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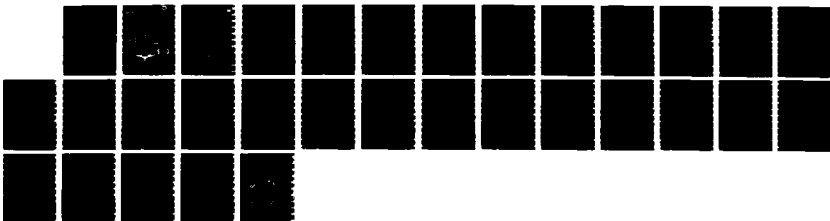
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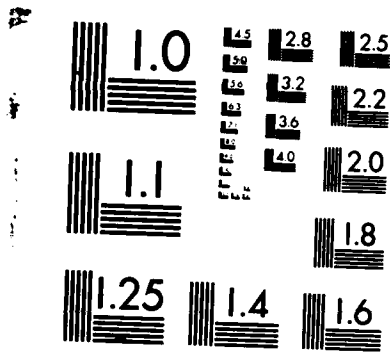
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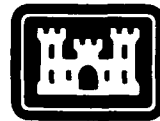
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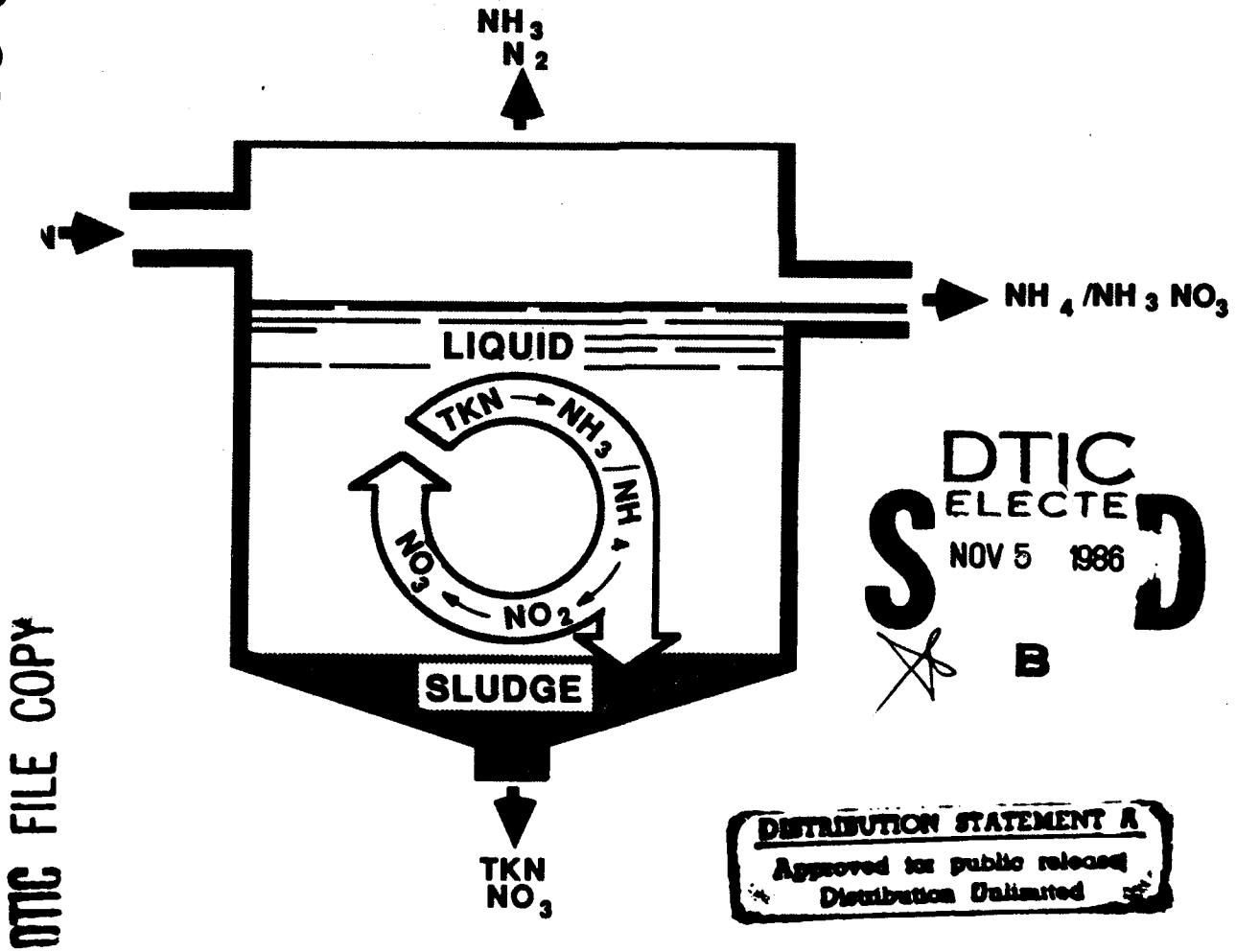


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Nitrogen control in wastewater treatment systems for military facilities in cold regions

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Nitrogen control in wastewater treatment systems for military facilities in cold regions

Sherwood C. Reed

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the activated sludge process. Trickling filters or RBC units are recommended for higher (> 1 mg/L) discharge limits. Pond systems are suitable for seasonal ammonia removal in cold climates.

PREFACE

This report was prepared by Sherwood C. Reed, Environmental Engineer, Civil Engineering Research Branch, Experimental Engineering Division, U.S. Army Cold Regions Research and Engineering Laboratory. Funding for this effort was provided by DA Project 4A762730AT42, *Design, Construction and Operations Technology in Cold Regions*; Task C, *Cold Regions Base Support; Maintenance and Operations*; Work Unit 009, *Water Supply and Wastewater Treatment at Cold Regions Facilities*. Technical monitor for this work unit was Walter Medding at the Office of the Chief of Engineers.

The report was technically reviewed by Dr. E.J. Middlebrooks, Tennessee Technological University, Cookeville, Tennessee, and Barry H. Reid, U.S. Environmental Protection Agency, Corvallis Environmental Research Laboratory, Corvallis, Oregon.

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Nitrogen Control in Wastewater Treatment Systems for Military Facilities in Cold Regions

SHERWOOD C. REED

INTRODUCTION

This report presents a summary of engineering design criteria for nitrogen control in those wastewater treatment processes in most common use at military facilities in the cold regions. These include trickling filters, wastewater treatment ponds (facultative and aerated), rotating biological contactors (RBC) and variations of activated sludge. In many cases nitrogen limitations are imposed as an additional requirement, so it is often necessary to add a new component or to retrofit an existing wastewater treatment system. The concepts described in this report can be used as an add-on for existing systems or as an integrated component in new systems.

The actual removal of nitrogen is only required when wastewater is discharged to very sensitive surface waters, if the effluent is to be used to recharge drinking water aquifers, or in other very special circumstances. It is unlikely that any present, or future, military facilities will be faced with these very stringent requirements. The most likely requirement will be for conversion of the less stable forms of nitrogen (organic N and ammonium) to nitrate to prevent environmental problems in receiving surface waters. The EPA estimates that by the year 2000, about 27% of the wastewater treated in the United States will incorporate some form of nitrogen control (EPA 1983a). Their projections indicated about 680 systems with this capability in 1982, and a need for over 2800 systems by the year 2000 (EPA 1983a). It is likely that a number of military facilities will be included in the latter group.

Three of the four concepts described in this report provide the necessary conversion of ammonia to nitrate, termed nitrification. The wastewater treatment ponds provide a significant actual removal of nitrogen via losses of gaseous ammonia to the atmosphere. The information in this report on trickling filters and wastewater treatment ponds is drawn from recent research here at CRREL (Reed 1984, Reed et al. 1986). Criteria for the other two processes are drawn from other recent publications.

BACKGROUND AND THEORY

The principal environmental impacts of concern are ammonia toxicity to fish and oxygen depletion in the receiving waters because of oxidation of ammonium to nitrate. Ammonium in wastewaters will also lower disinfection efficiency when chlorine is used because of the formation of chloramine compounds. However, ammonia removal is not typically required for this purpose. Nitrogen in wastewaters can promote eutrophication in surface waters, particularly lakes and ponds with a low volumetric exchange. Nitrification processes will not be effective in these special cases and near complete nitrogen removal may be required. Nitrification processes can be effective for controlling fish toxicity and oxygen depletion problems.

Nitrogen can exist in wastewaters in a number of different forms because of the various oxidation states represented, and it can readily change from one state to another depending on the oxidation or reduction reactions induced by the physical or biochemical conditions present at the time. The principal forms of concern in wastewater are:

1. *Organic nitrogen*—Present in the protein molecules in animal tissue and in fecal matter (typical assumption is about 8-10% nitrogen content). Microorganisms decompose the organic nitrogen and produce ammonia or ammonium ions.

2. *Ammonia*—Ammonia can be present as molecular ammonia (NH_3) or as ammonium ions (NH_4^+). The nitrogen in urine is initially urea, which is hydrolyzed by an enzyme to ammonium carbonate ($(\text{NH}_4)_2\text{CO}_3$). Microorganisms (*Nitrosomonas* and *Nitrobacter*) oxidize these ammonium compounds to nitrite (an unstable intermediate) and then to nitrate.

3. *Nitrate*—Nitrate (NO_3^-) is the stable end point of the oxidation reactions:

Organic N — Ammonia — Nitrite — Nitrate

Nitrate can be taken up by plants and algae and converted to organic nitrogen in cell tissue, or in an oxygen deficient (anaerobic) environment (with sufficient carbon present) it can be reduced to elemental nitrogen (N_2) and lost as a gas.

The nitrogen concentration in typical municipal wastewaters ranges from about 15 to over 50 mg/L. About 60% of this is in the ammonia form, about 40% in the organic form (Reed 1984). To avoid confusion it is conventional to express these concentrations as equivalent elemental nitrogen (as N) regardless of the particular ionic form. For example, 10 mg/L of nitrate nitrogen (as N) is equal to about 45 mg/L of the specific nitrate compound (as NO_3^-).

The presence of ammonia gas (NH_3) or ammonium ions (NH_4^+) in the wastewater strongly depends on the pH of the water and on temperature. Figure 1 illustrates this relationship. At pH 7 essentially only ammonium ions are present, at pH 12 only dissolved ammonia gas. This dissolved gas can be released from the liquid under proper conditions and this is the purpose of air stripping operations in mechanical treatment plants. Typically, the pH is elevated with lime, the sludge allowed to settle and then the clarified effluent aerated or allowed to cascade over a packed bed in a stripping tower. At the high hydraulic loading rates commonly employed, the concept is only effective in warm weather because of the temperature constraints illustrated in Figure 1.

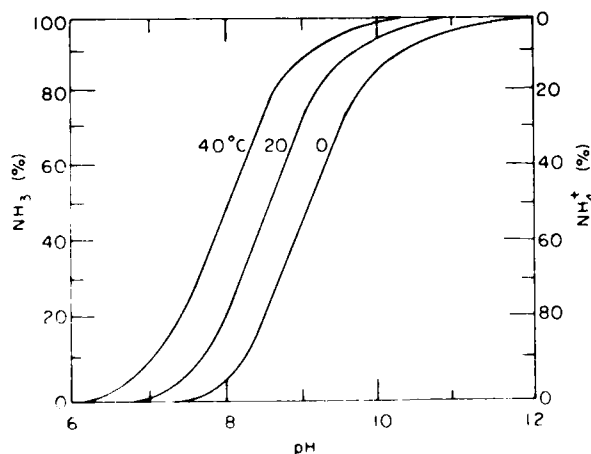


Figure 1. Distribution of ammonia and ammonium ions in water vs pH and temperature (after Gurnham et al. 1979).

The toxicity problem for fish is attributable to the dissolved ammonia gas (NH_3) present in the water. Acute toxicity levels range from 0.01 mg/L to over 2.0 mg/L depending on the type of fish, temperature, etc. The EPA recommends a limit of 0.02 mg/L based on salmonoid fish fry. As shown by the relationships in Figure 1, a slight increase in pH in the receiving water can result in a very significant increase in toxicity as the ammonium ions are converted to dissolved ammonia gas. It is therefore necessary to maintain low levels of ammonium in the effluent to ensure protection for the fish. Young fish are more susceptible than adults, and at a given pH the toxicity increases with temperature. Many regulatory agencies specify seasonal ammonia limits, with a very low value during spring and summer and a less stringent value, or no limit at all, during the winter.

Oxygen depletion in the receiving waters can significantly affect fish and other aquatic life as well as the aesthetic characteristics of the water body. The oxygen demand of unoxidized ammonium is about 4.6 mg/L of oxygen per mg/L of ammonium. A secondary effluent with a Biochemical Oxygen Demand (BOD) of 30 mg/L and ammonium of 15 mg/L would still exert an oxygen demand on the receiving water of about 114 mg/L. That could result in significant oxygen depletion during the warm summer months.

Wastewater composition is an important factor in the development, and maintenance, of a nitrifier population in the treatment system. The nitrifiers are autotrophic organisms (they use inorganic carbon) and have a low growth rate. They do not compete effectively with the rapidly developing heterotrophic organisms that depend on the simple carbonaceous organics (BOD) in the wastewater. If the BOD or the SS (suspended solids) of the wastewater is high, nitrification will be suppressed or eliminated. Trace metals and other toxic substances can destroy bacterial growths, and the nitrifying bacteria are more susceptible than the heterotrophic type. Water temperature and oxygen are critical parameters and sufficient alkalinity must be present in the wastewater to avoid inhibition of nitrification through pH depression. As shown by the expression below, nitrification releases hydrogen ions, which in turn depress the pH unless alkalinity is present to act as a buffer.



There have been successful experiences with both activated sludge and trickling filters in obtaining both BOD removal and nitrification in a single combined unit. This requires very careful management and operation to maintain all of the necessary conditions, and all of this is particularly difficult to sustain under winter conditions in cold climates. As a result, it is suggested that nitrification components, for activated sludge and trickling filters in cold climates, be designed as a separate unit dedicated for nitrification, and not as a combined unit. RBCs are typically designed in separate stages, with the final components used for nitrification, so the same principle prevails for this concept as well. The winter ammonia removal in cold regions pond systems will be relatively low. If stringent ammonia limits prevail in the winter then alternative management schemes, as described in the section to follow on ponds, may be necessary.

TRICKLING FILTERS

Introduction

Trickling filters are the most common type of wastewater treatment system operated by the U.S. Army (Poon et al. 1984). Treatment responses depend on the biological films growing on the support media in the filter bed. Until the 1950's the media were typically rock, but since that time they have been largely replaced in new systems by plastic media in various

Table 1. Characteristics of plastic and rock media (after Earnest et al. 1978).

Characteristic	Media	
	Plastic	Rock
Specific surface area*	84-341 m ² /m ³	62 m ² /m ³
Void ratio	95%	46%
Weight	44-112 kg/m ³	1400 kg/m ³

* The actual surface area of a particular media type per unit volume of those media.

Table 2. Summary of results from trickling filter studies.*

Location of filter and input wastewater type	Hydraulic [†] loading (m ³ /m ² min.)	BOD** loading (kg/m ³ day)	Temperature (°C)	Ammonia removal (%)
Hanover, New Hampshire				
Random fill plastic				
Primary	0.008	0.44	3-23	0-53
Secondary	0.007	0.12	3-23	0-76
Waldorf, Maryland				
Modular plastic secondary	0.08	0.42	8-28	47-88
Amherst, Massachusetts				
Modular plastic primary	0.08	2.1	8-23	51
Midland, Michigan				
Modular plastic secondary	0.02-0.08	0.06-0.27	7-20	85 at low end of loading range
Lima, Ohio				
Modular plastic secondary	0.03	< 0.01	9-24	84

* After Duddles et al. (1974), Schwarz (1976), Earnest et al. (1978), Sampayo (1980), Reed et al. (1986).

† Hydraulic loading is volume applied per unit area at the top of the filter.

** BOD loading is the mass applied per unit volume of the filter media.

configurations. The plastic has significant advantages because of its lower weight, greater surface area and greater void ratio. These characteristics allow for much higher hydraulic loadings, and less expense for structural supports and container walls. It is not uncommon to see plastic media in above-ground towers 4 to 6 m tall because of the improved conditions for biological growth and natural air flow, and the low cost for the container. Table 1 compares the characteristics of stone and plastic media. It is likely that any new construction or retrofits for nitrification will use plastic media systems.

Full-scale and pilot-scale trickling filter systems have been studied for their nitrification potential in New Hampshire, Maryland, Massachusetts, Michigan and Ohio (Duddles et al. 1974, Schwarz 1976, Earnest et al. 1978, Sampayo 1980, Reed et al. 1986). Selected results from all of these studies are presented in Table 2. The ammonia removals versus liquid temperature are plotted in Figure 2 for these systems. It is clear that the pilot units at Hanover, Amherst and Waldorf were more sensitive to temperature than the full-scale system at Lima

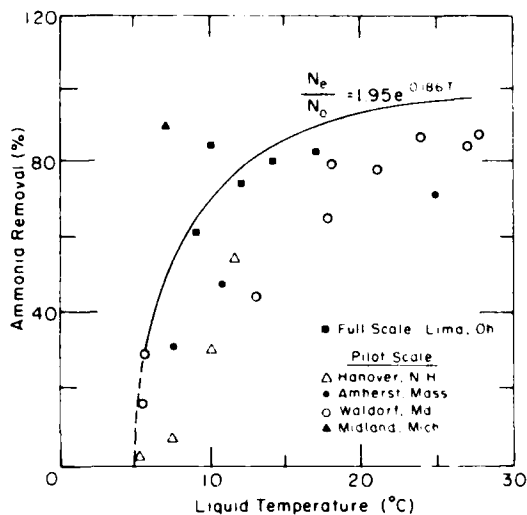


Figure 2. Ammonia removal vs temperature in full-scale and pilot-scale trickling filter systems.

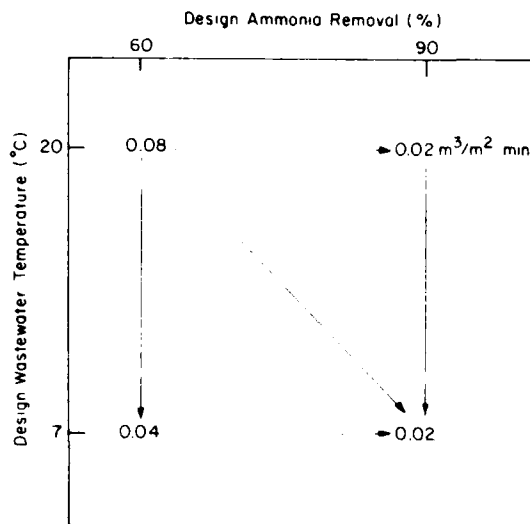


Figure 3. Ammonia removal in trickling filter systems vs temperature and hydraulic loading rate (after Duddles et al. 1974).

Table 3. Process requirements for significant nitrification in trickling filters.

Parameter	Required value
Influent ammonia	< 30 mg/L
pH	6.6 to 9.0
Temperature	7° to 35°C
Dissolved oxygen	> 0.5 mg/L
BOD	< 30 mg/L
Loading:	
Plastic media	< 0.35 kg/m³ day
Rock media	< 0.19 kg/m³ day
Alkalinity	7.14 mg CaCO ₃ /1 mg NH ₃
Metals and toxics	Low concentrations

or the pilot unit at Midland. This is probably because of the small diameter and the exposed location of the former units so that a greater portion of the applied flow was exposed to the extreme ambient temperatures around the perimeter. Results from all of the tests tend to converge on 5°C as the limiting temperature for nitrification in a trickling filter. Equation 1 below, derived from Figure 2, describes ammonia conversion in full-scale systems where the trickling filter unit is intended for nitrification only.

$$N_e/N_0 = 1.95e^{0.186T} \quad (1)$$

where N_e = ammonia in trickling filter effluent (mg/L)

N_0 = ammonia in trickling filter influent (mg/L)

T = liquid temperature in filter unit (°C).

The equation is valid for temperatures greater than 7°C. Graphical extrapolation can be used, as shown in Figure 2 for temperatures between 5° and 7°C. The process requirements to achieve significant nitrification in trickling filters are given in Table 3. A relationship between ammonia removal, temperature and hydraulic loading rate was developed in the Midland, Michigan, study (Duddles et al. 1974) and is shown in Figure 3. Ammonia removal was

not sensitive to temperature at the lowest hydraulic loading rate. It was necessary to reduce higher loading rates to just sustain performance at lower temperatures.

Design procedure

Every manufacturer of plastic media has design charts that give the specific surface area of filter media required for ammonia removal under various temperature conditions. An alternative approach has been developed recently (Reed et al. 1986), based on data from the full-scale system at Lima, Ohio, which were also the basis for eq 1. In this case the design parameter is expressed in terms of Total Kjeldahl Nitrogen (TKN). The TKN value includes both organic and ammonia nitrogen. In some systems (and at Lima, Ohio) a portion of the entering organic N was hydrolyzed, in effect producing "new" ammonia. Use of TKN as the loading parameter ensures a conservative basis for design.

1. Summer conditions ($T > 12^{\circ}\text{C}$):

$$\text{TKN} = 1.34 e^{0.09N_e} \quad (2)$$

where TKN = total Kjeldahl nitrogen that can be applied (kg/day per 1000 m² of specific surface area)

N_e = effluent ammonia nitrogen required by the system design (mg/L).

2. Winter conditions ($T > 7^{\circ}\text{C}$):

$$\text{TKN} = 0.987 e^{0.09N_e} \quad (3)$$

all terms as defined above.

Either equation is used to compute the TKN limits when the allowable effluent ammonia concentration is known. The volume of media required can then be calculated for a particular type of media. Since summer is usually the most critical period for ammonia, eq 2 should be solved first. The TKN value is then used to solve eq 3 to be sure that the winter discharge limit is satisfied. If the same limit prevails year-round, then eq 3 will control design in cold climates. These equations are valid within the range of other process requirements shown in Table 3. A thermal analysis should also be conducted as a routine part of design for systems above the 38th parallel in North America to be sure that the necessary 7°C temperature can be sustained in the filter bed. A cover and regulation of air flow may be required for some cases.

Design example

The following conditions apply for this example:

Average design flow = 1.3 m³/min.

TKN in wastewater: average = 15 mg/L; peak = 30 mg/L.

BOD and SS < 30 mg/L.

Wastewater temperature: 2-week average winter low = 8°C; summer average = 20°C.

Effluent ammonia: summer = 3 mg/L; winter = 9 mg/L.

Calculate the TKN loading:

Average conditions,

$$15 \text{ mg/L} = 15 \text{ g/m}^3$$

$$(1.3 \text{ m}^3/\text{min.})(15 \text{ g/m}^3)(1440 \text{ min./day})(10^{-3} \text{ kg/g}) = 28.1 \text{ kg/day.}$$

Peak conditions,

$$(1.3)(30)(1440)(10^{-3}) = 56.2 \text{ kg/day.}$$

Calculate the TKN factor with eq 2:

$$\text{TKN} = 1.34e^{(0.09)(3)} \\ 1.755 \text{ kg/day per } 1000 \text{ m}^2.$$

Use this value in eq 3 to be sure that winter requirements are satisfied:

$$1.755 = 0.892e^{(0.09)(N_e)} \\ N_e = 7.5 \text{ mg/L, acceptable; limit is } 9 \text{ mg/L.}$$

Calculate specific surface area required for peak conditions:

$$\frac{56.2 \text{ kg/day}}{1.755 \text{ kg/day per } 1000 \text{ m}^2} = 32,023 \text{ m}^2.$$

Assume media with $130 \text{ m}^2/\text{m}^3$ specific surface area. Calculate volume required:

$$\frac{32,023}{130} = 246 \text{ m}^3.$$

Use one tower; 8 m in diameter, 5 m tall = 251 m^3 . The hydraulic loading will be

$$\frac{1.3 \text{ m}^3/\text{min}}{50 \text{ m}^2} = 0.03 \text{ m}^3/\text{m}^2 \cdot \text{min.}$$

within range in Table 3, acceptable.

A filter tower of this type, designed for nitrification only, would not need to be followed by a clarifier to meet secondary treatment requirements for the input BOD and SS specified in Table 3. The hydraulic loading calculated in the example is the average daily rate. If the minimum hourly rate is below the media manufacturer's recommendations, then some recycling may be needed to keep the media wet. A higher hydraulic loading is also possible for the system described in the example by increasing the depth and decreasing the diameter of the tower. Recycle should be avoided if possible to avoid excess cooling in the tower during the winter.

Winter operations

Temperature measurements were made in a full-scale plastic media trickling filter tower at Tobyhanna, Pennsylvania, during the 1982-83 winter (Reed et al. 1986). The 4.9-m-deep, 18.5-m-diameter, uncovered unit was operating in the "combined" mode for both BOD and ammonia removal, with recycle at about 75% of the incoming flow. The results indicated that most of the heat losses occurred within the top 60 cm of the filter bed. The following relationships were developed from the thermocouple data.

$$T_o = 5.70 + (0.44)(T_A) \quad (4)$$

$$T_{60} = 2.94 + (0.74)(T_o) \quad (5)$$

$$T_E = 2.81 + (0.57)(T_o) \quad (6)$$

where T_o = average daily temperature at surface of trickling filter ($^{\circ}\text{C}$)

T_A = average daily ambient air temperature ($^{\circ}\text{C}$)

T_{60} = average daily temperature at a depth of 60 cm in filter bed ($^{\circ}\text{C}$)

T_E = average daily temperature of effluent from trickling filter ($^{\circ}\text{C}$).

These equations show that it was not possible to sustain acceptable nitrification temperatures in the bed (7°C) when the ambient air temperature was 3°C or less, and that general freezing would commence on the bed surface with the ambient air at about -13°C. This particular unit will be covered to reduce heat losses and freezing problems. These equations are only valid for this particular unit, but the trends demonstrated are probably applicable for uncovered units in other locations.

Heat losses can be reduced in uncovered trickling filter towers by closing some of the air vents at the base of the tower. The dissolved oxygen in the filter effluent should be at least 2 mg/L, but air vents can be closed until this value is approached. Assuming no recirculation, there will still be sufficient air flow if the water temperature does not drop more than 1°C as the water passes through the bed (EPA, in press).

POND SYSTEMS

Introduction

Wastewater ponds are the most common form of treatment in the north-central U.S., Alaska and Canada. Construction, labor and energy requirements are low, so where sufficient land exists they are usually the most cost-effective treatment alternative. Ponds can be used as the sole treatment method, or for sedimentation or polishing after other forms of treatment, or as preliminary treatment-storage prior to land application of wastewater. Ponds are also classified with respect to the degree of mixing or aeration provided as well as the operational mode. The major types of concern for this report are:

1. Oxidation—shallow, aerobic throughout, depending on algae and surface-re-aeration for oxygen.

2. Facultative—deeper than an oxidation pond, anaerobic at depth; surface zone has same oxygen sources as oxidation pond.

3. Controlled discharge—usually a set of facultative pond cells. Each cell is isolated and then discharged, usually in late spring and early fall. Similar concept for land treatment where the pond is discharged during growing season for site vegetation.

4. Partial mix aerated—usually deeper than facultative ponds; mechanical aeration is provided to satisfy the oxygen requirements of the system. However, there is not enough aeration intensity to keep all solids in suspension and completely mixed.

High intensity, "complete mix" aerated ponds are a variation of the activated sludge process. Some pond systems have a complete mix cell as their first component followed by one of the types described above. The nitrogen removal in that initial short-detention-time cell is minimal because of the high BOD and SS concentrations and the dominance of the heterotrophic organisms.

Typical nitrogen removal values are summarized in Table 4 for both facultative and partial mix aerated ponds. Taken as a group, the facultative ponds exhibit significantly higher nitrogen removal. The aerated ponds have a higher percentage of nitrate in the effluent, but for both types of ponds the major form of effluent nitrogen is ammonia. The better performance of the facultative ponds is in part ascribable to the longer detention time, but the major factor is believed to be the presence of algae. The algae-carbonate interactions result in an elevated pH, which in turn enhances volatile loss of ammonia at the pond surface.

Design procedure

Calculation methods for estimating ammonia removal from both facultative and aerated ponds can be found in Middlebrooks et al. (1982) and EPA (1983b). These methods are still valid. Another approach has been developed more recently (Reed 1984) that estimates am-

Table 4. Nitrogen removal in wastewater stabilization ponds.*

Location	Total nitrogen [†] (mg/L)		Percent removal
	Influent	Effluent	
Facultative ponds			
Peterborough, New Hampshire	17.8	10.1	43
Eudora, Kansas	50.8	9.1	82
Kilmichael, Mississippi	35.9	7.2	83
Corrine, Utah	15.3	3.0	80
Aerated ponds			
Windber, Pennsylvania	22.7	25.0	-10
Endora, Kansas	16.9	11.9	30
Pawnee, Illinois	51.0	6.4	87
Gulfport, Mississippi	26.2	13.9	47

* After Bowen (1977), Hill and Shindala (1977), McKinney (1977), Reynolds et al. (1977), Earnest et al. (1978), Gurnham et al. (1979), Polkowski et al. (1979), Englande (1980), Middlebrooks et al. (1982), EPA (1983b).
[†] Annual average values.

monia removal in terms of the total nitrogen (total N = TKN + NO₂ + NO₃) in the pond system. Total nitrogen was selected as the design parameter in recognition of the many nitrogen transformations that will naturally occur during the long detention times in pond systems. Organic nitrogen will, for example, be converted to ammonia and ammonia can be taken up by plants and animals and converted back to organic nitrogen. Calculations based on total nitrogen provide a more conservative basis for design than the earlier methods.

In addition to atmospheric losses, nitrogen can be removed in the pond by plants and animals, be incorporated in bottom deposits or leave with the effluent. Under favorable conditions the dominant mechanism is believed to be volatilization losses to the atmosphere (Middlebrooks et al. 1982, EPA 1983b, Reed 1984). It is clearly established that nitrogen removal is related to the detention time, the pH, and the water temperature in the pond system. Wastewater alkalinity is important for the algal-carbonate interactions that result in pH elevation. These interactions can raise the pH to 10 or greater on warm sunny days. At other times, with more moderate pH levels the unit rate of nitrogen removal may be low, but the long detention time in the pond compensates.

Data were collected on a frequent schedule from every cell at all of the facultative pond systems listed in Table 4 for at least a full annual cycle. This large body of data allowed quantitative analysis with all major variables included. Two design models were independently developed, and then validated with facultative pond data from sources not used in development. Both are first order models, dependent on pH, temperature and detention time in the system. Both predict the removal of total nitrogen, but it is implicit in the development of each that volatilization of ammonia is the major removal pathway.

Reed model (Reed 1984)

$$N_e = N_o e^{-K_T [t + 60.6(\text{pH} - 6.6)]} \quad (7)$$

where N_e = effluent total nitrogen (mg/L)
 N_o = influent total nitrogen (mg/L)
 K_T = temperature-dependent rate constant

t = detention time in system (days)
 pH = pH of near surface bulk liquid.

$$K_T = K_{20}(\theta)^{(T-20)} \quad (8)$$

where K_{20} = rate constant at 20°C = 0.0064

θ = 1.039

T = temperature of near-surface bulk liquid.

$$T = \frac{(0.5)(A)(T_a) + (Q)(T_i)}{(0.5)(A) + Q} \quad (9)$$

where A = surface area of pond (m²)

T_a = ambient air temperature (°C)

Q = influent flow rate (m³/day).

An acceptable approximation is to assume T = monthly average air temperature, with 2°C taken as the lowest value; pH conditions can be estimated from data from nearby ponds, or estimated with eq 10, which was derived from data in Bowen (1977), Hill and Shindala (1977), McKinney (1977) and Reynolds et al. (1977).

$$\text{pH} = 7.3 e^{(0.0005)(\text{ALK})} \quad (10)$$

where pH = pH of near surface bulk liquid and ALK = expected influent alkalinity (as CaCO₃) (mg/L).

Middlebrooks model (EPA, in press)

$$N_e = N_o \frac{1}{1 + t[0.000576T - 0.00028] e^{(1.080 - 0.0427)(\text{pH} - 6.6)}} \quad (11)$$

where N_e = effluent, total nitrogen (mg/L)

N_o = influent, total nitrogen (mg/L)

t = detention time (days)

T = temperature of near surface bulk liquid (see eq 9 above)

pH = pH of near surface bulk liquid (see eq 10 above).

A typical pond design will initially determine the minimum detention time required for the specified BOD removal. The nitrogen removal that will occur in that same time period can be calculated with eq 7 or 11. It will be conservative to assume that any nitrogen remaining in the effluent is all in the ammonia form. If further removal or conversion is required, the equations can be rearranged and solved for the time required to obtain this additional removal. The costs for this additional detention time can then be compared to the costs for alternative nitrogen removal methods. For this purpose, eq 7 takes the following form:

$$t = \frac{\ln(N_e/N_o)}{(-K_T)} - 60.6(\text{pH} - 6.6). \quad (12)$$

These equations are also very useful for the design and operation of land treatment systems since nitrogen is usually the limiting factor determining the land area required for treat-

ment. Under favorable conditions, even a few weeks of detention in a pond can have a significant effect on nitrogen, and therefore significantly influence the size and costs of the final land treatment component. Land treatment operations can also be affected. Corn, for example, has a variable need for nitrogen during the growing season (higher need in early stages). It should be possible to bypass the final cells in the system during the early part of the growing season and then use final pond effluent during the latter stages. This would more closely match the nitrogen needs of the crop and ensure groundwater protection at all times.

Design example

The following conditions apply for this example:

Average design flow = 1.3 m³/min.

Total nitrogen in wastewater = 20 mg/L.

BOD in wastewater = 250 mg/L.

Alkalinity in wastewater = 250 mg/L.

Effluent BOD = 30 mg/L.

Ammonia in wastewater:

Summer = 7 mg/L, winter = 12 mg/L.

Summer pond temperatures = 23°C.

Winter pond temperatures = 8°C.

Calculate pond size for BOD removal: Use plug flow model from EPA (1983b) (other methods acceptable also). Assume $k = 0.036$, BOD loading on first cell 77 kg/ha·day; assume 1.4-m effective depth.

Wastewater flow:

$$(1.3 \text{ m}^3/\text{min})(1440 \text{ min./day}) = 1872 \text{ m}^3/\text{day}$$

BOD load:

$$(1.3)(250)(1440)(10^{-3}) = 468 \text{ kg/day.}$$

Surface, first cell:

$$468/77 = 6.08 \text{ ha.}$$

Volume, first cell:

$$(1.4)(60,800) = 85,120 \text{ m}^3.$$

Detention time, first cell:

$$85,120/1872 = 45 \text{ days.}$$

BOD in effluent from this cell:

$$\begin{aligned} C_e &= C_o e^{-0.036(45)} \\ &= 250 e^{-1.62} \\ &= 49.5 \text{ mg/L.} \end{aligned}$$

Detention time in second cell:

$$\begin{aligned} e^{-0.036(t)} &= 30/49.5 \\ t &= 14 \text{ days.} \end{aligned}$$

Total detention time for BOD removal:

$$45 + 14 = 59 \text{ days.}$$

Calculate effluent nitrogen with eq 7, pH with eq 10, and K_T with eq 8:

$$\begin{aligned} \text{pH} &= 7.3 e^{(0.0005)(250)} \\ &= 8.27 \end{aligned}$$

$$\begin{aligned} K_T &= (0.0064)(1.039)^{(23-20)} \\ &= 0.0072 \end{aligned}$$

$$\begin{aligned} N_e &= (20) e^{-0.0072[59 + 60.6(8.27-6.6)]} \\ &= 6.3 \text{ mg/L.} \end{aligned}$$

Acceptable limit is 7 mg/L. Winter effluent nitrogen will be:

$$\begin{aligned}K_{T\text{winter}} &= (0.0064)(1.039)^{(8-20)} \\ &= 0.004 \\ N_e &= (20) e^{-0.004[59 + 60.6(8.27-6.6)]} \\ &= 10.5 \text{ mg/L.}\end{aligned}$$

Less than 12 mg/L is acceptable.

Winter operations

In northern climates it will be necessary to use the controlled discharge concept for a pond system if stringent limits for nitrogen prevail in the winter. A typical 180-day controlled discharge pond system, under the conditions described in the example above, would discharge about 6 mg/L nitrogen in the spring and less than 3 mg/L in the early fall.

Equations 7 and 11 predict the total nitrogen that will be present in the pond effluent. In most cases it is prudent to assume that all of that is in the ammonia form; however, aerated ponds with a light BOD loading and a long detention time can provide significant nitrification. The total nitrogen in the effluent from the pond system at Egerton, Wisconsin (Polkowski 1979), contained an average of about 30% nitrates. The concentration was very low in winter and ranged up to 60% of the effluent N in warm months. Similar results (average nitrate 20%) were observed at the pond system in Gulfport, Mississippi (Englande 1980). At the other more heavily loaded systems studied by EPA (Earnest et al. 1978, Gurnham et al. 1979), there was essentially no nitrate in the pond effluent.

ROTATING BIOLOGICAL CONTACTORS

Introduction

RBCs are a relatively new development. Treatment responses, as with trickling filters, depend on the biological films growing on the media. The media in this case are large disks on a central rotating shaft. The partially submerged (about 40% of diameter) disks rotate through the wastewater stream and then through the air. This exposure to the air, every rotational cycle, provides the oxygen necessary to sustain the process. There were mechanical and performance problems with some of the earlier RBC systems, but these difficulties have been overcome and the concept can be an effective wastewater treatment process.

The units are arranged in sets with the wastewater flowing through sequential stages in series. This hydraulic regime ensures plug flow conditions. The same relationship between the heterotrophic and autotrophic organisms described previously also prevails in RBC systems. It is possible, as with trickling filters, to obtain both BOD removal and some nitrification in the same RBC unit. For systems where significant ammonia conversion is a process requirement, the use of separate stages of RBC units dedicated for that purpose is recommended. RBC units have also been used in this way to nitrify effluents from conventional trickling filters and activated sludge systems.

Design procedure

Nitrification in pilot- and full-scale RBC systems has been studied in a variety of locations in the U.S., Canada and Japan (Hynek 1980, Miller et al. 1980, O'Shaughnessy 1980, Smith and Khettry 1980, Pano and Middlebrooks 1982). Selected results from these studies are presented in Table 5. The work at Logan, Utah (Pano and Middlebrooks 1983), and elsewhere indicates that nitrification essentially stops at 5°C. This is similar to the trickling filter results discussed previously, where 7°C was suggested as the minimum temperature for significant

Table 5. Nitrification in RBC units.*

Location	Wastewater type	Temperature (°C)	Hydraulic loading (m ³ /m ² day)	Ammonia (mg/L)	
				In	Out
Brampton, Ontario	Secondary	6.5-8.7	0.076	14	> 1
Cadillac, Michigan	Secondary	18	0.02	10	1
Miyazaki, Japan	Landfill leachate	27	0.02	129	4
Logan, Utah	Dilute sewage (= secondary)	5	0.05	9	9
		15	0.05	7.7	0.7
		20	0.05	10.0	0.2

* After Hynek and Jemura (1980), Smith and Khettry (1980), Pano and Middlebrooks (1983).

nitrification. The temperature dependence of RBCs has been defined by Pano and Middlebrooks (1983):

$$k_{NT} = k_{N20}(\theta)^{(T-20)} \quad (13)$$

where k_{NT} = temperature-dependent rate constant for RBCs

k_{N20} = rate constant at 20°C

$\theta = 1.10$

T = liquid temperature (°C).

In the early stages of a nitrification RBC, the ammonia oxidation proceeds at a zero order rate until the ammonia reaches a concentration of about 5 mg/L. At 20°C this maximum rate is 3 g/m²·day with the temperature dependence as described by eq 13 (EPA, in press). Below 5 mg/L the nitrification proceeds at first order rates as described by Monod growth kinetics, with half saturation constants of 2.80 mg/L at 20°C and 0.45 mg/L at 15°C (EPA, in press).

The ammonia removal in the early stages is given by eq 14:

$$N_S = \frac{(A_S)(R_N)}{1000} \quad (14)$$

where N_S = ammonia oxidized in the stage (kg/day)

A_S = surface area of disks in the stage (m²)

R_N = 3 g/m²·day at 20°C (use eq 13 to adjust to other temperatures).

When the calculations based on eq 14 indicate that the ammonia concentration is close to 5 mg/L in a particular stage, then eq 15 is used to predict the ammonia concentration in each of the remaining stages.

$$N_e = N_o - \frac{A_S k_N}{Q} \frac{N_e}{(N_e + C_N)} \quad (15)$$

where N_e = effluent ammonia in the stage (mg/L)

N_o = influent ammonia entering the stage (mg/L)

A_S = surface area in the stage (m²)

k_N = removal rate constant (g/m²·day)

= 3.74 at 20°C (use eq 13 for adjustment)

$$\begin{aligned}
 C_N &= \text{half saturation constant (mg/L)} \\
 &= 0.45 \text{ at } 15^\circ\text{C} \\
 &= 2.80 \text{ at } 20^\circ\text{C}.
 \end{aligned}$$

It is necessary to assume a hydraulic loading and the surface area per stage to solve eq 14 and 15 on a trial and error basis until ammonia design requirements are satisfied. Manufacturers' literature can provide information on both factors. The equations are also based on the assumption that the BOD entering the first stage of the nitrification RBC is low (< 30 mg/L). Higher organic loadings will inhibit nitrification and an adjustment factor is necessary (see Brenner et al. 1983).

Design example

The following conditions apply for this example:

Average design flow = 1.3 m³/min.

BOD and TSS < 30 mg/L.

Ammonia in wastewater: average = 20 mg/L; peak = 30 mg/L.

Wastewater temperature: summer = 20°C; winter = 10°C.

Effluent ammonia requirements: summer = 1.0 mg/L; winter = 15 mg/L.

Design flow = (1.3 m³/min.)(1440 min./day) = 1872 m³/day.

Ammonia loading:

average conditions = (1872 m³/day)(20 g/m³)(10⁻³ kg/g) = 37.4 kg/day

peak conditions = 56.2 kg/day.

Try hydraulic loading of 0.08 m³/m²·day.

Total surface area needed:

$$\frac{1872 \text{ m}^3/\text{day}}{0.08 \text{ m}^3/\text{m}^2 \cdot \text{day}} = 23,400 \text{ m}^2.$$

Assume a 4-m-long shaft with 4000 m² of surface area.

Use three stages with two shafts each:

$$(4000 \text{ m}^2/\text{shaft})(2 \text{ shafts}/\text{stage})(3 \text{ stages}) = 24,000 \text{ m}^2, \text{ acceptable.}$$

Calculate, with eq 14, the nitrification in the early stages at 20°C:

$$\frac{(4000 \text{ m}^2/\text{shaft})(2 \text{ shafts}/\text{stage})(3 \text{ g}/\text{m}^2 \cdot \text{day})}{1000 \text{ g}/\text{kg}} = 24 \text{ kg}/\text{day} \cdot \text{stage.}$$

Calculate first stage effluent with peak ammonia input:

$$\frac{(52.6 \text{ kg}/\text{day} - 24 \text{ kg}/\text{day})(1000 \text{ g}/\text{kg})}{(1.3 \text{ m}^3/\text{min.})(1440 \text{ min.}/\text{day})} = 17.2 \text{ mg}/\text{L.}$$

Calculate second stage effluent:

Ammonia entering:

$$(56.2 \text{ kg}/\text{day} - 24 \text{ kg}/\text{day}) = 32.2 \text{ kg}/\text{day.}$$

Effluent concentration:

$$\frac{(32.2 - 24)(1000)}{(1.3)(1440)} = 4.4 \text{ mg}/\text{L.}$$

To be conservative, assume the second stage can only produce 5 mg/L effluent. Calculate third stage effluent with eq 15:

$$\begin{aligned}
 N_e &= N_o - \frac{A_s k n}{Q} \frac{N_e}{(N_e + C_N)} \\
 &= 5 - \frac{(8000)(3.74)}{(1872)} \frac{N_e}{(N_e + 2.8)} \\
 &= 5 - \frac{16N_e}{(N_e + 2.8)}
 \end{aligned}$$

$$(N_e)^2 + 13.8N_e - 14 = 0$$

$$(N_e + 6.9)^2 = 61.6$$

$$N_e = 0.95 \text{ mg/L.}$$

Less than 1 mg/L, acceptable. Check for winter conditions at 10°C with peak ammonia concentrations; calculate nitrification rate at 10°C with eq 13:

$$(3 \text{ g/m}^2 \cdot \text{day})(1.10)^{(10-20)} = 1.16 \text{ g/m}^2 \cdot \text{day.}$$

Nitrification in early stages:

$$\frac{(4000)(2)(1.16)}{(1000)} = 9.3 \text{ kg/day} \cdot \text{stage.}$$

First stage effluent:

$$\frac{(56.2 - 9.3)(1000)}{1872} = 25 \text{ mg/L.}$$

Second stage effluent:

$$\frac{(46.9 - 9.3)(1000)}{1872} = 20 \text{ mg/L.}$$

Third stage effluent: at 10°C $k_N = (3.74)(1.10)(10-20) = 1.44$, use $C_N = 0.45$.

$$N_e = - \frac{(8000)(1.44)}{1872} \frac{N_e}{(N_e + 0.45)}$$

$$(N_e)^2 - 13.4N_e - 9 = 0$$

$$(N_e - 6.7)^2 = 53.6$$

$$N_e = 14 \text{ mg/L.}$$

Less than 15 mg/L, acceptable.

Winter operations

It is common design practice either to house each of the shafts or stages with a rigid cover or to put the units in a building. The purpose of the cover is to exclude sunlight, avoid dessication of the biological growth and to protect the unit from extreme ambient temperatures. As a result, RBC systems will operate at or near the wastewater temperature and will not be significantly influenced by low ambient air temperatures. The exception is the case where RBCs are added to upgrade the effluent from an existing system where winter conditions result in significant cooling of the wastewater. In these cases a thermal analysis should be part of the project design to be sure that suitable temperatures will prevail in the RBC stages designed for nitrification.

Dissolved oxygen levels should be maintained between 3 and 4 mg/L to support the nitrification reactions, and pH should be maintained at 7.0 or higher. The design example above was based on peak nitrogen levels. An alternative is to consider prior flow equalization to attenuate the peaks. A cost comparison will then indicate whether equalization or extra media surface area is more cost-effective.

As noted in the earlier section on trickling filters, the biological growths in attached growth nitrification result in minimal excess solids and sloughing so that a final clarifier is not required. Similar results were observed at the nitrification RBCs at Cadillac, Michigan (Miller et al. 1980). This suggests that final clarification should not be necessary for RBCs dedicated for nitrification as long as the BOD/TSS entering the units is less than about 20 mg/L. Final clarification is necessary for RBC units designed for the combined removal of BOD and nitrogen.

ACTIVATED SLUDGE SYSTEMS

Introduction

In contrast to trickling filters and RBCs, activated sludge systems depend on biological growth suspended in the wastewater. Aeration or mechanical mixing, or both, are used to keep the biological solids suspended in the liquid and to supply the necessary oxygen. A wide variety of process configurations exist, each with its own name and nomenclature, but the basic treatment responses are common to all. In essence, the suspended heterotrophic organisms remove the soluble BOD from the wastewater and thereby increase the total mass of sludge. The sludge is separated in a clarifier and a portion returned to the aeration tank to keep the biological population stable and active. The autotrophic organisms are also present, but as in the previous cases cannot grow and remove ammonia in the presence of high BOD concentrations.

In contrast to trickling filters and RBCs, a suspended growth nitrification reactor cannot contain only the autotrophic nitrifiers. These organisms prefer attachment to surfaces and do not readily flocculate and settle. If these conditions are not satisfied, there will be a continual "wash out" of organisms as individual cells, and nitrification will not proceed at acceptable rates. It is necessary to maintain a significant level of organic material in the wastewater entering the unit. Conversion of this material by the heterotrophs present will produce enough biomass to serve as the substrate for the nitrifiers and to encourage flocculation so that the biomass stays in the system. A BOD level of about 50 mg/L is recommended for wastewater entering the nitrification unit to ensure the necessary responses (EPA, in press). These nitrification reactors, if properly designed and operated, can produce an effluent ammonia concentration of less than 0.5 mg/L (EPA, in press).

The suspended growth reactor can follow activated sludge, trickling filter or RBC units designed for BOD removal. In the case of activated sludge, the unit would typically follow the

secondary clarifier. If trickling filters or RBCs are the preliminary step for BOD removal, a clarifier is not required between these units and the suspended growth nitrification reactor. A final clarifier, after the nitrification reactor, is required in all cases since suspended solids will exceed 30 mg/L.

Design procedure

The design method for these reactors uses the kinetic theory of organism growth and the "solids retention time" approach commonly used for activated sludge systems. The maximum growth rate for the nitrifier organisms is determined with eq 16:

$$k_{ST} = k_{S15} e^{(0.12)(T-15^{\circ}\text{C})} \quad (16)$$

where k_{ST} = temperature-dependent suspended growth rate for nitrifiers (days⁻¹)

k_{S15} = maximum rate at 15°C (days⁻¹), typically = 0.18 days⁻¹

T = temperature (°C).

The growth rate is also affected by the dissolved oxygen (DO) concentration and the pH in the reactor. These can vary with the type of system, flow pattern, etc. A pure oxygen system will, for example, have a high DO but may also have a low pH because of retained CO₂ in the covered unit. Equation 17 can be used to estimate the effects of these factors on the growth rate:

$$k_{ST}' = k_{ST} \left(\frac{\text{DO}}{\text{DO} + 1.3} \right) (f_{\text{pH}}) \quad (17)$$

where k_{ST}' = suspended growth rate for nitrifiers adjusted for DO and pH

DO = effective dissolved oxygen concentration in reactor (mg/L)

f_{pH} = reduction factor for pH effects

= 0.5 (pH T < 7.0)

= 0.8 (pH \approx 7.5)

= 1.0 (pH \approx 8).

It is then necessary to determine the solids retention time and the amount of solids wasted per day to determine the size and the detention time of the suspended growth reactor. The minimum solids retention time to prevent washout of the nitrifying organisms is the reciprocal of the growth rate:

$$x_t = (1/k_{ST}') (c_p) \quad (18)$$

where x_t = minimum solids retention time (days) and c_p = peaking factor. The peaking factor in eq 18 is necessary to ensure satisfactory performance under all conditions. It is determined by comparing the mass loading of ammonia under peak conditions to the mass loading under average conditions.

The amount of solids to be wasted daily is given by eq 19, 20, 21 and 22:

$$M_c = \frac{Q(v_c)(S_o - S_e)}{[1 + k_{ST}'(0.05)](f_v)} \quad (19)$$

where M_c = mass of carbonaceous biomass (g/day)

Q = design flow (m³/day)

y_c = yield coefficient (g Volatile SS/g BOD); 0.6 for typical cases
 S_o = influent BOD (mg/L)
 S_e = effluent BOD (mg/L)
 f_v = fraction of biomass that is volatile
 = 0.85, typical case.

$$M_N = \frac{Q(y_N)(N_o - N_e)}{f_v} \quad (20)$$

where M_N = mass of nitrifier biomass (g/day), y_N = yield coefficient (g VSS/g NH_4); 0.1 for typical cases and other terms as defined above.

$$M_I = Q(s_I) \quad (21)$$

where M_I = mass of inert solids (g/day), s_I = inert suspended solids in influent (mg/L), and other terms as defined above.

$$M_T = \frac{M_c + M_N + M_I}{1000} \quad (22)$$

where M_T = total mass of solids (kg/day) and other terms as defined above.

The solids retention time (x_t) can also be expressed as the ratio of the solids under aeration to the solids wasted per day:

$$x_t = \frac{(MLSS)(V)}{M_T} \quad (23)$$

where x_t = solids retention time (days)

MLSS = Mixed Liquor Suspended Solids in nitrification reactor (kg/m³)

V = volume of reactor tank (m³).

The MLSS to be adopted for design will depend on system type and temperature conditions. A value of 5000 mg/L would be appropriate for warm climates or pure oxygen systems. A range of 2500-3000 mg/L is recommended for conventional aeration in cold climates. Equation 23 can be rearranged to determine tank volume:

$$V = \frac{(x_t)(M_T)}{(MLSS)} \quad (24)$$

The hydraulic detention time is given by:

$$t = V/Q \quad (25)$$

where t = hydraulic detention time (days) and other terms as defined above.

It is necessary to design the aeration equipment, the final clarifier and the sludge recycle to complete the process design. The oxygen requirements can be determined with eq 26:

$$O_2 = (1.5 \times 10^{-1})[(Q)(S_o) + 2(Q)(N_o) + 0.81(M_c + M_N)] \quad (26)$$

where O_2 = oxygen required (kg/day) and other terms as defined previously.

Design of the aeration equipment to produce the required oxygen uses conventional procedures found in numerous text books and design manuals. Diffused aeration systems are recommended for these separate nitrification reactors in cold climates and when prior flow equalization is not provided in the system to permit greater flexibility of control and lower heat losses.

Clarifier design also follows conventional practice; typical values are 1 to 1.4 m/hr overflow rates (EPA, in press). The return sludge concentration from this clarifier will usually range from 8000 to 17,000 mg/L (pure oxygen systems at high end). The return sludge system is usually designed for 100% recycle capacity to ensure a reserve. Under normal conditions the recycle will be at about 50% of capacity.

Design example

The following conditions apply for this example:

Average design flow = 1.3 m³/min.

BOD entering reactor = 50 mg/L.

Inert suspended solids = 15 mg/L.

Ammonia in wastewater: average = 20 mg/L; peak = 30 mg/L.

pH in aeration tank = 7.5.

O₂ in aeration tank = 2 mg/L.

Wastewater temperature: summer = 20°C; winter = 10°C.

Effluent ammonia requirements = 3 mg/L.

Effluent BOD = 5 mg/L.

Design flow, $Q = (1.3 \text{ m}^3/\text{min.})(1440 \text{ min./day}) = 1872 \text{ m}^3/\text{day}$.

Determine maximum growth rate at lowest temperature with eq 16:

$$k_{ST} = 0.18 e^{(0.12)(10-15)}$$

$$= 0.10 \text{ days}^{-1}$$

Adjust for DO and pH with eq 17:

$$k'_{ST} = (0.10)[2/(2 + 1.3)](0.8)$$

$$= 0.05 \text{ days}^{-1}$$

Determine solids retention time with eq 18:

$$x_t = (1/0.05)(30/20) = 30 \text{ days}$$

Determine carbonaceous solids with eq 19:

$$M_d = \frac{(1872)(0.6)(50-5)}{[1 + (0.05)(0.05)](0.85)}$$

$$= 59,315 \text{ g/day}$$

Determine nitrifier biomass with eq 20:

$$M_N = \frac{(1872)(0.1)(20-3)}{0.85}$$

$$= 3744 \text{ g/day}$$

Determine inert solids with eq 21:

$$M_i = (1872)(15) = 28,080 \text{ g/day.}$$

Total solids will be:

$$M_T = \frac{59,315 + 3744 + 28,080}{1000} = 91.14 \text{ kg/day.}$$

Use eq 24 to determine reactor volume: assume mixed liquor solids = 3000 mg/l or MLSS = 3 kg/m³

$$V = [(30)(91.14)]/3 = 911.4 \text{ m}^3.$$

Hydraulic detention time:

$$t = 911.4/1872 = 0.49 \text{ days} \\ = 11.7 \text{ hours.}$$

Determine oxygen requirements with eq 26:

$$O_2 = (1.5 \times 10^{-3})[1872(50) + 3(1972)(20) + 0.81(59,315 + 3744)] \\ = 394 \text{ kg/day.}$$

The design approach illustrated above is very conservative and will ensure satisfaction of ammonia effluent requirements at all times.

Winter operations

The use of seed organisms from another nitrification source is recommended for startup of these suspended growth systems because the nitrifiers are present in low concentrations in most wastewaters. Startup and effective performance of a unit may take several months without use of seed organisms. Startup under winter conditions should be avoided, if possible, even if seed organisms are available.

Operational experience after startup will indicate the optimum solids retention time and MLSS for the system. EPA (in press) provides guidance on this aspect.

COMPARISON OF ALTERNATIVES

The temperature dependence of the processes discussed in earlier sections are compared in Figure 4. The curve for pond systems describes ammonia removal via volatilization; the other three systems depend on ammonia conversion to nitrate. The RBC curve follows eq 13 up to about 13°C. Beyond that point, field experience indicates minimal temperature dependence. Although the figure indicates a positive rate constant for all processes at 5°C, the actual removal or conversion of ammonia at that point is minimal to zero.

The figure clearly indicates that attached growth processes (RBC, trickling filters) are more effective per unit time than the suspended growth (activated sludge, ponds) systems. Wastewater ponds cannot compete with the other high rate processes on a unit time basis.

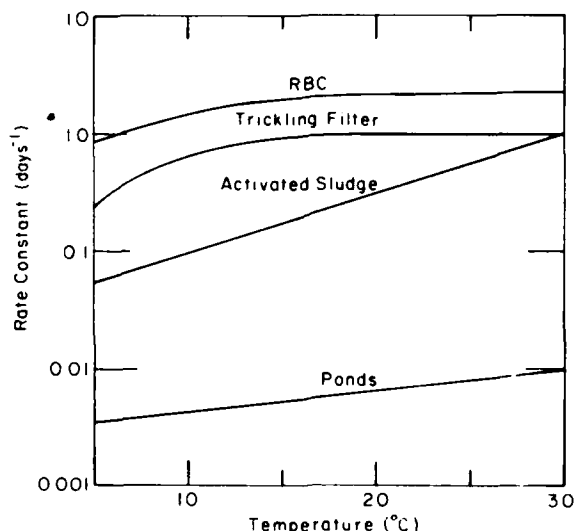


Figure 4. Rate constants vs temperature.

but the very long detention period in pond systems compensates. In locations where land is available at low costs, ponds can be a cost-effective alternative.

If very stringent (less than 1 mg/L) year-round ammonia requirements prevail, then a pond system, by itself, may not be feasible. However, in situations where new ammonia limits are imposed on an existing pond, or where the pond is a preliminary component in a land treatment system, the calculation procedures presented in this report and in Reed (1984) should be used to determine nitrogen removals in the pond prior to the design of any subsequent components.

Attached growth processes would seem to be the best choice for dedicated nitrification systems. The apparent difference between RBCs and trickling filters shown in Figure 4 may not be significant because of the limited source data and the accuracy of the calculation procedures. The final selection between the two processes should be based on cost and reliability. There may be an advantage to RBCs for winter operation in very cold climates because of lower potential heat losses. However, a covered trickling filter unit without recycle and with the air flow regulated can probably do as well.

The activated sludge suspended growth process has potential as an add-on retrofit to existing activated sludge systems where it may be important to retain continuity in mechanical equipment and operator experience. The maintenance requirements for the nitrification unit will then be essentially the same as the other existing components in the system.

If very stringent (less than 0.5 mg/L) ammonia limitations are required, it may be necessary to use the plug flow activated sludge process because it can be designed and operated to produce an average effluent with less than 0.5 mg/L (EPA, in press). The lower limits for RBCs are between 0.5 and 2 mg/L, and 1 to 3 mg/L for trickling filters (EPA, in press).

The attached growth processes are suggested for U.S. Army facilities when the discharge limits for ammonia are 1 mg/L or higher. These systems are less complex, easier to operate and produce less sludge than the suspended growth activated sludge alternative.

CONCLUSIONS

1. Significant levels of ammonia removal or conversion are possible with the four wastewater processes discussed in this report.

2. Most of the nitrogen removal in pond systems is attributable to ammonia losses to the atmosphere. The other processes depend on biological conversion of ammonia to nitrate.

3. It may be necessary to use the activated sludge process for nitrification when stringent effluent (less than 0.5 mg/L) standards prevail. It may be desirable to use it as an add-on to existing suspended growth systems to preserve continuity of equipment and to meet maintenance requirements. The attached growth processes are suggested for all other situations on U.S. Army facilities.

4. Diffused air should be used for aeration of activated sludge units in cold climates.

5. The final selection between RBCs and trickling filters should be based on cost and reliability factors. The trickling filter process is more likely to experience thermal stress and can be more difficult to operate in very cold climates.

6. Startup of nitrification units should not be attempted in the winter months in cold climates. The use of seed organisms to assist in startup is recommended.

LITERATURE CITED

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