An Assessment of Nitrification
in an RBC Plant

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ABSTRACT

To evaluate the nitrification efficiency in a rotating biological contactor (RBC) system, samples were collected from the wastewater treatment plant at Princeton, Illinois, twice a week for over a year. Significant nitrification occurred in stages 3 and 4. Stage 3 achieved an average soluble ammonia nitrogen (SNH_3-N) removal efficiency of 40% for an influent concentration ranging from 2 to 18 mg/L with a stage loading rate of 0.4 lb SNH_3-N/d/1000 sq ft. Stage 4 removed an additional 64% from the stage 3 effluent. A linear relationship was found between effluent and influent SNH_3-N concentrations at stages 3 and 4. These relationships are proposed for RBC design purposes.

Significant nitrification will occur when the soluble BOD_5 ($SBOD_5$) concentration is reduced to about 15 mg/L. With favorable conditions, both $SBOD_5$ and SNH_3-N removals took place at stages 1 and 2 for some periods. Practically no nitrification occurred in stage 5 or in the secondary clarifier.

INTRODUCTION

Wastewater effluent ammonia nitrogen (NH_3-N) limitations have been promulgated in Illinois and many other states. To comply with Illinois effluent standards, a rotating biological contactor (RBC) system was installed in 1979 at Princeton, Illinois. The facility was designed to meet effluent requirements of 10 mg/L 5-day total biochemical oxygen demand ($TBOD_5$), 12 mg/L total suspended solids (TSS), and 1.5 mg/L NH_3-N.

Objectives and Report Plan

The purpose of this study was to evaluate the performance of each stage in a train of an RBC system in terms of NH_3-N removal. The specific objectives of the study were: 1) to determine the removal efficiency of soluble ammonia nitrogen (SNH_3-N)

and soluble organic nitrogen (SOrg-N) for each stage; 2) to ascertain the relationship between SNH_3-N reduction and alkalinity, nitrate nitrogen (NO_3-N), pH, and other factors; 3) to develop a nitrogen budget for each stage; and 4) to characterize NH_3-N removal kinetics for design purposes.

This report describes the RBC system at Princeton and the procedures and results of the study. It also includes a proposed RBC design approach.

RBC SYSTEM

A layout of the wastewater treatment facility is shown in figure 1. The RBC installation consists of two 5-stage trains (figure 2) operating in parallel. Each of the first two stages in each train provides 100,000 sq ft (9290 m^2) of media (standard density) and is designed for BOD_5 removal. Each of the remaining six contactors has 150,000 sq ft (13,935 m^2) of media (high density) and is provided for NH_3-N removal. Each unit rotates at 1.6 rpm with a mechanical drive and is covered.

During dry weather flow the settled sewage from the primary clarifiers flows by gravity to the RBC units. During high flow periods a portion of the settled sewage is routed to a trickling filter and then back to the RBC units. There are two secondary clarifiers following the RBC units. The secondary clarifier effluent is further treated by rapid sand filters. Effluent from the filters is then chlorinated before being
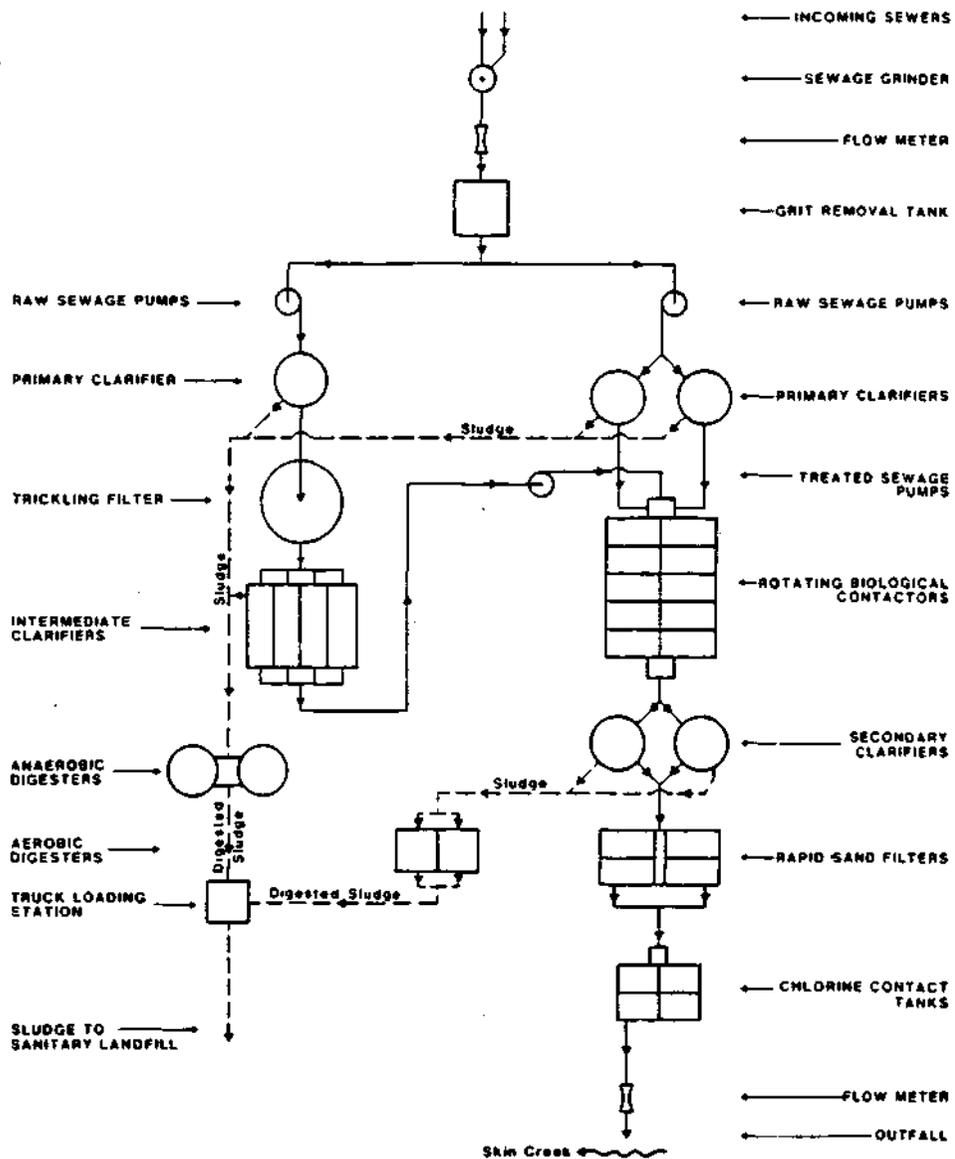
Figure 1. Schematic flow diagram of Princeton wastewater treatment plant
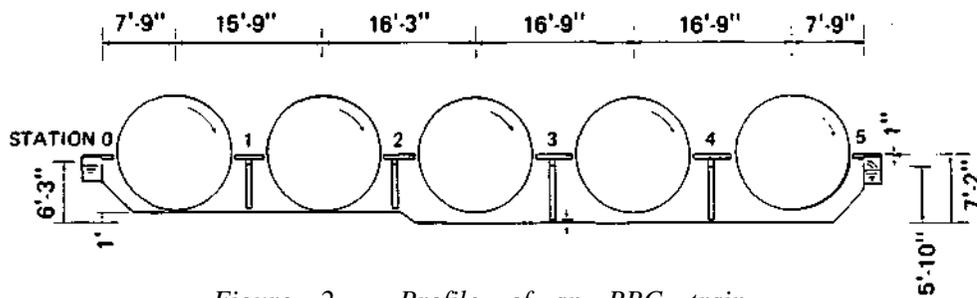
Figure 2. Profile of an RBC train

discharged to the receiving stream. The pertinent design features for the RBC system are as follows:

Design flow:	1.63 mgd	(6170 m^3/d)
Peak flow:	4.58 mgd	(17300 m^3/d)
Hydraulic loading:	1.25 gpd/sq ft	(51 L/m^2/d)
Total BOD_5 loading:	1.12 lb/d/1000 sq ft	(55 g/m^2/d)
Ammonia-N loading:	0.10 lb/d/1000 sq ft	(0.5 g/m^2/d)
Detention time:	2 hours	

METHODS

The south train was arbitrarily selected for the study. Wastewater samples were collected with ISCO automatic samplers from the RBC influent, the effluents of each stage of the train, and the effluent of a secondary clarifier. Samples were composited from hourly collections obtained over two 24-hour periods each week. Analyses were performed for total and soluble BOD_5, total and soluble ammonia nitrogen, organic nitrogen, nitrate nitrogen (NO_3-N), nitrite nitrogen (NO_2-N), and total alkalinity. Temperature, dissolved oxygen, and pH were recorded in the field. Analyses were performed on each composite sample in accordance with *Standard Methods* (1).

The period of study extended from January 1981 to the middle of January 1982. Except during the periods July 8-22, 1981, and January 1982 the wastewater flow was about equally divided between the two trains. During these two periods, the wastewater flow was intentionally routed only through the south train in order to stress the system.

RESULTS AND DISCUSSION

RBC Influent

A summary of wastewater quality for the RBC influent is shown in table 1. The temporal variations in wastewater temperature, flow rate, dissolved oxygen (DO), soluble BOD_5 ($SBOD_5$), and SNH_3-N for the RBC influent are depicted in figure 3. The temperature was generally low (7.5-12.1° C) from January through March 1981 with an average of 10.3° C. It increased steadily to a high of 21° C in July 1981 and then varied

Table 1. Water Quality of RBC Influent at Princeton

	Winter	Spring	Sumner	Fall	Shock Periods	
	1981					
	1/6-2/18					
	11/9-12/28	2/23-5/20	5/25-9/8	9/14-11/4	7/8-7/22/81	1/5-13/82
Temperature, °C	7.5-16.2	9.2-15.0	15.5-20.8	17.0-20.9	19.9-21.1	10.5-11.9
DO, mg/L	0.8-4.5	2.70-7.88	0.72-5.25	0.05-3.91	0.23-2.64	4.01-4.84
PH	7.38-7.95	7.65-7.92	7.52-7.81	7.00-7.86	7.52-7.72	7.86-8.01
Alkalinity, mg/L as $CaCO_3$	223-323	245-291	234-299	253-310	265-289	286-297
TNH_3-N, mg/L	8.33-20.84	2.26-16.01	5.23-11.99	4.41-16.05	10.00-15.08	14.60-16.66
SNH_3-N, mg/L	8.07-19.96	1.65-15.86	4.06-11.17	4.41-15.17	9.58-14.93	14.60-15.09
NO_3-N, mg/L	0.07-3.74	0.16-4.84	0.10-4.65	0.01-2.15	0.13-2.16	0.90-1.46
NO_2-N, mg/L	0.04-0.44	0.12-0.48	0.05-0.62	0.01-0.27	0.01-0.41	0.21-0.62
TKN, mg/L	12.82-32.24	5.82-22.05	6.82-19.52	12.29-25.05	12.58-24.40	22.61-29.38
SKN, mg/L	9.11-22.96	3.88-17.46	5.23-14.11	8.23-18.56	10.64-17.64	17-59-20.05
Solids, mg/L						
Dissolved	400-540	444-528	424-544	440-540	472-504	504-616
Suspended	52-160	32-214	30-228	50-132	52-184	76-128
Volatile Susp.	48-116	24-150	26-144	46-108	46-120	64-120
Settleable	0-2.4	0.2-5.0	0.1-7.5	0.02-1.60	0.4-1.20	0.05-0.28
$TBOD_5$, mg/L	47.6-126.7	25.5-107.8	24.9-74.6	28.8-96.5	32,2-93.3	91.6-108.6
$SBOD_5$, mg/L	12.7-56.8	6.1-32.7	3.8-28.6	12.8-44.0	8.6-37.4	35.9-55.8
Flow, mgd	0.45-0.87	0.54-2.04	0.71-1.51	0.56-1.01	1.0-1.12	1.15-1.24
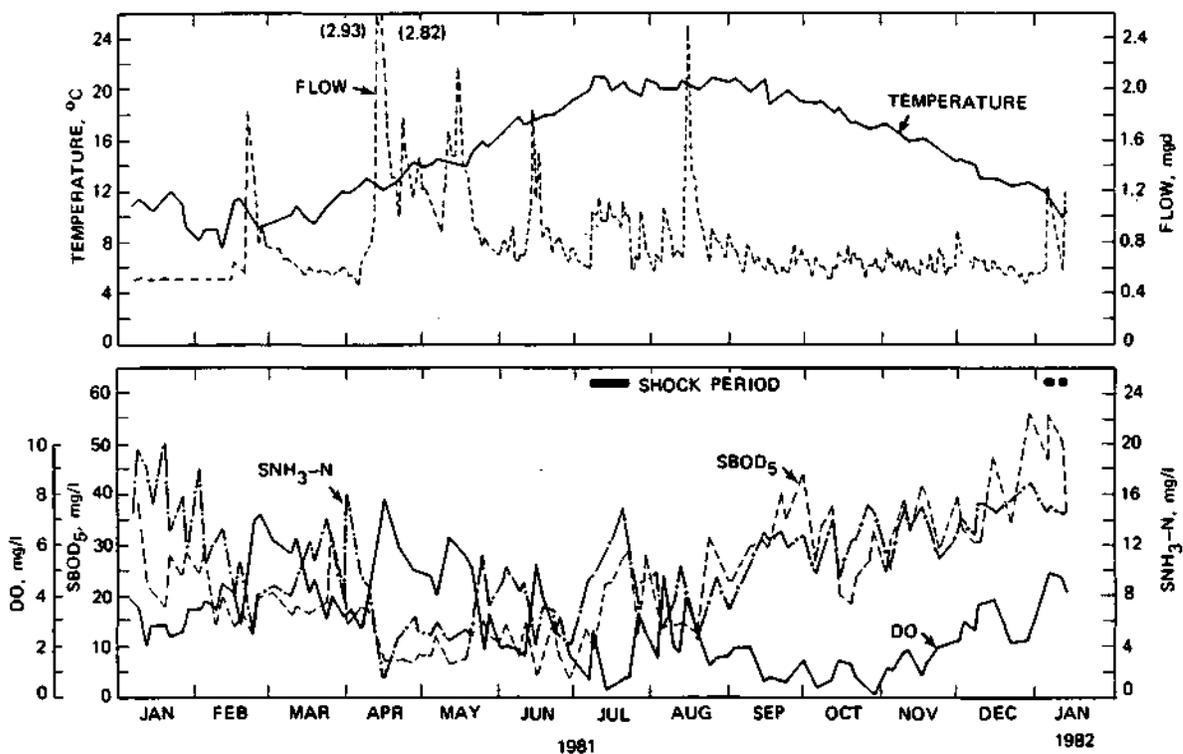
Figure 3. Temporal variations in observed wastewater flow, temperature, dissolved oxygen, soluble BOD_5, and soluble NH_3-N for RBC influent

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only $\pm 2^\circ$ C up through September 1981. After September 1981, temperatures steadily decreased. For this investigation 12° C and 19° C temperatures were found applicable for defining seasonal changes.

Flow rates generally ranged from 0.5 to 0.8 mgd (1900-3000 m^3/d) except for some peak flows (figure 3). High flows were due to precipitation and infiltration.

Dissolved oxygen concentrations in the RBC influent were frequently below 1.0 mg/L during the July shock period and during September and October 1981 (figure 3). The DO values were high during the spring and cold weather periods as well as during high flow periods. Both $SBOD_5$ (figure 3) and $TBOD_5$ concentrations were generally low during the spring, and high for the summer and fall seasons.

Ammonia and Kjeldahl nitrogen in both total and soluble forms were quite variable, with lower concentrations occurring in April and May 1981. This was probably a result of the spring snowmelt and infiltration. Ammonia nitrogen concentrations were high during winter and fall (figure 3).

The RBC influent contained high total alkalinity ranging from 220 to 300 mg/L as $CaCO_3$. The pH values ranged from 7.4 to 8.0. Soluble phosphate varied from 0.9 to 4.5 mg/L as P. While sulfate concentrations were for the most part in the range of 40 to 80 mg/L, the concentrations of TSS (30-230 mg/L) and volatile suspended solids (VSS 24-150 mg/L) displayed no seasonal patterns. However, settleable solids varied widely from sample to sample (0.02-7.50 mg/L).

SNH_3-N to TNH_3-N and SOrg-N to TOrg-N Ratios

The ratios of SNH_3-N to TNH_3-N and SOrg-N to TOrg-N were determined for each sample. For SNH_3-N/TNH_3-N the range was generally from 0.81 to 1.00. Many of the samples had ratios of 1.00. Low SNH_3-N/TNH_3-N ratios of about 0.7 were observed for the period April 15-22, 1981, and were the result of high flow. Under normal operations, the average SNH_3-N/TNH_3-N ratios were 0.95, 0.94, 0.93, 0.91, 0.85, and 0.69 for the RBC influent and the five stage effluents, respectively. The annual average ratios of SOrg-N/TOrg-N for the corresponding samples were 0.32, 0.28, 0.23, 0.20, 0.22, and 0.09. For practical purposes TNH_3-N and SNH_3-N can be considered to be the same in the RBC influent and the first four stages. SNH_3-N and SOrg-N have been used for the purposes of evaluation in this study because they are the more readily available forms of nitrogen for use in microbial synthesis and nitrification.

Loading

Table 2 summarizes the conventional design loading and observed loading rates for normal operations and the two shock periods in the south train of the RBC system. The average values for normal operations were slightly lower than design rates except for the $TBOD_5$ loading rate, which was about one-half of design.

The temporal variations in hydraulic and mass loading rates are shown in figure 4. Hydraulic loading was high in the spring (April-June 1981) and low (72% of design) during winter (January-March 1981). The average hydraulic loading for the summer and fall 1981 seasons were, respectively, close to and 18% lower than design values. During the shock periods the average hydraulic loadings exceeded design values by 30% and 40% for July 1981 and January 1982, respectively.

In general, SNH_3-N loading rates were low during spring and high during the fall (figure 4). The average loading rates for certain periods are listed in figure 5 in parentheses. Average SNH3-N loadings during the July 1981 and January 1982 shock periods were respectively 60% and 130% higher than design values.

Figure 4 also shows that $SBOD_5$ loading rates during the first half of 1981 ranged from 0.1 to 0.2 lb/d/1000 sq ft (0.49 to 0.98 $g/m^2/d$). The $SBOD_5$ loadings during the last half of 1981 were about double those of the first half of the year. During the shock periods, the $SBOD_5$ loadings were 1.6 to 3.3 times higher than those of normal operations (table 2).

Table 2.   Loading Rates in South Train of RBC System

Parameter	Design	Annual (normal operations) Average	Range	Shock periods 7/8-22/1981 Average	Range	1/5-13/1982 Average	Range
Flow, mgd	0.82	0.76	0.45-2.04	1.06	1.00-1.12	1.19	1.15-1.23
Detention time, hours	2.0	2.16	0.80-3.64	1.56	1.45-1.64	1.38	1.33-1.43
Hydraulic loading (gpd/sq ft)	1.25	1.17	0.69-3.14	1.62	1.53-1.72	1.83	1.78-1.90
SNH_3-N loading (lB/d/1000 sq ft)	0.10	0.092	0.043-0.14	0.16	0.13-0.21	0.23	0.22-0.23
$SBOD_5$ loading (lb/d/1000 sq ft)	-	0.21	0.07-0.41	0.34	0.11-0.54	0.70	0.54-0.83
$TBOD_5$ loading (lb/d/1000 sq ft)	1.12	0.62	0.31-1.02	0.98	0.42-1.29	1.51	1.43-1.63

Note: 1 gpd/sq ft = 40.74 l/m^2-d; 1 lb/d/1000 sq ft = 4.88 g/m^2-d
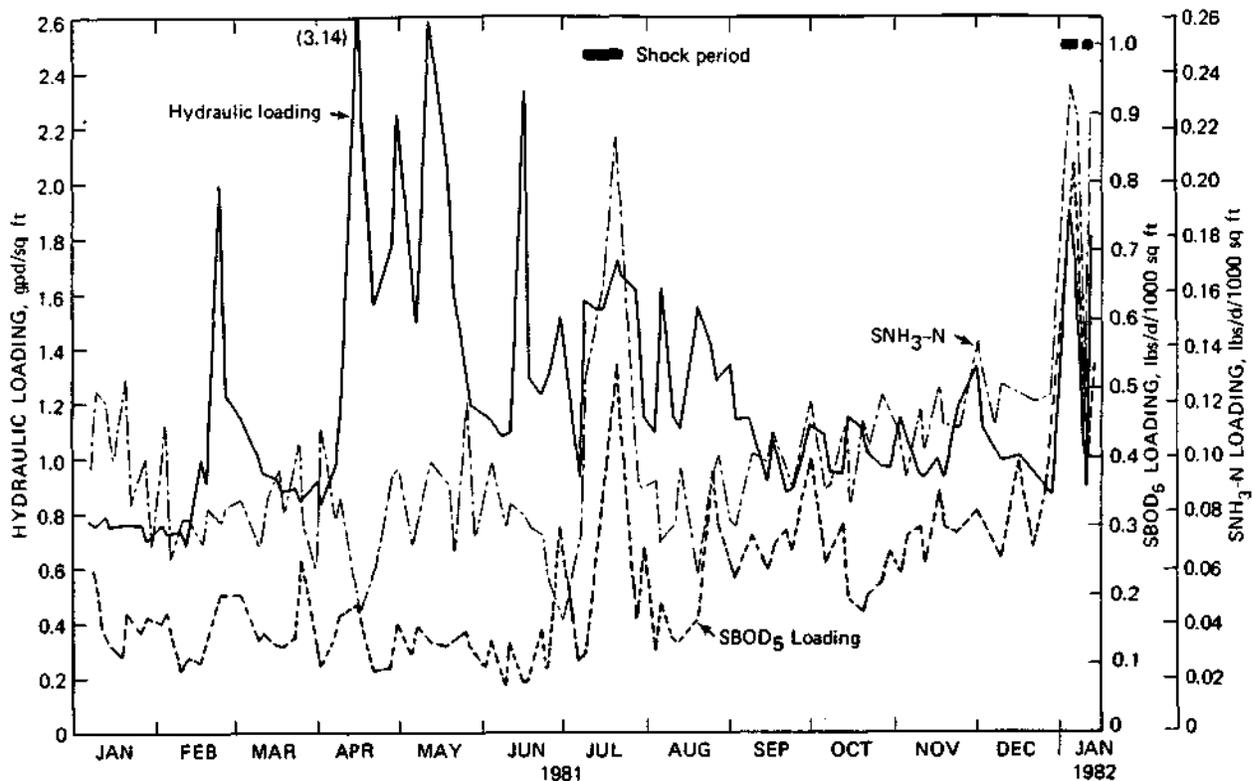
Figure 4. Temporal variations in soluble NH_3-N and BOD_5 loading rates and hydraulic loading rates

Stages 1 and 2

As previously stated, stages 1 and 2 are designated for BOD_5 removal. In addition to carbon, microorganisms require nitrogen and phosphorus for growth. During the bacterial oxidation of carbonaceous materials, some nitrogen and phosphorus from the wastewater are assimilated by the cells. The BOD_5:N:P ratio is about 100:5:1 (2,3). Thus, the removal of NH_3-N during the biological treatment of wastewater may be due, in part, to cell assimilation, and not necessarily only to nitrification.

Nitrification is a two-step oxidation process converting ammonia to nitrite and then nitrite to nitrate. On the other hand, ammonia is also generated by the hydrolysis of Org-N in biological growth. Therefore, cell assimilation and hydrolysis should be considered in the evaluation of nitrification by biological treatment.

The observed data indicated that SNH_3-N increased slightly on many occasions as wastewater passed through stage 1. These increases were due to the production of

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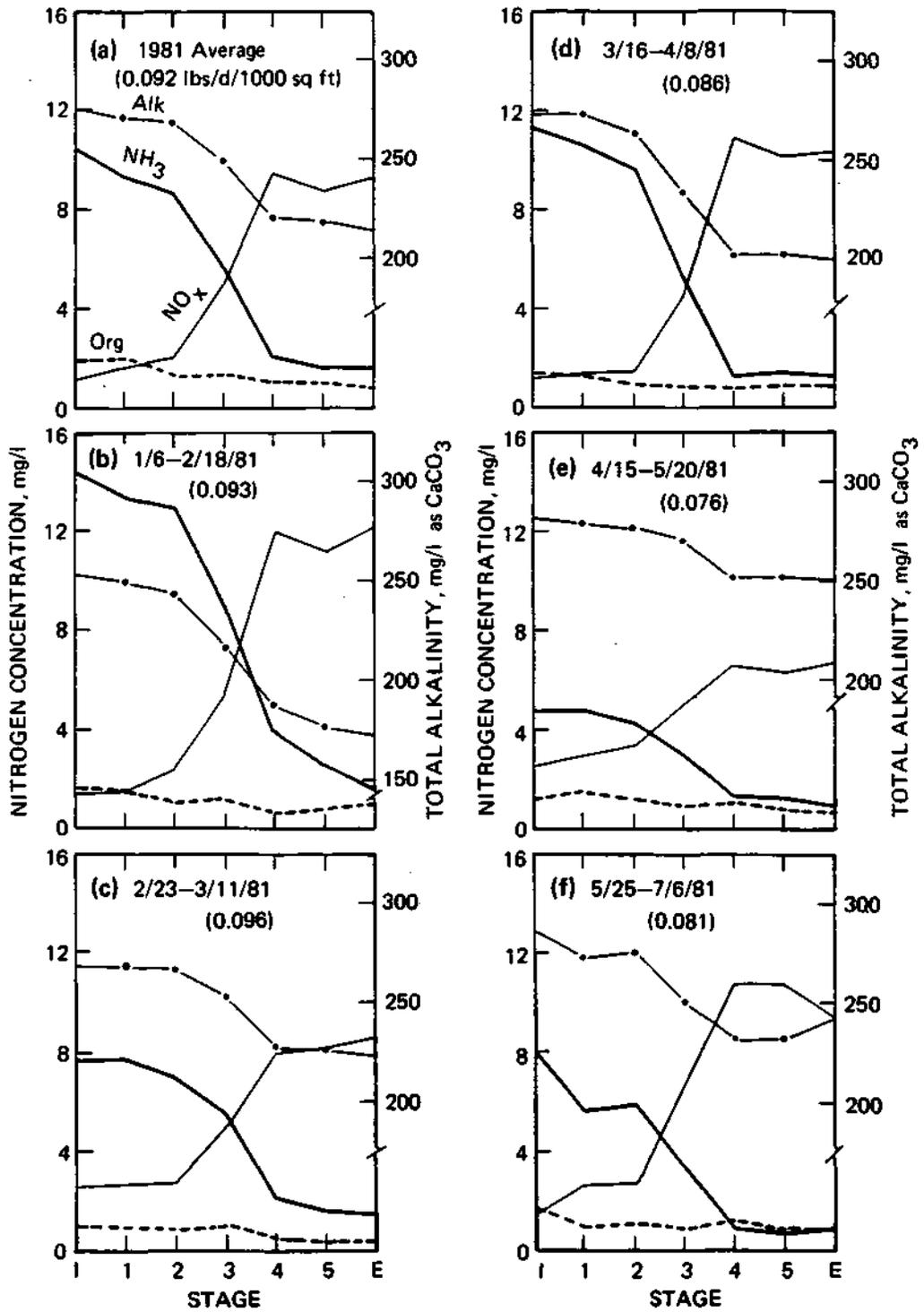
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*Figure   5.   Profiles   of   average   SNH₃-N,   soluble   organic-N,   NOₓ-N,
          and   alkalinity   concentrations   through   the   RBC   system
              (Average   loading   rates   shown   in   parentheses)*

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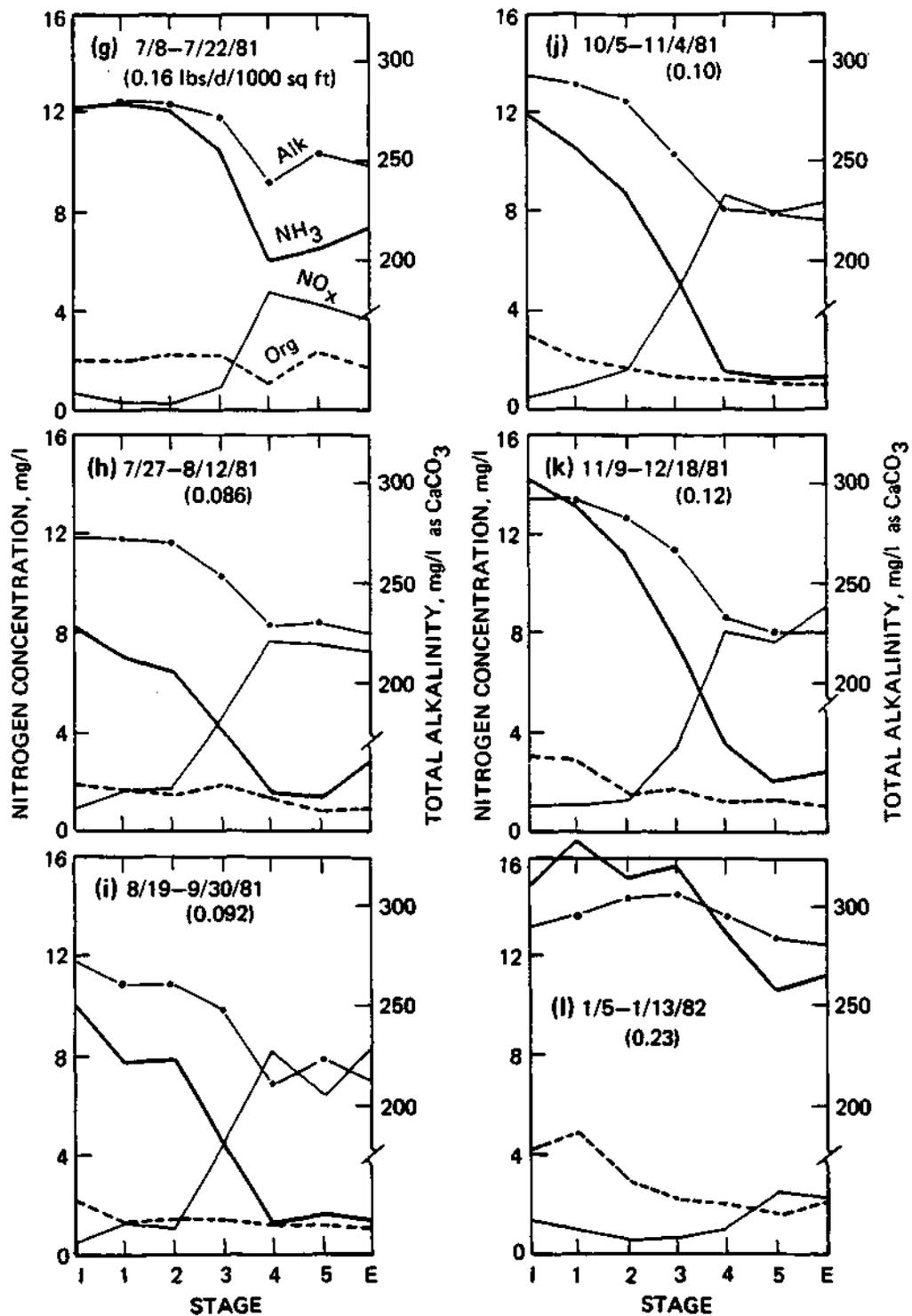
Figure 5. Concluded

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NH_3-N from the hydrolysis of Org-N. Similar observations have been reported by others (4-7). According to Mueller et al. (8), SOrg-N is hydrolyzed by both heterotrophic and autotrophic species.

Nevertheless, there was some SNH_3-N reduction in stage 1 during January 1981 and May through early November 1981, excluding the July shock period. The cumulative removal curves for stage 2, shown in figure 5, are also representative of stage 1 with the exception of November and December 1981. Nitrification occurred at stage 2 during October 5 - December 18, 1981 (figures 5j and 5k). On many occasions, however, the SNH_3-N concentration in stage 2 had also increased from that of stage 1. This indicated that hydrolysis of Org-N must also have occurred at stage 2 during certain periods.

For the period May 25 - July 6, 1981, an average of 4.7 mg/L $SBOD_5$ (1.1 mg/L - 6.4 mg/L) was removed from the wastewater during stage 1 (9). Five percent of 4.7 mg/L $SBOD_5$, or 0.24 mg/L N, would have been assimilated by heterotrophic organisms. An average of 2.31 mg/L SNH_3-N was removed at stage 1 (figure 5f) as a result of nitrification, assimilation, and hydrolysis. SOrg-N was reduced by 0.61 mg/L (figure 5f). Thus nitrification removed 2.68 mg/L SNH_3-N (2.31 - 0.24 + 0.61) during stage 1.

It was evident that carbonaceous oxidation and nitrification were occurring simultaneously in the first stage during this period. Miller et al. (7) observed a similar phenomenon. It may be concluded that under certain environmental conditions nitrifiers and heterotrophic bacteria can co-exist in an RBC unit. The relative concentrations of the two groups at any specific point in an RBC system are a function of the nutrient concentrations and other environmental factors.

According to the USEPA (10), if the $TBOD_5$/TKN ratio is greater than 5.0 the treatment system is classified as a combined carbon oxidation-nitrification process. If the $TBOD_5$/TKN ratio is less than 3.0, the system is classified as a separate stage nitrification process. There is no practical example for a range of $TBOD_5$/TKN between 3.0 and 5.0. Treatment facilities with the $TBOD_5$/TKN ratio between 3.0 and 5.0 could be considered to have an intermediate degree of separation between carbon oxidation and nitrification (10). This is the case for Princeton's RBC system. At Princeton the ratio of $TBOD_5$ to total Kjeldahl nitrogen (TKN) at the RBC influent ranged, for the most part, from 3.0 to 5.0 with an annual average of 3.7. The majority of ratios of $SBOD_5$ to soluble Kjeldahl nitrogen (SKN) fell between 1.0 and 2.0 with an average of 1.8.

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Nitrification in Stages 3 and 4

Biological nitrification, which converts ammonia to nitrate, is carried out by two predominant genera of autotrophic nitrifying bacteria that derive their energy from the oxidation of inorganic nitrogen under aerobic conditions. <u>Nitrosomonas</u> converts ammonia to nitrite, and <u>Nitrobacter</u> then transforms nitrite to nitrate. The development and predominance of nitrifying organisms in an RBC system is a natural phenomenon, requiring no special management except provision for appropriate hydraulic and mass loading rates.

Heterotrophic nitrification also can occur when nitrite and/or nitrate are produced from organic or inorganic compounds by heterotrophic organisms (more than 100 species including fungi). However, the amount of oxidized nitrogen formed by heterotrophic organisms is relatively small (11).

The growth rate of nitrifying bacteria is much less than that of heterotrophic bacteria. Nitrifying bacteria have a longer generation time of at least 10 to 30 hours (11,12). They are also much more sensitive to such environmental conditions as temperature, pH, DO, and substrate concentrations (13,14). The growth rate of nitrite-oxidizers is much greater than that of ammonia-oxidizers (10,11).

Figure 5 shows that SNH_3-N concentrations are significantly reduced as wastewater passes through stages 3 and 4. Under normal operations, Org-N concentrations are generally less than 2 mg/L, with only minimal concentration changes across stages 3 and 4. The assimilation of SNH_3-N for carbon oxidation is insignificant in these two stages. Therefore, for purposes of this study, a reduction in SNH_3-N is considered to be due only to nitrification.

A wide range, from a high of 19 mg/L SNH_3-N during the winter to a low of 2 mg/L for the spring, was observed in stage 3 influent. High SNH_3-N concentrations in the RBC influent during the July shock period were associated with the normal industrial wastewater dilutions due to industrial shutdown for vacations. The annual mean stage loading rate was 0.40 lb SNH_3-N/d/1000 sq ft (1.95 $g/m^2/d$).

An examination of figures 5 and 6 reveals that significant nitrification occurred in both stages 3 and 4. Table 3 summarizes the performance efficiencies at stages 3 and 4 based on their influent and effluent SNH_3-N concentrations under normal operations. Poor performance in both stages was observed during the period April 15 - May 20, 1981, due to the combination of low influent SNH_3-N concentrations, high flows, and
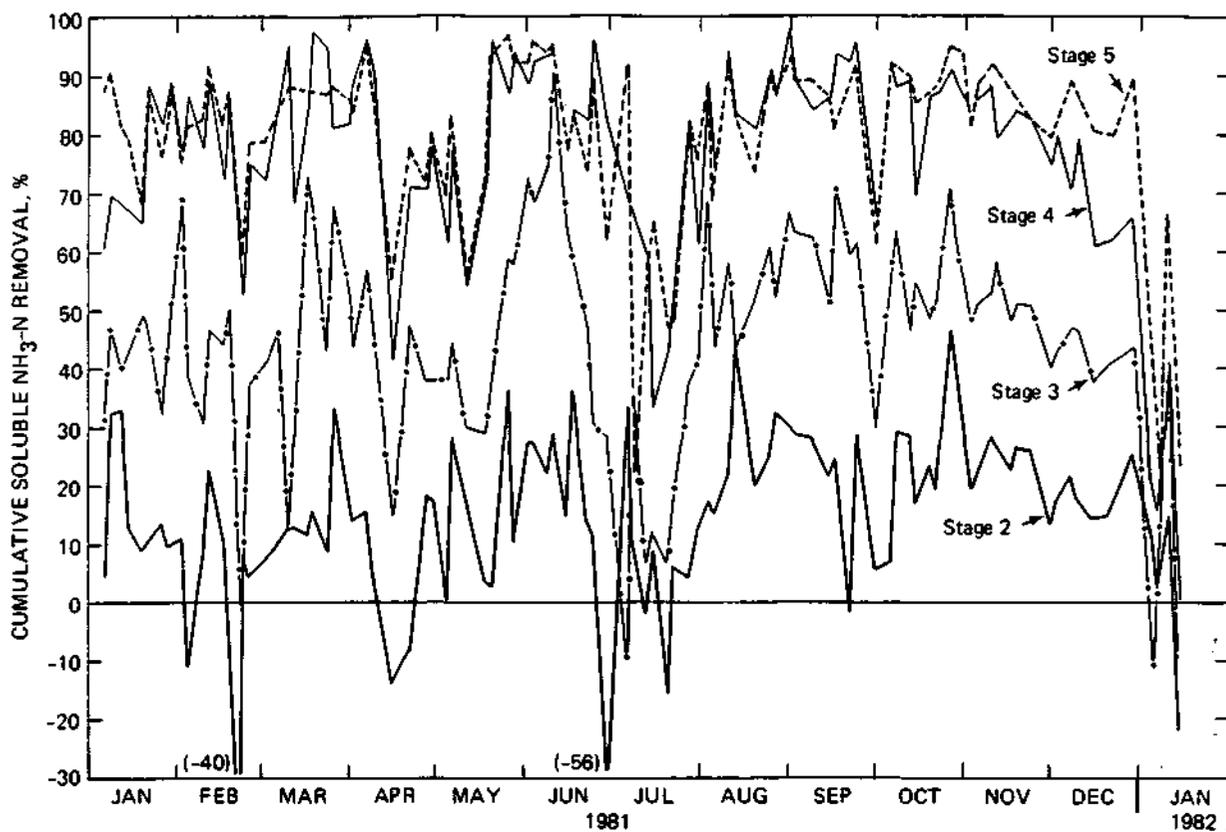
Figure   6.   Soluble   ammonia   nitrogen   removal   through   the   RBC   system

Table 3.   Soluble NH_3-N Removal in Stages 3 and 4

Period	Stage 3, %		Stage 4, %		Mean temp., °C
	Mean	Range	Mean	Range	
19 81 average (normal operations)	40	11-87	64	22-95	15.1
1/6-2/18/81	36	11-65	59	22-92	10.3
2/23-3/11/81	36	34-37	59	41-89	9.9
3/16-4/8/81	47	34-69	76	51-92	11.5
4/15-5/20/81	29	15-42	54	31-93	13.9
5/25-7/6/81	52	21-87	69	40-95	17.6
7/8-7/22/81*	13	3-26	43	24-57	20.2
7/27-8/12/81	42	32-62	63	33-87	20.1
8/19-9/30/81	44	25-62	73	52-97	20.0
10/5-11/4/81	40	24-50	69	34-80	17.7
11/9-12/28/81	31	25-42	57	36-75	14.4
1/5-1/13/82*	5	(-24) 2-8	17	11-29	11.1

* Induced shock periods

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short hydraulic detention times. The annual mean removal efficiencies were 40 and 64% of the incoming SNH_3-N concentrations at stages 3 and 4, respectively. Similar situations have been observed by others (7,14,15). In contrast, for most pilot studies, the majority of ammonia (72-97%) in wastewater is removed during the first stage of the RBC system (16-19). At Migazaki City, Japan, nitrification was reported to be nearly complete after only two of four RBC stages of treatment, with 70% removal at stage 1 (20).

Factors Affecting Nitrification

Effect of BOD$_5$

The literature indicates that nitrification is sensitive to organic loading and depends on the carbon removal RBC units. This is because nitrifiers have a much slower growth rate than heterotrophs. There are differences in the reported values for the critical concentrations of organic matter. They are as follows: TBOD$_5$ approaching 30 mg/L (4,20-22); TBOD$_5$ reaching 20 mg/L (6); SBOD$_5$ less than 20 mg/L (23); SBOD$_5$ reduced to 15 mg/L (22,24); SBOD$_5$ around 10 mg/L (4); COD less than 50 mg/L (14 mg/L TBOD$_5$) (5); and an organic loading below 20 g COD/m^2/d (25). These differences are due to the different wastewaters used with various bench- and pilot-scale setups.

At Princeton, significant nitrification generally commenced at stage 3. The SBOD$_5$ concentrations in the stage 3 influent ranged from 4.4 to 18 mg/L with an annual mean of 9.2 mg/L under normal operations. As previously mentioned, from late May 1981 through June 1981 nitrification also occurred during stage 1 when the SBOD$_5$ concentrations in the RBC influent were generally below 15 mg/L. It can be concluded from the Princeton data that nitrification can occur when the SBOD$_5$ concentration is reduced to 15 mg/L.

Temperature

The rate of biological nitrification is strongly influenced by temperature. Nitrification is much more sensitive to temperature than the carbonaceous BOD removal process. However, nitrification in the RBC process is generally not as sensitive to temperature as that in the activated sludge process.

Table 3 suggests the effect of wastewater temperature on RBC performance. Under normal operations, the lower the temperature, the lower the removal efficiency.

A pilot-plant study by Lue-Hing et al. (20) with very high strength NH_3-N wastewater indicated that the maximum NH_3-N removal rate in the first stage ranged from 95 lb/d/ 1000 cu ft (1.5 $kg/m^3/d$) at 9° C to 170 lb/d/1000 cu ft (2.7 $kg/m^3/d$) at 20° C. There was almost a two-fold increase at the higher temperature. This is in agreement with the suggestion in the literature that a nitrification rate decrease of about 50% occurs for each 10° C drop in wastewater temperature below 30° C (10). According to Murphy et al. (26), between 10° C and 20° C the Q_{10} (the ratio of reaction rates at two temperature values with a 10° C difference) is 2.33. The temperature sensitivity of an RBC system is similar to that of an activated sludge process with a 10-day solids retention time (SRT) (26).

The optimum temperature range for nitrification is 30-36° C (13,14,27). Bushwell et al. (27) reported that there is little growth of Nitrosomonas below 5° C. It has been shown through laboratory tests that at 5° C there is no NH_3-N removal by RBC units (28). Nitrobacter has not been found to grow at temperatures below 4° C or above 45° C (14). Nevertheless, there are no municipal wastewater treatment plants that can afford to adjust temperature to the optimum. At Princeton, the plant was operated in the range of 7.5° C to 21° C, which is below optimum temperatures for nitrification.

It is apparent from figures 5 and 6 and tables 3 and 4 that nitrification deteriorated during the January 1982 shock period from a combination of low temperature (11° C) and high hydraulic (1.83 gpd/sq ft; 75 $L/m^2/d$) and high SNH_3-N (0.23 lb/d/1000 sq ft; 1.12 $g/m^2/d$) loading rates. During this period, there was no nitrification occurring at stage 3 and little nitrification at stage 4. During the July 1981 shock period there was little SNH3-N removal occurring at stage 3 and somewhat more occurring at stage 4 (table 3 and figures 5 and 6). These were affected only by high hydraulic and mass loadings. Due to the shorter retention time, carbonaceous-removal bacteria took over as the major constituent of the biomass in stages 3 and 4, and nitrifying bacteria lost out due to the competition.

A report of an RBC pilot-plant study at Columbus, Indiana [Hao and Hendricks (29)] reported that during a cold period (January 26 - March 11, 1975) 59% of NH_3-N removal was achieved at 2.5 gpd/sq ft (102 $L/m^2/d$), and 80 to 93% NH_3-N removal was achieved at 1.5 gpd/sq ft (61 $L/m^2/d$) during other periods. Similarly, Richard et al. (30) claimed that NH_3-N removal by filtration through a submerged biological filter, using a pozzolana filter medium, was directly proportional to temperature and retention time.

Table 4. Cumulative Percent Removal of Soluble NH_3-N
across RBC System

Period	1	2	3	4	5	Secondary clarifier
1981 averaqe (normal operations)	9	17	44	77	78	83
1/6-2/18/81	6	8	36	70	77	86
2/23-3/11/81	1	7	18	71	67	74
3/16-4/8/81	5	17	55	89	86	89
4/15-5/20/81	-2	12	41	59	68	79
5/25-7/6/81	22	16	52	86	85	85
7/8-7/22/81*	-3	0.3	25	51	40	40
7/27-8/12/81	14	20	48	78	76	65
8/19-9/30/81	18	18	52	84	73	84
10/5-11/4/81	9	24	52	84	82	87
11/9-12/28/81	6	21	42	74	80	81
1/5-1/13/82*	-9	3	4	18	34	40
Winter (Jan - Mar)	4	11	38	74	77	84
Spring (Aor - June)	10	18	50	77	78	82
Summer (Jul - Sep)	13	14	40	75	68	70
Fall (Oct - Dec)	7	23	46	79	81	84

* Induced shock periods

Dissolved Oxygen

Theoretically, the oxygen demand for the biological oxidation of 1 mg/L of NH_3-N to NO_3-N is **4.57 mg/L.** This does not include any synthesis that may be brought about by the nitrifiers. At Princeton, the DO profiles across the RBC system followed a general pattern of rapid initial decline at stages 1, 2, and/or 3 with rapid recovery at each stage thereafter, especially stage 5. This pattern of a DO profile is similar to that reported by others (8,9,16,22).

The DO concentrations at stage 3 ranged from 0.6 to 5.3 mg/L with an annual mean of 2.3 mg/L. For stage 4, the DO concentrations varied from 1.4 to 4.8 mg/L with an annual mean of 2.6 mg/L. These values of DO concentrations at stages 3 and 4 show that DO was not a limiting factor for nitrification. On the average, DO concentration increased by 2.0 mg/L at stage 5.

pH

In the biological nitrification process, as ammonia is oxidized, DO and carbonate ($CO_3^=$) are used while nitrate and carbonic acid (H_2CO_3) are produced. This causes a destruction of alkalinity coupled with a depression in pH.

The pH has an important effect on the nitrification process. Normally the optimal pH for nitrite- and nitrate-forming bacteria is slightly alkaline, although no exact optimum pH value has been defined. Reported data show a wide range for optimum pH of 7.0 to 9.0 with maximum activity at an approximate pH of 8.5 (4,7,10-14,16-20,31-34). Below pH 7.0 the adverse effects on ammonia oxidation become pronounced (32,33). Nitrifiers, either in suspended growth or in attached thin film, are severely inhibited when there is a drop of pH below 6.8 (10). Nitrification ceases completely below a pH of 5.5 in submerged filters (13).

At Princeton the pH of the RBC influent ranged from 7.0 to 8.0. At the influent to stage 3, the pH ranged from 7.4 to 7.9. A slight decrease (7.1 to 7.8) in the stage 3 effluent was observed. These values are within the optimum range for ammonia oxidation.

The pH drop from stage to stage may be caused by the destruction of alkalinity by nitrification. Nevertheless, there was virtually no pH change in the wastewater through stage 4, where the maximum amount of nitrification occurred. The median values for pH at the seven sampling stations were all within a small range: 7.6 - 7.7. Typically, high buffering capacity (alkalinity: 220-320 mg/L as $CaCO_3$) of the RBC influent did not allow a significant pH depression due to alkalinity consumption during nitrification.

Alkalinity and NO_x-N

By stoichiometric calculations, 7.14 mg/L of alkalinity as $CaCO_3$ will be destroyed for every 1 mg/L of NH_3-N oxidized. However, Sherrard (35) claimed that this value was in error and should be less for biological nitrification because ammonia is incorporated into the biomass, resulting in a lesser quantity of NH_3-N available for oxidation to nitrate. The alkalinity destruction in a nitrifying activated sludge process is a function of the SRT and influent wastewater BOD_5:N:P ratio. Alleman and Irving (36) observed an extremely low value (2.73 mg/L of alkalinity as $CaCO_3$ per 1 mg/L NH_3-N oxidized) for nitrification in a sequencing (fill-and-draw configuration design) batch reactor. In the RBC system at Cadillac, Michigan, alkalinity declined 8.1 mg/L for each mg/L of NH_3-N oxidized (37,38).

The occurrence of nitrification in an RBC system can be determined from changes in alkalinity, pH, NO_x-N (NO_2-N plus NO_3-N), and biomass appearances. The significant reductions in alkalinity and the increases in NO_x-N concentrations shown in figure 5 support the fact that nitrification occurred in stages 3 and 4 during the July

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1981 shock period (figure 5g) and in stages 4 and 5 during the January 1982 shock period (figure 5ℓ).

The ratios of alkalinity reduction/SNH_3-N removal and NO_x-N formation/SN_3-N removal for each sample were calculated for stages 3 and 4. The average values for the selected periods are listed in table 5. A wide range of values was observed for both ratios at each of the two stages. Under normal operations, the ratios of alkalinity reduction to NH_3-N removal at stages 3 and 4 were 6.8 and 9.1, respectively. For stage 4, mean ratios were above the theoretical value of 7.1 except during the winter shock period.

Stoichiometrically, 1 mg/L of NO_x-N will be generated for each 1 mg/L of NH_3-N that is oxidized. The mean ratio of NO_x-N formation/SNH_3-N removal for stage 3 was 0.94, a reasonable value based on expectations, while for stage 4 it was 1.49 (table 5). The values in table 5 suggest that the actual amount of SNH_3-N oxidized at stage 4 was greater at all times than observed values, and the amount oxidized at stage 3 during the spring was greater than observed values. In contrast, in a pilot-plant study, Hao and Hendricks (18) found a low ratio (0.5) of NO_3-N formation/NH_3-N removal.

A question arises: where does the extra SNH_3-N come from? It is postulated that sloughed-off biomass from the previous stage breaks up and releases SNH_3-N as the

Table 5. Ratios of Alkalinity Destroyed and NO_x-N Formed to Soluble NH_3-N Removed

Period	Alkalinity destroyed/ SNH_3-N removed		NO_x-N formed/ SNH_3-N removed	
	Stage 3	Stage 4	Stage 3	Stage 4
1981 average	6.8	9.1	0.94	1.49
1/6-2/18/81	6.4	8.5	0.98	1.75
2/23-3/11/81	5.0	9.4	0.94	0.96
3/16-4/8/81	7.1	8.5	0.82	1.83
4/15-5/20-81	8.9	11.5	1.25	1.74
5/25-7/6/81	8.9	12.7	1.19	2.06
7/8-7/22/81*	-	8.3	-	1.04
7/27-8/12/81	5.8	9.2	0.89	1.40
8/19-9/30/81	4.9	11.4	1.14	1.31
10/5-11/4/81	7.4	8.5	0.90	1.03
11/9-12/28/81	4.9	8.0	0.66	1.15
1/5-1/13/82*	-	4.0	-	0.1.6

Induced shock periods

wastewater passes through each successive treatment unit. SH3-N also can be released from bottom sludge deposits. For stage 3, it seems that the use of alkalinity reduction or NO_x-N formation, rather than SNH_3-N removal, would be more useful in the evaluation of nitrification. Further discussion on this will be presented later.

Stage 5 and Secondary Clarifier

Inspection of table 4 and figures 5 and 6 shows that very little nitrification occurred at either stage 5 or the secondary ($2°$) clarifier. This was presumably due to the low influent SNH_3-N concentration. The average SNH_3-N for the stage 5 influent was 2.0 mg/L with a range of 0.2 to 14.5 mg/L. The stage 5 effluent averaged 1.6 mg/L SNH_3-N with a range of 0.2 to 6.4 mg/L. Under normal operations, the added benefit of stage 5 is not justifiable and can be said to be a wasteful unit operation without added removal efficiency. However, with the structural design of Princeton's RBC system, it would be very difficult to remove the fifth stage from service to save operational costs.

Figure 5 suggests that regardless of the variation in RBC influent SNH_3-N concentrations, under normal operations most secondary effluent SNH_3-N concentrations were less than 2.0 mg/L, with an annual mean of 1.5 mg/L and a range of values from 0.2 to 4.0 mg/L. High effluent SNH_3-N values were observed during the cold weather period (November through February). High values were also obtained during the July 1981 shock period. Recovery to lower normal effluent concentrations occurrred about one month after this shock period.

The overall SNH_3-N removal efficiency was only 83% (table 4). For the secondary clarifier, under normal operations there was no seasonal difference in the percent removal efficiencies. However, the effluent SNH_3-N concentrations were generally higher during cold weather.

During the July 1981 shock period, the hydraulic loadings exceeded design rate by 30%, and NH_3-N loading exceeded design by 60%. The average influent SNH_3-N was 12.2 mg/L. The average SNH_3-N concentration from the secondary clarifier was 7.3 mg/L. This resulted in an overall reduction of only 40%.

Total NH_3-N data obtained in this study were evaluated in terms of the 1.5 mg/L effluent standard. Forty-five out of 80 secondary effluent collections, under normal operations, were in compliance with the Illinois standard; i.e., 56% of the time the secondary effluent could have been discharged directly into the receiving stream.

Nitrogen Budget

The transformations of various nitrogen forms, with average concentration values, are depicted in figure 7 for Princeton's RBC system. The total nitrogen (sum of NH_3-N, Org-N, NO_2-N, and NO_3-N) entering the RBC system during normal operations remained somewhat constant at approximately 19 mg/L. Theoretically the total nitrogen concentration should remain constant as wastewater passes through the RBC units. In practice, though, some NH_3-N may be lost to the atmosphere by stripping. It has been reported that the degree of NH_3-N stripping in an RBC process is slight: approximately 4% of the total N budget (8,39). Ford et al. (14) reported that little NH_3-N stripping occurs in an activated sludge process when the mean pH value is less than 8.5. A noticeable loss of NH_3-N, most probably attributable to air stripping, was observed when the pH was 8.9. NO_2-N and/or NO_3-N produced from the aerobic biofilm may be partially converted to gaseous nitrogen (N_2, N_2O) by denitrifying bacteria and may be lost to the air if the DO concentration is very low.
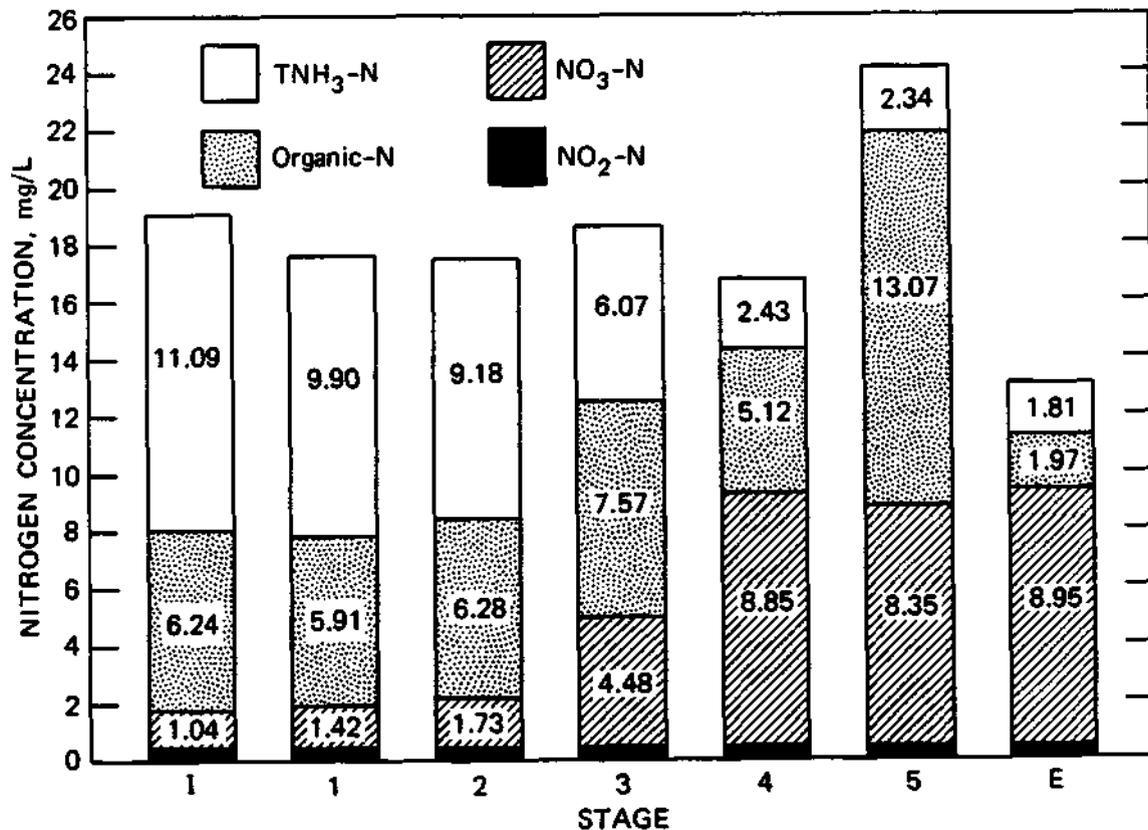
Figure 7. Changes of nitrogen forms in the RBC system

Murphy et al. (26) measured nitrogen balance. They found that 15% of total N entering an RBC system did not appear in the effluent. At Princeton, losses of 10 and 30% total N occurred as the wastewater passed stage 4 and the secondary clarifier effluent, respectively (figure 7). NO_2-N concentrations, represented by solid bars in figure 7, remained fairly constant with a variation of 0.21 - 0.38 mg/L.

It is apparent from figure 7 that changes in concentration occurred for the various forms of nitrogen as the wastewater flowed through the RBC system. There were small reductions in NH_3-N and NO_3-N with increases in the Org-N concentration when the wastewater passed through the system, except at stage 5. This was because of the con-figuration of the contactors — there is a 12" drop at the bottom of stages 3-5. The settled sludge was chopped and scoured in stage 5. A majority of the insoluble Org-N was removed by sedimentation in the clarifiers (figure 7).

Removal Kinetics

Many studies employ either the Monod growth kinetics or Michaelis-Menton enzyme kinetics for modeling NH_3-N. On the basis of steady-state conditions and a complete-mix chamber, the general reaction rate can be developed from the mass balance and can be written as:

$$Q(S_{m-1} - S_m) - A (K_n S_m^n) = 0 \tag{1}$$

$$\text{or } Q/A (S_{m-1} - S_m) = K_n S_m^n = r \tag{2}$$

where

r = reaction (nitrification) rate, lb/d/1000 sq ft ($g/m^2/d$)

Q = flow rate, mgd (m^3/d)

A = surface area of media, sq ft (m^2)

S_{m-1}, S_m = influent and effluent substrate concentrations at stage m, mg/L

K_n = nth-order constant

First-order kinetics can be written as:

$$r_1 = K_1 S_m \tag{3}$$

when the reaction rate is proportional to the substrate concentrations.

Half-order kinetics can be written as:

$$r_{1/2} = K_{1/2}S_m^{1/2} \tag{4}$$

when the reaction rate is proportional to the one-half power of the substrate concentration.

Zero-order kinetics can be written as:

$$r_o = K_o \tag{5}$$

when the reaction rate is independent of the substrate concentration.

The literature shows different opinions concerning the reaction rate for ammonia oxidation of wastewater treated by an RBC system or by other fixed-film processes. The rate of nitrification has been described with respect to NH_3-N concentration as first-order (4,7,22,24-28,32,37,40-47), zero-order (7,24,44-51), and half-order (47,52) kinetics. Nitrification rates vary from first-order at low substrate concentrations to zero-order at high concentrations (5,24,44,46). Reported concentration values for zero-order reaction rates vary among investigators: 2.5 mg/L (5), 5.0 mg/L (24), and 6.0 mg/L (44) of NH_3-N.

Recently a laboratory study by Poon et al. (47) indicated that both first-order and half-order kinetics applied equally well for SNH_3-N loadings from 0 - 1 lb/d/1000 sq ft (0-5 $g/m^2/d$) and beyond with $SBOD_5$ loadings within the range of 0 - 1.6 lb/d/1000 sq ft (0 - 7.8 $g/m^2/d$). Biofilm kinetics may involve three regions which are first-order followed by half-order and then by zero-order as NH_3-N concentrations and loadings increase (52). For the in-situ measurement of nitrifying bacteria kinetics in mixed culture, Hall and Murphy (45) reported that nitrification is zero-order with respect to substrate concentration and first-order with respect to the nitrifier bacterial concentration.

Other investigators (13,35,53-55) have developed conceptual models incorporating the fundamentals of substrate, oxygen diffusion, and biofilm growths. Mass transfer resistances have been applied for the reaction kinetics by others (56-59). Famularo et al. (51) proposed a model for an RBC system by applying oxygen and substrate transports with both liquid and biofilm resistants. An S-shaped curve was suggested by Poon et al. (60,61), showing that SNH_3-N removal is initially first-order, changing to half-order as the SNH_3-N loading increases, and finally reaching a maximum rate (zero-order) with very high SNH_3-N loadings.

Discrepancies in reaction rate kinetics may be due to varying wastewater types, substrate concentrations, and environmental conditions employed in bench- and/or pilot-scale studies. Assumptions of constant reaction rates within all stages and steady-flow conditions are not always valid. An RBC system has varying reaction rates for the various stages. The wastewater flow of a treatment plant is always in a non-steady state and is never constant. Therefore, it is necessary to investigate conditions within each stage of the RBC system.

Conventional hydraulic and mass loading rates are derived from the total media area for all stages combined and not just from the surface area of each stage. They are meaningless because actual loading rates are higher than conventionally calculated rates and vary from one stage to the other. Loading rates based on the surface area at each stage should be used.

In order to compare the manufacturer's design criteria with the observed data from stages 3 and 4, the data were plotted (not included) as percentage SNH_3-N removal versus stage hydraulic loading rate and effluent SNH_3-N concentration versus hydraulic loading rate. The plotted points were found to be scattered. It was impossible to obtain the curves with respect to SNH_3-N concentration as stated in the design manual. It can be concluded that SNH_3-N removal is unaffected by the hydraulic loading rate.

The expression "percent removal" or "percent remaining" is generally used in nitrification reactions and is not meaningful in the evaluation of reaction rates. The amount of SNH_3-N removed per unit area per unit time (lb/d/1000 sq ft), R, does correspond to reaction rates or unit rates of nitrification.

Figures 8 and 9 show that SNH_3-N removal increased as its loading increased up to a maximum limit. Both figures indicate that the SNH_3-N "stage" loading rates (L) over 0.6 lb/d/1000 sq ft (2.9 g/m^2/d) will cause a deterioration in RBC performance, which did occur during the shock periods. Ito and Matsuo (25) found that the maximum rate was 0.82 lb/d/1000 sq ft (4 g/m^2/d) for bench-scale RBC units. The maximum removal (reaction or nitrification) rate at Princeton was 0.35 lb/d/1000 sq ft (1.7 g/m^2/d) (figure 9).

The relationships for reaction (or removal) rate (R) versus influent SNH_3-N concentration (Si) and also for reaction rate versus SNH_3-N "stage" loading rate (L) for both stages 3 and 4 (excluding shock periods) were evaluated by 3-way (linear, semi-log, log-log) regression analysis. Occasionally log-log relationships produced the highest correlations, but for the most part linear relationships produced the best correlations.

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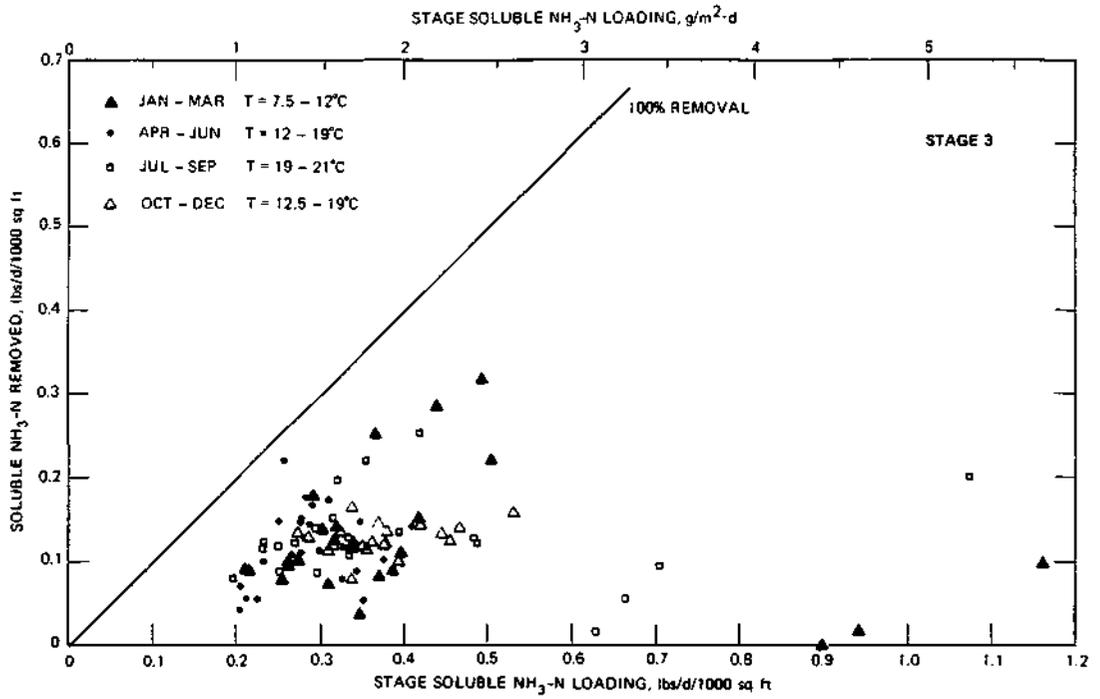
Figure   8.   Soluble   NH₃-N   removal   at   stage   3
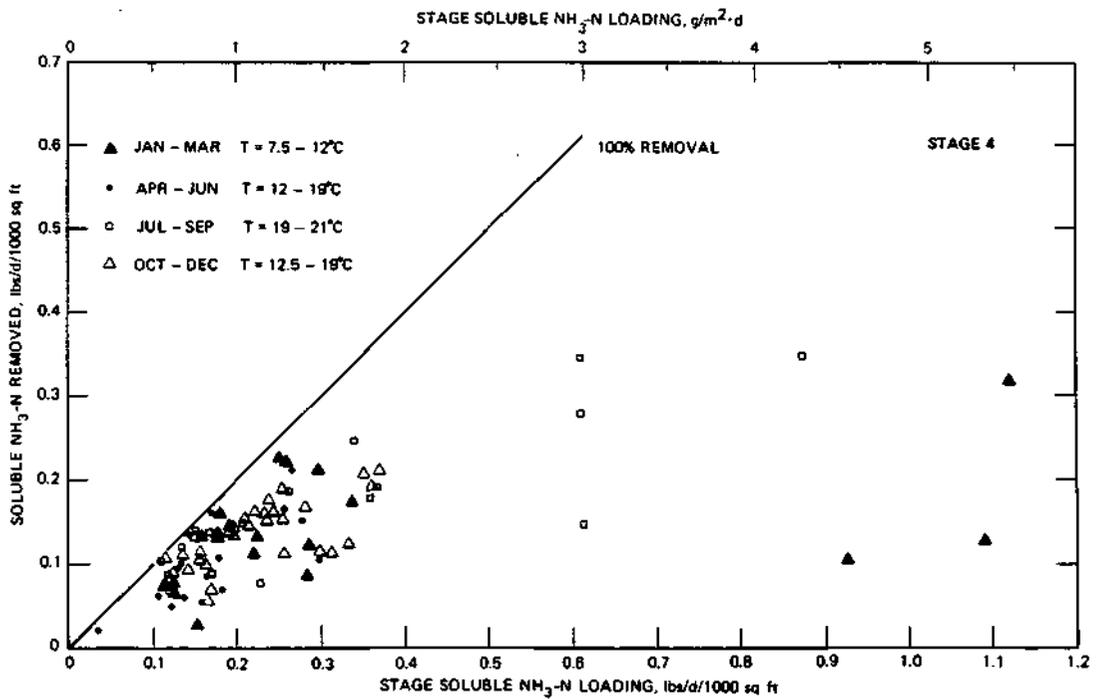
Figure   9.   Soluble   NH₃-N   removal   at   stage   4

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Table 6 suggests that, in general, the correlation coefficients for R vs. L regressions are slightly better than those for R vs. Si. The correlation coefficients were not great but were satisfactory for biological data. Regressions for R vs. Si will yield the popular first-order kinetics as expressed in equation 3. First-order kinetics are completely unaffected by hydraulic or mass loading rates, environmental conditions, wastewater characteristics (temperature, pH, DO, hydraulic detention time, etc.) or configuration or operation modes of RBC equipment. Recently, however, Marsh et al. (62) developed an empirical equation to predict NH_3-N concentration variation in an RBC system on the basis of temperature, influent NH_3-N, BOD concentrations, and flow rate.

There are numerous uncontrollable factors in the RBC process. The dynamic changes in microbial populations and their metabolic activities within a treatment process are never consistent and do not follow any simple or complex model assumptions. Several models describe NH_3-N removal relationships in RBC systems, but none is applicable to the Princeton data. To determine the reaction rate for nitrification, the

Table 6. Results of Regression Analyses
of Reaction Rates (R, lb/d/1000 sq ft)
vs. Influent SNH_3-N Loadings (L) and Concentrations (Si)

Stage and period	R vs. L (lb/d/1000 sq ft)		R vs. Si (mq/L)
	Linear relationship	Corr. coeff.	Correlation coefficient
Stage 3			
1981 average	R = 0.031 + 0.30L	0.44	0.46
Jan-Mar 1981	R = 0.069 + 0.61L	0.67	0.50
Apr-Jun 1981	–	0.19	0.56
Jul-Sep 1981	R = 0.035 + 0.32L	0.51	0.65
Oct-Dec 1981	–	0.24	–0.10
Stage 4			
1981 average	R = 0.030 + 0.49L	0.71	0.62
Jan-Mar 1981	R = 0.025 + 0.49L	0.62	.0.47
Apr -Jun 1981	R = 0.018 + 0.53L	0.72	0.81
Jul-Sep 1981	R = 0.027 + 0.53L	0.82	0.83
Oct-Dec 1981	R = 0.062 + 0.35L	0.54	0.38

empirical relationship between SNH_3-N "stage" loading rates and removal rates, as shown in table 6, may be more useful.

The observed SNH_3-N data from stages 3 and 4 were also subjected to evaluation using the relationships between stage influent SNH_3-N loading conditions (rate, L, and concentration, Si) and effluent concentrations (Se). Better relationships existed between Se and Si for each stage, with good correlation coefficients (table 7). This meant that the effluent SNH_3-N concentration increased as its influent concentration increased.

On the basis of the coefficients of correlation values in tables 6 and 7, one may conclude that the relationship between Se and Si is more useful in design considerations than the relationship between R and L. The current manufacturer's design curves are based on the relationship between Se and hydraulic loading rate. However, one study of SBOD5 removal (9) suggested that there is a linear relationship between the $SBOD_5$ removal rate and stage $SBOD_5$ loading rate, with no relationship between effluent and influent $SBOD_5$ concentrations.

Table 7.   Results of Regression Analyses
of Effluent SNH_3-N Concentrations (Se)
vs. Influent SNH_3-N Loadings (L) and Concentrations (Si)

Stage and period	Se(mq/1) vs. L (lb/d/1000sq ft) Correlation coefficient	Se(mg/L) vs. Si(mg/L) Corr. coeff.	Linear relationship
Stage 3			
1981 average	0.68	0.85	Se = 0.62 Si - 0.14
Jan-Mar 1981	0.45	0.66	Se = 0.52 Si + 0.91
Anr-Jun 1981	0.64	0.78	Se = 0.52 Si + 0.29
Jul-Sep 1981	0.82	0.79	Se = 0.51 Si + 0.32
Oct-Dec 1981	0.77	0.96	Se = 0.97 Si - 0.31
Stage 4			
1981 average	0.65	0.72	Se = 0.44 Si - 0.44
Jan-Mar 1981	0.51	0.53	Se = 0.79 Si - 0.22
Apr-Jun 1981	0.63	0.20	
Jul-Sep 1981	0.83	0.77	Se = 0.39 Si - 0.31
Oct-Dec 1981	0.70	0.80	Se = 0.71 Si - 0.21

Design Considerations

For the design of nitrification in an RBC system, $SBOD_5$ and SNH_3-N concentrations and loading rates with temperature, DO, pH, and alkalinity should be carefully examined. If an RBC process is designated for both carbonaceous and ammonia removals, some reduction of SNH_3-N will occur within the first few stages that are designed primarily for $SBOD_5$ removal. For example, at Princeton an average 17% of the SNH_3-N was reduced in the first two stages (table 4). In considering the SNH_3-N concentration entering the first nitrification stage, one needs to take into account the reduction of SNH_3-N occurring within the carbonaceous stages from the RBC system influent. Stage loading rates should be used instead of the conventional overall loading rate.

Engineers should select a size for the first nitrification unit on the basis of availability and engineering experiences. A contactor with a surface area of 100,000 or 150,000 sq ft (9290 or 13,935 m^2) is commonly used. The effluent SNH_3-N concentration from the first nitrification stage can be estimated by using the relationship between effluent and influent SNH_3-N concentrations (Se vs. Si) developed from similar RBC systems or from a laboratory-scale model. Scale-up is needed, though, from the laboratory units. This procedure can be repeated for the second nitrification unit and so on (size can be smaller) until the design effluent limitations are met.

SUMMARY AND CONCLUSIONS

Collection of operational data at the Princeton wastewater treatment plant's RBC system was conducted on a twice-a-week basis over a one-year period from early January 1981 to the middle of January 1982. This study was conducted to gain insights into the phenomenon of NH_3-N oxidation occurring within each stage of an RBC system. The following conclusions can be drawn from the data obtained.

1. Hydraulic and SNH_3-N loading rates, under normal operations, were only slightly lower than those of design (conventional); while $TBOD_5$ loading rates averaged much less at about one-half of design.

2. Hydrolysis or organic nitrogen occurred in stages 1 and 2 on many occasions.

3. With favorable environmental conditions during June 1981, both $SBOD_5$ and SNH_3-N removals occurred simultaneously in stage 1. This was also the case . for stage 2 during November and December 1981.

4. Stage 3 gave an average SNH_3-N removal efficiency of 40% for an influent concentration ranging from 2 to 18 mg/L SNH_3-N with a stage loading of 0.4 lb SNH_3-N/d/1000 sq ft (1.95 $g/m^2/d$). Furthermore, stage 4 achieved an additional 64% SNH_3-N removal from the stage 3 effluent.

5. Nitrification occurring in stages 3 and 4 deteriorated during shock loading periods. It took about one month before these stages returned to their normal operational status after the 3-week July shock period.

6. Nitrification took place as the $SBOD_5$ concentration was reduced to 15 mg/L. It was affected by the wastewater temperature, which ranged from 7.5 to 21° C.

7. DO concentrations within stages 3 and 4 were not a limiting factor for ammonia oxidation.

8. Normally the pH in the RBC system was within the optimum, ranging from 7.5 to 7.8 pH units. The drop of pH due to nitrification was insignificant.

9. The average ratios of alkalinity destruction/NH_3-N removal for stages 3 and 4 were 6.8 and 9.1, respectively. The mean ratios of NO_x-N formation/SNH_3-N removal were 0.94 and 1.49, respectively, for stages 3 and 4. A release of SNH_3-N from a break-up of biomass and scouring of bottom deposits was suspected in stage 4.

10. Practically no nitrification occurred in stage 5 or the secondary clarifier. Normally, the SNH_3-N concentrations for the secondary clarifier effluents were below 2.0 mg/L irrespective of RBC influent concentrations. Overall, the RBC system removed 83% of the SNH_3-N regardless of the seasonal variations in influent SNH_3-N concentrations, but effluent concentrations were highest during cold weather. About 56% of the time, TNH_3-N in the secondary effluent met the 1.5 mg/L NH_3-N effluent standard.

11. Only about 10% of total nitrogen of all forms in the RBC system's influent was not accounted for at the effluent for stage 5.

12. Soluble NH_3-N removal was not found to be related to hydraulic loading rates. The loading rate should be calculated on the basis of the surface area at each stage within the RBC system and not for all stages combined.

13. The maximum stage loading rate was limited to 0.6 lb SNH_3-N/d/1000 sq ft (2.9 $g/m^2/d$). The maximum nitrification rate was found to be 0.35 lb SNH_3-N/d/1000 sq ft (1.7 $g/m^2/d$).

14. A linear relationship existed between effluent and influent SNH_3-N concentrations at stages 3 and 4 where nitrification occurred. This relationship is proposed for use in RBC process design.

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