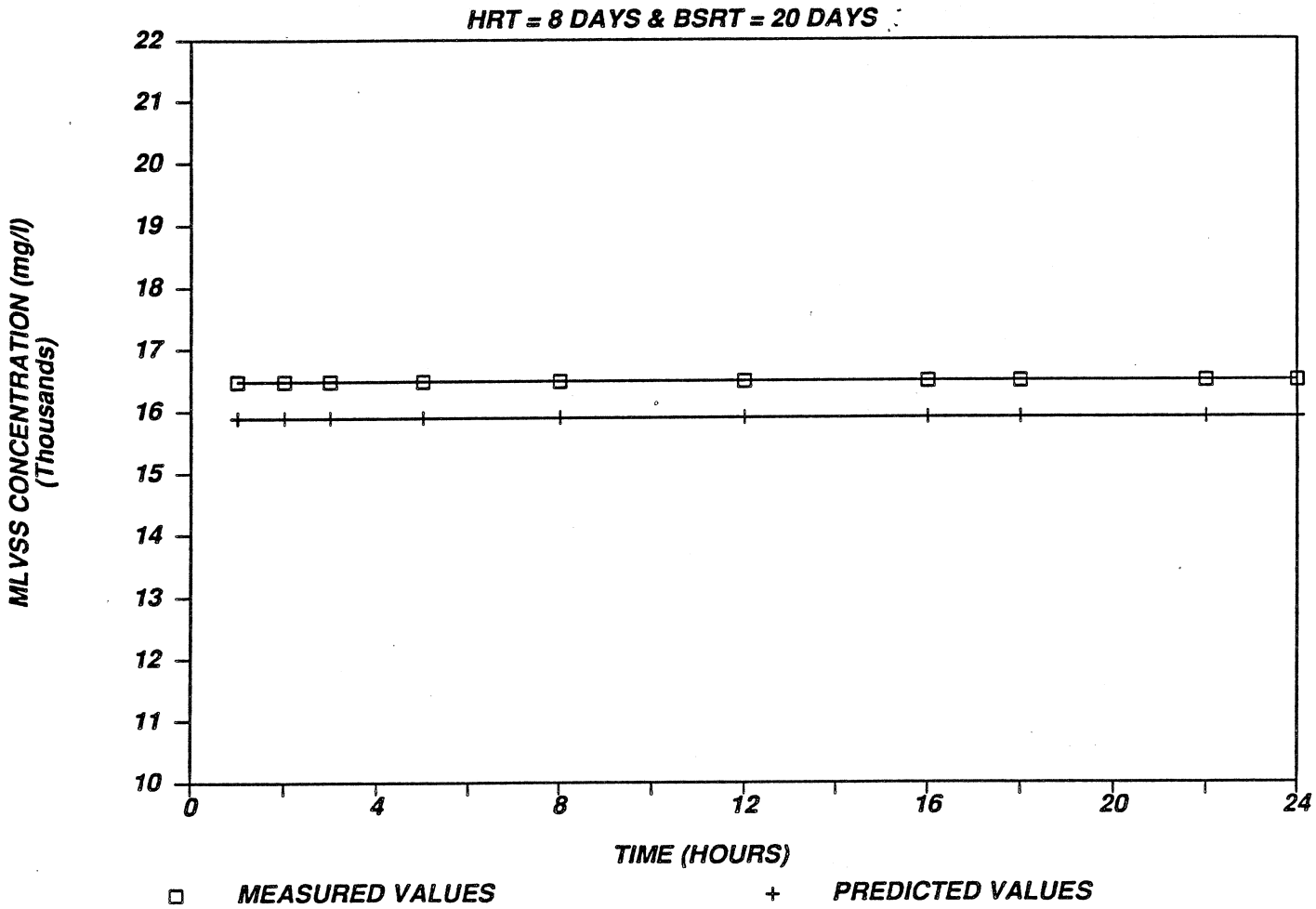


Figure B.9 Measured and predicted average MLVSS concentration profiles (4:4)

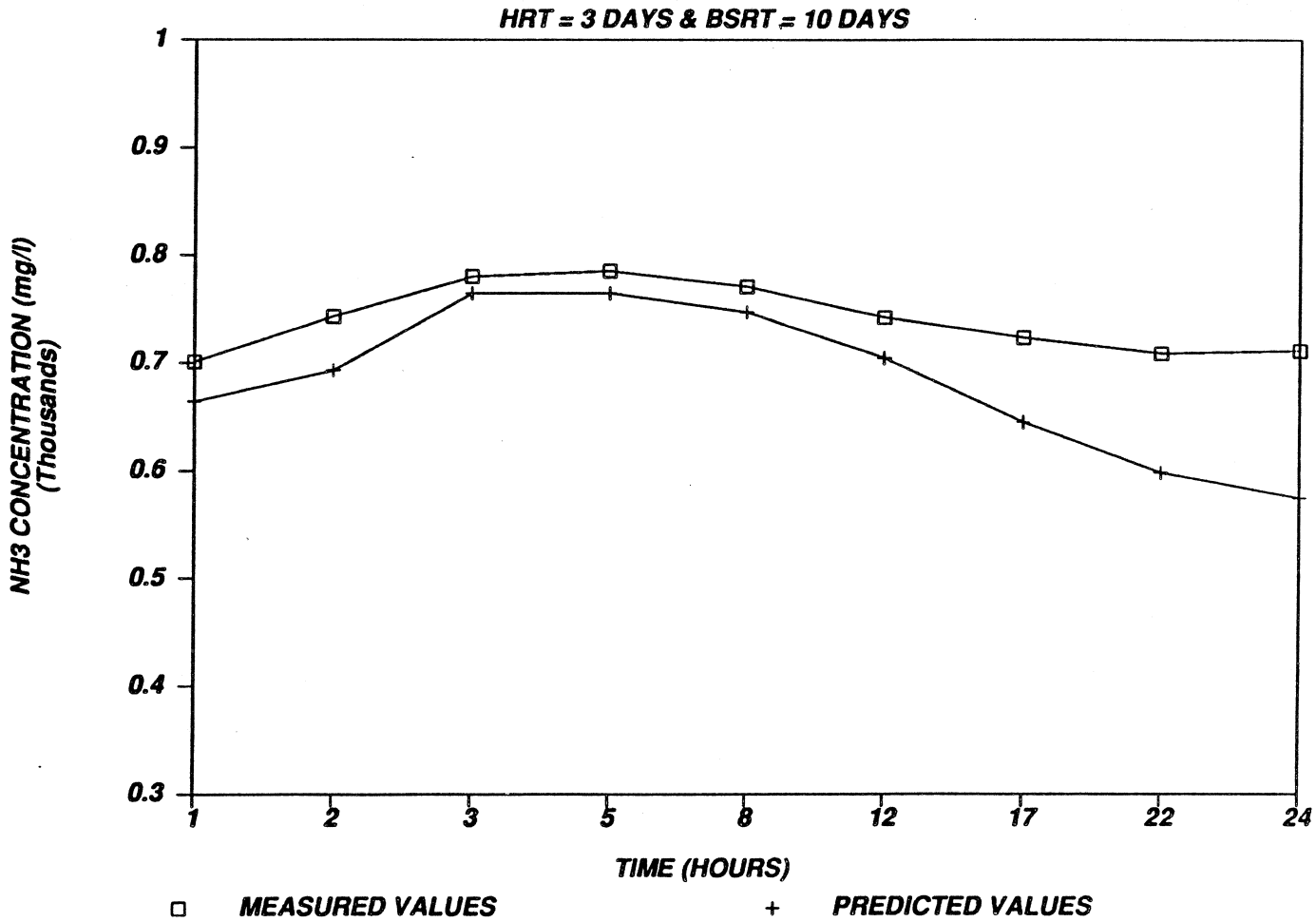


Figure B.10 Measured and Predicted NH_3-N Concentration Profiles for Reactor Operating at 3/10 (HRT/BSRT)

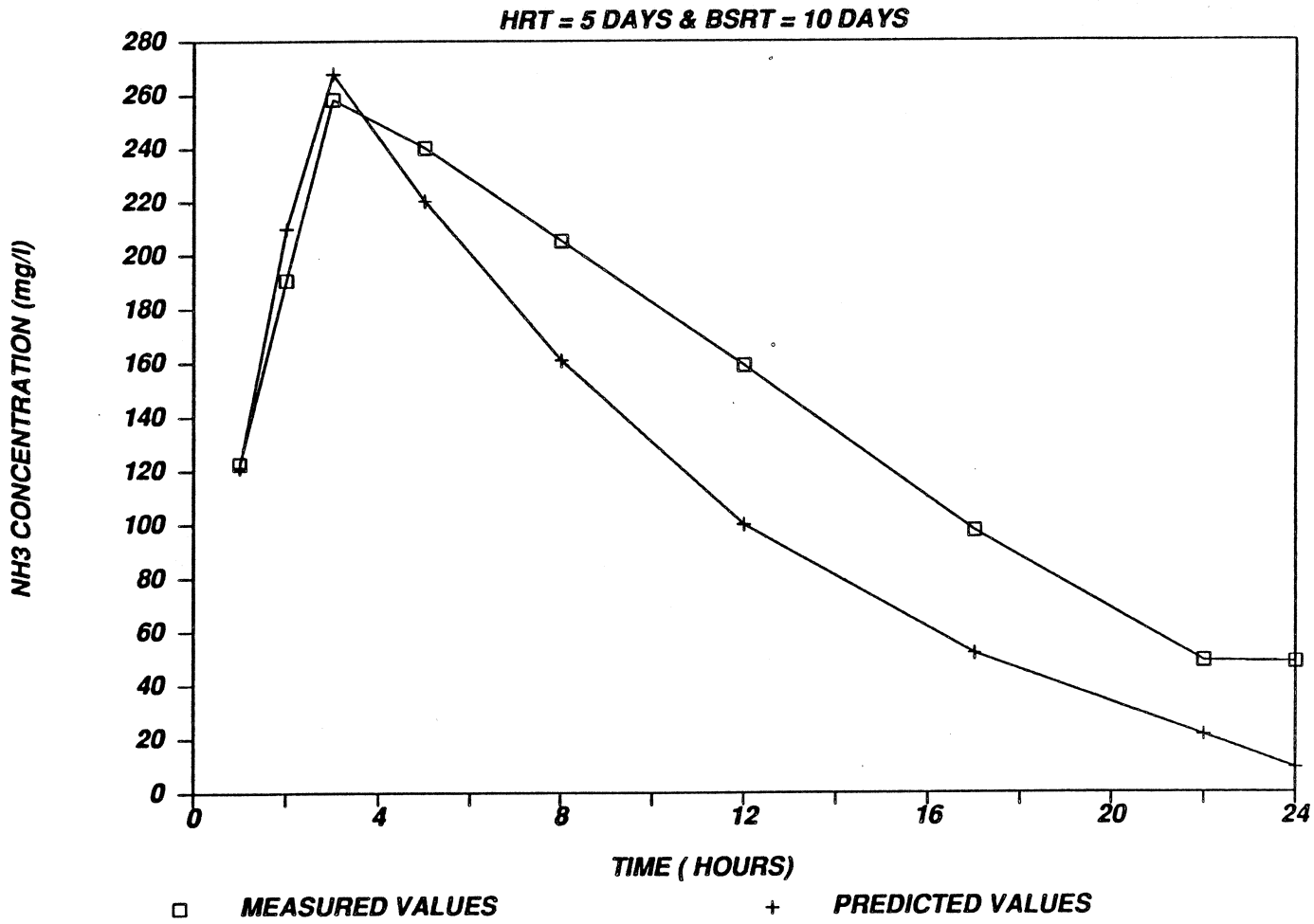


Figure B.11 Measured and Predicted NH_3-N Concentration Profiles for Reactor Operating at 5/10 (HRT/BSRT)

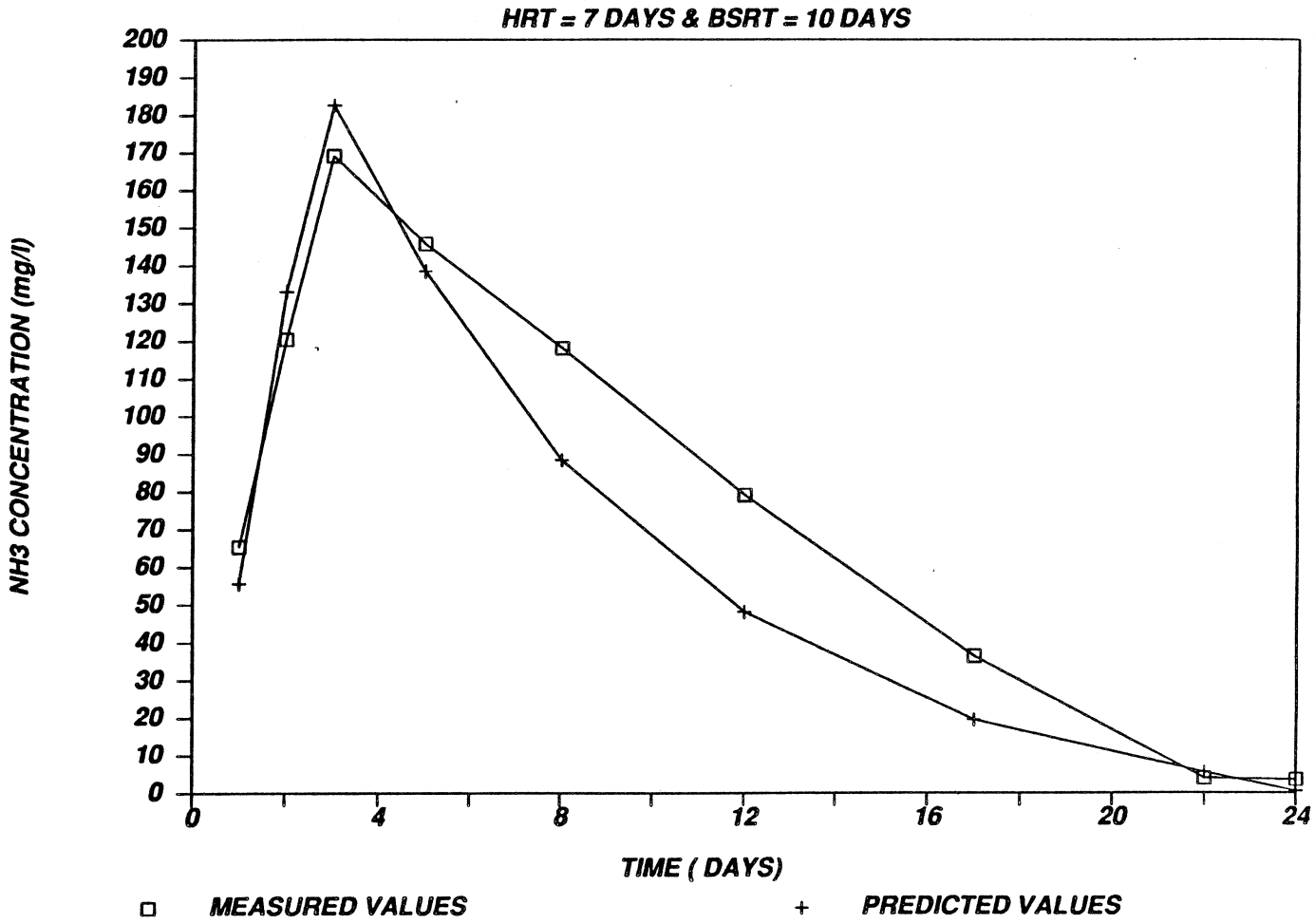


Figure B.12 Measured and Predicted NH_3-N Concentration Profiles for Reactor Operating at 7/10 (HRT/BSRT)

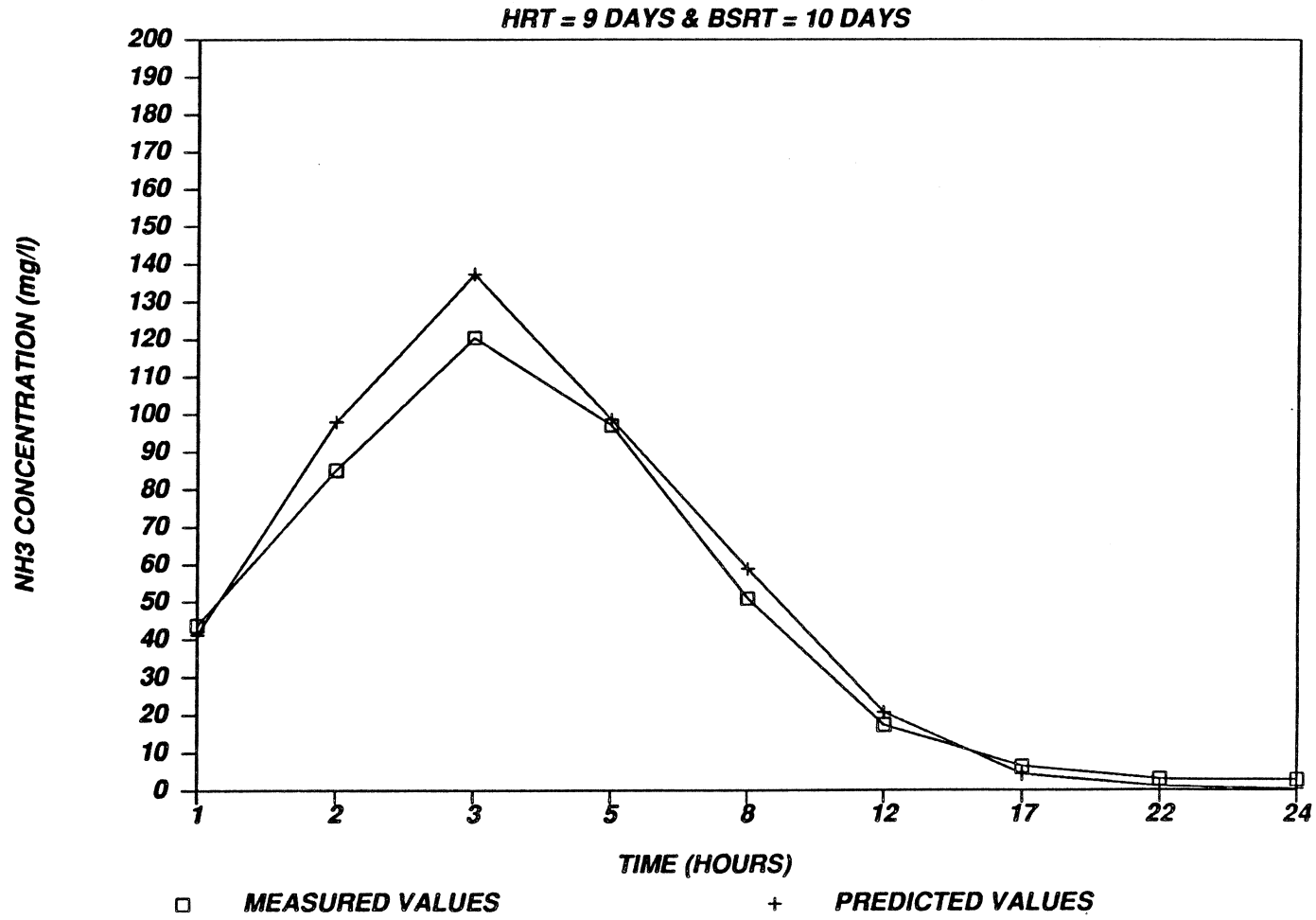


Figure B.13 Measured and Predicted NH_3-N Concentration Profiles for Reactor Operating at 9/10 (HRT/BSRT)

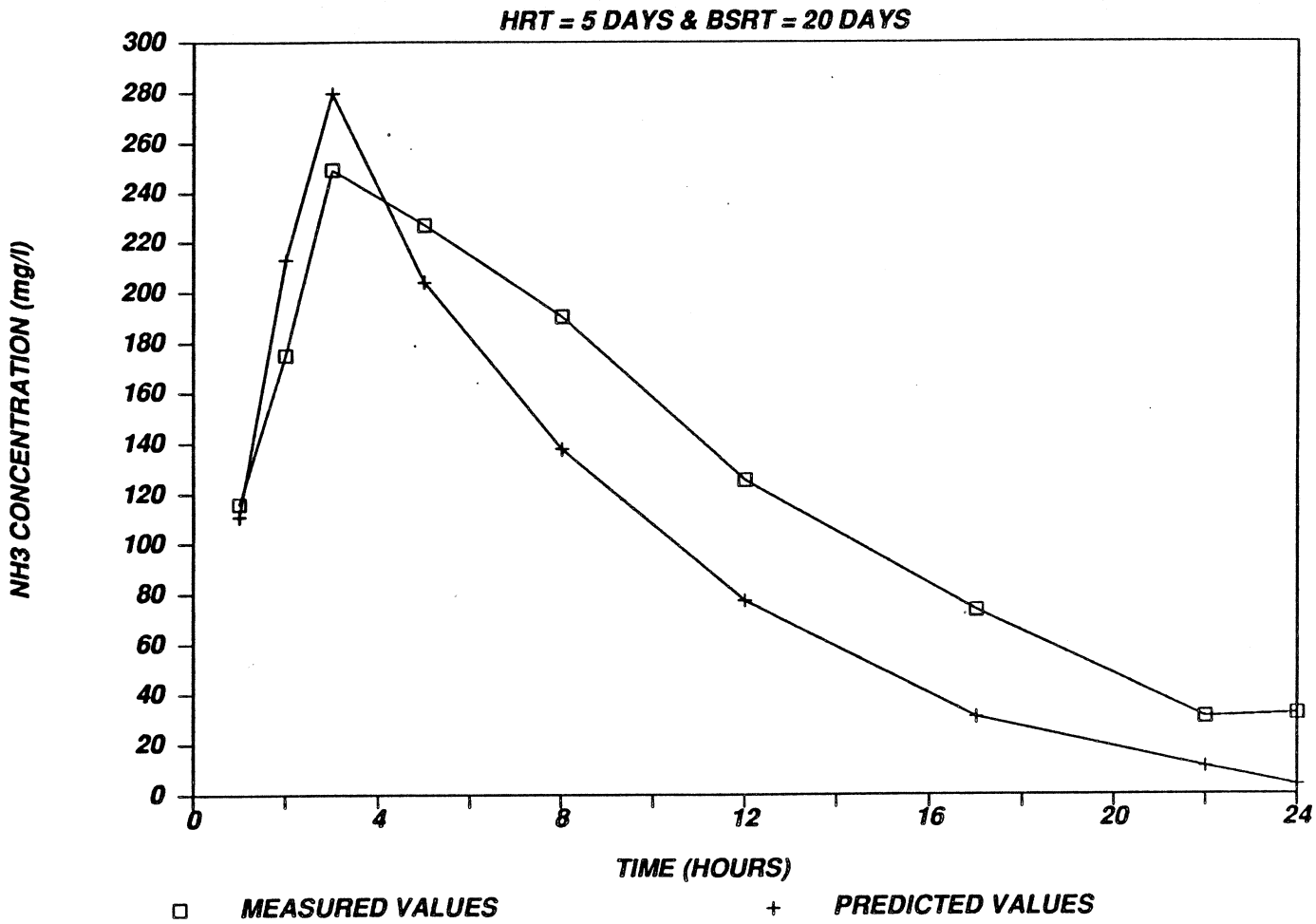


Figure B.14 Measured and Predicted NH_3-N Concentration Profiles for Reactor Operating at 5/20 (HRT/BSRT)

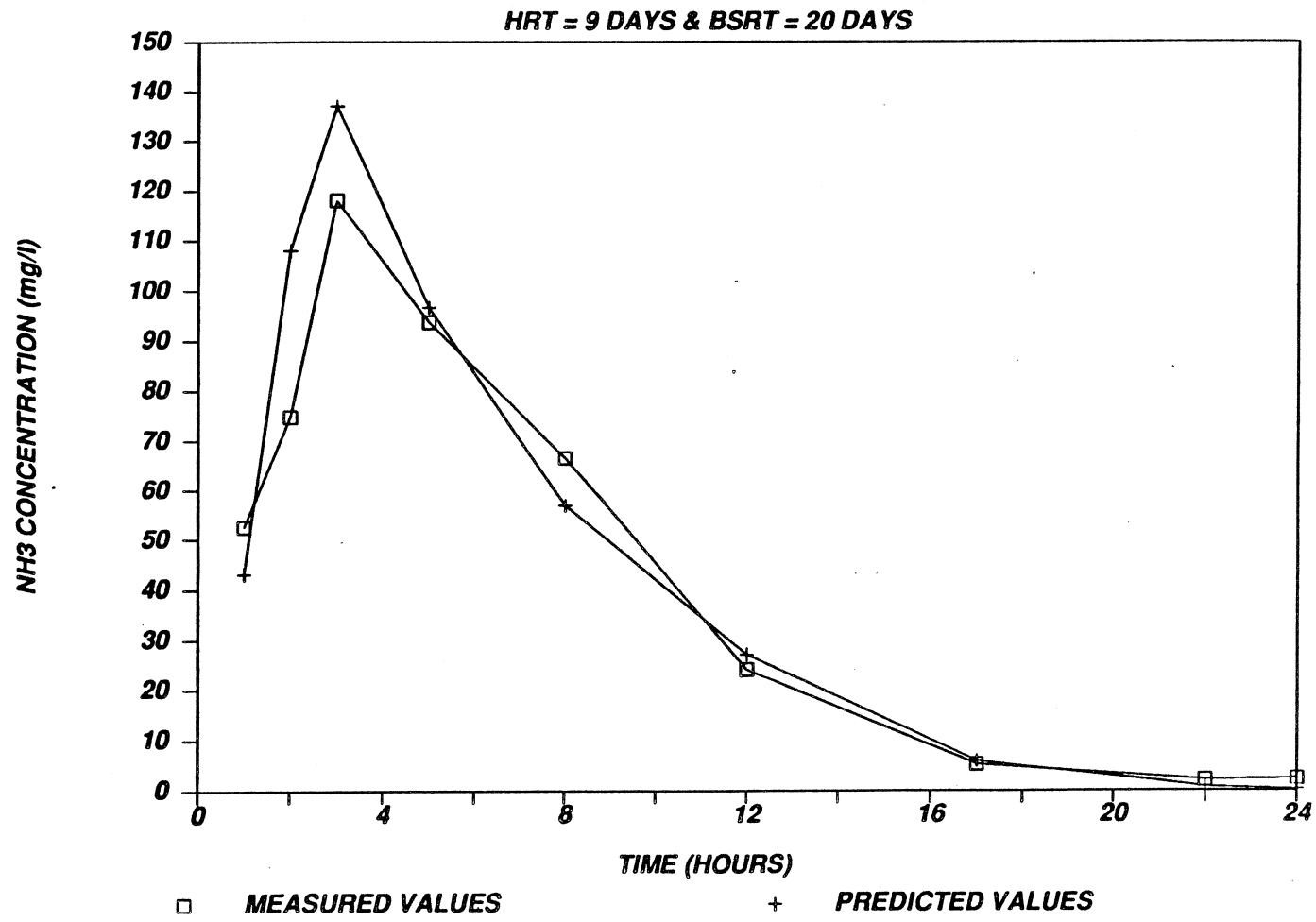


Figure B.15 Measured and Predicted NH_3-N Concentration Profiles for Reactor Operating at 9/20 (HRT/BSRT)

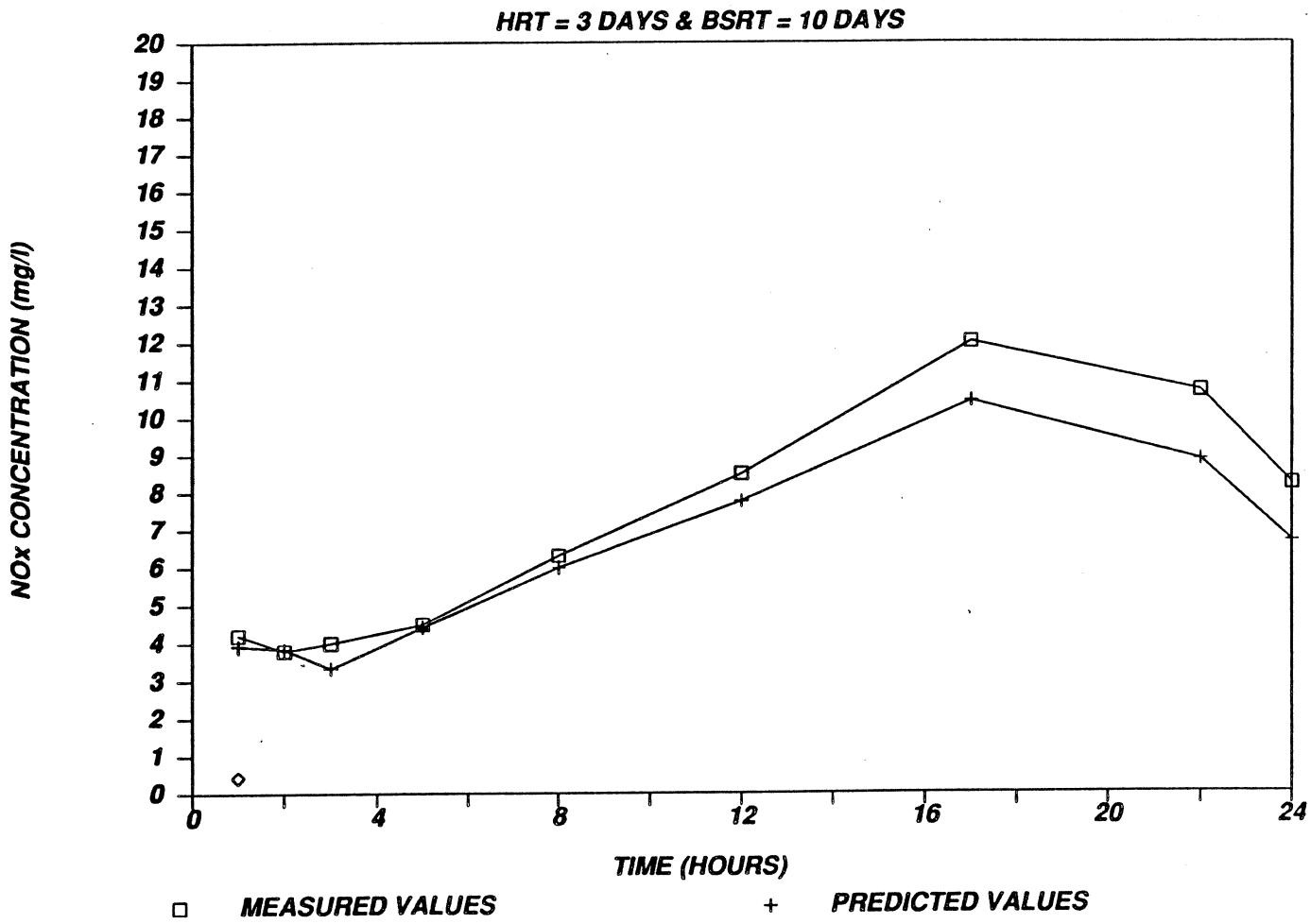


Figure B.16 Measured and Predicted NOx Concentration Profiles for Reactor Operating at 3/10 (HRT/BSRT)

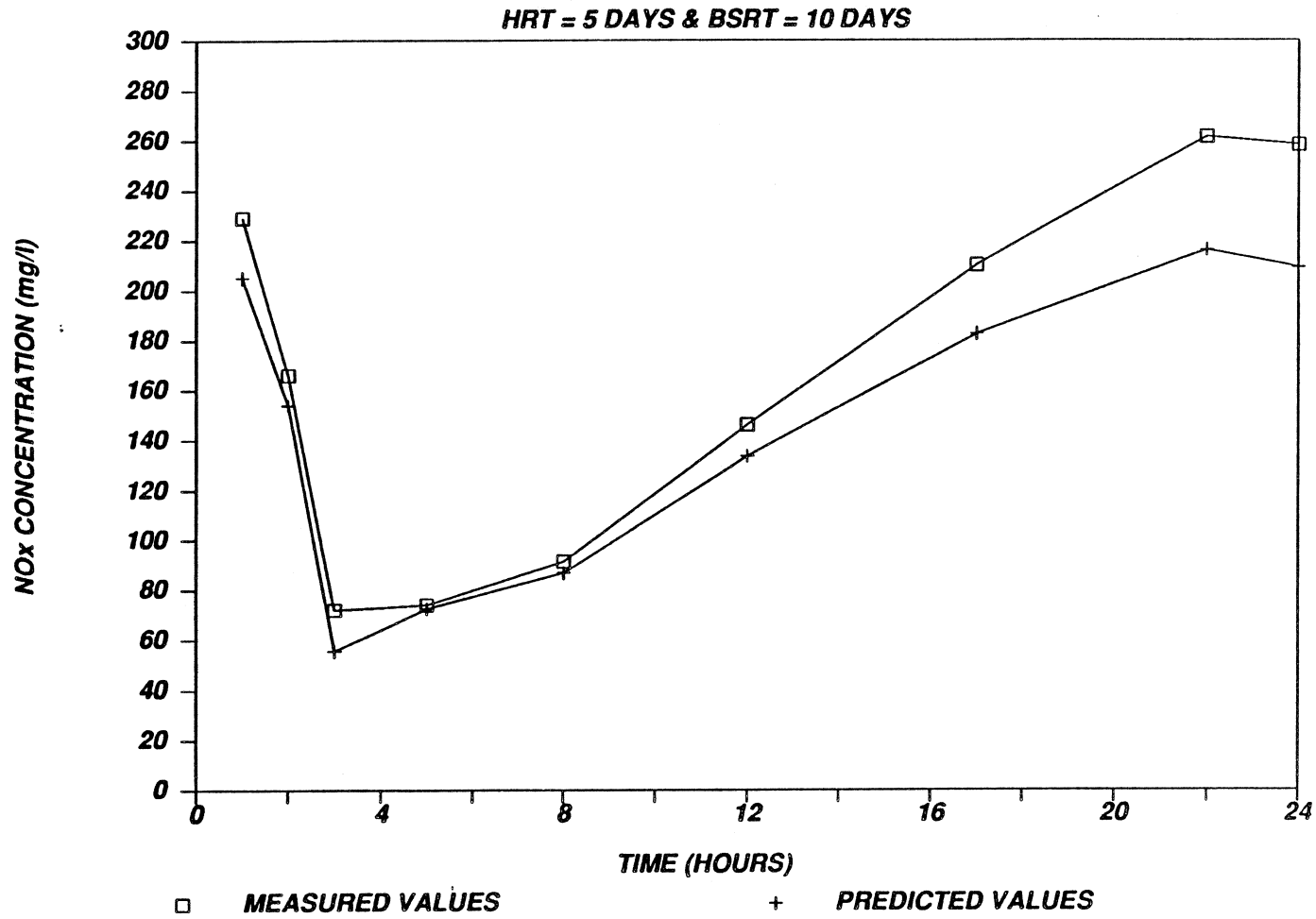


Figure B.17 Measured and Predicted NOx Concentration Profiles for Reactor Operating at 5/10 (HRT/BSRT)

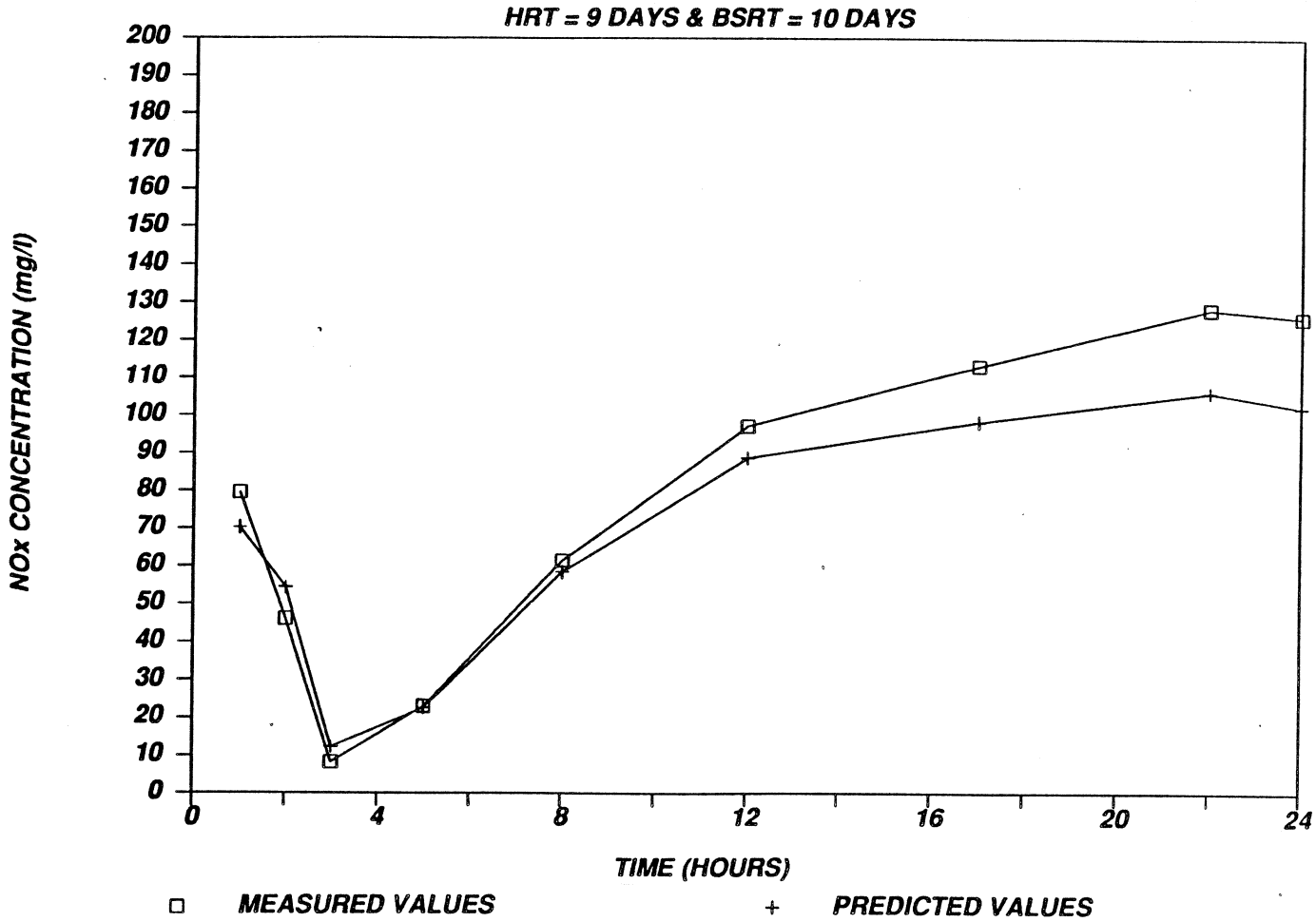


Figure B.18 Measured and Predicted NOx Concentration Profiles for Reactor Operating at 9/10 (HRT/BSRT)

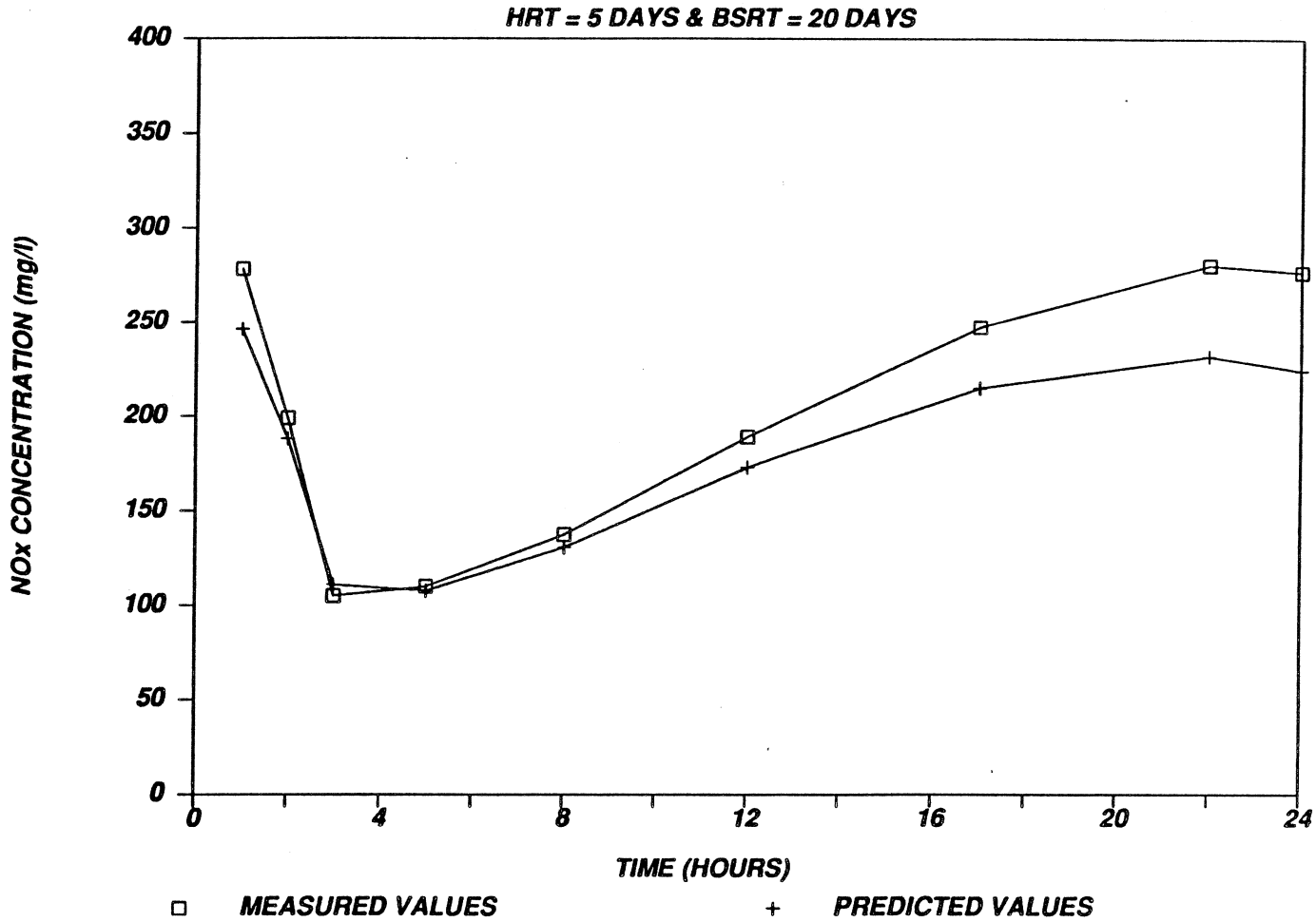
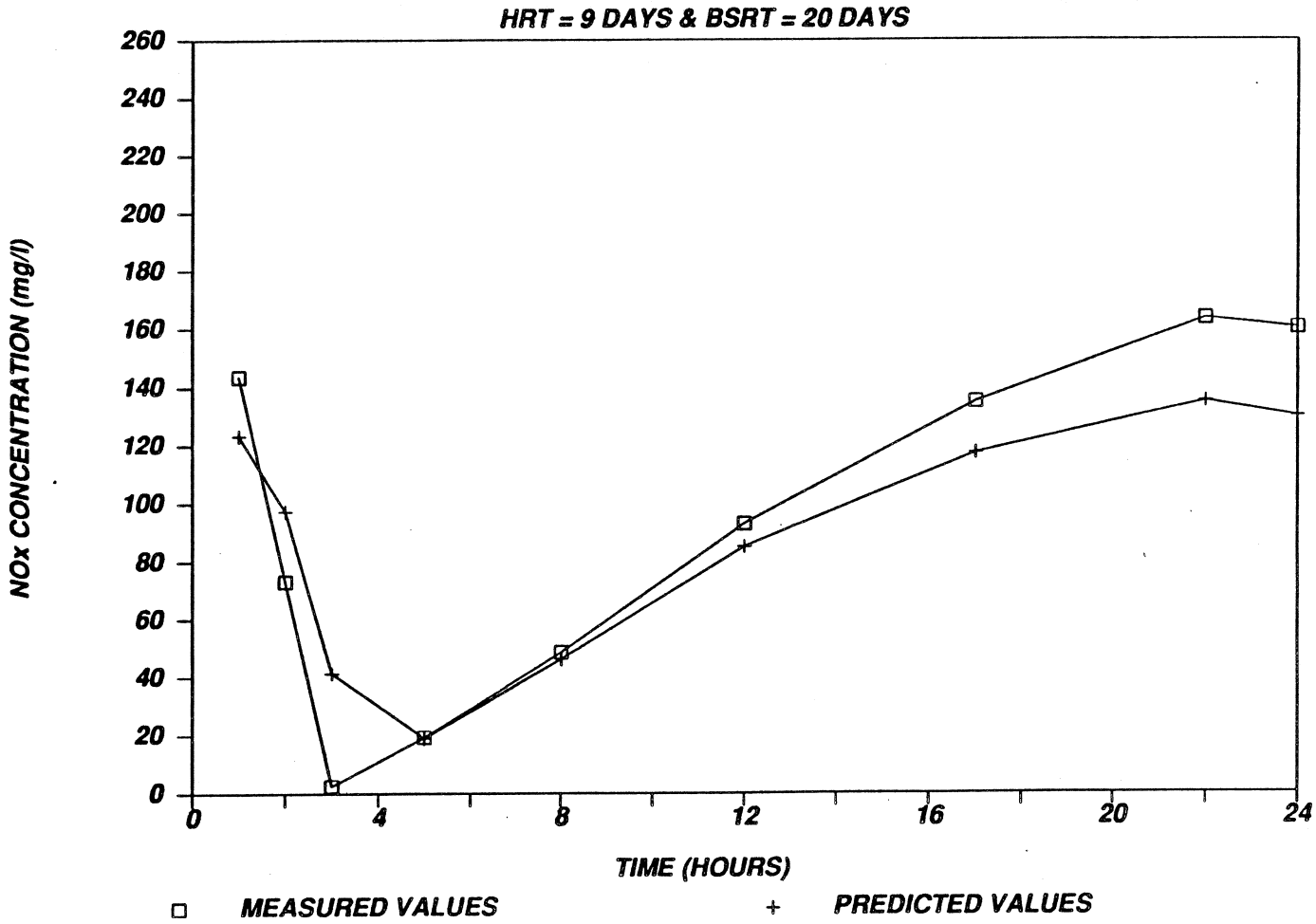


Figure B.19 Measured and Predicted NH_3-N Concentration Profiles for Reactor Operating at 5/20 (HRT/BSRT)



B.20 Measured and Predicted NOx Concentration Profiles for Reactor Operating at 9/20 (HRT/BSRT)

APPENDIX C

TABLE C1. METHODS USED FOR PHYSICAL AND CHEMICAL ANALYSES

PARAMETER	METHOD
Total Solids (TS)	209 A. Total Solids Dried at 103 - 105° 209 F. Total, Fixed and Volatile Solids in Solid and Semisolid Samples
Total Suspended Solids (TSS)	209 C. Total Suspended Solids Dried at 103 - 105°C
Volatile Suspended Solids (VSS)	209 D. Fixed and Volatile Solids Ignited at 550°C
Sludge Volume Index (SVI)	213 C. Sludge Volume Index
Dissolved Oxygen (DO)	421 F. Membrane Electrode Method
Oxygen Uptake Rate (OUR)	213 A. Oxygen Consumption Rate
Biological Oxygen Demand (BOD_5)	507 Oxygen Demand (Biochemical)
Chemical Oxygen Demand (COD)	508 B. Oxygen Demand (Chemical) Closed Reflux, Titrimetric Method
Total Kjeldahl Nitrogen	420 A. Macro - Kjeldahl Method
Ammonia Nitrogen (NH_3-N)	417 E. Ammonia-Selective Electrode Method
Nitrite Nitrogen (NO_2-N)	419 Nitrogen (Nitrite)
Nitrate Nitrogen (NO_3-N)	418 B. Nitrate Electrode Screening Method
Total Phosphorus (TP)	424 Preliminary Digestion Steps for Total Phosphorus Sulfuric Acid - Nitric Acid Digestion 424 E. Stannous Chloride Method
pH and Temperature	423 pH Value. Corning Portable Digital pH/Temperature Meter No. 4 from Corning Science Products, USA.