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Date: 1/24/78

Project Title: Nitrification in the Waste Treatment Process by Attached Microbial Films

Project No: E-20-691 (continued by E-20-698)

Project Director: Dr. F.M. Saunders

Sponsor: Office of Water Research and Technology

Effective Termination Date: 9/30/76

Clearance of Accounting Charges: All Clear

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NITRIFICATION WITH ROTATING BIOLOGICAL CONTACTOR SYSTEMS



F. MICHAEL SAUNDERS
RODNEY L. POPE

SCHOOL OF CIVIL ENGINEERING

in cooperation with the

ENVIRONMENTAL RESOURCES CENTER

GEORGIA INSTITUTE OF TECHNOLOGY

ATLANTA, GEORGIA

Nitrification with Rotating
Biological Contactor Systems

F. Michael Saunders
Rodney L. Pope

Technical Completion Report
OWRT Project Number A-058-GA
Nitrification by Attached Biological Films
Initiated: July 1974 Completed: October 1978

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ABSTRACT

NITRIFICATION WITH ROTATING BIOLOGICAL CONTACTOR SYSTEMS

A laboratory bench-scale rotating biological contactor (RBC) system was utilized to evaluate the effect of microbial growth rates, expressed as cell residence time (θ_c), on biological nitrification of a simulated secondary wastewater effluent. The effect of influent organic-nitrogen was examined as well as the effect of influent wastewater organic strength at a value of θ_c near that for washout of nitrifying microorganisms. θ_c values were controlled by continuous circulation of RBC mixed liquor to remove sloughed biomass from suspension and by periodic scraping of attached biomass from disc surfaces.

Kinetic relationships developed for the nitrifying microorganisms in the RBC system were found to be in agreement with those developed for pure cultures of Nitrosomonas and Nitrobacter as well as nitrification data for mixed cultures in suspended growth biological wastewater treatment systems. The critical θ_c value for washout of nitrifying microorganisms was found to be approximately 1.5d. The hydrolysis of organically-bound nitrogen was not found to be a rate-limiting step in nitrification. Organic loadings as high as 13.7 gCOD/m²·d were found to have no effect on nitrification at nitrogen and hydraulic loading rates of 1.4 gN/m²·d and 70 l/m²·d, respectively.

The results of the completed research indicate that RBC systems can be effectively utilized for nitrification of domestic wastewaters with and without prior biological treatment. Further research should be conducted on pilot- and full-scale systems to determine the appropriate scale-up parameters.

Saunders, F. Michael, and Rodney L. Pope
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KEYWORDS--*nitrification, *wastewater treatment, *attached biological films, *rotating biological contactor systems, *cell residence time, *nitrifying bacteria, *domestic wastewater, *Nitrosomonas, *Nitrobacter.

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Mr. Rodney L. Pope is an Environmental Engineer with Engineering Science, Inc. and was a Graduate Research Assistant on the project during a portion of the period in which the project was funded.

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INTRODUCTION

With increasing regulation of wastewater discharges to the nation's waters, increasing emphasis has been placed on the oxidation and removal of ammonia-nitrogen in domestic and industrial wastewater effluents. While many wastewater treatment systems can effectively meet these new effluent nitrogen regulations with conventional biological treatment systems, complete and effective ammonia-nitrogen oxidation may not be plausible in many instances. A treatment system which is easily installed, cost-effective and available for immediate addition to such treatment systems is a rotating biological contactor (RBC). These treatment systems can be used effectively for the removal of residual organic matter and the concurrent oxidation of effluent ammonia-nitrogen, i.e. nitrification. The effectiveness of the RBC system for use in wastewater nitrification is due to an enhanced retention of microorganisms within the system as an attached biofilm. This then allows for the retention of the slow-growing nitrifying microorganisms. While design may typically be based on influent wastewater flow and BOD_5 and ammonia-nitrogen loading rates which are normalized to disc surface area, more fundamental design techniques and procedures are needed for RBC systems.

Accordingly, the primary emphasis of this research project was placed on a fundamental investigation of the growth kinetics of nitrifying organisms in a RBC system. The treatment system investigated was one which simulated that to be used for the treatment of a secondary effluent containing both residual organic matter and reduced nitrogen. A primary objective was to establish the growth kinetics of an attached nitrifying population which could ultimately be incorporated into a fundamental design procedure.

LITERATURE REVIEW

Rotating biological contactor (RBC) systems consist of a tank or series of tanks in which discs, mounted vertically and supported on a horizontal shaft, are partially immersed and rotated in the contained wastewater. Some typical characteristics of full-scale RBC systems are presented in Table 1. As the discs are rotated, they are sequentially exposed to the wastewater being treated and atmosphere above the wastewater. Microorganisms naturally contained in the influent wastewater attach to and grow on the submerged portion of the rotating discs. This attached microbial growth, i.e. biofilm, constitutes the biological medium in which biological oxidations occur.

RBC systems have several advantages over other similar biological systems. Power requirements, for example, are extremely low due to the balanced growth of the attached biofilm on the discs and the buoyancy of the plastic discs typically utilized. Power requirements therefore for the RBC process are generally less than 50% of those associated with the activated sludge process (Antonie, 1976). While the power requirements may be much less for a trickling filter system, depending upon the extent of recirculation, RBC systems also have some advantages over the trickling filter process. For example, the headloss through the RBC system is typically less than 0.3m, which is consistently less than the 1.5-3 m headloss through a trickling filter system. In addition, nuisances associated with trickling filters such as filter clogging, Psychoda larvae and flies and objectionable odors are also absent with the RBC system. RBC systems are therefore viable treatment alternatives due generally to their modular design, low power requirements and minimal headloss values.

Attached Microbial Films

While extensive studies have not been performed on the attached biofilms associated with RBC systems, numerous investigators have examined attached

Table 1

Physical Characteristics of Rotating
Biological Contactor Systems

<u>Parameter</u>	<u>Dimension</u>	<u>Reference</u>
Disc Diameter	1 - 4 m	Antonie, 1976; Birks and Hynek, 1971; Crittenden and Wells, 1971; Steels, 1974
Disc Shaft Length	1.8 - 7.6 m	Antonie, 1976; Steels, 1974
Depth of Submergence	40%	Antonie, 1976; Birks and Hynek, 1971
Rotational Speed	0.015 - 0.08 rev/sec	Antonie, 1976; Steels, 1974; Birks and Hynek, 1971
Disc Tip Speed	0.033 m/s	Antonie, 1976
Specific Surface Area	50 - 120 m ² /m ³	Antonie, 1976; Stover and Kincannon, 1976

biological films with various types of attachment media including discs, pipes, drums and fixed plates (e.g. Kornegay and Andrews, 1969; Tomlinson and Snaddon, 1966; Hoehn, 1975; Grieves, 1972; Atkinson et al., 1967). Through the research of these above investigators, biofilms have conceptually been considered to be composed of two separate layers, i.e. an active and an inactive layer. The activity or inactivity of these layers is related specifically to the removal of soluble organic matter (i.e. substrate) from a wastewater. As shown in Figure 1, the active layer constitutes the outer-most portion of the biofilm which is in direct contact with the wastewater liquid. The inactive film portion, if present, constitutes the remaining portion of the biofilm and is attached directly to the disc media.

Sanders (1966) examined the thickness of active layers of biofilms and found that substrate uptake rates increased as the depth of the biofilm increased to a critical film depth. No further increase in substrate uptake occurred as the depth of the biofilm increased beyond the critical depth. The existence of the active layer and critical depth concepts has been confirmed by others including Hoehn (1970), Kornegay and Andrews (1969), Eckenfelder (1961), Wuhrmann (1963), and Tomlinson and Snaddon (1966). Tomlinson and Snaddon (1966) suggested that the active portion of a biofilm was the aerobic region. However, Kornegay and Andrews (1969) indicated that active layer thickness and aerobic film thickness were not necessarily the same and that the active film thickness was a function of the particular substrate being examined. Atkinson and Fowler (1974) further suggested that active film thickness was equal to the depth of penetration of the substrate into the biofilm since substrate uptake rates increased until the film thickness was equal to the critical depth.

Therefore it can be generally stated that the critical depth concept establishes the limits for the active film layer. Furthermore, the remaining

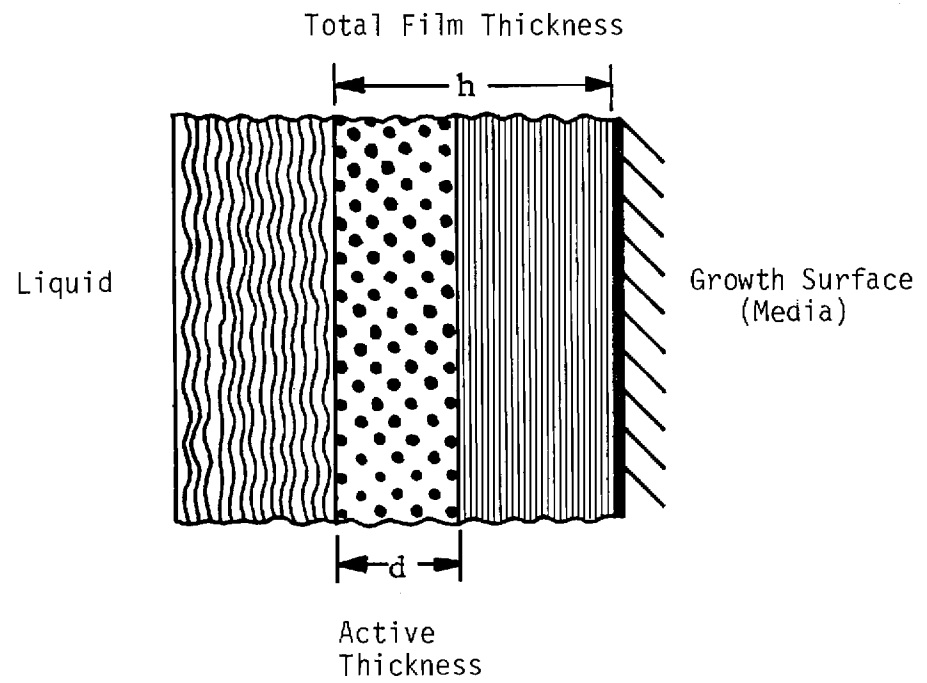


Figure 1. Cross-Section of a Biofilm (Kornegay and Andrews, 1969)

portion of the biomass, while capable of oxidizing substrate, is inactive due primarily to the lack of substrate. The critical depths for a number of biofilms are presented in Table 2. Since critical depth will vary with the nature of the wastewater being treated and the associated microbial population, the variations in critical depth indicated in Table 2 should be considered to be reasonable. It is furthermore apparent from the above information that the critical depth for an aerobic biofilm is relatively small, ranging from 27 to 200 μm . This range of critical depths is an extremely small portion of a total biofilm which may develop to a depth of 1000 μm or more in a biodisc system.

Further examination of the critical depth concept is required when it is considered that microbial metabolism within a biofilm can be controlled by the concentration of an electron donor, an electron acceptor or both at different levels within the biofilm. Howell and Atkinson (1976) presented five conditions controlling microbial kinetics that could occur in a biofilm, including the following: substrate-limitation, oxygen limitation, concurrent oxygen and substrate limitation and limitation by either oxygen or substrate with partial limitation by the other complementary component. The concentration of substrate and oxygen in a biofilm is controlled by two competing mechanisms, i.e. diffusional transport and microbial metabolism. Therefore, the rate of utilization of oxygen or substrate may then be controlled by the rate of transport to the reaction site or by the kinetics of the associated biochemical reactions.

While much information is available on the kinetics of the microbial reactions with both suspended and attached growth systems, the information regarding dissolved oxygen transport and uptake in biofilms is not as well developed. Lee et al., (1976) indicated that substrate oxidation kinetics in biofilms were not dependent upon dissolved oxygen concentration when it

Table 2
Critical Depths of Active Microbial Biofilms

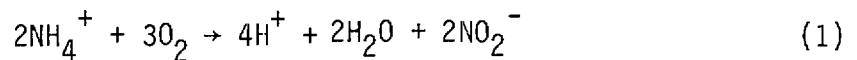
<u>Substrate</u>	<u>Media Supporting Biofilm</u>	<u>Critical depth of active biofilm</u>	<u>Reference</u>
Domestic Wastewater	Inclined rotating tube	120 μm	<u>Water Poll. Res.</u> , 1957
Domestic Wastewater	Inclined rotating tube	200 μm	Tomlinson and Snaddon, 1966
Glucose	Rotating Cylinder	65 μm	Kornegay and Andrews, 1969
Synthetic Wastewater	Rotating Cylinder	150 μm	Hoehn, 1973
Synthetic Wastewater	Culture chamber with an acrylic plate	27 - 62 μm	Sanders, 1966

is above a critical value. Sanders (1966), however, indicated that the transfer of oxygen into the liquid phase of a biofilm may control oxygen availability and therefore the rates of substrate utilization. Furthermore, Tomlinson and Snaddon (1966) concluded that higher oxygen concentrations above a biofilm increased the rate of transport of oxygen into a biofilm and the rate of oxygen utilization. Mehta et al. (1972) developed a performance equation for plastic media trickling filters and confirmed that liquid phase oxygen transport could limit the removal of organic matter with attached biofilms. Williamson and McCarty (1976) developed an extensive model for substrate removal kinetics in a biofilm including parameters for the diffusion of metabolic reactants into a biofilm, utilization of substrate by the microorganisms and diffusion of metabolic waste products through the biofilm into the wastewater. Their extensive model was developed and examined with nitrifying microorganisms with primary emphasis placed on the conversion of a nitrite to nitrate by Nitrobacter species. Although very comprehensive, the model did not allow for a change in the primary limiting reactant within the biofilm, i.e. the biofilm was either substrate-limited or oxygen-limited.

As substrate is continually removed in an attached film system, the biofilm invariably tends to increase in depth. Howell and Atkinson (1976) stated that, as a biofilm increased in thickness, sloughing began to increase due to the deterioration of the adhesive properties of the lower layers. Tomlinson and Snaddon (1966) presumed that sloughing was caused by anaerobic breakdown in the film base. In addition, an increase in biofilm thickness necessarily results in an increase in the mass of the biofilm which would then result in an increased shear being placed on the biofilm. Sloughing would occur when the shearing forces become greater than the adhesive forces. Following sloughing the remaining organisms attached to the film or those organisms suspended in the wastewater media continue to grow and produce a new attached biofilm.

Nitrification Microorganisms

Nitrification is mediated by autotrophic bacteria that use ammonium and nitrite ions as electron donors and oxygen as an electron acceptor. While a wide variety of heterotrophic bacteria as well as autotrophic bacteria are able to oxidize inorganic nitrogen, Painter (1970) concluded that nitrification in wastewater treatment processes is primarily due to the activity of two genera, Nitrosomonas and Nitrobacter. These autotrophic nitrifying bacteria utilize the reducing power of inorganic nitrogen species for energy production and carbon from carbon dioxide or bicarbonate ion for cellular synthesis. The stoichiometric reactions for the complete oxidation of ammonia-nitrogen by Nitrosomonas and Nitrobacter are as follows

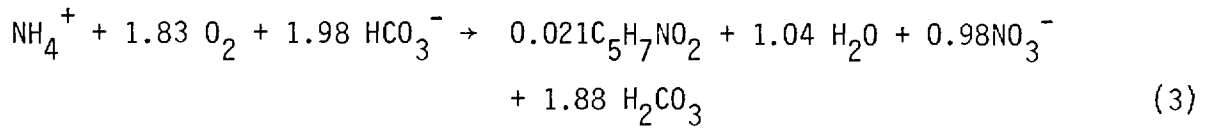


and



Free energy changes reported for the two above equations vary between 58-84 kcal/M and 15.4-20.9 kcal/M, respectively (Baas-Becking and Parks, 1972; Haug and McCarty, 1971; Lees, 1954; Nicholas, 1963; Painter, 1970). In addition, Kluyver and Donker (1926) postulated that two intermediates were involved in the oxidation of the ammonium ion to nitrate ion while Hofman and Lees (1953) reported the first intermediate to be hydroxyl amine (NH_2OH).

Dissolved Oxygen. The respective oxygen requirements for nitrification, as indicated in equations 1 and 2, are 3.43 g $\text{O}_2/\text{g NH}_4^+-\text{N}$ and 1.14 g $\text{O}_2/\text{g NO}_2^--\text{N}$, for a total oxygen requirement of 4.57 g $\text{O}_2/\text{g NH}_4^+-\text{N}$. These values, however, do not consider the utilization of nitrogen by the nitrifying bacteria for cellular synthesis. Gujer and Jenkins (1974) used the following equation



to determine the oxygen requirements for complete nitrification to be 4.19 g O₂/NH₄⁺-N. Using BOD incubation bottles, Wezernek and Gannon (1967) determined the total nitrogen requirement to be 4.33 g O₂/g NH₄⁺-N while Jeffrey and Morgan (1959) determined oxygen requirements to be approximately 4.5 g O₂/g NH₄⁺-N with long term BOD tests.

A critical variable with respect to the growth of nitrifying microorganisms is dissolved oxygen concentration. Nagel and Haworth (1969) found that nitrification increased as dissolved oxygen was increased from 0.25 to 2.0 mg/ℓ. In addition, Downing and Bayley (1961) and Jenkins (1969) reported the limiting dissolved oxygen concentrations to be 0.5 mg/ℓ, while Downing and Scragg (1958) reported the limiting concentration to be 0.3 mg/ℓ. Wuhrmann (1963) noted that nitrification was severely inhibited in high-rate activated sludge pilot plants operated with dissolved oxygen concentrations of 1 mg/ℓ while nitrification was virtually complete at dissolved oxygen concentrations of 4 and 7 mg/ℓ. Downing et al. (1964) indicated that dissolved oxygen concentrations greater than 1 mg/ℓ should not limit the rate of nitrification in activated sludge suspensions. With respect to high dissolved oxygen concentrations, Haug and McCarty (1971) reported that dissolved oxygen concentrations of 60 mg/ℓ did not affect the growth of nitrifying bacteria while Okun (1949) reported that nitrification was not affected by dissolved oxygen concentrations of 33 mg/ℓ.

pH. Numerous factors have been shown to affect the growth of nitrifying bacteria, with pH being a central variable. As typically found with single species or narrow cultures of organisms, a narrow range of pH values is reported for optimum growth with decreased nitrification rates occurring both above and below this range. For example, Hall (1973), Rimer and Woodward (1972) and Wild et al. (1972) reported optimum pH ranges for nitrification

in activated sludge suspensions of 7.0-9.4, 8.4-8.5 and 8.0-8.6, respectively, while Huang and Hopson (1974) indicated the optimum pH range for nitrification in an attached growth system was 8.4-8.8. While the above pH ranges are not mutually exclusive, it has been shown that nitrifying bacteria can acclimate to new pH conditions, after sufficient contact time has been allowed. For example, Haug and McCarty (1971) reported that an attached growth system underwent complete acclimation to a pH of 6.0, from an initial pH of between 7 and 8.5 following a 10 day acclimation period. Furthermore, Stankewich (1972) observed high degrees of nitrification between pH values of 5.8 and 6.0.

As indicated in equation 1, nitrification results in a release of hydrogen ions and a possible decrease in the pH of a wastewater being treated. Since low pH values can significantly affect the rates of nitrification, the buffer capacity of a wastewater is an important variable. From equation 3 it can be determined that the alkalinity requirement for the complete oxidation of ammonium ion to nitrate ion requires 7.01 g alk (as CaCO_3)/g NH_4^+ -N. Other values reported for suspended growth systems include 6.4, 6.0 and 7.1 g alk (as CaCO_3)/g NH_4^+ -N (Mulbarger, 1971; Horstkotte et al., 1974; Newton and Wilson, 1973). Gasser et al. (1974), Osborn (1965) and Haug and McCarty (1971) reported values of 6.5, 7.4 and 7.3 mg alk (as CaCO_3)/mg NH_4^+ -N, respectively, for attached growth cultures.

Temperature. Temperature is an additional variable which may affect the metabolic growth rates of nitrifying organisms. Numerous investigators have reported the optimum temperature for nitrification to be 30°C for both attached and suspended growth systems with decreases in nitrification rates resulting from either an increase or decrease from 30°C. For example, Gibbs (1920) indicated that exposure to 53-55°C for 10 min inactivated Nitrosomonas cultures and an equal exposure to 56-58°C inactivated cultures of Nitrobacter.

Growth Kinetics. While carbon dioxide or bicarbonate ions, ammonium ions and nitrite ions are required by a nitrifying population, the latter two components typically limit the rate of growth of nitrifying microorganisms in wastewater treatment systems. Growth kinetics for nitrifying bacteria have been shown to follow the relationship utilized by Monod (1950) in expressing the effects of substrate concentration on growth rate i.e.,

$$\mu = \hat{\mu} \frac{S}{K_S + S} \quad (4)$$

where μ = net specific growth rate coefficient, $\hat{\mu}$ = maximum net specific growth rate coefficient, S = concentration of limiting substrate and K_S = half-velocity coefficient. The growth constants, $\hat{\mu}$ and K_S , have been shown to be temperature dependent as indicated for Nitrosomonas,

$$\hat{\mu} = \exp(0.0951(T) - 2.174) \quad (5)$$

$$K_S = \exp(0.117(T) - 1.666) \quad (6)$$

and Nitrobacter,

$$\hat{\mu} = \exp(0.0587(T) - 1.133) \quad (7)$$

$$K_S = \exp(0.145(T) - 2.646) \quad (8)$$

where $\hat{\mu} = d^{-1}$, $K_S = \text{mg N/l}$ and $T = ^\circ\text{C}$ (Knowles et al., 1965).

In addition, Poduska (1973) presented a comparison of heterotrophic microbial populations and the autotrophic nitrifying populations using typical values for $\hat{\mu}$, K_S and yield coefficient, Y , as presented in Table 3. From the data in Table 3, it is apparent that nitrifying microorganisms have much slower growth rates than heterotrophic microorganisms, as indicated by

Table 3
Yield and Kinetic Coefficients for
Heterotrophic and Autotrophic Nitrifying Bacteria
(Poduska, 1973)

Parameter	Heterotrophic Bacteria	Autotrophic Bacteria	
		Nitrosomonas	Nitrobacter
Substrate	BOD ₅	NH ₄ ⁺ -N	NO ₂ ⁻ -N
Y(g VSS/g substrate)	0.4 - 0.6*	0.03 - 0.10	0.02 - 0.08
$\hat{\mu}$ (d ⁻¹)	3.6 - 4.8*	1.0 - 1.9	1.0
K _s (mg substrate/l)	100 - 200*	0.18 - 1.0	0.25 - 1.0

*Apparent values for activated sludge systems.

the significantly lower values for $\hat{\mu}$ and the extremely low yield coefficients, Y , especially when based on oxygen equivalents. In addition, the lower K_s values indicate that the nitrifying microorganisms would be growing at or near their maximum growth rates when substrate concentrations were in the concentration range of ≥ 1 mg N/l. These data therefore indicate that nitrifying organisms are slow-growing microorganisms which, when present, are growing at or near their maximum growth rates in the wastewater treatment system.

In addition to the above parameters which determine the growth rate characteristics of nitrifying microorganisms, available data indicate that substrate inhibition is possible if substrate concentrations are raised to significantly high levels. For example, Meyerhoff (1916, 1917) determined that the maximum respiration rates for Nitrosomonas and Nitrobacter occurred at 110 mg NH_4^+ -N/l and 210 mg NO_2^- -N/l, respectively, and that above these concentrations respiration rates decreased. These data, however, are in conflict with those of Engel and Alexander (1958) which reported no inhibition for Nitrosomonas in pure culture at a pH of 8.0 and an ammonia concentration of 640 mg NH_4^+ -N/l.

Antonisen et al. (1976) stated that inhibition of nitrifying microorganisms was directly related to the concentration of free ammonia (NH_3) and nitrous acid (HNO_2). Inhibition by free ammonia occurred at concentrations of 10-150 mg/l and 0.1-1 mg/l, respectively for Nitrosomonas and Nitrobacter. Nitrous acid inhibited only Nitrobacter at concentrations of 0.2-2.8 mg/l. These data indicate furthermore the major impact of pH on ammonia and nitrous acid concentrations and the potential resulting effects on growth rates. Components other than ammonia and nitrous acid have been reported to be inhibitory or toxic to nitrifying organisms as discussed by Pope (1978).

RBC System Performance

An RBC system is an extremely complex biological reactor. The discs with attached biofilm are continuously rotated through a cycle of complete submersion in the liquid wastewater followed by exposure of the biofilm and attached layer of wastewater to the atmosphere. This sequential exposure of the biofilm liquid and gaseous phases presents a large number of parameters which must be evaluated if process performance is to be adequately described.

Numerous kinetic relationships may determine the extent of treatment. For example, the two primary waste components of concern are dissolved oxygen and organic substrate, i.e. the electron acceptor and electron donors, respectively. When considering dissolved oxygen, its transport into the liquid phase takes place at the liquid surface in the reactor tank and at the surface of the liquid film attached to the biofilm during exposure of the biofilm to the atmosphere. While both play an important role in transport, the majority of the oxygen transferred into the system is that transported through the liquid film attached to the biofilm during exposure to the atmosphere. This is due primarily to the much greater air-liquid interface associated with the discs as opposed to the liquid contents of the reactor. Therefore, kinetic processes associated with oxygen transport include transport by diffusion across a liquid film to the surface of a biofilm, transport into a biofilm to the surface of a microbial cell and the subsequent transport across cell membranes and utilization by cells as the final electron acceptor. With respect to substrate, mixing of the bulk liquid must be considered with respect to substrate and oxygen, including the oxygen coming into the liquid phase either through transport directly through the liquid-gas interface or due to that oxygen contained in the liquid film attached to the biofilm as it enters the liquid phase. Substrate must be transported through a liquid boundary layer, associated with the biofilm exposed to the liquid

phase, through the biofilm to the surface of the cell and into a microorganism cell. Finally, the kinetics of biochemical oxidation by microorganisms both with respect to oxygen and substrate must be considered and evaluated. Transport of gaseous and soluble waste products must also be considered since these materials are produced within the depths of the biofilm and must therefore be transported through the cell wall, through the biofilm into the liquid boundary layer and ultimately into the liquid phase for discharge with the effluent or gas-stripping from the reactor.

In considering the above transport and metabolic processes, the ultimate objective is to determine the rate at which substrate (e.g. organic matter or inorganic nitrogen) can be oxidized to achieve effective substrate removal. The kinetic parameters associated with the transport and utilization of both oxygen and substrate can determine the maximum efficiency of the process. Variables which will effect these parameters include disc submergence and spacing, disc rotational speed, wastewater temperature, wastewater pH, the natural ecological changes which occur in biofilm microbial composition, as well as wastewater characteristics and the concentration of organic matter in the liquid phase. Much of the research performed with RBC systems has been with bench- and pilot-scale systems which are not totally indicative of the performance achievable with full-scale systems due to problems associated with the scale-up of small rotating disc systems. While some detailed experimental work has been done under highly controlled and idealized conditions, the vast majority of the information available in the literature has been focused on wastewater treatability with RBC systems and has not been normalized to a specific set of well controlled operational conditions. This information, however, is of use in evaluating overall process performance and indicating key variables which may affect process efficiency. The following is a brief discussion of some of the pertinent parameters which pertain to treatment performance with attached film systems.

Temperature. Temperature is an extremely important parameter in any biological treatment process due to its effect on metabolic rates of microorganisms, substrate and oxygen diffusivity, mass transfer coefficients and saturation concentrations for dissolved oxygen, for example. Furthermore, Ellis and Bananga (1976) reported on the overall performance of an RBC system due to temperature changes. It was found that a change in temperature from 11 to 27°C increased BOD₅ removal rates from 90 to 94% when the average influent BOD₅ was 240 mg/l and hydraulic retention time was 3 h. Antonie (1976) indicated that temperature increases above 13°C, however, did not improve organic removal based on conclusions drawn from numerous studies. In addition, he found that there was no improvement in ammonia oxidation above a temperature of 16°C. Lue-Hing et al. (1976), however, reported that a 10° increase in temperature to 20°C increased ammonia oxidation rates from 250 to 697 g NH₄⁺-N/m³-d at an average influent ammonia concentration of 780 mg/l.

Wastewater pH. The pH value of a wastewater can affect the performance of an RBC system. While pH has rarely been examined as a separate independent variable, numerous authors have observed various pH effects in RBC systems. Hao and Hendricks (1975) observed a simultaneous increase in pH and decrease in wastewater alkalinity. Birks and Hynek (1971) reported an increase in pH of 6.8 to 7.8 through a four-stage RBC system while Hudson et al. (1976) indicated that pH changed from 5.2 to 7.7 through a two-stage RBC system. The apparent justification for the above system response is the stripping of carbon dioxide (CO₂) from the wastewater by disc rotation. The removal of CO₂ would shift carbonate equilibrium, increase pH and thereby decrease carbonate alkalinity. Therefore the pH changes in a RBC system can be significantly affected by those parameters controlling the removal of CO₂ as well as those controlling the production of CO₂ and H⁺ ions. Disc rotational speeds furthermore affect the degree of contact between the biomass and the waste-

water and also the rate of oxygen transfer. Increased rotational speeds have been shown by numerous authors to enhance the removal of organic matter (Welsh, 1969; Welsh, 1968; Ellis and Banaga, 1976; Chittenden and Wells, 1971). Antonie (1976) furthermore, stated that a rotational tip speed of approximately 0.3 m/sec was the optimal peripheral velocity for an RBC system treating a domestic waste.

Dissolved Oxygen. The availability of dissolved oxygen may determine the maximum capability of a particular RBC system to remove organic matter. Therefore RBC systems operating at elevated temperatures with high strength wastes may generally be controlled by oxygen transfer rates due to the associated higher metabolic rates and lower dissolved solubility. Bintanja et al. (1975) indicated that enhancing the oxygen supply through the use of an oxygen-rich atmosphere resulted in increased removal of organic matter, decreased sludge production and an improved settleability of the sloughed biomass. Welsh (1968) indicated that the removal of organic matter (measured as COD) increased with increasing dissolved oxygen in the mixed liquor up to a concentration of 1.5 mg/l and then remained stable as dissolved oxygen was increased. Torpey et al. (1972) also observed increased organic removals in the RBC process using an oxygen-enriched atmosphere. Tomlinson and Snaddon (1966) concluded that the use of an oxygen-enriched atmosphere above a biofilm increased the rate of diffusion of oxygen into the film and enhanced organic removal rates.

Hydraulic Loading. A variable response has been indicated for the accumulation of biomass on disc surfaces through bench-scale RBC systems. Pretorius (1971) indicated that higher biomass accumulations occurred on the initial discs on a multistage system and decreased in subsequent stages. Hudson et al. (1976) observed an increase in fixed biomass as hydraulic retention time was decreased, reaching a maximum at a hydraulic retention time of 2 h and remaining constant with subsequent increases.

Many researchers have observed the effects of hydraulic loading on the performance of an RBC system. Removal of organic material was reported to increase with increased hydraulic retention time by numerous investigators including Antonie, 1970; Chittenden and Wells, 1971; Ellis and Banaga 1976; Hudson et al. 1976; and Welsh, 1968. Birks and Hynek (1971) and Welsh (1968) each observed a decrease in the performance of a RBC as influent COD concentrations were increased under similar operating conditions. Ellis and Banaga (1976) concluded that influent organic concentration and hydraulic retention time were not independent variables and used this pair of parameters, as an organic loading term, to evaluate process performance. As organic loading increased, removal percentages decreased slightly until the system reached its maximum hydraulic loading and increased removals were not observed (Stover and Kincannon, 1976; Welsh, 1968).

Summary

Considerable experimental research has been performed with attached films and RBC systems. However, an extremely limited amount of experimental research has been performed on the determination of the kinetic relationships for the oxidation of substrates and biomass production. Furthermore, while considerable progress has been made through the use of cell residence time concepts in the design, evaluation and operation of suspended growth systems, this concept has seen little application with attached growth systems. This research project was therefore undertaken as an initial step in the direct application of microbial growth kinetics to the design evaluation and operation of attached growth systems.

RESEARCH OBJECTIVES

The primary objective of this research project was to examine the rotating biological contactor system and determine the effect of microorganism growth rate on the nitrification process as mediated by an attached biofilm. Initially, the objective was to determine the critical net specific growth rate, or cell residence time, at which washout of nitrifying organisms occurred and to determine the extent of the nitrification at higher values of cell residence time. Subsequent to this initial phase of the experimental study, a second phase was initiated to determine the effect of an organically-bound nitrogen source on nitrification. The primary purpose was to determine if hydrolysis of organically-bound nitrogen was a rate-limiting reaction and if it had any significant effect on the nitrification process.

The final phase of the research project was an investigation of the effect of influent organic strength on the nitrification process. This phase was performed to determine the sensitivity of the organisms to shock loads of organic matter which may be experienced in wastewater treatment practice.

Therefore using cell residence time as a primary control mechanism on microorganism growth rate, this research project examined the effect of microorganism growth rate on nitrification and the extent to which hydrolysis of organically-bound nitrogen and increased organic loadings affected nitrification. The primary thrust of the research was oriented towards the treatment of a secondary effluent with the requirements that complete nitrification and additional organic removal be achieved.

EXPERIMENTAL SYSTEMS AND PROCEDURES

Experimental RBC System

The bench-scale RBC unit used in this study is shown schematically in Figures 2 and 3. The physical characteristics of the reactor system are presented in Table 4 and in greater detail by Pope (1978) and Cruz (1977). The top of the reactor was covered with a cylindrical lid to minimize evaporative losses. The contents of the RBC system were pumped continuously through an external loop which contained a heat exchanger and a series of two sieves (i.e. U.S. No. 20 and 80 sieves) to remove sloughed biomass contained in the recycle flow. The recycle flow rate was maintained at 1 l/min to maintain a reactor turnover time of approximately 6 min.

The effluent from the reactor system was controlled with a effluent overflow device from which composite samples were taken at 15 min intervals with an automatic sampler. The composite sample reservoir was located in a refrigerator.

Substrate

A synthetic wastewater was utilized to simulate the soluble portion of a typical secondary effluent. Based on data obtained from Mueller et al. (1958), Painter et al. (1961), Painter (1971) and Rebhum and Manka (1971) and others, representative organic and inorganic compounds were selected, as presented in Tables 5 and 6. The inorganic and organic substrates were prepared in large concentrated solutions and stored to assure the continuous consistency of the synthetic wastewater and to minimize the pumping of large volumes. The COD of the concentrated organic substrate was typically 45 g/l which was then diluted prior to introduction into the reactor system.

Three liquid flows were pumped into the RBC reactor system through a

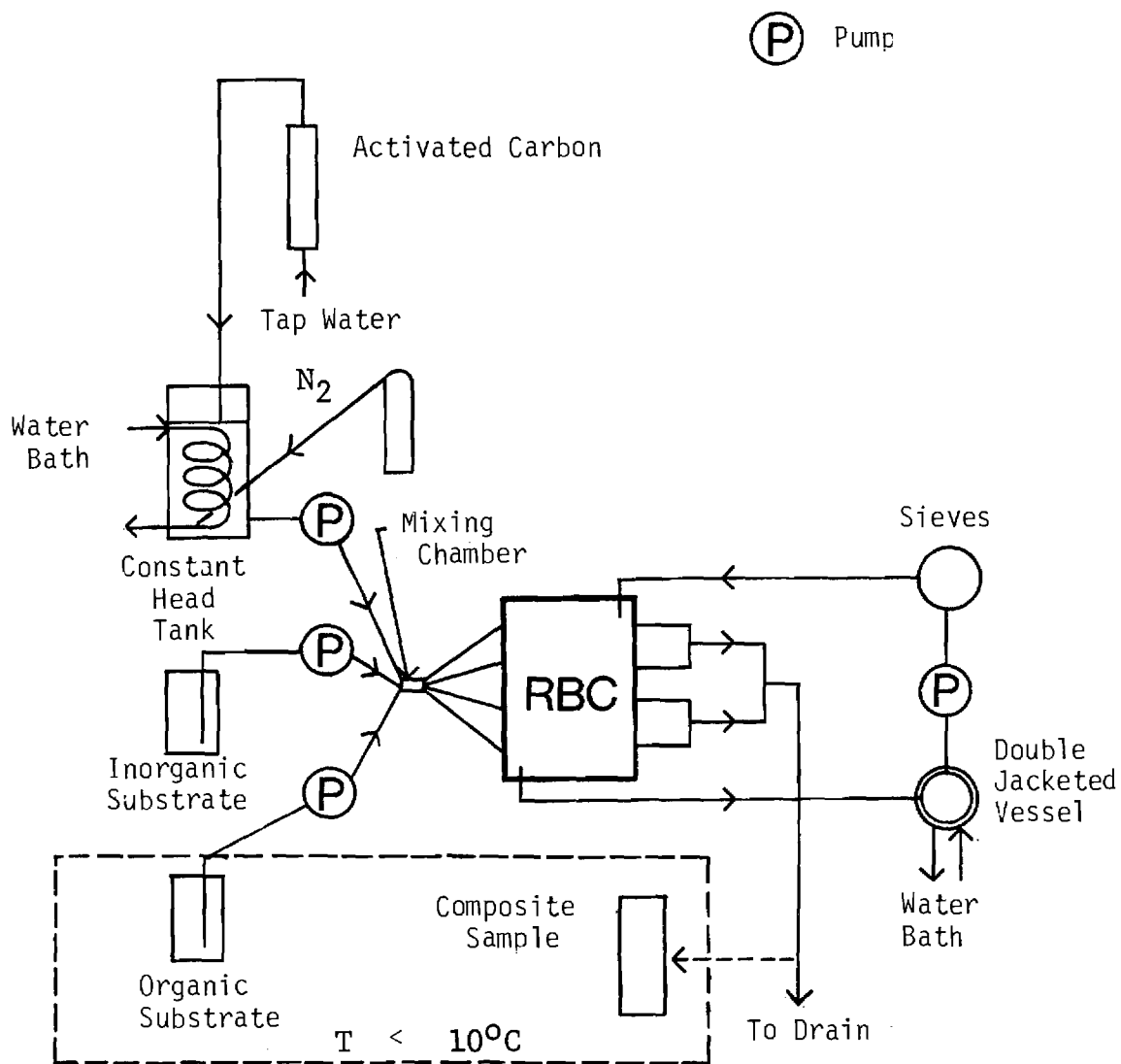


Figure 2. Schematic Diagram of the RBC System

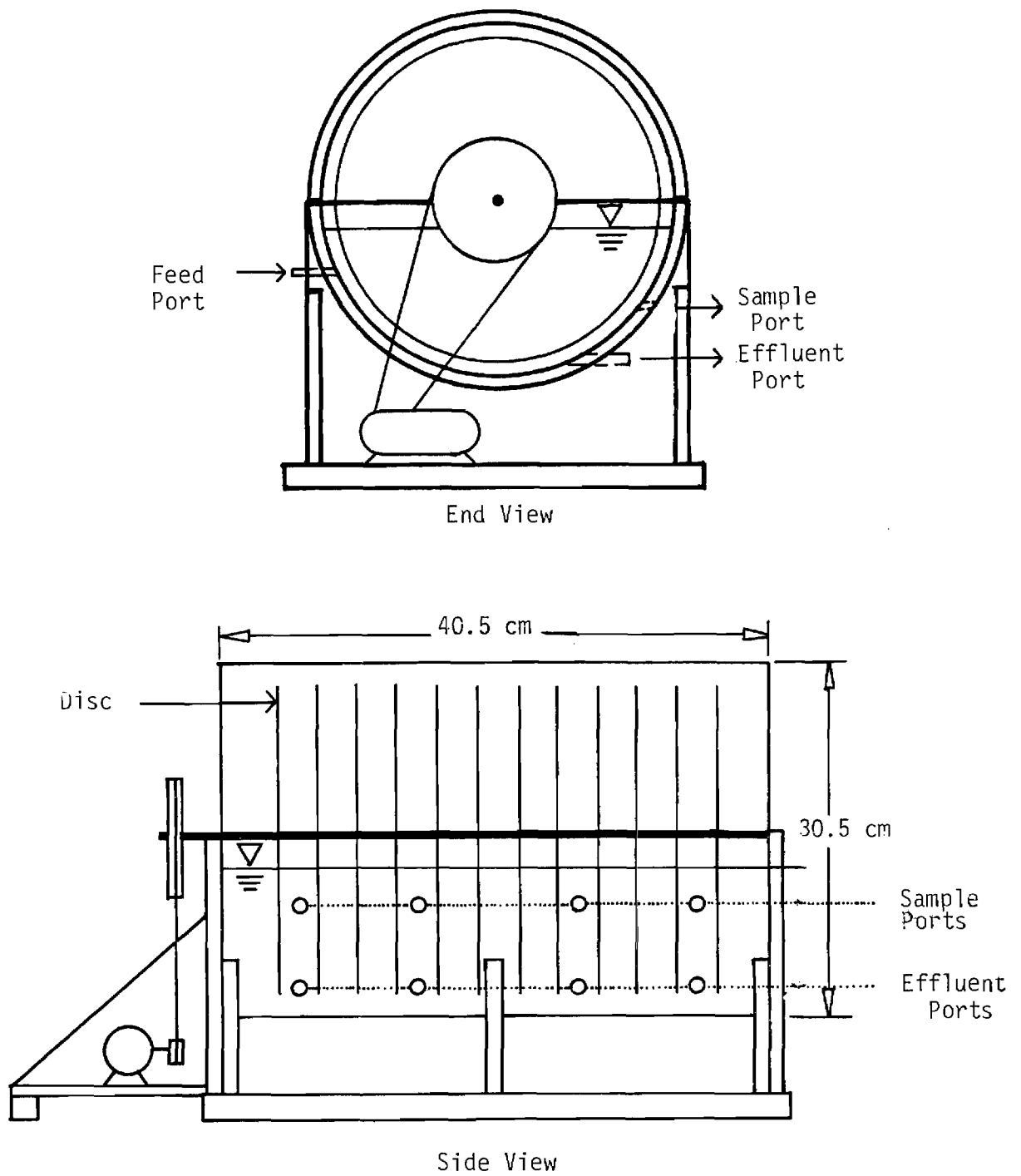


Figure 3. Schematic Diagram of the Laboratory-Scale Rotating Biological Contactor

Table 4

Characteristics of Experimental RBC Reactor

<u>Parameter</u>	<u>Quantity</u>
Disc diameter	25.4 cm
Disc thickness	0.32 cm
Disc Spacing	2.5 cm
Wetted Disc Surface Area	1.0 m ²
Disc Submergence	7.6 cm
Hydraulic Retention Time	2. h
Hydraulic Loading	72 l/d · m ²
Disc Rotational Speed	0.5 rev/s

Table 5

Composition of Synthetic Organic Substrate

<u>Compound</u>	<u>Mass Fraction</u>
Acetic Acid	1.7
Arabinose	10.3
Benzoic Acid	1.4
Butyric Acid	1.7
Citric Acid	4.6
Formic Acid	3.2
Galactose	10.3
Lactic Acid	3.2
Phenol	0.2
Propionic Acid	1.8
Sucrose	26.0
Valeric Acid	1.7
Xylose	10.3
Instant Tea*	22.6
FeCl ₃ · 6H ₂ O	1.0

*Nestea, Nestle Co., Inc., New York

Table 6
Composition of Synthetic Inorganic Substrate

<u>Compound</u>	<u>Mass Fraction</u>
$\text{MnSO}_4 \cdot \text{H}_2\text{O}$	0.07
ZnCl_2	0.13
$\text{CuSO}_4 \cdot 5\text{H}_2\text{O}$	0.04
$\text{CoCl}_2 \cdot 6\text{H}_2\text{O}$	0.05
$\text{Na}_2\text{B}_4\text{O}_7 \cdot 10\text{H}_2\text{O}$	1.16
KH_2PO_4	1.12
K_2HPO_4	2.87
$\text{Na}_2\text{HPO}_4 \cdot 7\text{H}_2\text{O}$	4.40
$(\text{NH}_4)_2\text{SO}_4$	12.15
NH_4Cl	0.22
CaCl_2	0.66
MgSO_4	0.66
NaHCO_3	22.15
Na_2CO_3	27.95
Na_2SO_4	26.37

small mixing chamber. The flow from this mixing chamber constituted the total synthetic influent wastewater. The major influent flow to the substrate mixing chamber was a 40 mL/min flow of deoxygenated tap water pumped from a constant head reservoir. The reservoir contained a heat exchanger to maintain an even influent temperature. The reservoir was also continuously agitated by the addition of nitrogen gas through a gas dispersion tube to maintain low influent dissolved oxygen concentrations, e.g. ≤ 3 mg/L. Dissolved oxygen was stripped from the dilution water influent to assure that the influent to the reactor was low in dissolved oxygen and typical of a secondary effluent. A second influent flow contained the synthetic organic substrate which was pumped from a reservoir maintained at a temperature of $<10^{\circ}\text{C}$. This flow was set at a constant rate of 5 mL/min throughout all experiments. Changes in the net concentration of the influent organic wastewater were achieved by altering the COD concentration in the organic reservoir. The third and final flow into the mixing chamber was the inorganic substrate. This fraction was maintained at room temperature and pumped at a constant rate of 5 mL/min into the mixing chamber to give a total chamber effluent flow of 50 mL/min. The COD concentration in the RBC influent flow was approximately 60 mg/L in the earlier phases of the study and was increased in the subsequent phases as indicated later. The nitrogen concentration in the RBC influent flow was maintained at approximately 20 mg-N/L. The influent COD/N ratio ranged from 3.03 to 14.3 while the influent COD/P ratio ranged from 6.25 to 33.3 in the eight phase study.

RBC Operation and Monitoring

The single stage RBC system used in this study employed recycle of the reactor contents to maintain a stable reactor temperature and to minimize the accumulation of sloughed biomass within the reactor.

Temperature was controlled at 21-25°C with maximum fluctuations of $\pm 1^\circ\text{C}$ during one operational phase. The pH of the mixed liquor was controlled primarily by adjusting the pH in the organic and inorganic substrates to achieve a mixed liquor pH of 7.2. The alkalinity of the wastewater was approximately 27 mg CaCO_3/ℓ .

Since the objective of the study was to evaluate the effects of microbial growth rate on nitrification, all reactor operational variables were maintained at constant levels. Disc rotational speeds were held constant at 0.5 rev/s with a peripheral disc velocity of 0.4 m/sec. An examination of the RBC system with respect to oxygen transfer indicated that the mass transfer coefficient, K_L , was equal to 1.1×10^{-5} m/sec. This was determined with the RBC system at a temperature of 20°C using tap water with discs containing no attached biofilm. The maximum transfer capabilities of the system therefore were equal to 8.65 g O_2/d (i.e. $T = 20^\circ\text{C}$, $\text{DO}_{\text{satn.}} = 9.0$ mg/ ℓ , wetted disc area = 1 m^2). This value for oxygen transfer was equivalent to approximately 85% of the total waste strength (both COD and NOD) being applied to the system in the initial phases and was considered to be satisfactory. The hydraulic retention time for the reactor was maintained constant at 2 h with a constant influent flow of 50 mL/min. This resulted in a constant hydraulic loading of approximately 71 $\ell/\text{m}^2 \cdot \text{d}$.

Influent nitrogen concentrations were maintained at approximately 20 mg/ ℓ throughout the study. Since the hydraulic loading for the reactor system was constant, the nitrogen loading was therefore maintained at approximately 1.4 g-N/ $\text{m}^2 \cdot \text{d}$. The organic loading in the reactor system was not constant through all phases and ranged from approximately 60 to 190 mg/ ℓ . The organic loading therefore varied from 4.0 g-COD/ $\text{m}^2 \cdot \text{d}$ to 13.7 g-COD/ $\text{m}^2 \cdot \text{d}$. The composition of the wastewater remained constant throughout the duration of the study with one exception. The nitrogen source was normally ammonia-nitrogen except during

one experimental phase where an amino acid, glycine, was supplied as the sole nitrogen source. The influent nitrogen concentration however was always adjusted to approximately 19 mg-N/l.

The growth rate of the attached microorganisms was the sole remaining variable for the reactor system. Two basic approaches were used in controlling microbial growth. In the first experimental phase, the biofilm was not disturbed and the system was allowed to achieve a steady state of operation. Waste solids which accumulated within the reactor system were removed with the effluent flow and through periodic removal of solids contained within the reactor system. In the remaining experimental phases, the growth rate of the attached microorganisms was controlled using a solids removal technique in which attached microbial films were scraped directly from disc surfaces as indicated later.

Analysis of influent reactor samples were performed with samples withdrawn from the respective substrate reservoirs and analyzed typically for COD, ammonia-nitrogen, organic-nitrogen and pH. Concurrently, 24 h effluent composite samples, which were maintained under refrigeration, were analyzed for pH, COD, ammonia-nitrogen, nitrate-nitrogen, nitrite-nitrogen, total suspended solids (TSS) and volatile suspended solids (VSS).

Influent wastewater flow rates for the organic and inorganic substrates were determined by weighing the substrate reservoirs daily and converting the change in mass to a volumetric flow rate. In addition, the two pumps were monitored and calibrated concurrently. Influent dilution water flows were measured using a graduated cylinder and stopwatch.

Control of Cell Residence Time

The cell residence time of the RBC system was controlled by scraping portions of attached biofilm from the discs with concurrent removal of sloughed biomass. Since the actual cell residence time was a function of

the total mass of solids within the system normalized by the rate at which solids were wasted from the system, a fixed value for cell residence time could not be established prior to an experimental phase. Therefore, an operational cell residence time, θ_{op} , was developed and used. This θ_{op} parameter, however, was used primarily for communicative purposes and to establish the frequency of scraping cycles.

To control the RBC system at a specified θ_{op} value, the following procedures were utilized. The 12 discs of the RBC system were divided into four sets containing three discs each. Each side of a disc within a set was assigned a number from 1 to 6. All disc sets being numbered in a similar ordered manner as shown in Figure 4. θ_{op} values of 6, 12 and 18 d were evaluated during the course of the studies reported herein. For a θ_{op} value of 6 d, a schedule was established in which each numbered disc was scraped once during a 6 d cycle. That is, the four disc sides numbered 1 (i.e. one disc in each of four sets) were each scraped of total attached biomass with a sharp-edged scraper. The biomass solids were collected and held separately for later analysis. On the second day of a $\theta_{op} = 6$ d cycle, all disc sides numbered 2 were scraped. This schedule was followed until on the seventh day the cycle was repeated with disc sides numbered 1 being scraped again. The procedure was very similar for θ_{op} values of 12 and 18 d with the exception that the interval between scrapings were 2 and 3 d, respectively.

In addition to scraping attached biomass from disc surfaces, sloughed biomass was routinely collected in one of three ways. The biomass which was removed from the system in the effluent was collected in an effluent composite, which was routinely monitored for suspended solids. Due to the size of the sloughed biomass, a portion of it was, however, not typically removed from the RBC system in the effluent. Therefore sloughed biomass was also removed from the mixed liquor on two sieves (U.S. No. 20 and 80) placed in series in the mixed

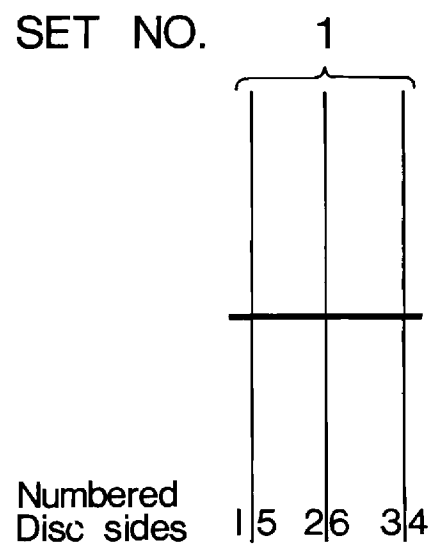


Figure 4. Schematic diagram of one of four sets of RBC discs

liquor recycle system. The accumulated biomass on these sieves was removed as frequently as disc surfaces were scraped, i.e. every 1, 2 or 3 d, and analyzed for total and volatile suspended solids. Finally, those sloughed solids not removed in the effluent or in the recycle system were collected manually by passing the entire mixed liquor volume through a U.S. No. 100 sieve. This was performed at the time of disc scraping by completely removing the mixed liquor from the RBC system and passing it through the sieve upon the return of the mixed liquor to the reactor. The scraping of discs, collection and sieving of the mixed liquor and removal of collected biomass on the sieves in the recycle system were performed within a 1 h period with the RBC system out of operation for only a 15-30 min time interval.

Analytical Methods

Analytical methods and procedures described in Standard Methods (1975) were generally followed, when possible, in the monitoring of wastewater, biomass and mixed liquor characteristics.

Chemical Oxygen Demand. Organic matter concentrations in wastewater samples were measured by the Chemical Oxygen Demand (COD) test as described in section 508 of Standard Methods (1975). A micro-COD procedure using 0.025 N $K_2Cr_2O_7$ was typically used for all effluent samples and all samples were monitored in duplicate. The precision of the test in the range of 10 - 50 mg COD/l was characterized by a coefficient of variation of 2 - 5%, (Pope, 1978) which was well within acceptable limits.

Nitrogen Species. Ammonia nitrogen was determined using an Orion Model 95-10 ammonium ion electrode coupled with a Leeds and Northrup pH/mv meter and a strip chart recorder (Cruz, 1977; Pope, 1978). A standard addition procedure was used to determine all ammonia nitrogen concentrations in duplicate. The details of the procedure are presented elsewhere by Cruz (1977) and Pope (1978). Effluent nitrite nitrogen concentrations were

determined with the diazotization test (Section 420, Standard Methods, 1975) on duplicate filtered composite samples. Effluent nitrate nitrogen concentrations were determined using the chromotrophic acid method (Section 419E, Standard Methods, 1975). Total Kjeldahl Nitrogen (TKN) was determined with an automated method (i.e. Industrial Method 28-69A, Technicon Corp., New York, N.Y.) on samples containing concentrated biomass fractions and on influent and effluent wastewater samples. The specific procedures utilized are outlined by Cruz (1977) and Pope (1978).

Suspended Solids. Suspended solids concentrations were determined with Gooch crucibles containing glass fiber filters as described in Section 208D of Standard Methods (1975). Total and volatile suspended solids analyses were performed on the effluent composite, solids scraped from the discs, solids retained on the recycle sieves and mixed liquor solids. A volume of 500 ml was filtered for analysis of effluent composite samples. Other solid samples were placed in a container upon collection, brought to a volume of 200 ml and mixed well. A volume of 10 ml of each solids fraction was then filtered. All samples were analyzed in duplicate.

EXPERIMENTAL RESULTS

Experimental Format. Project experimentation was subdivided into three phases. The primary objective for Phase A was to determine the effect of cell residence time on the growth rate of nitrifying organisms and the extent to which nitrification could be achieved. Four experimental runs were performed at four cell residence time values to determine the effect on reactor performance. The four experimental runs were performed with average influent ammonia nitrogen concentrations of 15.6-19.8 mg/l and influent COD concentrations of 49-62 mg/l. Experimental run A1 was performed with no disc scraping, while experimental runs A2, A3 and A4 were performed at θ_{op} values of 6, 12 and 18 d. Influent wastewater characteristics and θ_{op} values for the experimental runs in Phase A, as well as Phases B and C, are presented in Table 7.

Phase B was focused on the determination of the effect of an organically-bound nitrogen source on nitrification. Glycine, an amino acid, was chosen as the nitrogen source since it was typical of some of the proteinaceous matter which may be contained in secondary effluents. Influent nitrogen and COD concentrations were 20.3 mg/l and 84 mg/l, respectively, with the higher COD concentration being attributable to the glycine in the influent wastewater. The θ_{op} value utilized during this experimental phase was 18 d.

The extent to which organic strength of the influent waste affected the nitrification process was examined in Phase C. The influent ammonia-nitrogen concentrations were maintained at 18.0-20.5 mg/l. Influent COD concentration during steady state and transitory periods varied from 90 to 190 mg/l. The θ_{op} value for this phase was 12 d, a value previously justified in Phase A.

Initial System Start-Up. The four initial months of the project were devoted to construction of the reactor system and establishing the operational procedures that were to be employed during the remainder of the study. A

Table 7

RBC Operating Conditions for
Experiments in Phases A, B and C

PHASE	Experimental Run	INFLUENT CONDITIONS			θ_{op} (d)
		Nitrogen Source	Nitrogen Conc. (mg-N/l)	COD Conc. (mg/l)	
A	1	Ammonia	19.6	49	(no disc scraping)
A	2	Ammonia	17.5	54	6
A	3	Ammonia	19.8	59	12
A	4	Ammonia	15.6	62	18
B	1	Glycine	23.1	84	18
C	1	Ammonia	19.2	90	12
C	2	Ammonia	20.5	136	12
C	3	Ammonia	20.2	172	12
C	4	Ammonia	18.0	190	12

pH monitoring and control system was also placed in the reactor during this initial period to maintain pH at a constant level. As indicated later, the operation of this unit was not sufficient to adequately control pH and pH adjustments were subsequently made manually in the influent wastewater flows. During the following 7 month time period, the reactor system was operated using numerous operational parameters to establish techniques for sampling of the effluent and mixed liquor, removal of attached biomass from disc surfaces and suspended biomass in the mixed liquor. In addition, an oxygen stripping system was installed in the influent dilution water to maintain the influent dissolved oxygen concentration ≤ 3 mg/l.

General Operating Conditions and Results

Following the initial start-up period, the reactor was placed in continuous operation for the following 13 month period. The dates and periods of operation for each of the three phases are indicated in Table 8. Except for the terminal portions of Phase C (i.e. Phases C3 and C4), the general procedure was to operate the RBC system for a period of time equal to at least $3(\theta_{op})$ until a steady state of operation was achieved. During each phase, the influent flow to the RBC system was monitored and effluent composite samples were taken daily.

During Run A1 no biomass was scraped from disc surfaces and the unit was allowed to reach a state of equilibrium characteristic of the reactor system and attached biofilm. During subsequent phases, a solids inventory procedure was performed with a frequency equal to $\theta_{op}/6$. This solids inventory procedure constituted the following steps:

1. The discs with attached biomass were removed from the reactor by removing the supporting shaft. The appropriately numbered disc sides were then scraped completely of attached biomass. The biomass removed

Table 8

Operational Parameters and Loading Rates for the RBC System

Run	Phase A				Phase B		Phase C		
	1	2	3	4	1	1	2	3	4
Period of Operation (d)	80	38	38	49	59	36	13	13	11
Dates of Operation (year:day no.)	76:250- 329	77:18- 55	77:61- 98	77:100- 148	77:149- 207	77:208- 243	77:244- 256	77:257- 269	77:270- 280
θ_{op} (d)	*	6	12	18	18	12	12	12	12
Temperature (°C)	24	21	25	23	25	24	24	23	21
pH	7.3	7.3	6.8	7.3	7.3	7.3	7.3	7.3	7.2
Dissolved Oxygen (mg/l)									
Influent	3.1	3.4	2.4	2.6	2.2	2.2	1.5	3.1	3.3
Mixed Liquor	6.0	8.4	6.4	6.0	6.6	5.6	5.2	4.2	4.8
Hydraulic Retention Time (h)	2.1	2.0	1.9	2.0	2.0	2.0	2.0	2.0	2.0
Hydraulic Loading (l/d·m ²)	68	71	73	71	70	71	70	70	72
Organic Loading (g-COD/d·m ²)	4.1	4.0	4.6	4.4	5.9	6.4	9.5	12.0	13.7
Nitrogen Loading (g-N/d·m ²)	1.33	1.24	1.45	1.11	1.62	1.36	1.44	1.41	1.30

*Scraping of biomass from disc surface was not performed during Phase A1.

from the disc surfaces was then collected in a plastic container and stored for analysis. The interval between disc scrapings for the 6, 12 and 18 d values for θ_{op} were 1, 2 and 3 d, respectively. Therefore each disc side in each set of three discs was scraped once every θ_{op} cycle. The rate of removal of biomass solids from disc surfaces was expressed as an average wastage rate (r_d) over the scraping interval ($\theta_{op}/6$) with units of g/d.

2. The mixed liquor contained within the RBC reactor was pumped into a carboy. The contents were then passed through a U.S. No. 100 sieve to remove any suspended matter contained within the mixed liquor phase. This solids fraction obtained from the mixed liquor was referred to as r_{ml} (g/d). Following placement of the mixed liquor in the RBC system, the shaft with attached discs was returned and the system placed in operation.
3. The pump in the mixed liquor recycle system was inactivated and the solids collected on the two sieves (U.S. No. 20 and 80) were flushed into a collection vessel. This solids fraction, r_r (g/d), was then stored for subsequent analysis. Following cleaning of the sieves and retention of the solids, the sieves were returned to the recycle system and the recycle pump was again activated.

In addition to the three above fractions, r_d , r_{ml} , r_r , a sample of the effluent was monitored for suspended matter and was referred to as r_e (g/d). This routine monitoring of the solids within the reactor system allowed for a constant examination of the rate of production of solids within the reactor system. The solids fractions were examined individually to determine the mass of both total and volatile suspended solids collected. The solids data were then subsequently analyzed to determine the average cell residence time for each phase of reactor operation. In addition, the solids were examined for nitrogen content to determine the portion of nitrogen leaving the reactor

system in the biomass produced within the RBC system.

The temperature of the RBC system was maintained at relatively constant levels using a heat exchanger system included in the recycle system. As indicated in Table 8, the temperature ranged from 21 to 25°C. Generally the temperature within any given operational phase varied by a maximum of $\pm 1^\circ\text{C}$. The reactor pH was generally equal to 7.2-7.3 except during Run A3 when an average value of 6.8 was achieved.

The mixed liquor dissolved oxygen concentration was monitored during all phases to assure that dissolved oxygen was available to the attached microbial film. In addition, an effort was made to assure that the influent dissolved oxygen was well below saturation levels. As seen from the data in Table 8, the average influent dissolved oxygen concentrations during Runs A1, A2, C3 and C4 were maintained at approximately 3.2 mg/l while influent dissolved oxygen concentrations for the remaining phases were maintained between 1.4 and 2.2 mg/l. Variations in these controlled influent values were due primarily to malfunction of the nitrogen regulator in the dilution water reservoir and the lack of sufficient quantities of nitrogen gas. Influent dissolved oxygen concentrations, however, were maintained at levels near those typical of activated sludge systems using the pure oxygen process or those with efficient aeration systems. Furthermore, the dissolved oxygen concentration in the mixed liquor was at all times greater than the influent dissolved oxygen concentration. This further indicated that the oxygen transfer rate of the RBC system was, in fact, greater than the rate at which oxygen was being utilized by the attached microorganisms.

The process loading factors for the RBC system are also presented in Table 8. The organic and nitrogen loadings were calculated using the average hydraulic retention time data and submerged disc surface area of 1.0 m². A constant retention time, 2.0 ± 0.1 h, was maintained throughout each phase.

The hydraulic loading was therefore constant at values of 68-73 $\ell/d \cdot m^2$. The nitrogen loading was also constant at 1.11 to 1.62 $g N/d \cdot m^2$ since the influent nitrogen concentration was constant. The organic loading during Phase A was relatively constant ranging from 4.0 to 4.6 $g COD/d \cdot m^2$. Due to the additional input of organic matter in the form of glycine, the organic loading during Phase B was 5.9 $g COD/d \cdot m^2$. Finally, organic loading was the primary variable in Phase C and was varied from 6.4 to 13.7 $g COD/d \cdot m^2$ during the four runs of this phase.

Phase A Results

This phase consisted of four experimental runs designed to determine the effects of cell residence time on the performance of an RBC system. Primary emphasis was placed on rates of solids production, the removal of organic matter and the extent of effluent nitrification. The four experimental runs consisted of one run with no controlled wastage of biomass and three runs operated at θ_{op} values of 6, 12 and 18 d.

Experimental Run A1. The biomass attached to the discs was not intentionally scraped or wasted from the reactor system. Solids collected in the mixed liquor recycle system and those suspended within the reactor were periodically removed as indicated previously. Effluent suspended solids concentrations were also routinely determined for the reactor effluent.

The performance of the RBC system with respect to organic matter was excellent as shown in Figure 5. Effluent COD concentrations ranged from 2-16 mg/ℓ with an average effluent COD concentration of 7 mg/ℓ . Furthermore, also as indicated in Figure 5, a slight increase in influent COD concentration from approximately 45 mg/ℓ to 60 mg/ℓ resulted in no significant increase in effluent COD concentration. Sufficient biomass was therefore contained with the RBC system to sustain the increased COD loading. Data

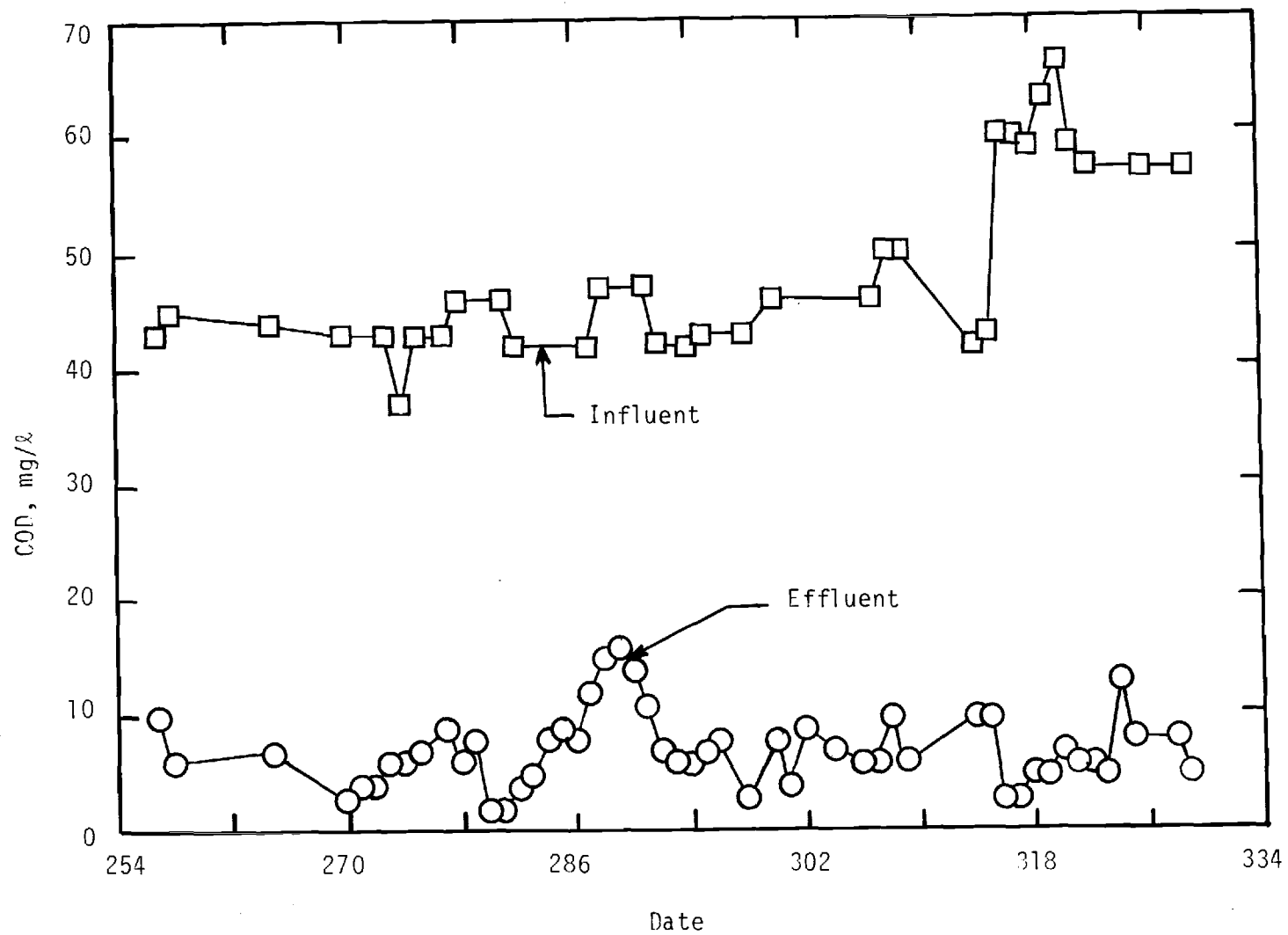


Figure 5. Soluble COD Concentrations in the Influent and Effluent Wastewaters During Experimental Run A1

presented in Figure 6 indicate that ammonia removal was quite effective with effluent ammonia concentrations typically less than 0.1 mg/l at influent concentrations of 11-26 mg NH_4^+ -N/l. While only limited data are available for effluent nitrate concentrations on days 315-321, it appeared that the removal of ammonia-nitrogen was almost totally the result of nitrification.

Experimental Run A2. This and all subsequent experimental runs were performed using the above mentioned disc scraping procedures to control cell residence time. Due to the nature of the biomass as accumulated during Run A1 and its associated sloughing rates, it was expected that the biomass sloughing rates would be erratic. As evidenced by the random nature of the solids wastage data presented in Figure 7, biomass sloughing was not consistent, even with the daily scraping cycle associated with the θ_{op} value of 6 d used during Run A2. The wide variation in the daily wastage data required that an extended testing period be employed to allow the system to attain a steady state level. Accordingly, this run was continued for approximately 38 d.

A severe system disturbance was experienced during the middle of this Experimental Run. On day 34 the automatic pH titration system, which was being used to control mixed liquor pH, malfunctioned and approximately 0.7 equiv. of 0.4 N H_2SO_4 were discharged into the mixed liquor over a short time interval. When this equipment failure was detected 2 h later, the pH of the mixed liquor was pH = 2.0. The mixed liquor pH was immediately adjusted to pH = 7 with sodium hydroxide and the system was placed back in continuous operation. As shown in Figure 7, the biomass wasted on the previous day had been at a low point, indicating an apparent internal accumulation, or reduced production, of biomass. Following day 34, the biomass wastage rate generally increased over the remaining period of the run but did not show a drastic change as a result of this severe pH perturbation.

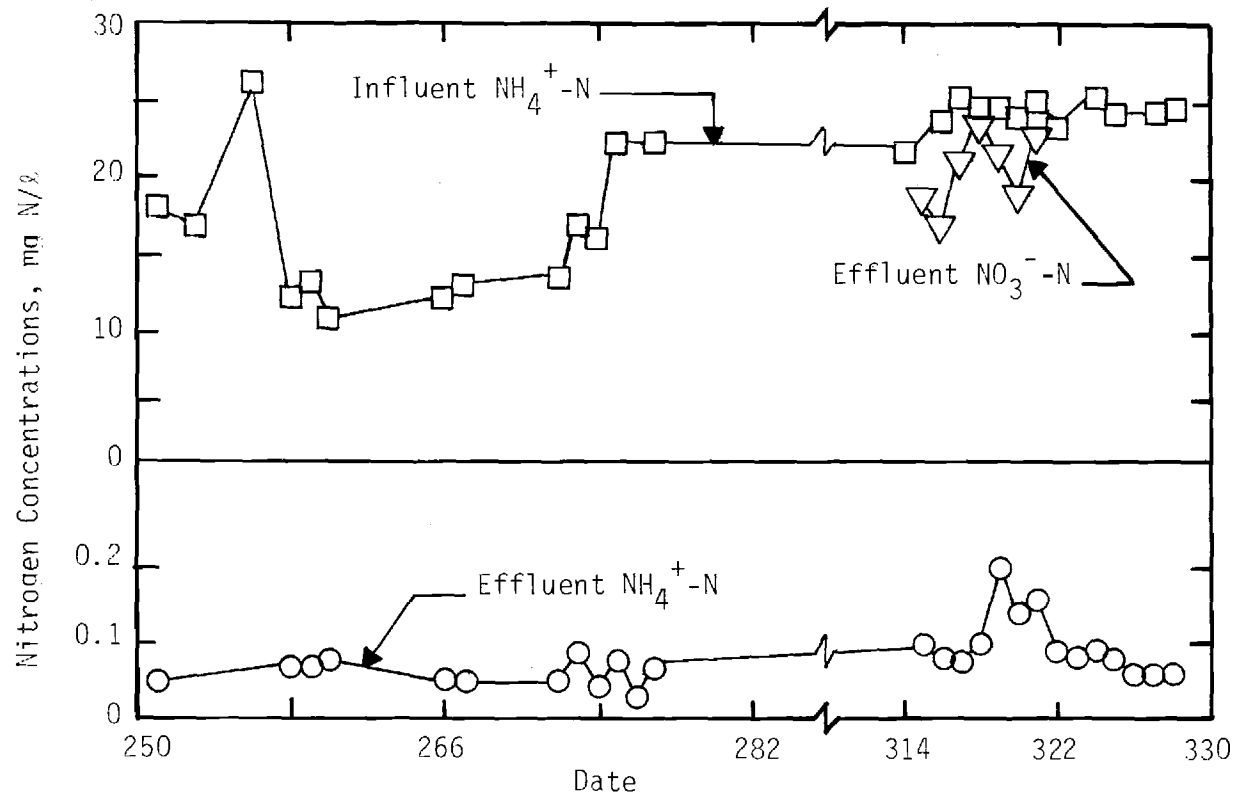


Figure 6. Influent and Effluent Ammonia-Nitrogen and Effluent Nitrate-Nitrogen Concentrations during Experimental Run A1.

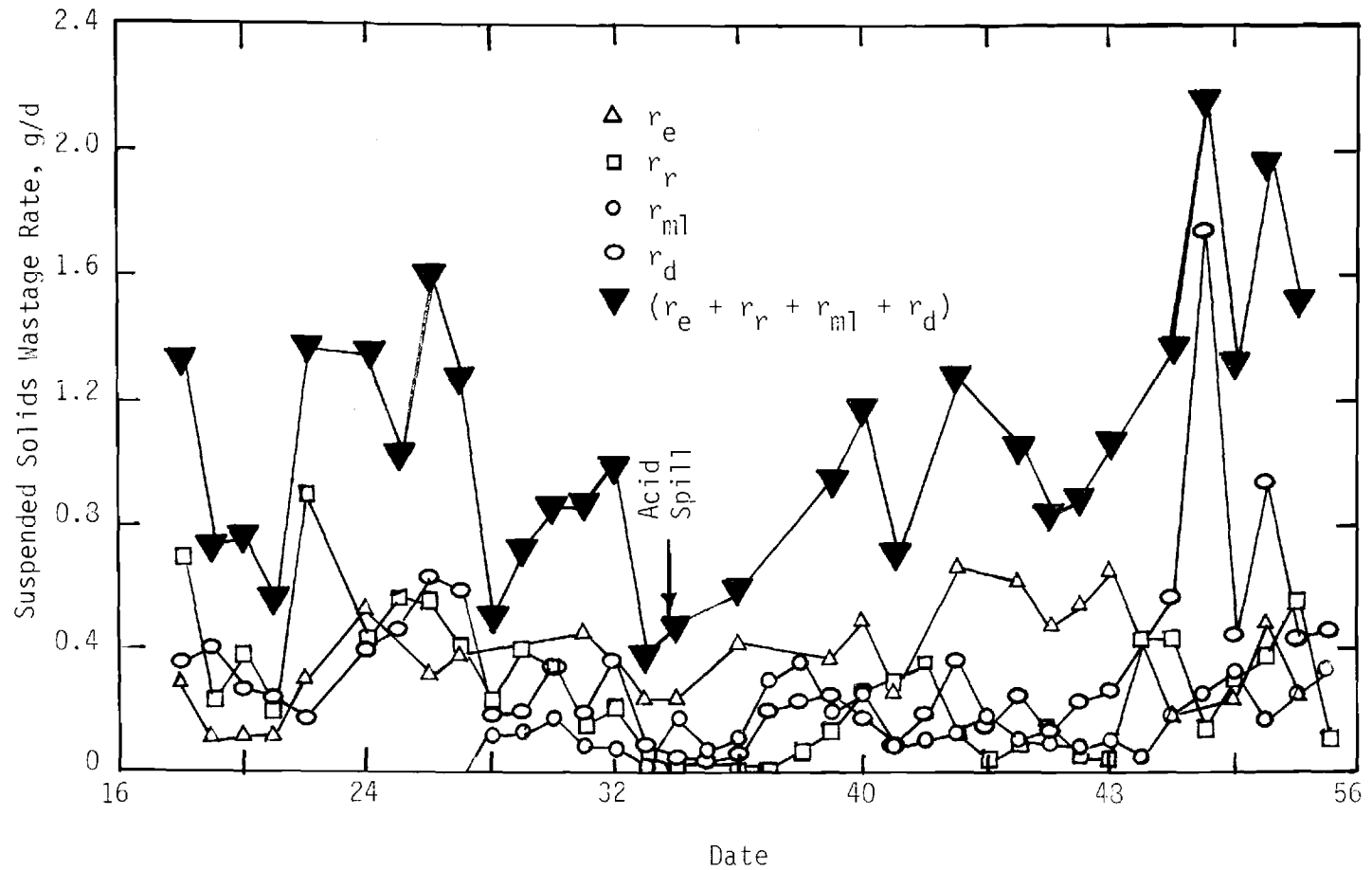


Figure 7. Rate of Wastage of Suspended Solids from the RBC System during Experimental Run A2

The performance of the RBC system during this run (i.e. $\theta_{op} = 6$ d) was similar to that of the initial Experimental Run A1. As shown in Figure 8, effluent COD concentrations averaged 10 mg/l (with the exception of day 34-35) at an average influent COD concentration of 54 mg/l. The effluent COD concentration on day 34 approached that of the influent, while significant improvement was apparent on the subsequent day. Finally, two days following the severe pH perturbation, the RBC system was again operating at or near an average COD effluent of 10 mg/l. The RBC system, therefore, demonstrated a remarkable response to a drastic pH shock and indicated an apparent advantage of attached biofilm systems under shock loading conditions.

The performance of the RBC system with respect to nitrification during Run A2 was erratic. During the initial 16 d of the run only partial effluent nitrification was achieved as shown in Figure 9. In addition, the effluent ammonia-nitrogen data during day 28-33 strongly indicated that the nitrifying microorganisms were being washed-out of the RBC system. Furthermore, the pH shock load of day 34 proved to be extremely toxic to the nitrifying microorganisms. Effluent nitrate-nitrogen concentrations after the pH shock remained at minimal detectable levels and ammonia-nitrogen concentrations were approximately equal to those of the influent wastewater. The system was therefore operated for 3.5 additional θ_{op} cycles (i.e. 21 d) after the pH shock to allow for the redevelopment of the nitrifying population. Since the nitrifying population did not redevelop in the RBC system it was apparent that the cell residence time for this run was less than the critical cell residence time for the nitrifying population and washout was experienced.

Experimental Run A3. Immediately following the conclusion of Run A2, θ_{op} was changed from 6 to 12 d while all other operational parameters remained unchanged. As indicated in Figure 10, the wastage of biomass from the RBC system proceeded in a manner similar to that experienced during Run A2.

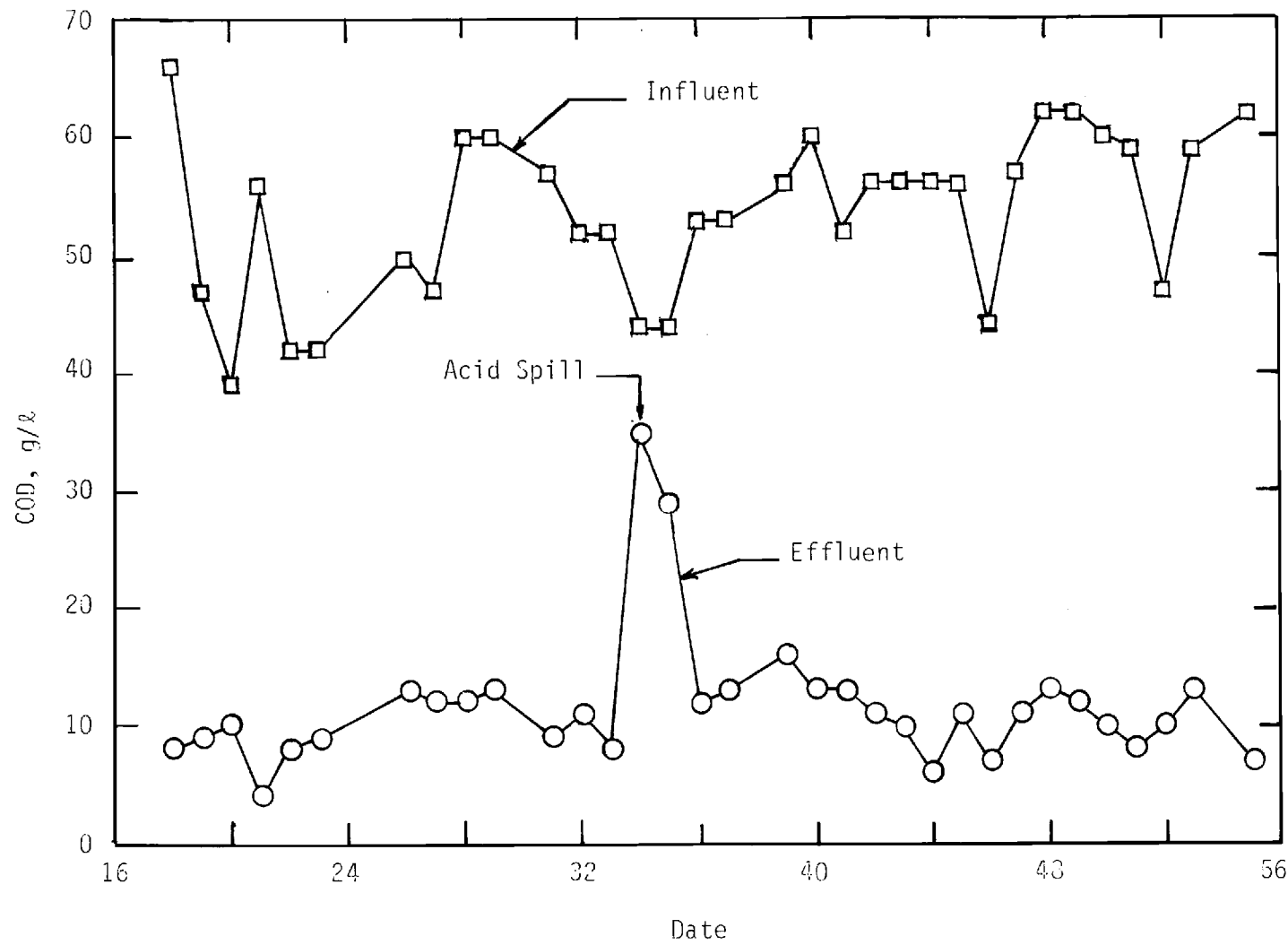


Figure 3. Soluble COD Concentrations in the Influent and Effluent Wastewaters during Experimental Run A2

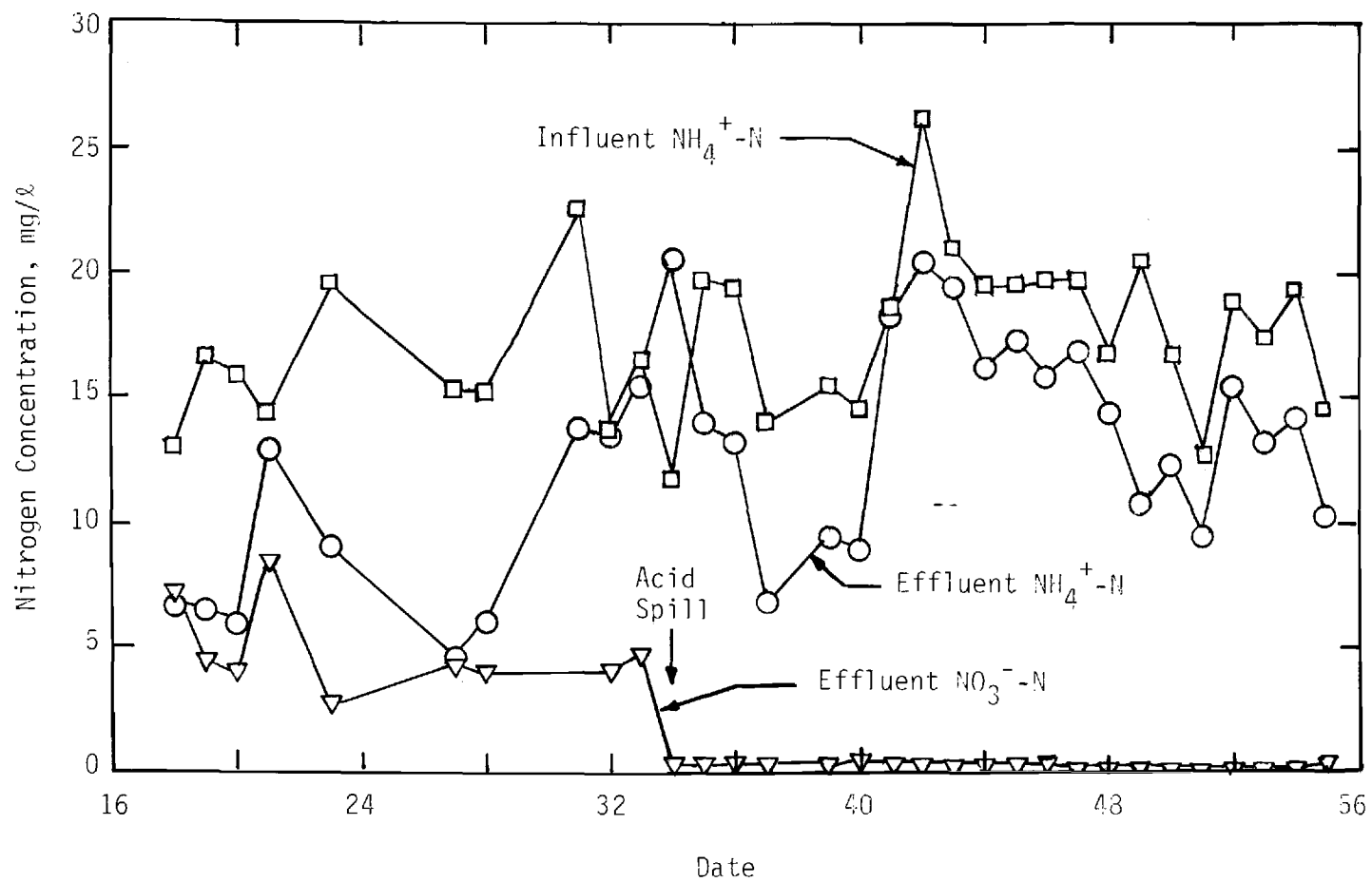


Figure 9. Influent Ammonia-Nitrogen and Effluent Ammonia- and Nitrate-Nitrogen Concentrations During Experimental Run A2

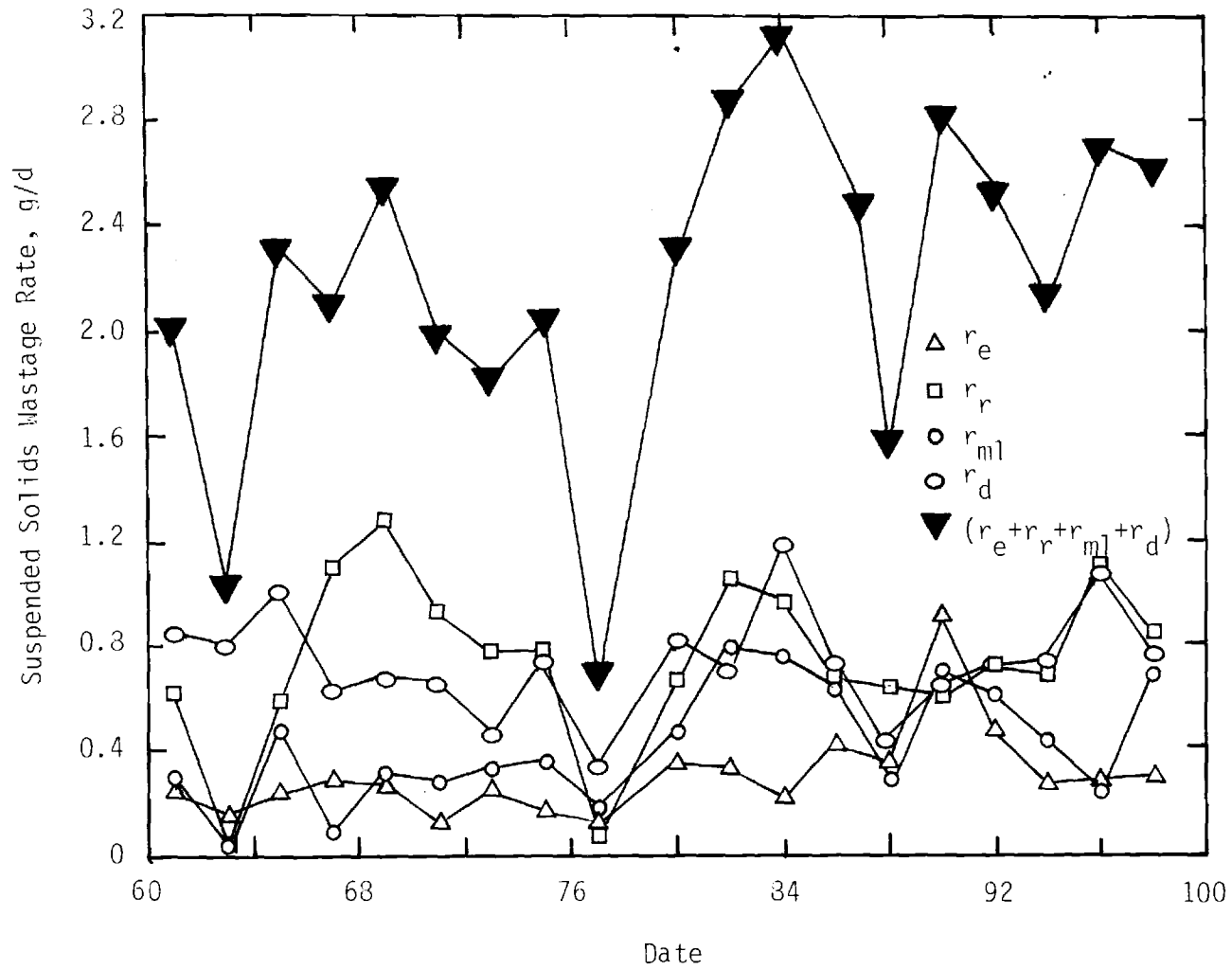


Figure 10. Rate of Wastage of Suspended Solids from the RBC System During Experimental Run A3

Organic removal efficiency improved slightly and equalled that experienced during Run A1, as shown in Figure 11.

As presented in Figure 12, there was an immediate increase in effluent nitrate concentration during the initial portion of Run A3 from the trace quantities detected at the end of Run A2 to $3.4 \text{ mg NO}_3^- \text{-N/l}$ on day 73. In addition, a trace of nitrite-nitrogen was contained in the effluent, however the major portion of effluent nitrogen was in the form of ammonia-nitrogen. From day 73 to 86, a drastic increase in nitrification occurred with effluent nitrate-nitrogen concentrations rising quickly to those of the influent ammonia-nitrogen. In addition, during day 75 to 84 nitrite-nitrogen initially increased and then decreased drastically. This response of the RBC system is typical of the classical evolution of nitrifying populations in which the growth rates of Nitrosomonas determine the kinetics of the nitrification process (Sawyer and McCarty, 1967). At day 88, nitrification was virtually complete with effluent nitrite- and ammonia-nitrogen concentrations at trace levels and effluent nitrate-nitrogen concentrations approximately equal to influent ammonia-nitrogen concentrations.

The nitrification data for the previous experimental run indicated that the critical cell residence time at which washout of nitrifying microorganisms occurred was greater than that achieved with $\theta_{op} = 6 \text{ d}$. The experimental data for Run A3 confirmed this conclusion since nitrifying microorganisms were able to immediately develop in the attached biofilm when θ_{op} was changed abruptly to 12 d with no other changes occurring in the reactor system. Furthermore, the data for Run A3 indicated that the cell residence time at $\theta_{op} = 12 \text{ d}$ was one at which virtually complete nitrification occurred.

Experimental Run A4. At $\theta_{op} = 18 \text{ d}$, the performance of the RBC system did not change significantly with respect to biomass wastage or organic removal. Effluent COD concentrations ranged from 4-10 mg/l and averaged 7 mg/l. Effluent nitrification continued as previously experienced during

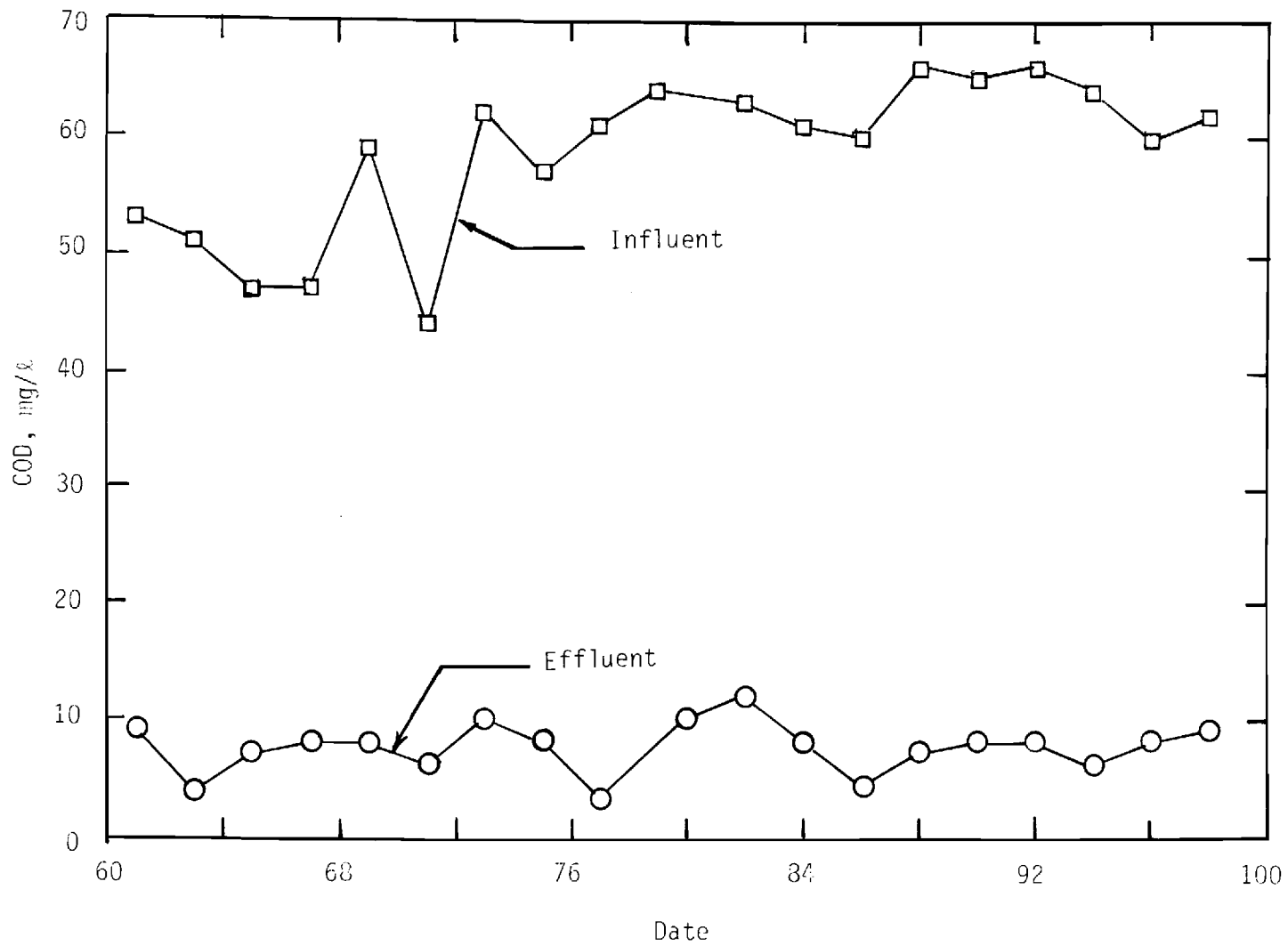


Figure 11. Soluble COD Concentrations in the Influent and Effluent Wastewaters During Experimental Run A3

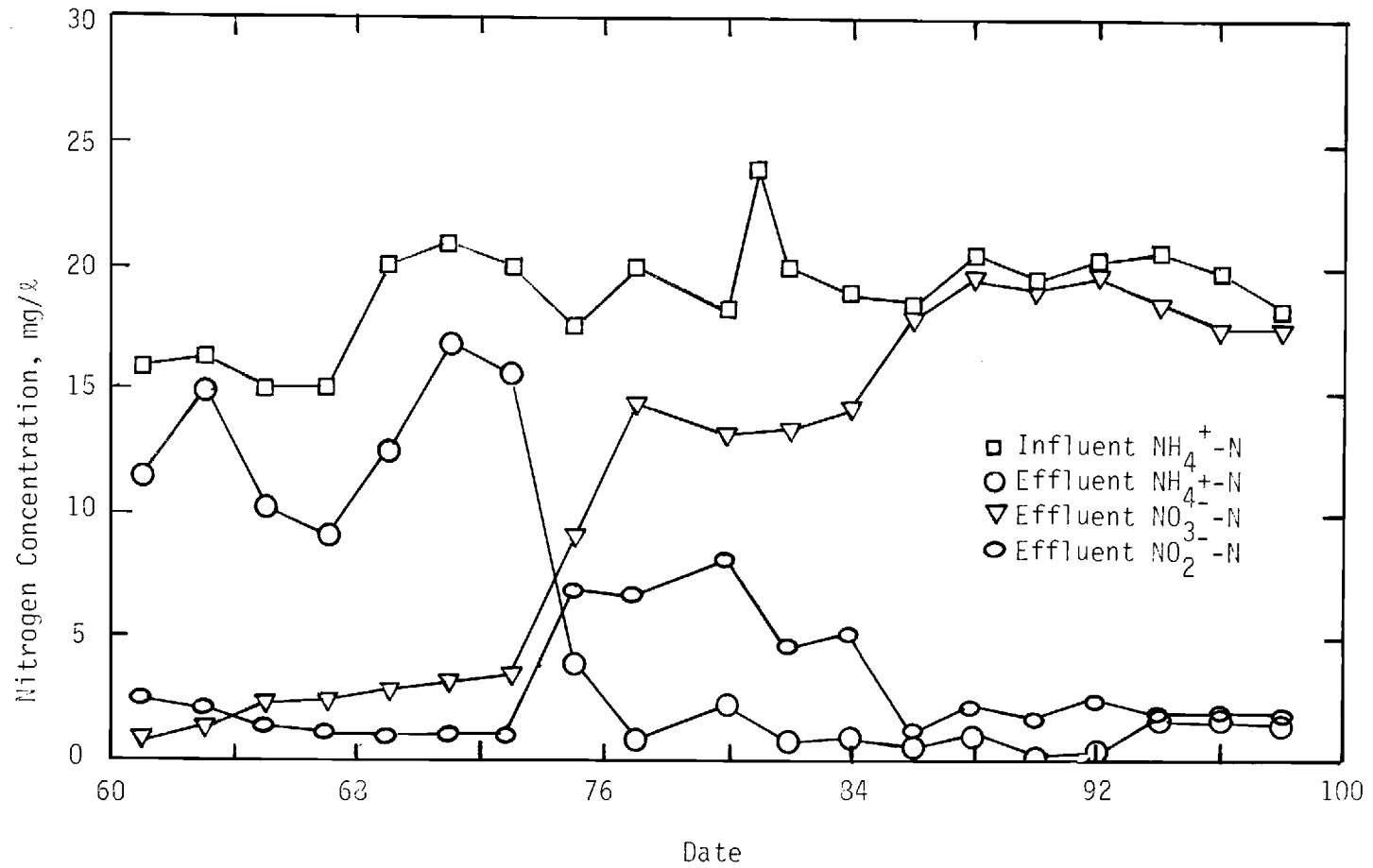


Figure 12. Influent Ammonia-Nitrogen and Effluent Ammonia-, Nitrate-, and Nitrite-Nitrogen Concentrations During Experimental Run A3

the terminal portion of Run A3. As shown in Figure 13, nitrification was virtually complete with only trace levels of ammonia-nitrogen appearing in the effluent. Effluent nitrate-nitrogen concentrations were approximate equal to those for influent ammonia-nitrogen.

Phase B Results

The primary objective of this phase was to determine the effect of influent nitrogen source on nitrification. Specifically, the amino acid glycine was utilized as the primary source of nitrogen during this run with influent ammonia-nitrogen concentration equal to zero. The θ_{op} employed during Run A4 was also employed in this phase, which was initiated immediately following the termination of Experimental Run A4 on day 150.

The influent organic loading was increased as a result of the additional organic matter associated with the influent organic-nitrogen source. As shown in Figure 14, however, the increased organic loading had only a minor effect on the performance of the system during the initial 2 to 3 d of operation. Subsequent to this initial period, effluent COD concentrations averaged 7 mg/l, a value previously experienced at this θ_{op} value. As indicated in Figure 15, nitrification continued to precede with no major changes resulting from the conversion to the influent organic nitrogen source. Effluent ammonia-nitrogen concentrations were at trace levels and effluent nitrate-nitrogen concentrations approached influent organic-nitrogen concentrations.

The results of Phase B indicated that the performance of the RBC system at $\theta_{op} = 18$ d was not drastically different from that experienced previously with the influent nitrogen source in the form of ammonia-nitrogen. It was also concluded that the data indicated that the RBC system would be an effective system for treating secondary effluents containing both ammonia and organic-nitrogen.

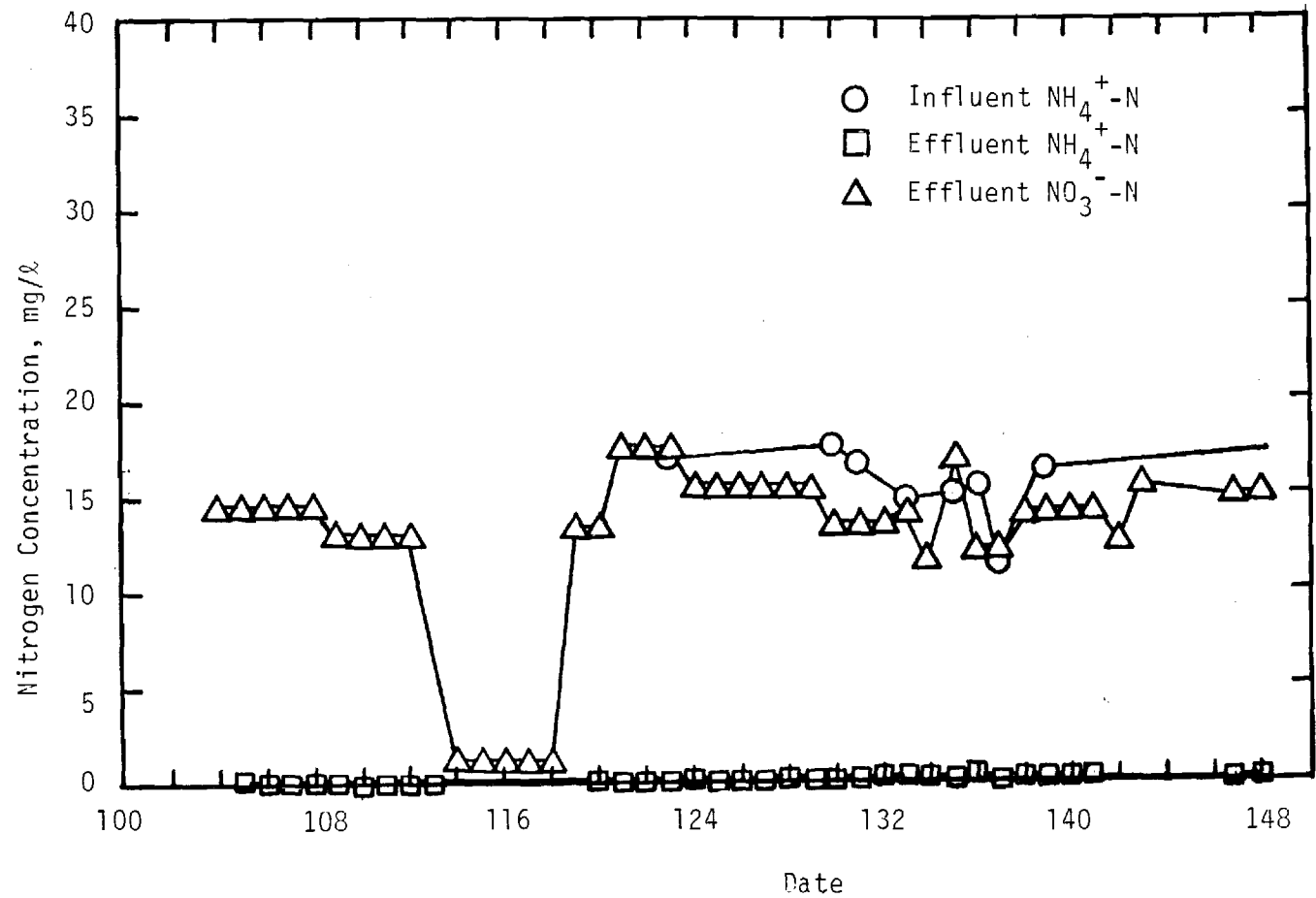


Figure 13. Influent Ammonia-Nitrogen and Effluent Ammonia- and Nitrate-Nitrogen Concentrations During Experimental Run A4

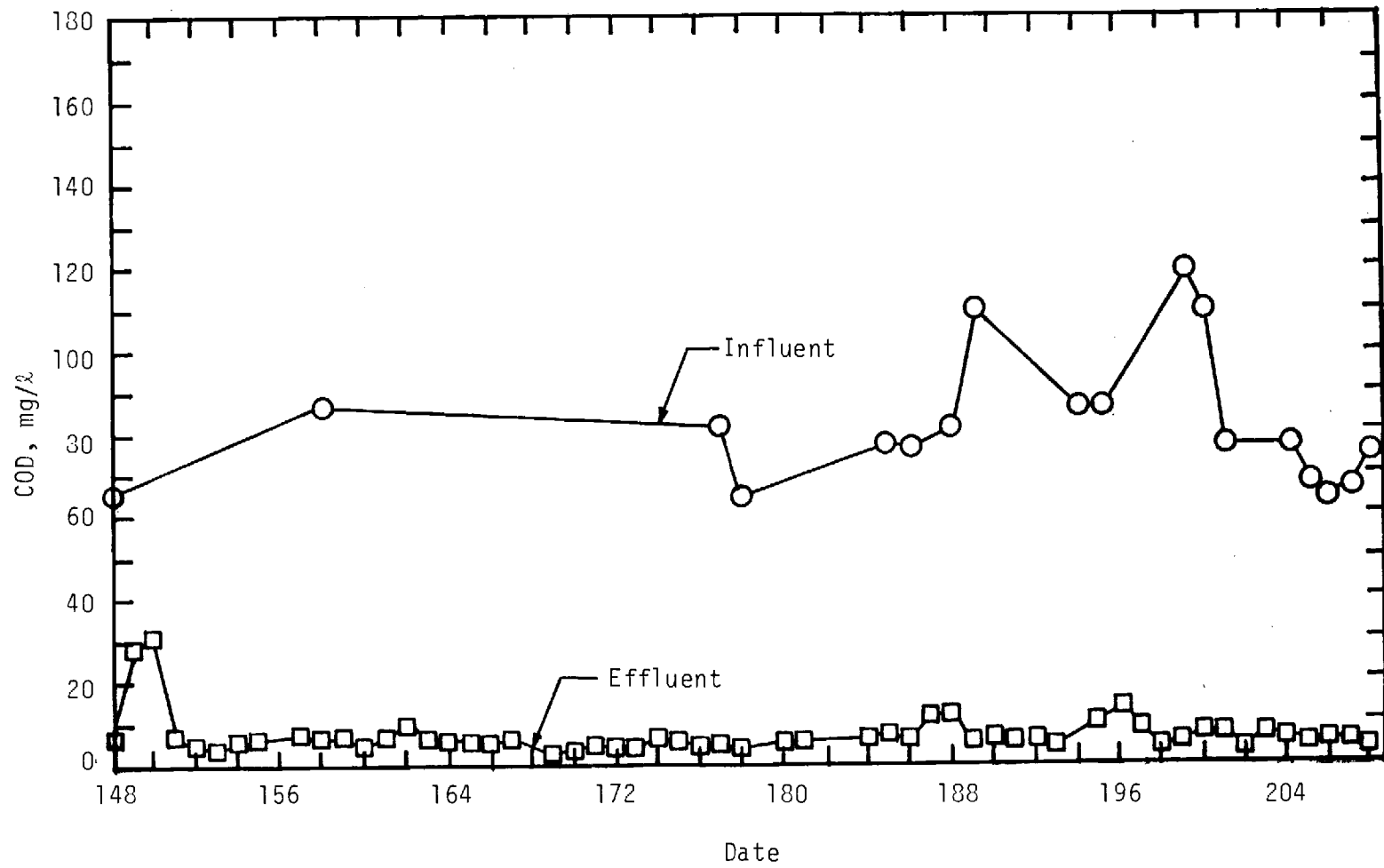


Figure 14. Soluble COD Concentrations in Influent and Effluent Wastewaters During Phase B

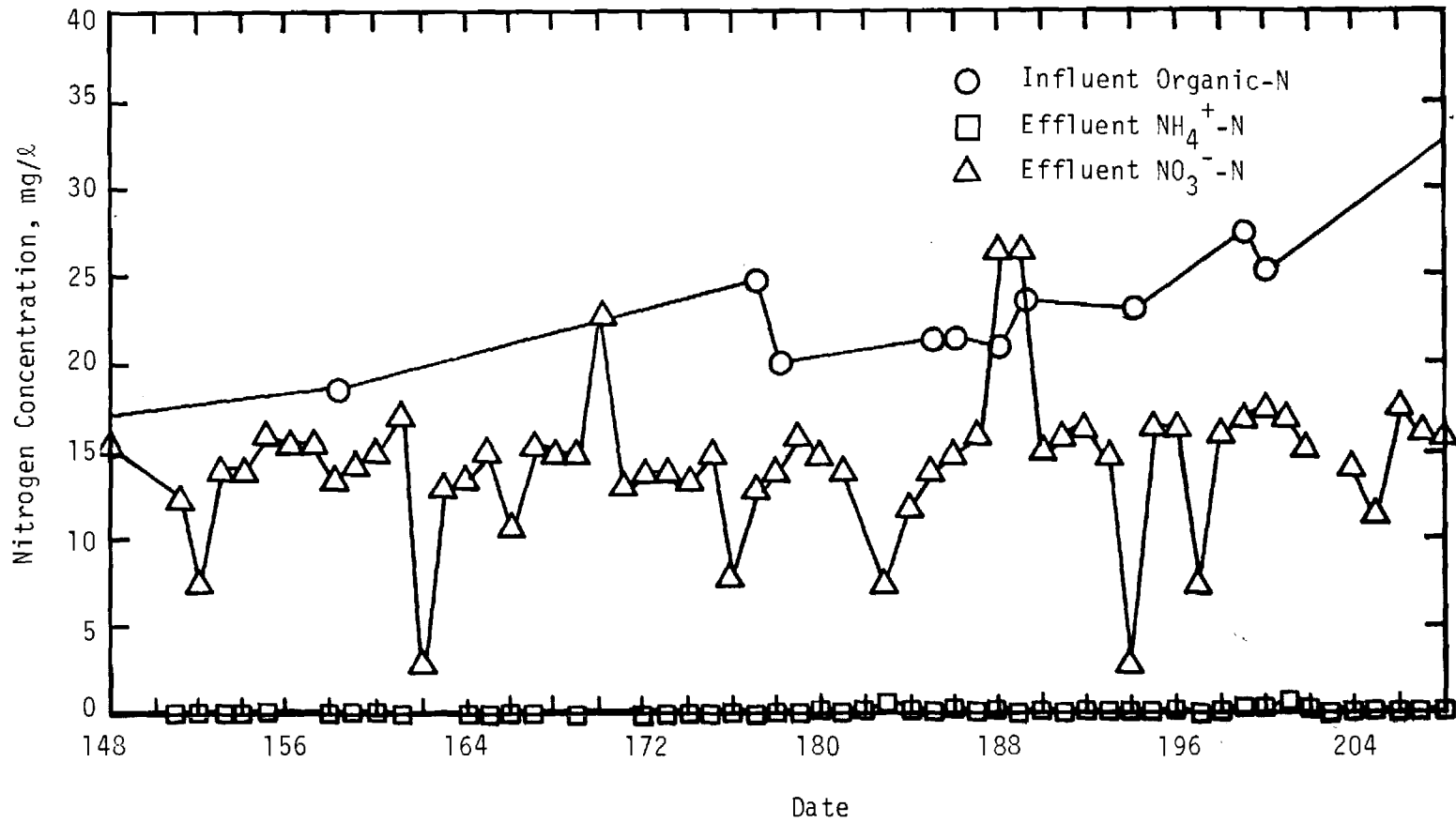


Figure 15. Influent Organic-Nitrogen and Effluent Ammonia and Nitrate-Nitrogen Concentrations During Phase B

Phase C Results

The primary objective of this Phase was to evaluate the effect of organic loading on nitrification. The influent nitrogen source was ammonia-nitrogen and $\theta_{op} = 12$ d. This θ_{op} value was utilized since data collected during Phase A indicated it to be the minimum θ_{op} value at which nitrification occurred. Therefore, the effect of organic loading was to be evaluated at a cell residence time at which the nitrifying population would be most susceptible to changes in biomass wastage rates and wastewater characteristics. The effects noted at such rapid growth rates would therefore present a conservative estimate of the effect of shock organic loadings on the nitrifying population.

Experimental Run C1. Influent COD concentration for this experimental run was increased slightly from 84 mg/l to 90 mg/l. During an initial 2 d time period, the value of θ_{op} was decreased from 18 d to 12 d. The RBC system was then operated continuously for 44 d to allow for the development of steady state conditions with respect to effluent COD and nitrogen concentrations.

As indicated in Figure 16, no significant adverse effects were observed with respect to effluent COD concentrations as a result of the altered operating conditions. The results were typical of previous runs in which similar changes in organic loading occurred. Furthermore, as shown in Figure 17, effective nitrification was achieved throughout the run. During the initial period of operation, effluent ammonia-nitrogen concentrations were at trace levels. However, on day 215 effluent ammonia-nitrogen concentrations approached 5 mg/l but then decreased to trace levels as effluent nitrate-nitrogen concentrations approached influent ammonia-nitrogen concentrations. This decrease in nitrification efficiency could have been the result of the decrease in the θ_{op} value or the increase in influent COD

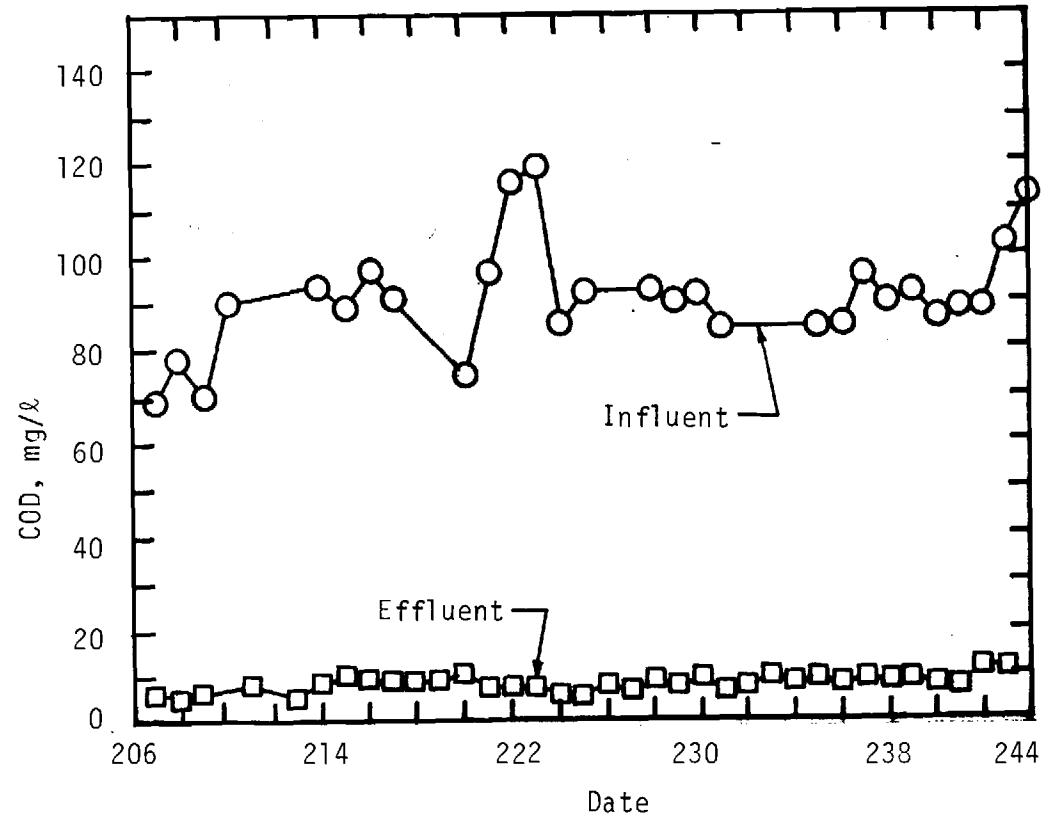


Figure 16. Soluble COD Concentration in Influent and Effluent Wastewaters During Experimental Run C1

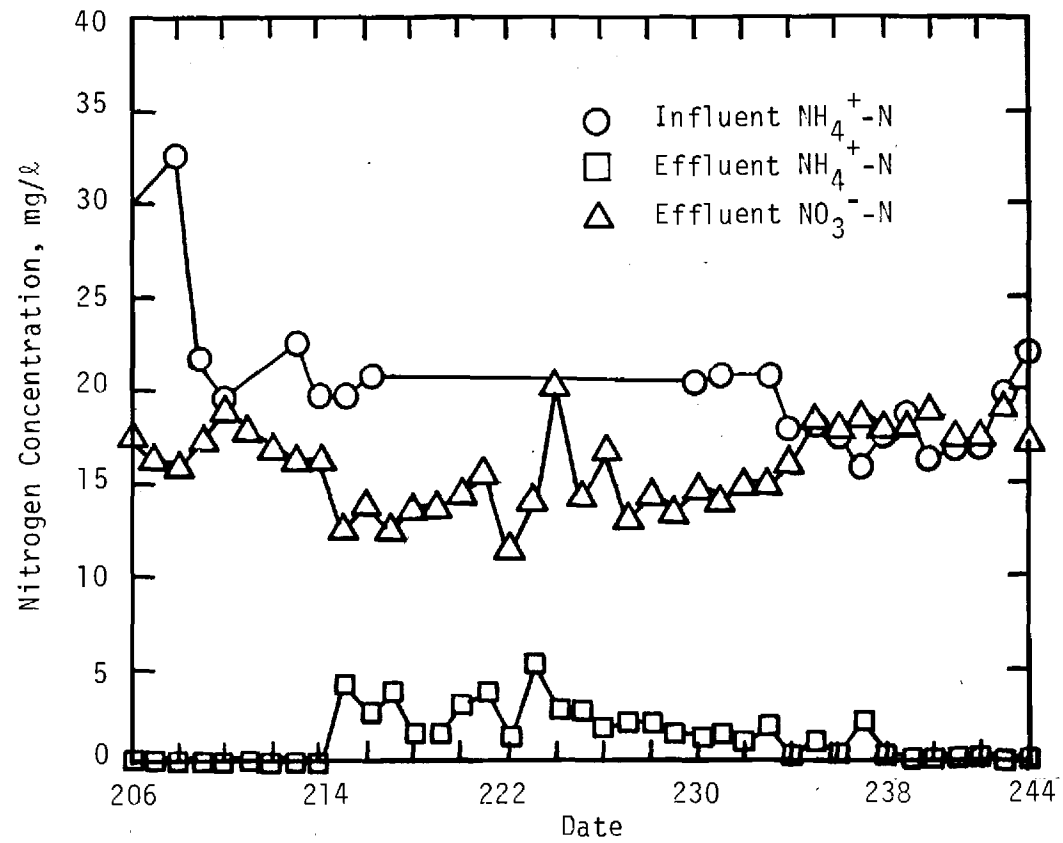


Figure 17. Influent Ammonia-Nitrogen and Effluent Ammonia- and Nitrate-Nitrogen During Experimental Run C1

concentration. However, the RBC system did ultimately achieve a steady state of operation in which effluent nitrification was virtually complete.

Experimental Runs C2, C3 and C4. Experimental Runs C2, C3 and C4 are presented collectively since these three runs were made with significant increases in influent COD concentration over short time periods and were not operated for extended time periods. Influent COD concentrations for the three runs were altered as indicated in Figure 18 to constant levels of 136, 172 and 190 mg/l, respectively. The effluent COD concentration from the RBC system during Runs C2 and C3 averaged 10 mg/l, indicating no significant effect of influent COD concentrations on effluent COD concentrations. As the influent COD concentration for Run C4 was increased from 172 to 190 mg/l over an initial 8 d period, the effluent COD concentration averaged approximately 15 mg/l. Therefore the performance of the RBC system with influent COD concentrations as high as 190 mg/l was excellent and no significant effects were noted for effluent COD concentrations.

During Phase C2 the increase in influent COD concentration to 136 mg/l did not significantly affect nitrification. That is, effluent ammonia-nitrogen concentrations were at the trace levels, and effluent nitrate-nitrogen concentrations were approximately equal to influent ammonia-nitrogen concentrations. The increase in COD concentration to 172 mg/l during Run C3 resulted in no significant effects on nitrification, as shown in Figure 19. Finally, the limited data for Run C4 in Figure 19 indicated that nitrification was not severely inhibited as evidenced by low effluent ammonia-nitrogen concentrations. Nitrate-nitrogen concentrations, however, did decrease significantly below influent ammonia-nitrogen levels. This was attributed to increased nitrogen requirements for biomass synthesis resulting from the increased organic loads for this run. Therefore the experimental results obtained over the 36 d period for Runs C1, C2, C3 and C4 indicated that nitrification was virtually complete at influent COD concentrations up to 190 mg/l.

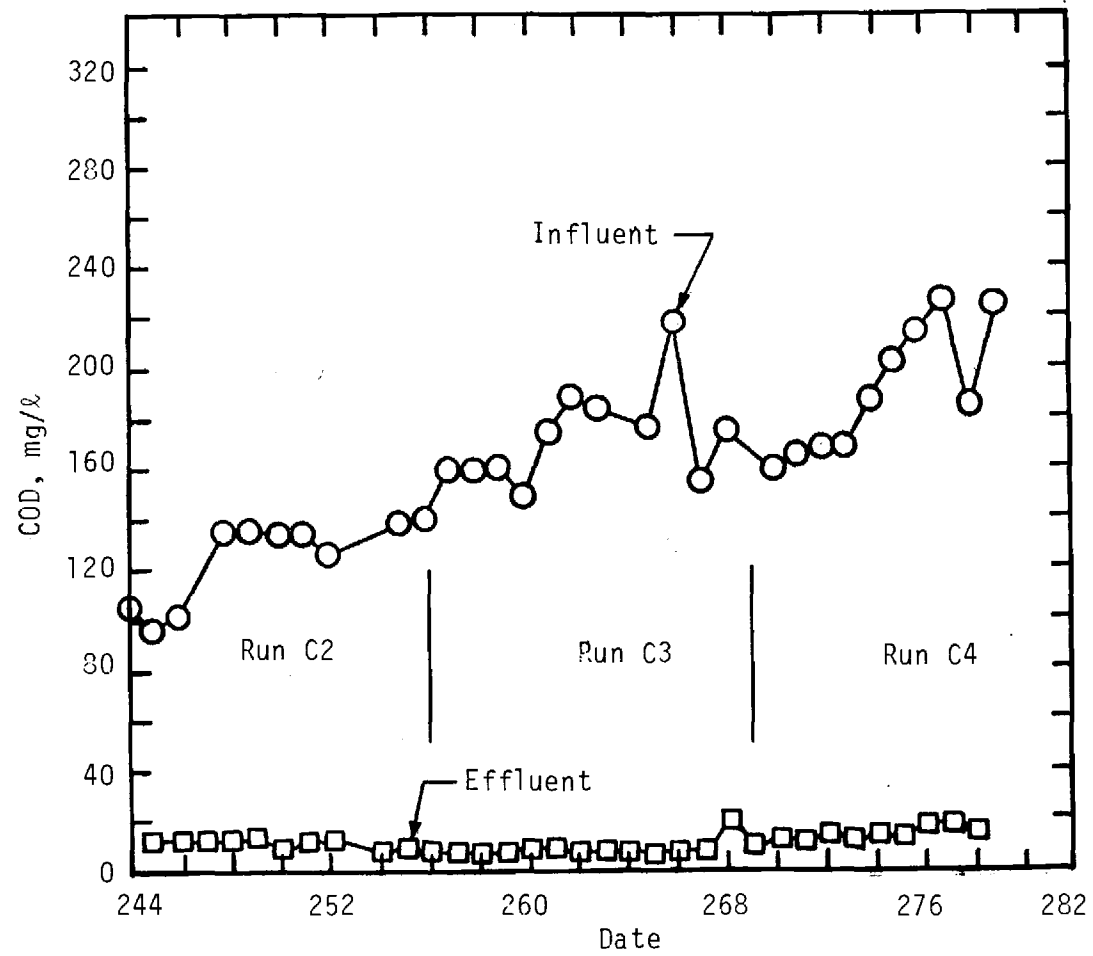


Figure 18. Soluble COD Concentration in Influent and Effluent Wastewaters During Experimental Runs C2, C3 and C4

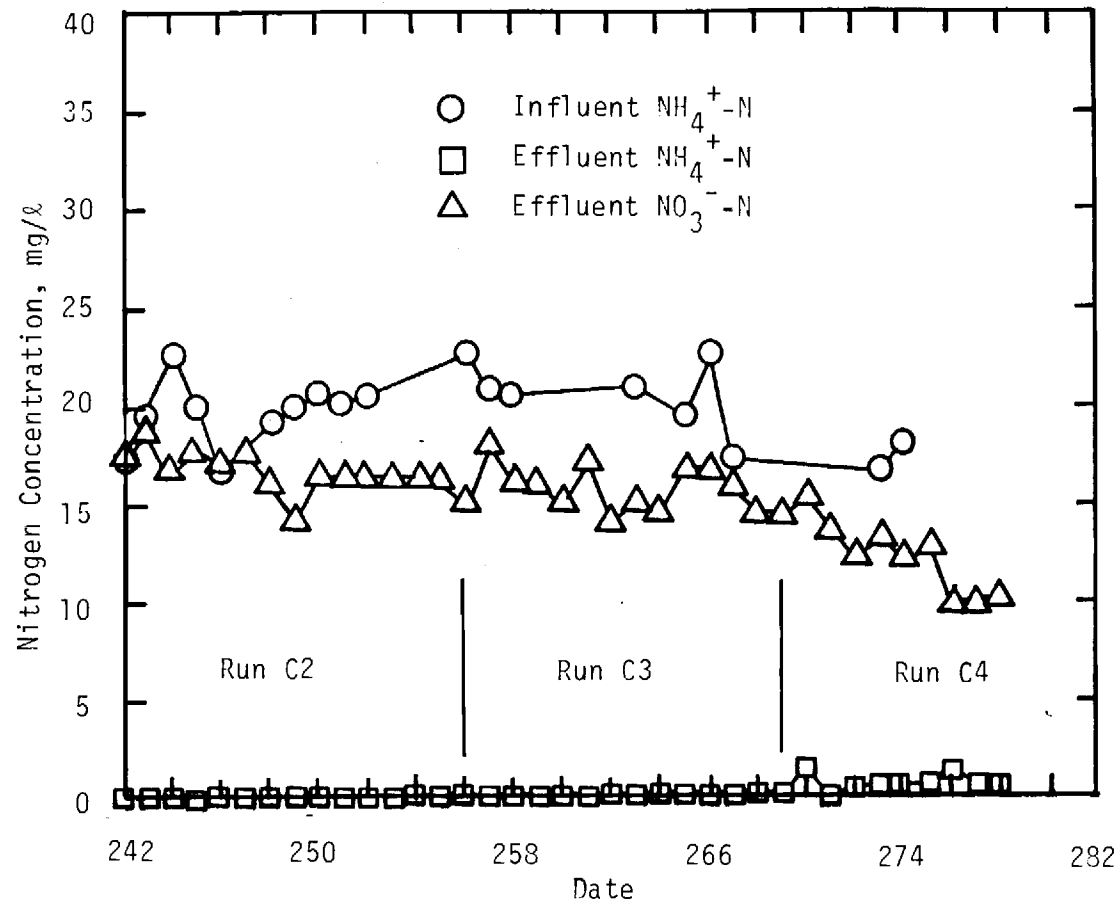


Figure 19. Influent Ammonia-Nitrogen and Effluent Ammonia- and Nitrate-Nitrogen Concentrations During Experimental Runs C2, C3 and C4

DISCUSSION OF EXPERIMENTAL RESULTS

Cell Residence Time

As indicated earlier, θ_{op} , i.e. operational cell residence time, values were used to establish the frequency of scraping RBC disc surfaces. It is, furthermore, obvious that biomass was wasted from the system in ways other than disc scraping, e.g. through the collection of naturally sloughed biomass in the mixed liquor recycle system and the accumulation of sloughed biomass in the reactor. Furthermore, sloughed biomass was discharged from the reactor in the effluent. Therefore direct correlation of θ_{op} values with actual cell residence time values for the RBC biomass was not possible.

The data required to properly calculate cell residence time values for the biomass included (1) an accurate measure of the quantity of biomass contained within the RBC system and (2) the rate at which biomass was wasted. As indicated previously in Figures 7 and 10, the total biomass wasted from the system varied considerably throughout even the most stable of steady state operational periods. This is further evident from an examination of Figure 20. These data generally indicate that the biomass removed from the oldest disc during any experimental run contained a higher level of biomass as θ_{op} increased. Furthermore, the data in Figure 20 indicate that biomass wastage was also responsive to increased organic loadings as indicated by the data for Runs A4 and B1 and Runs C2, C3 and C4 where organic strength was increased with each successive Experimental Run, respectively.

The data presented in Figure 20 are representative of the biomass wasted from the oldest disc surface in each set. To determine the total mass within the reactor system, it was necessary to estimate the total biomass attached not only to the oldest disc but also to the 5 remaining disc surfaces in each set whose age was less than θ_{op} . To determine the distribution of biomass within the disc system, a solids analysis procedure was initiated

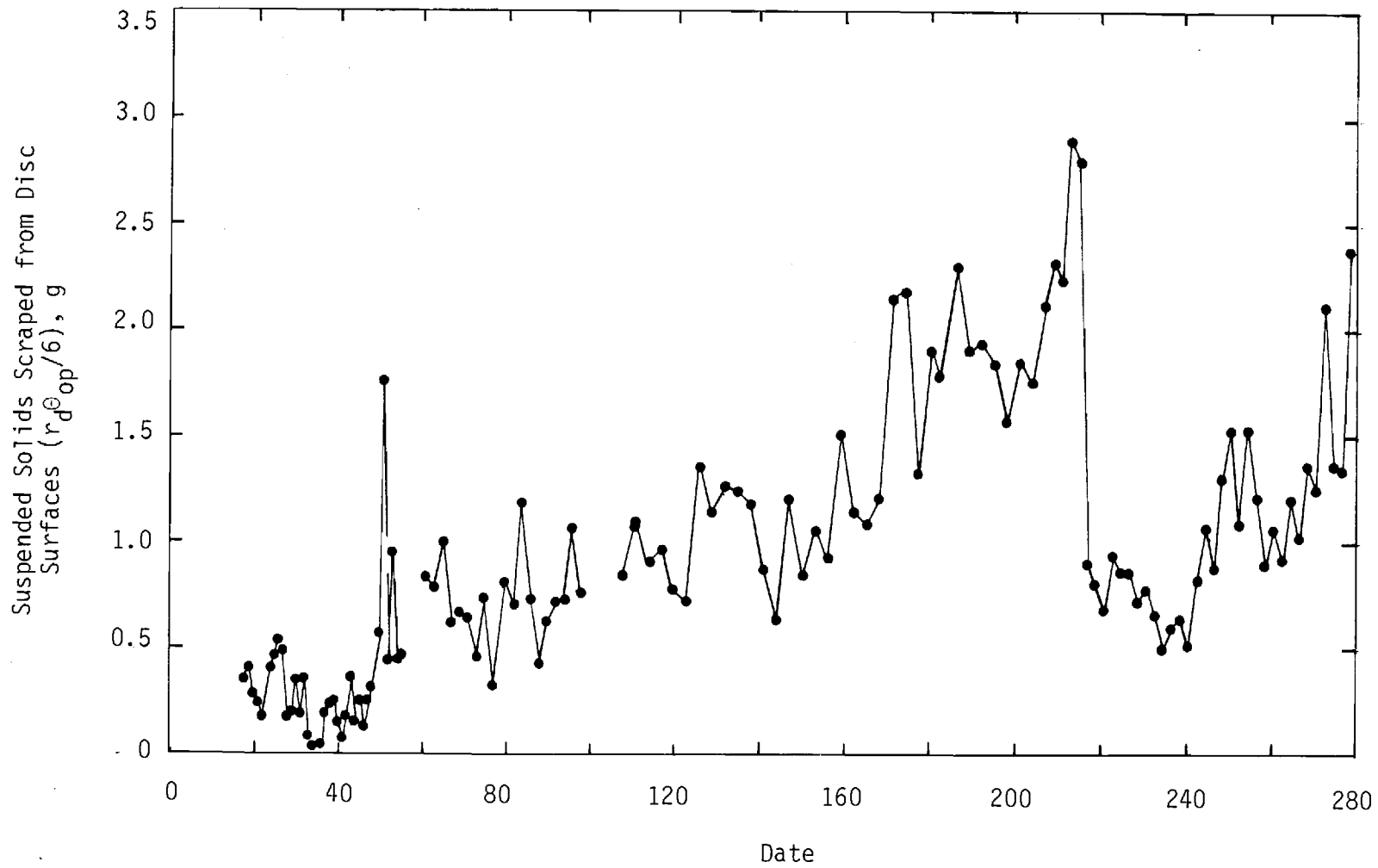


Figure 20. Biomass Wasted from RBC Disc Surfaces During Experimental Runs A2-A4, B1, C1-C4

during the latter portions of Experimental Runs A4, B1 and C1. Portions of disc surfaces with biomass age ranging from $\theta_{op}/6$ to θ_{op} d were collected and analyzed to determine the total biomass attached to the respective disc surfaces as a function of the time since the last scraping, i.e. biomass age. The data for Experiment Runs A4 and B1 are presented in Figure 21.

The quantity of attached biomass on the respective disc surfaces was generally a linear function of biomass age. This linear response was considered to be reasonable when the nature of the attached biofilms was considered. Initially, it was assumed that a biofilm consisted of an active and an inactive layer and that the active layer had a defined minimum depth which was substrate-limited. Then the rate of accumulation of biomass would be directly proportional to the rate at which substrate was being supplied to the reactor system. Since substrate addition was constant during all experimental runs, it was reasonable to expect that biomass accumulation rates would be a linear function of biomass age if sloughage rates were constant and invariant from set to set.

The total quantity of biomass accumulated on disc surfaces was not totally consistent with substrate loading and θ_{op} values as shown in Figure 21 for Runs A4 and B1. Both of these experimental runs were made at θ_{op} values of 18 d with organic loading rates of 4.1 and 5.4 g-COD/m²·d, respectively. The quantities of attached biomass in the RBC reactor system varied significantly. For example, biomass accumulation rates for Runs A4 and B1 were approximately 0.52 and 0.92 g/m²·d, respectively. This 77% (i.e. $(0.92-0.52)100/0.52$) increase in biomass accumulation rate could not however be attributed to the differences in organic loading rates which increased only 32%. Biofilm sloughage rates therefore were assumed to be different for these two Runs.

Data presented for Run C1 in Figure 22 indicate that the biomass accumulation rate was different from those determined for Runs A4 and B1. This experimental run was apparently not continued for a sufficient time period

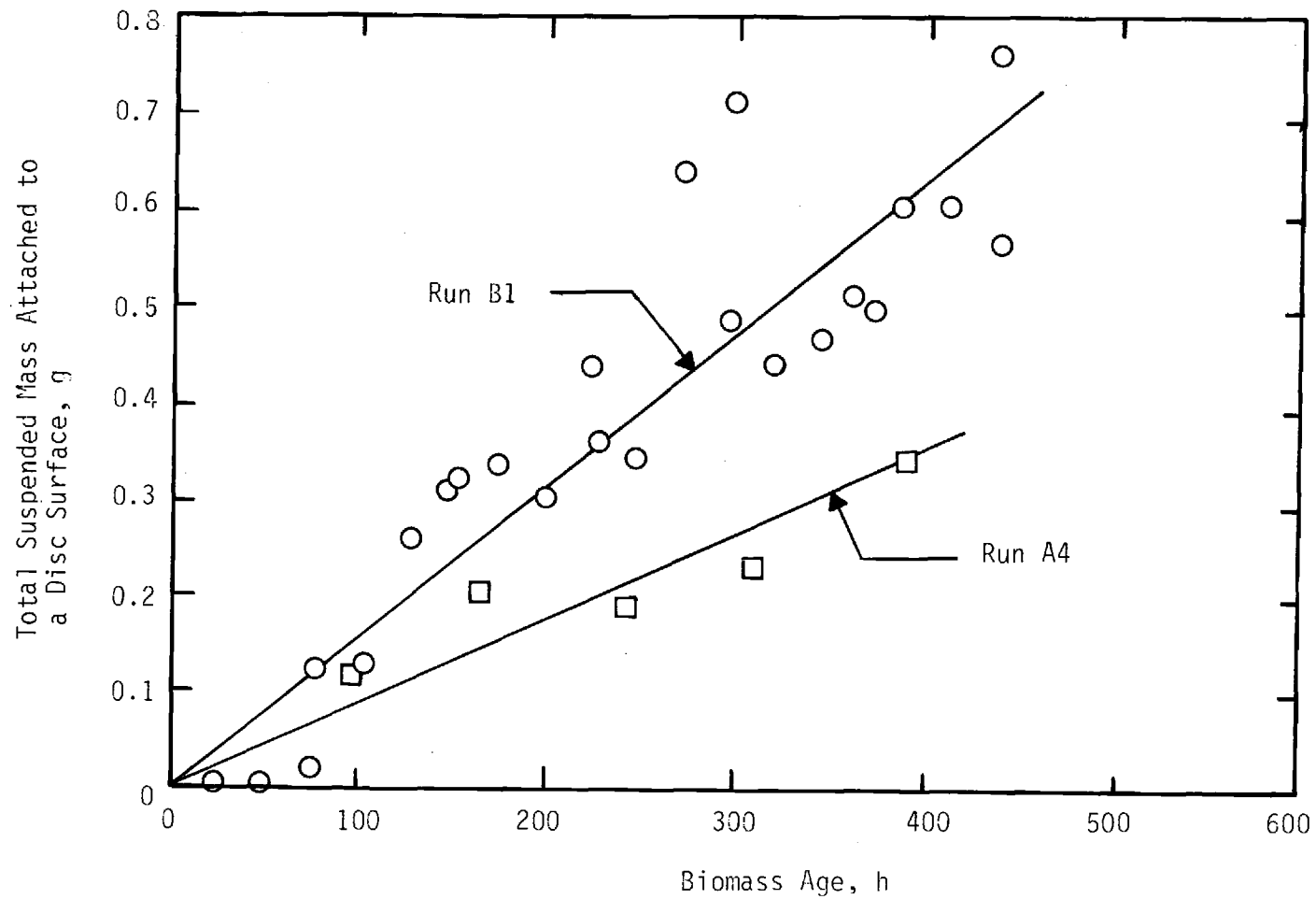


Figure 21. Mass of Total Suspended Solids Attached to a Disc Surface as a Function of Biomass Age for Experimental Run A4 and B1

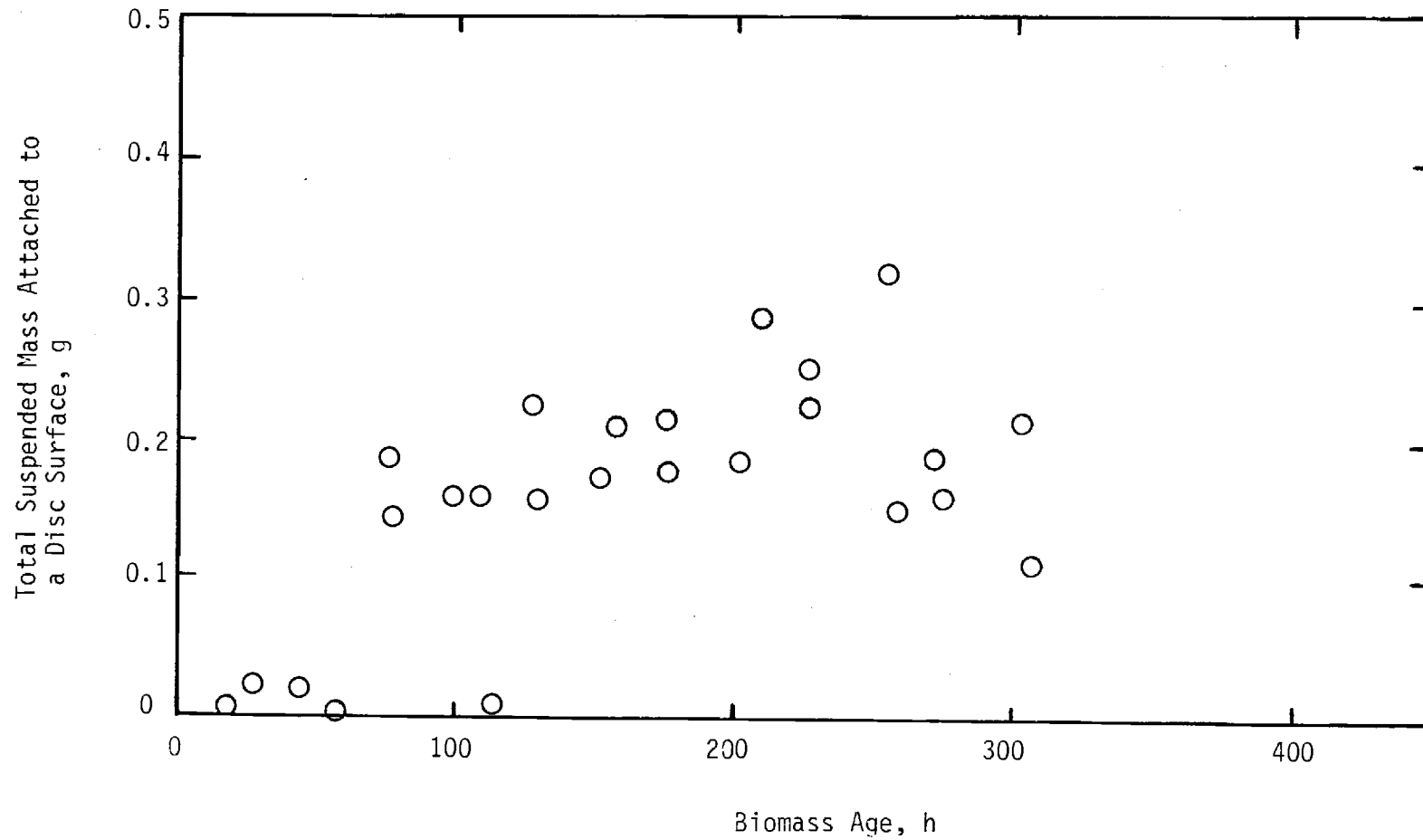


Figure 22. Mass of Total Suspended Solids Attached to a Disc Surface as a Function of Biomass Age for Experimental Run C1

to allow for the development of a stable biofilm following the change from $\theta_{op} = 18$ d to $\theta_{op} = 12$ d. The rate of accumulation during the initial period (i.e. biomass age < 150 h) was similar to the response indicated for Runs A4 and B1. However, there appeared to be no significant increase in biomass at age > 150 h. The response for Run C1 apparently was significantly affected by the transitory nature of the run as a result of the abrupt changes in COD concentration and θ_{op} . Furthermore, additional data presented by Cruz (1977) and Pope (1973) confirm that the responses indicated in Figure 21 were most typical of biomass accumulation rates on disc surfaces.

The total biomass contained within the reactor system was therefore assumed to be contained in two fractions. The first fraction was the biomass attached to disc surfaces. The second fraction consisted of the sloughed biomass solids contained in the mixed liquor. The total biomass, M_T , within the reactor system was therefore

$$M_T = M_{ML} + M_D \quad (9)$$

where M_{ML} = sloughed biomass which was suspended in the RBC mixed liquor and M_D = biomass attached to disc surfaces. In the calculation of M_D , it was assumed that the rate of accumulation of biomass on the disc surfaces was a linear function of biomass age. The average value of M_D was then estimated for each scraping cycle using the rate of wastage of attached biofilm (i.e., r_d) and θ_{op} in the following equation

$$M_D = [r_d/6] \left(\sum_{i=1}^6 \bar{t}_i \right) \quad (10)$$

where $\bar{t}_i = t_i - \frac{1}{2} \left(\frac{\theta_{op}}{6} \right)$, \bar{t}_i = average age of attached biomass on the i th disc surfaces, and t_i = elapsed time since i th disc surface was last scraped.

For example, following substitution of appropriate values into equation 10 for $\theta_{op} = 12$ d, M_D then equals

$$M_D = [(r_d)/6](\Sigma((2-1) + (4-1) + (6-1) + (8-1) + (10-1) + (12-1))) = 6r_d \quad (11)$$

Finally, equation 10 can be reduced to the following for all θ_{op} values

$$M_D = 0.5 r_d(\theta_{op}) \quad (12)$$

Therefore, total attached biomass could be determined with the solids data generated during the routine disc scraping cycle.

The remaining fraction of the average total biomass in the system, i.e. the suspended biomass in the mixed liquor, was estimated using biomass wastage data collected for the mixed liquor (r_{ml}) and for the recycle sieve system (r_r). The total mixed liquor mass, M_{ML} , was then assumed to be

$$M_{ML} = \frac{(r_{ml} + r_r)(\theta_{op}/6)}{2} \quad (13)$$

This relationship was based on the assumption that the rate at which sloughed biomass accumulated in the mixed liquor and recycle systems was a linear function of time and M_{ML} was to be estimated in the middle of a scraping interval. Therefore the average total biomass, M_T , in the RBC system was estimated to be equal to

$$M_T = 0.5[r_d \theta_{op} + (r_{ml} + r_r)(\theta_{op}/6)] \quad (14)$$

The remaining data required for the calculation of the average cell residence time at the end of a scraping cycle was the rate at which biomass was wasted from the RBC system. The total biomass wastage rate, \bar{R}_w , was equal to

$$\bar{R}_w = (r_d + r_{ml} + r_e + r_r) \quad (15)$$

Calculation of cell residence time, θ_c , was therefore possible using equations 4 and 5 as follows

$$\theta_c = \frac{M_T}{\bar{R}_W} = \frac{[r_d \theta_{op} + (r_m + r_r)(\theta_{op}/6)]}{2(r_d + r_m + r_e + r_r)} \quad (16)$$

Data used in the calculation of the respective θ_c values are presented in Table 9. Although considerable scatter is evident in the data for M_T , M_D and \bar{R}_W , these broad variations were normalized through the calculation of the cell residence time values. This is indicated in Figure 23 where a direct relationship between θ_{op} and θ_c is obvious. These θ_c values then provided an effective means of normalizing the response of the experimental runs with regard to nitrification, organic removal and biomass production.

Nitrification

Prior to an examination of nitrification efficiency, it was necessary to determine the extent to which several possible reactions involving the pertinent nitrogen species occurred. For example, the fate of influent nitrogen was to be determined by three metabolic processes. These included (1) inclusion into cell mass as a result of synthesis reactions by autotrophic and heterotrophic microorganisms, (2) utilization of ammonia-nitrogen as an electron source by autotrophic nitrifying microorganisms and (3) utilization of nitrate- and nitrite-nitrogen, produced through nitrification, as a final electron acceptor by denitrifying microorganisms.

To determine the fate of influent ammonia- or organic-nitrogen, nitrogen data were collected and analyzed for the soluble portion of effluent wastewaters and for the biomass routinely wasted from the RBC system. These data were normalized to disc surface area and are presented in Table 10. The data reported for ammonia-, nitrate-, nitrite and soluble organic-nitrogen

Table 9

Cell Residence Time Values and Associated Data

Experimental Run	Phase A			Phase B		Phase C		
	A2	A3	A4	B1	C1	C2*	C3*	C4*
Time Interval for θ_c calculation	42-55	86-98	124-148	166-207	235-243	243-256	257-269	270-280
θ_{op} (d)	6	12	18	18	12	12	12	12
M_D (g)	1.43	4.37	3.32	5.49	1.85	4.07	3.29	5.04
M_{ML} (g)	0.2	1.27	1.45	1.74	1.19	2.18	1.98	2.92
M_{Total} (g)	1.63	5.64	4.77	7.23	3.04	6.25	5.27	7.96
\bar{R}_w (g/d)	1.26	2.4	1.43	2.03	1.66	3.18	2.66	4.04
θ_c (d)	1.3	2.4	3.3	3.6	1.8	2.0	2.0	2.0

*Calculations are for highly transitory periods for these experimental runs.

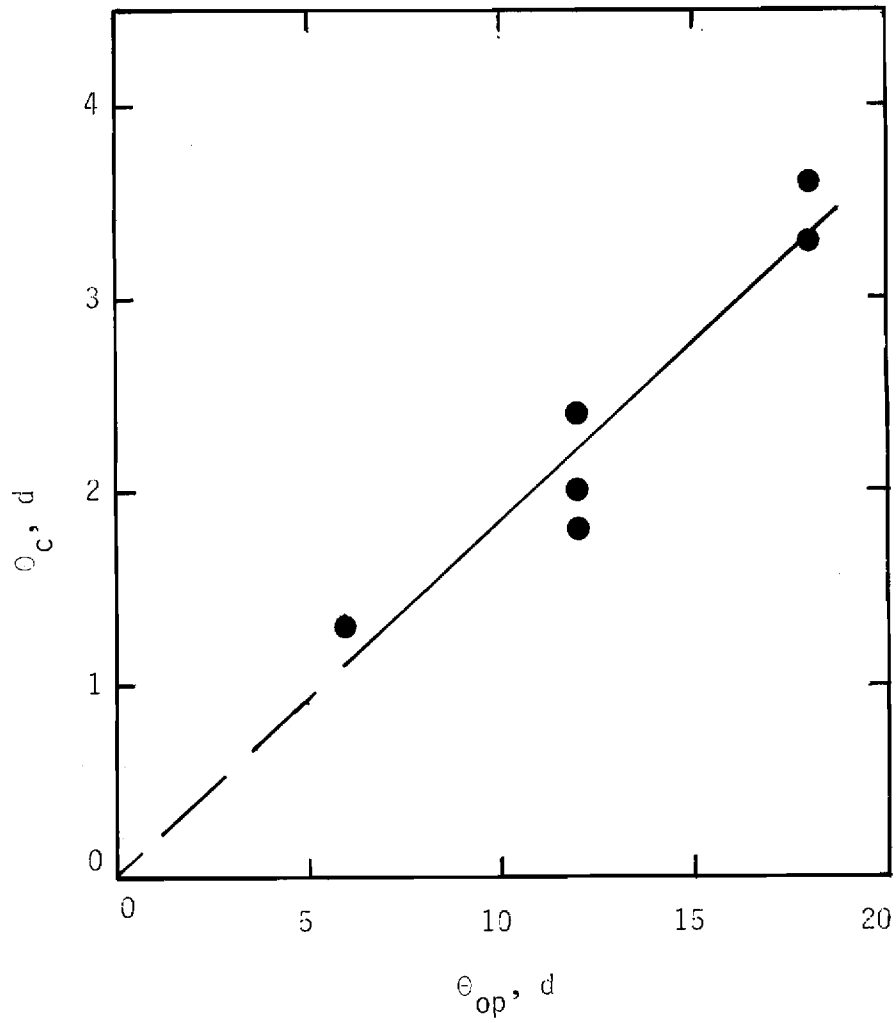


Figure 23. θ_C Values for Phases A, B and C As A^C Function of θ_{op}

Table 10

Nitrogen Data for Influent and Effluent Wastewaters and Wasted Biomass

Experimental Run	A1	A2	A3	A4	B1	C1	C2	C3	C4
Influent Wastewater									
NH_4^+ -N ($\text{g}/\text{m}^2 \cdot \text{d}$)	1.33	1.24	1.45	1.11	0	1.36	1.44	1.41	1.30
Organic-N ($\text{g}/\text{m}^2 \cdot \text{d}$)	0	0	0	0	1.62	0	0	0	0
Effluent Wastewater									
NH_4^+ -N ($\text{g}/\text{m}^2 \cdot \text{d}$)	0.01	0.97	0.07	0.01	0.02	0.03	0.01	0.02	0.05
Soluble Organic-N ($\text{g}/\text{m}^2 \cdot \text{d}$)	-	-	-	-	0.06	-	-	-	-
NO_3^- -N ($\text{g}/\text{m}^2 \cdot \text{d}$)	1.08	0.01	1.35	1.02	1.05	1.25	1.12	1.11	0.88
NO_2^- -N ($\text{g}/\text{m}^2 \cdot \text{d}$)	-	0.001	0.06	-	0.002	-	-	-	-
Wasted Biomass (\bar{R}_w)									
Suspended Organic-N ($\text{g}/\text{m}^2 \cdot \text{d}$)	-	0.12	0.20	0.19	0.30	0.19	0.39	0.30	0.48
Nitrogen Balance*	82	89	116	111	88	108	106	101	108
Nitrification Efficiency**	99	<1	91	99	93	98	99	98	95

* Percent of Influent-N measured in effluent and wasted biomass

**Percent of soluble effluent nitrogen in nitrate form.

in effluent wastewaters and suspended organic-nitrogen associated with the wasted biomass were then evaluated to determine the fate of the influent ammonia-nitrogen.

For Experimental Runs A2-A4, B and C1-C4, the nitrogen balance data in Table 10 indicate that the rate of nitrogen discharge in the effluent wastewaters and wasted biomass was approximately equal to influent nitrogen loading rates. The fact that the balance was occasionally significantly greater than 100% (e.g. Runs A3, C1 and C4) was attributed to experimental errors involved in monitoring trace quantities of effluent ammonia, nitrate and soluble organic nitrogen. The nitrogen balance of 82% for Run A1 was much lower than that for other runs primarily because suspended organic-nitrogen was not included in balance. If the average value for this fraction from Runs A2-A4 was assumed to be indicative of that wasted during Run A1, the nitrogen balance value would be 95%.

Therefore the nitrogen data indicated that the nitrogen balance was virtually complete for each Experimental Run with the nitrogen species identified. In addition, although it was not determined whether anaerobic conditions developed in an inactive portion of the biofilm, no significant denitrification appeared to have occurred in the biofilms of the RBC system. Denitrification could, however, have been encouraged if disc rotational speeds were decreased to lower the dissolved oxygen concentration in the mixed liquor allowing for anaerobic conditions to develop in the biofilm.

The extent of nitrification was determined through an examination of the soluble portions of effluent wastewaters. Accordingly, nitrification efficiency was calculated by determining the portion (i.e. percentage) of total effluent nitrogen in the nitrate-nitrogen form. As indicated in Table 10, nitrification was virtually complete, i.e. nitrification efficiency = 91-99%, in all Experimental Runs with the exception of Run A2 which was operated at

the lowest θ_{op} and θ_c values examined.

The effect of θ_c on nitrification is indicated directly in Figure 24. The response of the RBC system at $\theta_c > 1.3$ d was that of virtually complete nitrification. The highest θ_c value at which nitrification did not occur was 1.3 d while the lowest θ_c value at which nitrification did occur was 1.8 d. These data therefore indicate that the minimum value of θ_c required for nitrification to occur, i.e. $(\theta_c)_{min}$, was in the range of 1.3-1.8 d. The maximum net specific growth rate ($\hat{\mu}$) for the nitrifying population was therefore in the range of 0.56-0.77 d^{-1} , i.e. $\mu = \theta_c^{-1}$. At the temperatures for Runs A2 and C1 at $\theta_c = 1.3-1.8$ d, the maximum net specific growth rates for Nitrosomonas (the rate-limiting organism) were calculated to be $\hat{\mu} = 0.84$ and 1.12 d^{-1} with equation 5. The predicted critical cell residence time for washout, i.e. $(\theta_c)_{min}$, according to equation 5 was therefore in the range of 0.9-1.2 d. The predicted range for $(\theta_c)_{min}$ of 0.9-1.2 d and the experimentally determined range of 1.3-1.8 d were highly compatible and confirmed the use of θ_c in modelling nitrification in the RBC system. In addition, the experimental data were compared with nitrification data obtained by numerous authors using both attached and suspended growth biological systems. As presented in Figure 25, the experimental data were also highly compatible with these data collected from both suspended growth and attached film reactor systems treating numerous synthetic and domestic wastewaters. It can therefore be conclusively stated that the nitrifying population associated with the attached-film RBC system examined could be modelled with kinetic data available for pure cultures of nitrifying microorganisms and for mixed culture systems treating a wide diversity of wastewaters. The use of such a modelling technique will greatly advance the design procedures used for RBC systems, enhance the utilization of these cost-effective systems and assure a higher degree of successful nitrification of wastewater effluents with RBC systems.

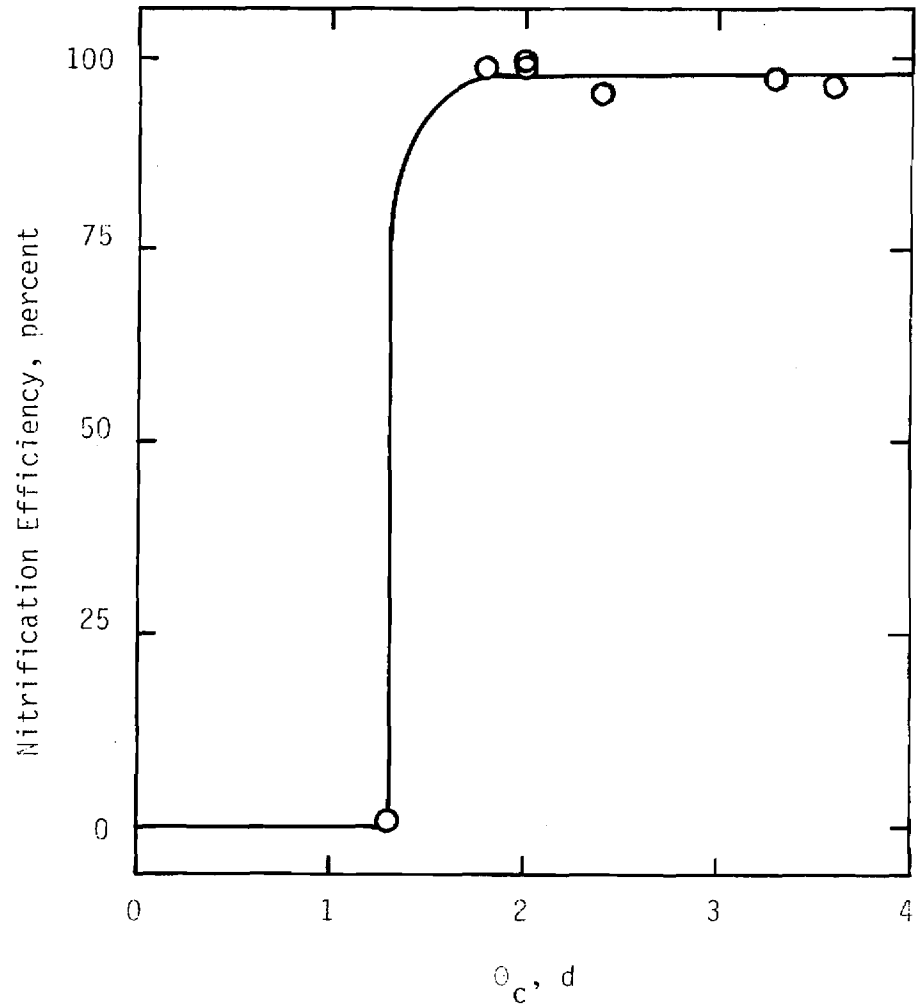


Figure 24. Effect of θ_c on Nitrification Efficiency of RBC System

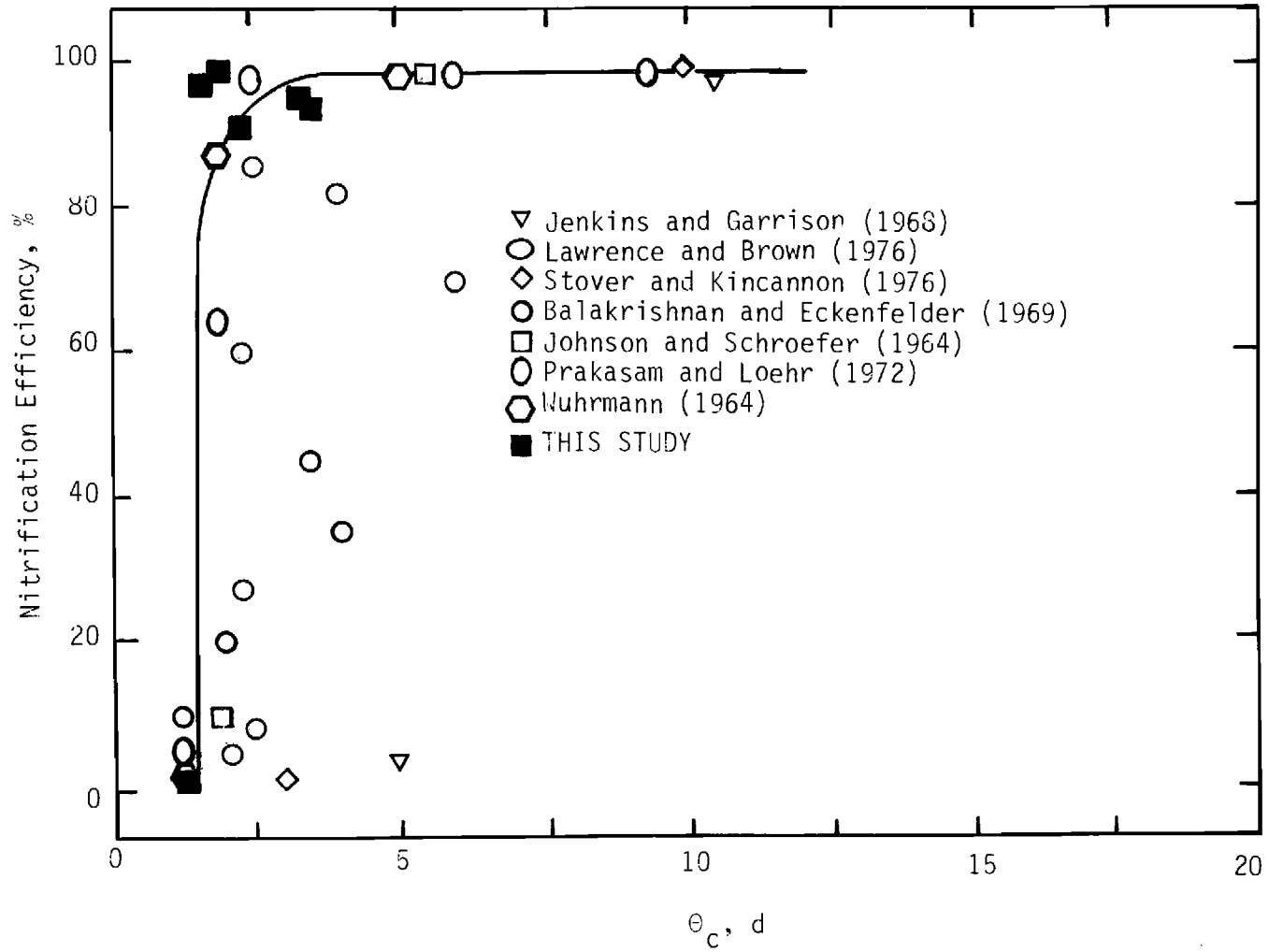


Figure 25. Comparison of Nitrification Efficiency and θ_c for Suspended- and Attached-Growth Biological Treatment Systems

Data presented for Phase B indicated that the hydrolysis of organically bound nitrogen, i.e. glycine, did not inhibit the nitrification process since virtually complete nitrification occurred. It is therefore indicated that nitrification can be achieved in RBC systems with influent organic-nitrogen much the same as when nitrification occurred with ammonia-nitrogen as the primary nitrogen source.

Oxidation of Carbonaceous Organic Matter

As indicated in Figure 26, the performance of the RBC system with respect to the removal of organic matter was excellent. Organic removals during Phase A ranged from 83-88% at an average influent COD concentration of 55 mg/l. As θ_c was increased from 1.3 to 3.3 d during Phase A effluent COD concentrations decreased from approximately 10 to 6 mg/l. In addition, the performance of the system remained at similar levels as influent organic loading increased from approximately 4.3 g COD/m²·d during Phase A to 13.7 g COD/m²·d during Experimental Run C4. When examined as presented in Figure 26, the soluble effluent COD concentrations for all experimental runs generally decreased with increasing θ_c . There, however, was no noticeable effect of influent COD concentration on effluent quality. This response was the apparent result of the biofilm in the RBC system being substrate-limited and not oxygen-limited. As influent organic loading increased, substrate diffused further into the biofilm, expanding the total depth of the active layer. The biofilm however did not apparently become oxygen-limited as a result of the depth of the active portion of the biofilm.

Biomass Production

Sludge production data presented in Table 11 indicate that yield coefficients ranged from 0.29 to 0.53 g TSS/g COD removed, values typical of those for domestic wastewaters. The nitrogen content of the wasted biomass, in addition,

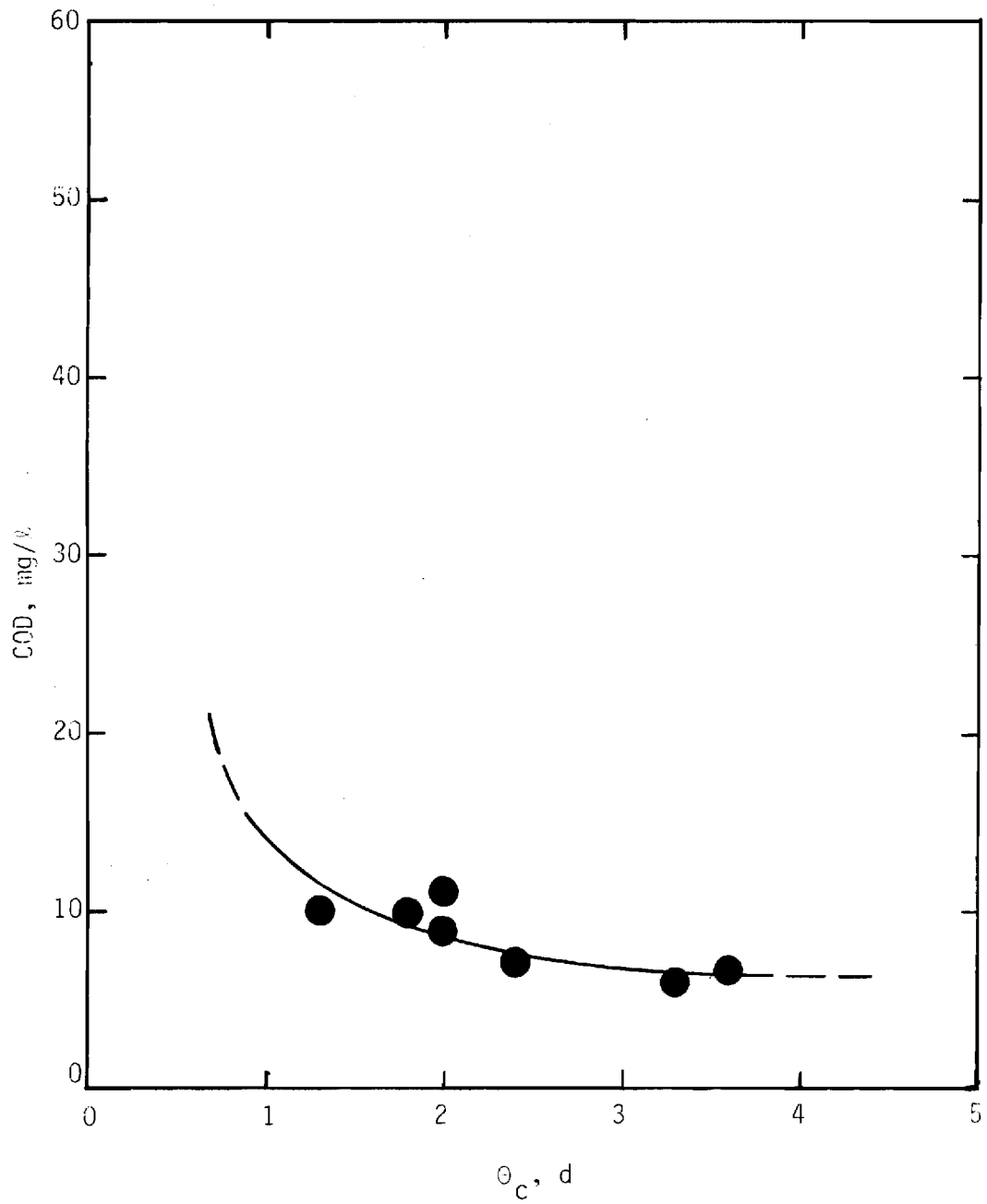


Figure 26. Soluble Effluent COD Concentration as a Function of θ_c

Table 11

Characteristics of Wasted Biomass

Experimental Run	A2	A3	A4	B1	C1	C2	C3	C4
Biomass Production Rate (g/d m ²)	1.23	2.33	1.4	1.97	1.61	3.09	2.58	3.92
Yield Coefficient (gTSS/gCOD _R)	0.40	0.53	0.37	0.39	0.29	0.36	0.23	0.31
Biomass Nitrogen Content (%)	9.7	8.7	13.5	12.3	11.9	12.7	11.8	12.2
Average MLSS (mg/l)	33	212	242	290	198	363	330	487

ranged from 8.7-13.5%. The data also indicate a slight trend of increasing nitrogen content of sludge solids at increasing values of θ_c .

The average mixed liquor suspended solids concentrations increased as θ_c increased, as evidenced by the increases noted for Runs A2-A4. MLSS values also increased with organic loading as indicated by the data for Runs C1-C4. Finally the MLSS concentrations were typical of those found in pilot and full scale RBC systems (Antonie, 1976).

CONCLUSIONS

1. The laboratory reactor system was utilized effectively to evaluate nitrification in RBC systems due primarily to the close control provided on biomass solids within the reactor. As a result, the use of the cell residence time concept was effective in characterizing the microbial population within the reactor system.

2. Nitrification was experimentally shown to occur at $\theta_c \geq 1.8d$. This was in agreement with values predicted by data for pure cultures of Nitrosomonas and Nitrobacter and by operational data for attached- and suspended-growth biological treatment systems.

3. Concurrent removal of carbonaceous matter was achieved with nitrification at organic loading as high as $13.7 \text{ g COD/m}^2 \cdot \text{d}$ at hydraulic and nitrogen loading of $70 \text{ l/m}^2 \cdot \text{d}$ and $1.4 \text{ g N/m}^2 \cdot \text{d}$, respectively.

4. Hydrolysis of organically-bound nitrogen, as glycine, did not limit nitrification kinetics.

5. The θ_c concept was confirmed as a viable technique for modeling RBC reactor systems. Additional parameters, such as dissolved oxygen, disc rotational speed and temperature, should however be examined in greater detail before an effective system model can be developed.

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