CHAPTER 6
NUTRIENT REMOVAL

6.1 INTRODUCTION
While the reliability of pond systems to remove BOD$_5$ and suspended solids is well-documented, the N and P removal capability of wastewater ponds has been given little consideration in any type of system design until recently. As more stringent nutrient standards are adopted, nutrient removal processes must be included in design for new systems and added to existing systems. Nitrogen removal can be critical in many situations since NH$_3$, even at low concentrations, can adversely affect aquatic life in receiving waters, and the addition of NO$_3$ to surface waters is a major contributor to eutrophication. Nitrate is often the controlling parameter for design of land treatment systems. Any N removal in the primary pond units can result in very significant savings in acreage required for final land treatment. Phosphorus, which is limiting for algal growth, is present at concentrations in municipal wastewater that stimulate that growth and must be reduced to control eutrophication.

The dominant forms of N coming into a conventional facultative wastewater treatment pond system are referred to as the Total Kjeldahl Nitrogen (TKN), which is the sum of the organic N, NH$_3$ and ammonium ions (NH$_4^+$). In biological systems, such as aerobic ponds, where the pH is usually less than 8.0, the majority of the NH$_3$ is in the ionic form. TKN can be reduced through several processes, including gaseous NH$_3$ stripping to the atmosphere, NH$_3$ assimilation into the biomass, biological nitrification/denitrification and sedimentation of insoluble organic N. These processes are affected by temperature, DO concentration, pH value, retention time and wastewater characteristics. Within bottom sediments under anoxic conditions in facultative ponds, denitrification can take place. Temperature, redox potential and sediment characteristics affect the rate of denitrification. In well-designed aerated ponds with good mixing conditions and distribution of DO, however, the effect on the rate of denitrification will be negligible.

The capacity of conventional facultative and aerated ponds to convert NH$_3$ is discussed in the following sections. Several commercial processes that have been developed for N removal are also described. There is, however, little operational data available to demonstrate the effectiveness of the commercial units.

6.2 FACULTATIVE PONDS

6.2.1 Removal Mechanisms
Nitrogen loss from streams, lakes, impoundments, and wastewater ponds has been observed for many years. Data on N losses in pond systems were not sufficient to conduct a comprehensive analysis until the early 1980’s, and even then there was no agreement as to the mechanisms of removal. Investigators have suggested algal uptake, sludge deposition, adsorption by bottom soils, nitrification, denitrification, and loss of NH$_3$ as a gas to the atmosphere (volatilization). Evaluations by Pano and Middlebrooks (1982), Reed (1984b) and Reed et al. (1995) indicate that a combination of factors may be responsible. The dominant mechanism, under favorable
conditions, is thought to be loss by volatilization to the atmosphere. The several mechanisms are depicted in Figure 6-1.

![Figure 6-1. Generalized N pathways in wastewater ponds.](image)

**6. 2.2 Performance**

EPA undertook a number of studies of facultative wastewater pond systems in the late 1970's (Bowen, 1977; Hill and Shindala, 1977; McKinney, 1977 and Reynolds et al., 1977). The results verified the hypothesis that significant N removal occurs in pond systems. Their findings from those studies are summarized in Table 6-1. Nitrogen removal is related to pH, detention time, and temperature. pH fluctuation resulting from the interaction of algae and $HCO_3^-$ changes the alkalinity and is an important parameter to monitor. Under ideal conditions, up to 90 percent $NH_3$ removal can be achieved in facultative wastewater treatment ponds.
Table 6-1. Annual Values from EPA Facultative Wastewater Pond Studies

<table>
<thead>
<tr>
<th>Location</th>
<th>HRT (d)</th>
<th>Temp (°C) Ave. range</th>
<th>pH (median)</th>
<th>Alkalinity (mg/L)</th>
<th>N Removal (percent)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Peterborough</td>
<td>107</td>
<td>-7 – 20</td>
<td>7.1</td>
<td>85</td>
<td>43</td>
</tr>
<tr>
<td>3 cells</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Kilmichael</td>
<td>214</td>
<td>4.5 – 26</td>
<td>8.2</td>
<td>116</td>
<td>80</td>
</tr>
<tr>
<td>3 cells</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Eudora</td>
<td>231</td>
<td>1.1 – 27</td>
<td>8.4</td>
<td>284</td>
<td>82</td>
</tr>
<tr>
<td>3 cells</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Corinne</td>
<td>42</td>
<td>-3.9 – 23</td>
<td>9.4</td>
<td>555</td>
<td>46</td>
</tr>
<tr>
<td>First 3 cells</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Several studies of N removal have been completed more recently, but the quantity of data is still limited. A study of 178 facultative ponds in France showed an average N removal of 60 to 70 percent (Racault et al., 1995). Wrigley and Toerien (1990) studied four small-scale facultative ponds in series for 21 months and observed an 82 percent reduction in NH₃.

Shilton (1995) quantified the removal of NH₃ from a facultative pond treating piggery wastewater, and found that the rate of volatilization varied from 355 to 1534 mg/m²/d (0.07 - 0.314 lb/1000 ft²/d). The rate of volatilization increased at higher concentrations of NH₃ and TKN.

Soares et al., (1995) monitored NH₃ removal in a wastewater treatment pond complex of different geometries and depths in Brazil, and found that the NH₃ concentrations were lowered to 5 mg/L in the maturation ponds so that the effluent could be discharged to surface waters. The NH₃ removal in the facultative and maturation ponds could be modeled by the equations based on the volatilization mechanism proposed by Pano and Middlebrooks (1982).

Using ¹⁵N-labelled NH₃, Camargo Valero and Mara (2007) demonstrated the uptake of NH₃ by the algal biomass in the pond, followed by assimilation into the suspended organic fraction (85 percent in the effluent), and movement into the pond benthos by sedimentation. A study of ponds in Kansas (Tate et al., 2002) showed increased NH₃ in August, which would tend to confirm that it is taken up by algae, and its movement to the benthos, from which it is released under late summer conditions.

6.2.3 Theoretical Considerations

It is hypothesized that NH₃ removal in facultative wastewater treatment ponds occurs via three mechanisms: gaseous NH₃ stripping to the atmosphere, NH₃ assimilation in algal biomass and biological nitrification.

The low concentrations of NO₃⁻ and NO₂⁻ measured in pond effluents indicate that nitrification generally does not account for a significant portion of NH₃ removal. Ammonia assimilation in algal biomass depends on the biological activity in the system and is affected by temperature, organic load, detention time, and wastewater characteristics. The rate of gaseous NH₃ losses to
the atmosphere depends mainly on the pH value, temperature, and the mixing conditions in the pond. Alkaline pH shifts the equilibrium equation $NH_3^+ + H_2O \leftrightarrow NH_4^+ + OH^-$ toward gaseous $NH_3$, whereas the mixing conditions affect the magnitude of the mass transfer coefficient. Temperature affects both the equilibrium constant and mass transfer coefficient.

At low temperatures, when biological activity decreases and the pond contents are generally well mixed due to wind effects, stripping will be the major process for $NH_3$ removal in facultative wastewater treatment ponds. The $NH_3$ stripping process in ponds may be expressed by assuming a first-order reaction (Stratton, 1968; 1969). The mass balance equation will be:

$$VdC/dt = Q(C_o - C_e) - kA(NH_3) \quad (6-1)$$

where:

- $Q$ = flow rate, m$^3$/d
- $C_o$ = influent concentration of $(NH_4^+ + NH_3)$, mg/L as N
- $C_e$ = effluent concentration of $(NH_4^+ + NH_3)$, mg/L as N
- $C$ = average pond contents concentration of $(NH_4^+ + NH_3)$, mg/L as N
- $V$ = volume of the pond, m$^3$
- $k$ = mass transfer coefficient, m/d
- $A$ = surface area of the pond, m$^2$
- $t$ = time, d

The equilibrium equation for $NH_3$ dissociation may be expressed as

$$K_b = [NH_4^+][OH^-]/[NH_3] \quad (6-2)$$

where:

- $K_b = NH_3$ dissociation constant.

By modifying Equation 6-2, gaseous $NH_3$ concentration may be expressed as a function of the pH value and total $NH_3$ concentration $(NH_4^+ + NH_3)$ as follows:

$$[H^+] = KW [OH^-] \quad (6-3)$$

$$C = NH_4^+ + NH_3 \quad (6-4)$$

$$NH_3 + \frac{C}{1 + 10^{pKW - pH - pKb}} \quad (6-5)$$

where:

- $pKW = -\log KW$
- $pKb = -\log K$
Assuming steady-state conditions and a completely mixed pond where $C_e = C$, Equations 6-4 and 6-5 will yield the following relationship:

$$
\frac{C_e}{C_o} = \frac{1}{1 + \frac{AK}{Q} \left[ \frac{1}{1 + 10^{pK_W - pK_b - pH}} \right]}
$$

(6 - 6)

This relationship emphasizes the effect of $pH$, temperature ($pK_W$ and $pK_b$ are functions of temperature) and hydraulic loading rate on $NH_3$ removal.

Experiments on $NH_3$ stripping conducted by Stratton (1968, 1969) showed that the $NH_3$ loss-rate constant was dependent on the $pH$ value and temperature ($T = °C$) as shown in the following relationships:

$$
NH_3 \text{ loss rate constant } \propto e^{1.57(pH-8.5)}
$$

(6 - 7)

$$
NH_3 \text{ loss rate constant } \propto e^{0.13(T-20)}
$$

(6 - 8)

King (1978) reported that only four percent $N$ removal was achieved by harvesting floating Cladophora fracta from the first pond in a series of four receiving secondary effluents. The major $N$ removal in the ponds was attributable to $NH_3$ gas stripping. The removal of total $N$ was described by first-order kinetics, using a plug flow model ($N_t = N_0 e^{-0.03t}$ where $N_t$ = total $N$ concentration, mg/L, $N_0$ = initial total $N$ concentration, mg/L and $t$ = time, d).

During windy seasons, or large-scale facultative steady-state conditions, will well-designed ponds approach completely mixed conditions. Moreover, when $NH_3$ removal through biological activity becomes significant, or $NH_3$ is released from anaerobic activity at the bottom of the pond, the expressions for $NH_3$ removal in the system must include these factors along with the theoretical consideration of $NH_3$ stripping. A mathematical relationship for total $N$ removal based on the performance of three full-scale facultative wastewater treatment ponds is developed here that considers the theoretical approach and incorporates temperature, $pH$ value, and hydraulic loading rate as variables. In this case, Equation 6-9 for TKN removal in facultative ponds is substituted for the theoretical expression for $NH_3$ stripping:

$$
\frac{C_e}{C_o} = \frac{1}{1 + \frac{AK \times f(pH)}{Q}}
$$

(6 - 9)

where:

$K = \text{removal rate coefficient (L/t)}$

$f(pH) = \text{function of pH}$.

The $K$ values are considered to be a function of temperature and mixing conditions. For a similar pond configuration and climatic region, the $K$ values may be expressed as a function of temperature only. The function of $pH$, which is considered to be dependent on temperature, affects the $pK$, and $pK_b$ values, as well as the biological activity. Based on a statistical analysis of the data when incorporated into the equation, the $pH$ function was found to describe an
exponential relationship. As many reaction rate and temperature relationships are described by exponential functions, such as the Van’t Hoff-Arrhenius equation, it is not unreasonable to assume that such a relationship would apply to the application of the theoretical equation to a practical problem.

6.2.4 Design Models
Data were collected on a frequent schedule from every cell in the pond system listed in Table 6-1 for at least a full annual cycle. A quantitative analysis of all major variables was performed and several design models were developed. Both plug flow and complete mixing models were useful in predicting $N$ removal in facultative pond systems (Tables 6-2 and 6-3). They are first-order models that depend on $pH$, temperature and HRT. They have been validated using data from other sources. Further validation of the models can be found in Crites et al. (2006), Reed et al. (1995), Reed (1985), and Reed (1984b).

Both models assume that volatilization of $NH_3$ is the major pathway of $N$ removal from wastewater treatment ponds. The application of the models is shown in Figure 6-2, and the predicted total $N$ in the effluent is compared to the actual monthly average values measured at the Peterborough, New Hampshire ponds. The models are expressed in terms of total $N$, and should not be confused with the equations reported by Pano and Middlebrooks (1982) that are limited to the $NH_3$ fraction.

![Figure 6-2. Predicted versus actual effluent $N$, Peterborough, New Hampshire.](image-url)
Table 6-2. Model 1, N Removal in Facultative Ponds – Plug Flow Model (Reed, 1985)

\[ N_e = N_0 e^{-K_T [t + 60.6 (pH - 6.6)]} \]  
\[ (6 - 10) \]

where:

- \( N_e \) = effluent total nitrogen, mg/L
- \( N_0 \) = influent total nitrogen, mg/L
- \( K_T \) = temperature dependent rate constant
- \( K_T = K_{20} (\theta)^{T-20} \)
- \( K_{20} = \text{rate constant at 20}^\circ\text{C} = 0.0064 \)
- \( \theta = 1.039 \)
- \( t \) = detention time in system, d
- \( pH = \text{pH of near surface bulk liquid} \)

See Reed (1984b) for typical pH values or estimate with:

\[ pH = 7.3e^{0.0005ALK} \]

Use the Mancini and Barnhart Equation (1976) for pond water temperature determination.

where:

- \( ALK = \text{expected influent alkalinity, mg/L [derived from data in U.S. EPA (1983a) and Reed (1984b)]} \)

\[ T = \frac{0.5AT_a + QT_i}{0.5A + Q} \]

where:

- \( A \) = surface area of pond, m\(^2\)
- \( T_a \) = ambient air temperature, °C
- \( T_i \) = influent temperature, °C
- \( Q \) = influent flow rate, m\(^3\) / d

A high rate of NH\(_3\) removal by air stripping in advanced wastewater treatment depends on a high (> 10) chemically adjusted pH. The algae-carbonate interactions in wastewater ponds can elevate the pH to similar levels for brief periods during the day. At other times, at lower pH levels, the rate of removal may be slower, but the long detention time results in lower N concentrations in the effluent.

Table 6-3. Model 2, N Removal in Facultative Ponds – Complete Mix Model (Middlebrooks, 1985).

\[ N_e = \frac{N_0}{1 + t(0.000576T - 0.00028)e^{(0.0880 - 0.0427)(pH - 6.6)}} \]  
\[ (6 - 11) \]
where:

\[
\begin{align*}
N_e & = \text{effluent total nitrogen, mg/L} \\
N_i & = \text{influent total nitrogen, mg/L} \\
\tau & = \text{detention time, d} \\
T & = \text{temperature of pond water, °C} \\
pH & = \text{pH of near surface bulk liquid}
\end{align*}
\]

Use the Mancini and Barnhart (1976) Equation (Table 6-2) to determine pond water temperature.

Figure 6-3 (Crites, 2006) illustrates the validation of both models using data from pond systems from other sources. The diagonal line represents the best fit of predicted versus actual values. The close fit and consistent trend demonstrate that either model can be used to estimate N removal. In addition, the models have been used in the design of several ponds systems and have been found to work as predicted. It is nevertheless likely that other removal mechanisms are at work (Camargo Valero and Mara, 2007) and therefore this model should be used only as a first step in designing for NH₃ removal in ponds.

![Figure 6-3. Verification of design models.](image)

6.2.5 Applicability
Nitrogen removal occurs in facultative wastewater treatment ponds, and may be reasonably predicted for design purposes with either of the two models. Nitrogen removal in ponds may be more cost-effective than other alternatives for removal and/or NH₃ conversion. These models should be useful for new or existing wastewater ponds when N removal and/or NH₃ conversion is required. The design of new systems would typically base detention time on the BOD₅ removal requirements. The N removal that will occur during that time can then be calculated with either model. It is prudent to assume that the remaining N in the effluent will be NH₃ and to design any
further removal/conversion for that amount. A final step would be to compare the cost of additional detention time in an expanded pond system for N removal with other removal alternatives.

Use of these models is particularly important when ponds are used as a component in land treatment systems, since total N is often the controlling design parameter. A reduction in pond effluent N will often permit a significant reduction in the area needed for land treatment.

6.2.6 Limitations
Other than the requirement for sufficient pond acreage, the facultative pond system can be expected to provide maximum NH₃ conversion during the summer months, occasionally exceeding 90 percent. This level of treatment cannot be expected during winter months.

6.2.7 Operation and Maintenance
Operation and maintenance are the same as for facultative ponds designed to remove BOD₅.

6.3 AEROBIC PONDS

6.3.1 Introduction
The same conditions that apply to push the equilibrium of NH₄⁺ to NH₃ in conventional facultative pond systems applies to aerobic ponds.

6.3.2 Performance
EPA sponsored comprehensive studies of aerated pond systems between 1978 and 1980 that provided information about N removal (Tables 6-4 and 6-5). The results verify the consensus of previous investigators that N removal is related to pH, detention time and temperature in the pond system.

Table 6-4. Wastewater Characteristics and Operating Conditions for Five Aerated Ponds (Earnest et al., 1978; Englande, 1980; Gurnham et al., 1979; Polkowski, 1979; Reid and Streebin, 1979; Russell et al., 1980)

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Pawnee a</th>
<th>Bixby b</th>
<th>Koshkonong c</th>
<th>Windber d</th>
<th>North Gulfport e</th>
</tr>
</thead>
<tbody>
<tr>
<td>BOD, mg/L</td>
<td>473</td>
<td>368</td>
<td>85</td>
<td>173</td>
<td>178</td>
</tr>
<tr>
<td>COD, mg/L</td>
<td>1026</td>
<td>635</td>
<td>196</td>
<td>424</td>
<td>338</td>
</tr>
<tr>
<td>TKN mg/L</td>
<td>51.41</td>
<td>45.04</td>
<td>15.3</td>
<td>24.33</td>
<td>26.5</td>
</tr>
<tr>
<td>NH₃-N mg/L</td>
<td>26.32</td>
<td>29.58</td>
<td>10.04</td>
<td>22.85</td>
<td>15.7</td>
</tr>
<tr>
<td>Alkalinity mg/L</td>
<td>242</td>
<td>154</td>
<td>397</td>
<td>67</td>
<td>144</td>
</tr>
<tr>
<td>pH</td>
<td>6.8-7.4</td>
<td>6.1-7.1</td>
<td>7.2-7.4</td>
<td>5.6-6.9</td>
<td>6.7-7.5</td>
</tr>
<tr>
<td>Hydraulic loading rate mgd</td>
<td>0.0213</td>
<td>0.0285</td>
<td>0.0423</td>
<td>0.0663</td>
<td>0.0873</td>
</tr>
</tbody>
</table>

See p.xiv for conversion table.
Parameter | Pawnee | Bixby | Koshkonong | Windber | North Gulfport  
|----------|-------|-------|-----------|--------|-----------------  
| Organic loading rate (kg BOD/ha/d) | 151 | 161 | 87 | 285 | 486  
| Detention time, d | 143 | 107 | 72 | 46 | 22  

*a*Illinois; *b*Oklahoma; *c*Wisconsin; *d*Pennsylvania; *e*Mississippi

**Table 6-5. N Removal in Aerated Ponds (adapted from U.S. EPA 1983a)**

| Location | Pawnee, Illinois | Bixby, Oklahoma | Koshkonong, Wisconsin  
|----------|-----------------|----------------|-------------------------  
| Parameter, mg/L | Influent | Effluent | Influent | Effluent | Influent | Effluent  
| TKN | 51.4 | 5.0 | 45.0 | 8.4 | 15.3 | 7.6  
| NH₃ | 26.3 | 1.3 | 29.6 | 3.5 | 10.0 | 5.3  
| NO₃⁻ | - | 0.8 | - | - | 1.7 | 4.4  
| NO₂⁻ | - | 0.1 | - | - | 0.1 | -  
| Alkalinity | 242 | 161 | 154 | 70 | 397 | 382  
| pH | 6.8-7.4 | 7.8-9.3 | 6.1-7.1 | 6.7-9.2 | 7.2-7.4-7.9  
| Temp °C | - | 3-22 | - | 5-29 | - | 1-25  
| DO | - | 1.9-16.0 | - | 3.9-13.5 | - | 7.6-15.3  

| Location | Windber, Pennsylvania | N. Gulfport, Mississippi | Mt. Shasta, California  
|----------|------------------------|--------------------------|--------------------------  
| Parameter, mg/L | Influent | Effluent | Influent | Effluent | Influent | Effluent  
| TKN | 24.3 | 23.6 | 26.5 | 10.8 | 15.7 | 11.1  
| Range | 13.2-46.0 | 14.4-34.1 | 20.6-30.9 | 7.2-13.3 | 10.1-20.9 | 6.8-14.2  
| NH₃ | 22.9 | 22.9 | 15.7 | 5.1 | 10.3 | 5.4  
| NO₃⁻ | - | 0.72 | - | 2.36 | 0.3 | 0.7  
| NO₂⁻ | - | 0.2 | - | 0.6 | 0.2 | 0.5  
| Alkalinity | 67 | 82 | 144 | 102 | 93 | 74  
| pH | 5.6-6.9 | 6.8-8.5 | 6.7-7.5 | 6.8-7.5 | 6.5-7.6 | 7.4-9.7  
| Temp °C | - | 2-24 | - | 11-29 | - | 2.7  
| DO | - | 5.7-15.0 | - | 0.8-9.3 | - | 10.9-14.0  

**6.3.3 Empirical Design Equations**

Table 6-6 contains a summary of selected equations developed to predict NH₃ and TKN removal in diffused air aerated ponds (Middlebrooks and Pano, 1983). All of the equations draw values from the same database. Different combinations of data were chosen and combined to develop several of the equations. The “system” column in Table 6-6 indicates which ponds or series of ponds were used to develop each equation. These data were analyzed statistically and the equations were selected based upon the best statistical fit for the various combinations. It should be noted that the combinations of data sets are not directly comparable.
An examination of the HRT calculated using the various formulas for TKN removal show that the greatest difference between the maximum and minimum detention times calculated from the equation is 14 percent. In view of the variation in methods used, this variability is reasonable. All of the relationships are statistically significant at levels higher than one percent. As a result, any of them may be applied to estimate TKN removal in an aerated pond design. Given the simplicity of the plug flow model and the fraction removed model, it is recommended that one or both be used along with the theoretical models to verify that there will be adequate removal in the event that unusual BOD₅ loading rates are encountered.

Table 6-6. Comparisons of Various Equations to Predict NH₃ and TKN Removal in Diffused-Air Aerated Ponds (Middlebrooks and Pano, 1983).

<table>
<thead>
<tr>
<th>Equation Used to Estimate HRT or Effluent Concentration</th>
<th>Correlation Coefficient</th>
<th>HRT, d</th>
<th>Compared w Max RT % Difference</th>
<th>System</th>
</tr>
</thead>
<tbody>
<tr>
<td>TKN Removal</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>( \ln \frac{C_e}{C_0} = 0.0129(\text{Detention Time}) )</td>
<td>0.911</td>
<td>125</td>
<td>5.3</td>
<td>Ponds 1, 2 and 3</td>
</tr>
<tr>
<td>TKN Removal Rate</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>( \text{TKN}_{\text{rr}} = 0.809(\text{TKN Loading Rate}) )</td>
<td>0.983</td>
<td>132</td>
<td>0</td>
<td>Total System</td>
</tr>
<tr>
<td>TKN Removal Rate</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>( \text{TKN}_{\text{rr}} = 0.0946(\text{BOD}_5 \text{ Loading Rate}) )</td>
<td>0.967</td>
<td>113</td>
<td>14.4</td>
<td>Total System</td>
</tr>
<tr>
<td>TKN Fraction Removed</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>( \text{TKN}_{\text{fr}} =0.0062(\text{Detention Time}) )</td>
<td>0.959</td>
<td>129</td>
<td>2.3</td>
<td>Ponds 1, 2 and 3</td>
</tr>
<tr>
<td>NH₃ Removal</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>( \ln \frac{C_e}{C_0}=-0.0205(\text{Detention Time}) )</td>
<td>0.798</td>
<td>79</td>
<td>40.2</td>
<td>All Data</td>
</tr>
<tr>
<td>NH₃ Removal Rate</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>( \text{NH}<em>3</em>{\text{rr}} = 0.869(\text{NH}_3-\text{N Loading Rate}) )</td>
<td>0.968</td>
<td>92</td>
<td>30.3</td>
<td>Total System</td>
</tr>
<tr>
<td>NH₃ Removal Rate</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>( \text{NH}<em>3</em>{\text{fr}} = 0.0606(\text{BOD}_5 \text{ Loading Rate}) )</td>
<td>0.932</td>
<td>132</td>
<td>0</td>
<td>Total System</td>
</tr>
<tr>
<td>NH₃ Fraction Removed</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>( \text{NH}<em>3</em>{\text{fr}} = 0.0066(\text{Detention Time}) )</td>
<td>0.936</td>
<td>121</td>
<td>8.3</td>
<td>Ponds 1, 2 and 3</td>
</tr>
</tbody>
</table>
Using any of the above expressions will result in an estimate of the TKN removal that is likely to occur in diffused-air aerated ponds. Unfortunately, data are not available to develop similar relationships for surface aerated ponds.

While the relationships developed to predict $NH_3$ removal are significant for all of the equations presented in Table 6-6, the agreement between the calculated HRT for $NH_3$ removal differed significantly from that observed for the TKN data. This is not surprising in view of the variety of mechanisms involved in $NH_3$ production and removal in wastewater ponds, but it does complicate the use of the equations to estimate $NH_3$ removal in aerated ponds.

Statistically, any of the expressions may be used to calculate the HRT required to achieve a certain percent reduction in $NH_3$. Perhaps the best equation to use in designing $NH_3$ removal is the relationship between the fraction removed and the HRT. The correlation coefficient for this relationship is higher than the correlation coefficient for the plug flow model, and both equations are relatively straightforward to use.

### 6.3.4 Nitrogen Removal in Continuous Feed Intermittent Discharge (CFID) Basins

#### 6.3.4.1 Description
Rich (1996, 1999) has proposed CFID basins for use in aerated pond systems for nitrification and denitrification. The systems are designed to use in-basin sedimentation to uncouple the solids retention time from the HRT. The influent flow is continuous. A single basin has a dividing baffle to prevent short-circuiting.

#### 6.3.4.2 Applicability of CFID
Some CFID systems have experienced major operational problems with short-circuiting and sludge bulking; however, by minimizing these problems with design changes the systems can be made to function properly. CFID design modifications can be made to overcome most difficulties and details are presented by Rich (1999).

#### 6.3.4.3 Advantages and Disadvantages
When designed and built correctly, the system is capable of producing an effluent in a pond system comparable with activated sludge systems designed for nitrification/denitrification. Experience with the system, however, is limited.

#### 6.3.4.4 Design Criteria
The basic CFID system consists of a single reactor basin divided into two cells with a floating baffle. The two cells are referred to as the influent (Cell 1) and effluent cell (Cell 2). Mixed liquor is recycled from Cell 2 to the head-works to provide a high ratio of soluble biodegradable organics to organisms and the $O_2$ source is primarily $NO_3^-$. This approach is used to control bulking. Although some nitrification will occur in the influent cell, the system is designed for nitrification to occur in the effluent cell. Further details of the operation of the CFID systems, see Rich (1999).

#### 6.3.4.5 Performance

---

6-12
Performance data were not available.

6.3.4.6 Limitations
There is little proven design information and operational difficulties have been encountered.

6.3.4.7 Operation and Maintenance
It is expected that maintenance would be the same as that required for other aerobic ponds.

6.3.4.8 Costs
Construction costs would be the same as those for conventional aerobic pond systems. Considerable savings would accrue when comparing the cost to produce an effluent quality with a CFID system with an equivalent effluent produced by an activated sludge system designed for nitrification/denitrification.

6.3.4.9 General Applicability
The Rich (1999) method is a design for nitrification in an aerobic pond. The equations in Table 6-6 are empirical and may or may not apply to a general design. That said, they show what might be expected in terms of $N$ removal. Designing a pond system to nitrify wastewater is not difficult if the water temperature and detention time are adequate to support nitrifiers and sufficient DO is supplied. Recycling of the mixed liquor is a significant benefit. As with all treatment methods, an economic analysis should be performed to determine whether this system is cost effective.

6.3.5 Nitrification Using Fixed Film Media
In addition to the proprietary systems described later in this chapter, Reynolds et al., (1975), Polprasert and Agarwalla, (1995), and Ripple (2002) have conducted studies using baffles and suspended materials as media for attached growth for nitrifying organisms. Nitrification is a function of temperature, and where temperature was a factor in the studies, there was a significant decline in nitrification. It is not clear that the impact of winter temperatures can be overcome when the water temperature drops below 10 °C.

6.3.6 Pump Systems, Inc. Batch Study
6.3.6.1 Description
In 1998, a solar-powered circulator (equivalent to the SolarBee Model SB2500) was installed in an 11.7 ha pond with a depth of 4.5 m at Dickinson, North Dakota with no incoming wastewater. The circulator flow rate was 9463 L (2500 G) per minute. The $NH_3$ concentration at the beginning of the experiment was approximately 20 mg/L. Dissolved oxygen, $pH$, BOD$_5$, TSS, $NH_3$, water temperature and other parameters were measured over a 90-day period at various locations and depths. Over 1500 samples were collected over the test period. Average data for the various locations and depths are shown in Table 6-7. The average water temperature was 20.5 °C. DO was present throughout the pond at all depths, but on occasion dropped to 0.4 mg/L at the bottom. These occasional low DO concentrations may have had an adverse effect on the results presented below, but they do provide some guidance as to how to estimate the expected conversion of $NH_3$ in a partial mix aerobic pond system (see Equation 6.9).
Table 6-7. Average Values for Batch Test in Pond 4 at Dickinson, North Dakota Area = 11.7 Ha (29 Ac), No Inflow. (Pump Systems, Inc., 2004).

<table>
<thead>
<tr>
<th>Days</th>
<th>Ln Ce/Co</th>
<th>pH</th>
</tr>
</thead>
<tbody>
<tr>
<td>0</td>
<td>0</td>
<td>7.7</td>
</tr>
<tr>
<td>1</td>
<td>0.15</td>
<td>7.7</td>
</tr>
<tr>
<td>7</td>
<td>0.21</td>
<td>7.7</td>
</tr>
<tr>
<td>12</td>
<td>0.16</td>
<td>8</td>
</tr>
<tr>
<td>13</td>
<td>0.11</td>
<td>8.1</td>
</tr>
<tr>
<td>19</td>
<td>-0.06</td>
<td>8.4</td>
</tr>
<tr>
<td>26</td>
<td>-0.36</td>
<td>8.8</td>
</tr>
<tr>
<td>29</td>
<td>-0.36</td>
<td>8.7</td>
</tr>
<tr>
<td>36</td>
<td>-0.6</td>
<td>8.8</td>
</tr>
<tr>
<td>42</td>
<td>-0.67</td>
<td>8.6</td>
</tr>
<tr>
<td>48</td>
<td>-0.79</td>
<td>8.5</td>
</tr>
<tr>
<td>50</td>
<td>-0.82</td>
<td>8.5</td>
</tr>
<tr>
<td>57</td>
<td>-0.76</td>
<td>7.7</td>
</tr>
<tr>
<td>62</td>
<td>-0.75</td>
<td>8</td>
</tr>
<tr>
<td>70</td>
<td>-0.48</td>
<td>8.1</td>
</tr>
<tr>
<td>76</td>
<td>-0.62</td>
<td>8</td>
</tr>
<tr>
<td>84</td>
<td>-0.74</td>
<td>7.8</td>
</tr>
<tr>
<td>90</td>
<td>-0.91</td>
<td>8.1</td>
</tr>
</tbody>
</table>

$C_o = \text{influent concentration of (NH}_4^+ + \text{NH}_3\text{), mg/L as N}$

$C_e = \text{effluent concentration of (NH}_4^+ + \text{NH}_3\text{), mg/L as N}$

The reduction in $NH_4^+$ with time was directly related to the variation in pH value (Table 6-7). When the pH exceeded 8.0, the reduction in $NH_4^+$ increased, implying a greater loss of the $NH_3$ to the atmosphere.

The results of this experiment show the low reaction rate for nitrification that occurs in partial mix aerobic ponds. The reaction rate of 0.0107/d obtained at an average temperature of 20.5 °C in the Dickinson experiments agrees with results obtained with data collected in an aerobic pond located in Wisconsin (Middlebrooks et al., 1982). At 1 °C, the $NH_3$ conversion reaction rate for the Wisconsin partial mix pond ranged between 0.0035 and 0.0070/ d. Using the average value of 0.005/d at 1 °C and the value of 0.0107/d obtained at Dickinson at 20.5 °C, an approximate value of 1.04 can be inserted into the $\theta$ in the classic temperature correction equation: $k_T = k_20(1.04)^{(T-20)}$. Example C-6-1 in Appendix C illustrates the effects of reaction rates and temperature on the performance of partial mix pond systems.
6.3.7 Nitrogen Removal in Ponds with Wetlands and Gravel Nitrification Filters

The nitrification filter bed (NFB) was developed by Sherwood C. Reed, and the following material was extracted from Reed et al. (1995). The NFB has been installed at three locations in the United States. The system was developed as a retrofit for existing free water surface (FWS) and subsurface flow (SF) wetland systems having trouble meeting \( NH_3 \) effluent standards. Schematic diagrams of both FWS and SF wetlands fitted with NFBs are shown in Figure 6-5. The NFB is a vertical-flow gravel filter bed located on top of existing wetlands. When applied to the FWS wetland, the fine gravel bed is supported on a coarse gravel layer to ensure aerobic conditions in the NFB.

NFB units can be located at the front or near the end of the wetland where wetland effluent is pumped to the top of the NFB and distributed evenly over the surface. Introducing the wetland effluent to the NFB at the head of the system has the advantage of mixing the influent wastewater with the highly nitrified NFB, effluent which results in denitrification and removal of nitrogen from the system. In addition the BOD\(_5\) is reduced, and some of the alkalinity lost during nitrification will be recovered. If the NFB is placed at the end of the wetland, nitrification will occur, but denitrification will be limited and the \( NO_3^- \) will pass out of the system. This will require less pumping capacity, but the advantages of denitrification could easily offset the cost of pumping.

Although similar to a recirculating sand filter, the NFB uses gravel rather than sand and can process a much higher hydraulic loading rate than the sand filter. Hydraulic loading rates, including a 3:1 recycle ratio, for a NFB located in Kentucky is 4 \( m^3/m^2/d \) (100 G/ft\(^2\)/d), in contrast to loadings on recirculating sand filters of 0.2 \( m^3/m^2/d \) (5 G/ft\(^2\)/d).

Trickling filter and rotating biological contactor attached-growth concepts were used to develop a design relationship for the NFB (Equation 6-12). The relationship in Equation 6-12 was derived from curve fitting performance data and should be used with caution. The equation should give reasonable estimates of the specific surface area to produce effluent \( NH_3 \) concentrations between 0 and 6 mg/L. Equation 6-12 has been verified at a 2 MGD system in Mandeville, Louisiana (Reed et al., 2003).

\[
A_v = \frac{2713 - 1115(C_e) + 204(C_e)^2 - 12(C_e)^3}{k_T} \quad (6-12)
\]

where:

\(A_v\) = specific surface area, \( m^2/kg \) \( NH_3/d\)
\(C_e\) = desired NFB effluent \( NH_3\), mg/L
\(k_T\) = temperature-dependent coefficient
\(\theta (NH_3) = 1.048\)
\(\theta (NH_4^+\) = 1.15

At temperatures \( \geq 10 \) \(^\circ\)C, \(k_T = 1 (1.048)^{(T-20)}\)
At temperatures 1-10 °C, $k_T = 0.626(1.15)^{(T-10)}$

The following conditions are necessary for good nitrification performance:
- BOD$_5$:TKN ratio must be less than one
- Sufficient oxygen must be present
- Surface must be moist at all times
- Sufficient alkalinity must be available to support nitrification (approximately 10g of alkalinity per gram of NH$_3$)

The NFB bed depth ranges from 0.3 - 0.6 m and the bed extends across the entire width of the wetland cell to ensure mixing with the influent wastewater (Fig 6-4). Sprinklers are used to distribute the wetland effluent over the surface of the NFB. In cold climates it may be necessary to enclose the NFB to prevent freezing.

The Benton, Kentucky NFB has been operating successfully at a hydraulic design flow of 3.79 x 10$^6$ L/d (1 MGD) per day with a wetland input of 20 mg/L NH$_3$ and an output of 2 mg/L. Performance data for approximately three years for the Benton facility are shown in Table 6-8 (Reed, 2000). Pond influent carbonaceous BOD$_5$ and NFB effluent carbonaceous BOD$_5$ are shown in Figure 6-6. The very large concentration values are probably analytical errors because the TSS concentrations for these days were very low. NFB effluent carbonaceous BOD$_5$, TSS and NH$_3$ concentration variability are shown in Figure 6-7. Ammonia effluent concentrations were well below 5 mg/L with very few exceptions.
Table 6-8. Benton, Kentucky Recirculating Gravel Filter / Constructed Wetland Operational Data (Reed, 2000).

<table>
<thead>
<tr>
<th>Date</th>
<th>Carbonaceous BOD₅ (mg/L) Influent</th>
<th>Effluent</th>
<th>TSS (mg/L) Influent</th>
<th>Effluent</th>
<th>NH₃ (mg/L) Influent</th>
<th>Effluent</th>
</tr>
</thead>
<tbody>
<tr>
<td>April 92</td>
<td>137 (3)</td>
<td>4.2 (3)</td>
<td>89 (4)</td>
<td>4.5 (4)</td>
<td>1.5 (4)</td>
<td></td>
</tr>
<tr>
<td>May</td>
<td>180 (4)</td>
<td>2.6 (4)</td>
<td>273 (4)</td>
<td>2.8 (4)</td>
<td>2 (4)</td>
<td></td>
</tr>
<tr>
<td>June</td>
<td>160 (4)</td>
<td>4.6 (4)</td>
<td>124 (4)</td>
<td>7 (4)</td>
<td>1.5 (4)</td>
<td></td>
</tr>
<tr>
<td>July</td>
<td>208 (5)</td>
<td>6.7 (5)</td>
<td>113 (5)</td>
<td>13.8 (5)</td>
<td>1.9 (5)</td>
<td></td>
</tr>
<tr>
<td>August</td>
<td>202 (4)</td>
<td>4 (4)</td>
<td>144 (4)</td>
<td>4.5 (4)</td>
<td>1.3 (4)</td>
<td></td>
</tr>
<tr>
<td>September</td>
<td>270 (5)</td>
<td>4.9 (5)</td>
<td>164 (5)</td>
<td>5.6 (5)</td>
<td>1.4 (5)</td>
<td></td>
</tr>
<tr>
<td>October</td>
<td>153 (3)</td>
<td>5.7 (3)</td>
<td>131 (3)</td>
<td>2.7 (3)</td>
<td>0.7 (3)</td>
<td></td>
</tr>
<tr>
<td>November</td>
<td>99 (4)</td>
<td>18 (4)</td>
<td>195 (4)</td>
<td>9 (4)</td>
<td>0.5 (4)</td>
<td></td>
</tr>
<tr>
<td>December</td>
<td>216 (3)</td>
<td>54.5 (4)</td>
<td>124 (4)</td>
<td>2.5 (4)</td>
<td>0.9 (4)</td>
<td></td>
</tr>
<tr>
<td>January 93</td>
<td>173 (4)</td>
<td>5 (4)</td>
<td>148 (4)</td>
<td>4 (4)</td>
<td>6.8 (4)</td>
<td></td>
</tr>
<tr>
<td>February</td>
<td>115 (4)</td>
<td>5.5 (4)</td>
<td>129 (4)</td>
<td>7 (4)</td>
<td>1 (4)</td>
<td></td>
</tr>
<tr>
<td>March</td>
<td>73 (5)</td>
<td>3.8 (5)</td>
<td>170 (5)</td>
<td>5 (5)</td>
<td>3.7 (5)</td>
<td></td>
</tr>
<tr>
<td>April</td>
<td>178 (4)</td>
<td>6.3 (4)</td>
<td>158 (4)</td>
<td>6.8 (4)</td>
<td>3.8 (4)</td>
<td></td>
</tr>
<tr>
<td>May</td>
<td>174 (5)</td>
<td>7.4 (5)</td>
<td>212 (5)</td>
<td>4.8 (5)</td>
<td>1.1 (5)</td>
<td></td>
</tr>
</tbody>
</table>

Number in parentheses = number of samples.

Figure 6-5. Benton performance data for pond + wetland + NFB (Reed, 2000).
When retrofitting an existing pond-wetland system for nitrification and $N$ removal, the NFB appears to have economic advantages and simplicity of construction and operation. It also is likely that the NFB would be a viable alternative for $N$ removal in the initial design of a pond-wetland system.

### 6.4 COMMERCIAL PRODUCTS

There are numerous products and processes that are offered as a means to improve pond performance and remove $N$. Several options are presented below (Burnett et al., 2004).

#### 6.4.1 Description of Options

**6.4.1.1 Add Solids Recycle**

Adding solids recycle can be a reliable method of producing an effluent that can meet stringent $NH_3$ limits. With the addition of solids recycle, a pond is converted to a low-mixed liquor suspended solids (low-MLSS) activated sludge system. This can be accomplished using an external clarifier and adding a pump to return solids to the headworks. The BIOLAC® process uses an internal clarifier. Effluent from the clarifier is discharged to disinfection or routed through the subsequent cells of the pond system.

Successful operation of low-MLSS activated sludge system requires that the recycled solids be kept in suspension. The aerobic pond must be kept completely mixed. In most cases, a portion of the existing pond is partitioned into a complete mix cell because the energy required to mix the cell is far greater than that required to reduce the BOD$_5$ or nitrify the $NH_3$. The remaining portion of the system is used for polishing the effluent or storing the water before discharge.

Because the recycle system is an activated sludge variation, it can be designed and operated with traditional activated sludge design methods. Floating baffle curtains with exit ports are
frequently used for cell partitioning. Excess sludge wasting can be accomplished in a separate holding pond, or downstream cells of the existing pond can be used to store and treat sludge for disposal.

A complete mix section can be located anywhere in the flow train of an aerobic pond system. If the complete mix cell is placed near the end of the flow train nitrification occurs after carbonaceous BOD₅ has been removed. With the complete mix zone first in the process, sludge can easily be returned to a manhole or other suitable location upstream of the plant influent. By recycling sludge to the headworks, anoxic conditions and a high food-to-microorganism ratio will help control sludge bulking, provide some denitrification, and recover alkalinity.

6.4.1.2 Converting to a Sequencing Batch Reactor (SBR) Operation
Converting an aerobic pond to an activated sludge system can be accomplished by operating the aerobic pond as an SBR. A portion of the aerobic pond is partitioned into two or more complete mix SBR zones.

SBRs operate in a sequence of fill, react, settle, and decant. In a single-train SBR, flow into the basin will continue through all four cycles. Where parallel systems exist, the SBR can be operated as a typical SBR system; however, the construction costs will be higher. Rich (1999) has referred to this operation as a CFID process, but it is the same as the commercial SBR system marketed by Austgen-Biojet.

In SBR mode, aeration is used intermittently, and a decanting process transports the settled wastewater to downstream facultative cells or to a disinfection chamber before discharge. Decanting is accomplished with pumps, surface weirs, or floating decanting devices. A portion of the low-MLSS must be wasted during the reaction (mixing and aeration) phase to keep the process in balance. Rich (1999) suggests adding a recycle pump station with mixed liquor returned to the influent sewer to provide an anoxic environment for control of sludge bulking.

6.4.1.3 Install Biomass Carrier Elements
The addition of baffles and suspended fabrics for attached growth to accumulate and reduce pollutants has been suggested for many years (Reynolds et al., 1975; Polprasert and Agarwalla, 1995). The availability of commercial fabrics for the removal of NH₃ is a relatively new development. The carriers are plastic ribbons or wheels that are installed in the aerated zone to provide surface area for the growth of microorganisms. Provided that there is adequate surface area, nitrifying microorganisms can grow and multiply on the plastic surfaces and achieve NH₃ removal. The aerated cell does not have to be completely mixed, which is required in the recycle and SBR approaches, but the increased O₂ demand of the attached microorganisms must be met. Solids that drop from the biomass carriers settle or pass to next pond cells. Sludge buildup will increase, but will be reduced by anaerobic digestion, which will minimally affect the frequency at which sludge will need to be removed.

6.4.1.4 Commercial Pond Nitrification Systems
The following is a partial list of pond nitrification systems offered commercially:
1. ATLAS IST™ – Internal clarifier system by Environmental Dynamics, Inc.
2. CLEAR™ Process – SBR variant by Environmental Dynamics, Inc.
3. Ashbrook SBR – SBR system by Ashbrook Corporation

6.4.1.5 Applicability of Commercial Systems
Experience with all of the systems mentioned above is limited, and it is difficult at this time to predict the applicability and performance.

6.4.1.6 Advantages and Disadvantages
Until more design and operational data become available, it is difficult to delineate differences in the various commercial systems.

6.4.1.7 Design Criteria
Although design criteria were not available, the SBR systems would be designed the same way as a typical SBR. The quantities of plastic biomass required apparently are proprietary information. The design of the MBBR™ process is proprietary, as is the Poo-Gloo™ system.

6.4.1.8 Performance
The limited performance data are presented with the descriptions of the individual processes.

6.4.1.9 Limitations
Because of the limited experience and associated data, it is difficult to assess the effectiveness of the various processes.

6.4.1.10 Operation and Maintenance
With the addition of any of these treatments, it is expected that there will be more operation and maintenance time.

6.4.1.11 Costs
It is expected that the base costs associated with the type of pond into which the commercial equipment was placed in would continue. Additional costs would include the commercial product and any associated operation and maintenance.

6.4.1.12 ATLAS-IS™
Description
The Advanced Technology Pond Aeration System with Internal Separator (ATLAS-IS™) is offered by Environmental Dynamics, Inc. (EDI). It is designed to provide a high level of treatment with minimal operation and maintenance requirements. The process consists of a fine bubble floating lateral aeration system that contains a series of internal clarifiers or settlers. The settlers are constructed of a plastic material and may contain lamellate baffles. The units are installed within a complete mix zone of the aerated pond system. Mixed liquor enters the settling chamber through the bottom. A slight concentration of the low-MLSS takes place in the settler as the mixed liquor rises and spills over a weir into an effluent pipe. No return activated sludge
(RAS) or waste activated sludge (WAS) is required. Over time the low-MLSS will build up to a level adequate to grow nitrifying microorganisms. Some solids are carried downstream so no separate sludge wasting is necessary.

6.4.1.12.1 Performance
The ATLAS-IS™ system has been tested at Ashland, Missouri and has been successful in building up low-MLSS and in achieving nitrification. A schematic of the system is shown on Figure 6-8.

6.4.1.13 CLEAR™ Process
EDI also offers an SBR variant known as the Cyclical Pond Extended Aeration Reactor (CLEAR™). A completely mixed aerated pond cell is partitioned into three zones using floating baffle curtains. Influent is fed to each of the three zones in sequence. Aeration is applied to the zone receiving influent wastewater and, for part of this cycle, one of the other two zones. While the inflowing zone is aerated, the other two zones cycle between settling and decanting. WAS is removed using airlift pumps, either to downstream facultative ponds for storage or for further processing and disposal. A control system is provided to operate the motorized wastewater influent valves and decanters. There are currently no full-scale installations of the CLEAR™ process. A depiction of the process is shown in Figure 6-9.

![Floating IS Module](image)

Figure 6-7. EDI ATLAS – IS Internal Pond Settler.
6.4.1.14 Ashbrook Sequencing Batch Reactor

Description
Ashbrook Corporation (Houston, Texas) Sequencing Batch Reactor (SBR) system consists of decanters, motorized valves, and a control system. A facility has been installed in a pond system in Quincy, Washington. The aerated pond was portioned into sections and air was provided for complete mixing in two or more SBR cells. Operation is similar to a conventional SBR process, and the system in Quincy has been working well. Performance data are presented in Table 6.9. Figure 6-10 is a photograph of the system.

Table 6.9. Data for the Quincy, Washington SBR System (Ashbrook Corporation).

2002-2003 Average Influent/Effluent Data

<table>
<thead>
<tr>
<th></th>
<th>BOD, mg/L</th>
<th>TSS, mg/L</th>
<th>NH₃, mg/L</th>
</tr>
</thead>
<tbody>
<tr>
<td>Influent</td>
<td>145</td>
<td>159</td>
<td>19</td>
</tr>
<tr>
<td>Effluent</td>
<td>14</td>
<td>6</td>
<td>1.7</td>
</tr>
</tbody>
</table>
6.4.1.15 AquaMat® Process Description
AquaMat® is a biomass carrier system marketed by Nelson Environmental, Inc., Winnipeg, Manitoba. Plastic ribbons slightly more dense than water are connected to a plastic float, and ribbons extend into the waste stream three feet or more and provide additional surface area for bacteria to grow. When used with pond systems, the application is referred to as the Advanced Microbial Treatment System (AMTS).

6.4.1.15.1 Performance
Year-round nitrification has been achieved in an aerated pond in Laurelville, Ohio, and in Canada. Performance data are shown in Table 6-10 and two views of the AquaMat® are shown in Figure 6-11.
Figure 6-10. The AquaMat® Process

Table 6-10. Nelson AquaMat® Biomass Carrier, Larchmont, Georgia (Burnett et al., 2004).

<p>| | |</p>
<table>
<thead>
<tr>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Reported Average Effluent Quality</td>
<td></td>
</tr>
<tr>
<td>BOD₅ (mg/L)</td>
<td>6</td>
</tr>
<tr>
<td>TSS (mg/L)</td>
<td>10</td>
</tr>
<tr>
<td>NH₃ (mg/L)</td>
<td>0.1</td>
</tr>
</tbody>
</table>

6.4.1.16 Moving Bed Biofilm Reactor™ Process

6.4.1.16.1 Description
The Moving Bed Biofilm Reactor™ (MBBR™) is marketed by AnoxKaldnes North America, Inc., Providence, Rhode Island. The process is similar to the AquaMat® except that thousands of small polyethylene wheels are suspended in the pond (Figure 6-12). With a sufficient number of the “wheels”, adequate surface area is provided for growth of nitrifiers. An aerated pond in Johnstown, Colorado has been successfully upgraded using the MBBR™.
Figure 6-11. A MBBR™ “wheel.”

6.4.1.17 Poo-Gloo™ (Wastewater Compliance Systems, Inc.)
This patented device, developed at the University of Utah, consists of several concentrically nested domes that provide substrate for bacteria. They are placed on the bottom of a pond, creating a dark environment with robust air and wastewater mixing which removes contaminants from the water. Figure 6-13 shows a diagram of the device.

Figure 6-12. Schematic of a Poo-Gloo device cross-section.

6.4.2 Other Processes
Partial denitrification has been achieved by most of the systems described above, although the nitrogen removal pathways are not well understood. Several other commercial SBR systems and biomass carrier systems are available. Their use in ponds appears to be limited. The principle is the same and it appears reasonable to expect these proprietary systems would work. The manufacturers of the products have unique experience working with pond systems. In addition to the products, the companies have experience with floating baffle curtains for partitioning, installation of equipment without removing existing ponds from service, cost-effective and
efficient aeration systems for large surface area installations, and optimizing complete mix and partial mix aeration regimes.

6.5 REMOVAL OF PHOSPHORUS
In general, removal of phosphorus \((P)\) has not been required for wastewaters that receive pond treatment, but there are a number of exceptions for systems in the north central United States and Canada. It is expected that \(P\) removal will become a nation-wide requirement. The following sections present what has been done to control the discharge of \(P\) from wastewater treatment ponds.

6.5.1 Batch Chemical Treatment
In order to meet a \(P\) requirement of 1 mg/L for discharge to the Great Lakes, an approach using in-pond chemical treatment in controlled-discharge ponds was developed in Canada. Alum, ferric chloride and lime were tested using a motorboat for distribution and mixing of the chemical. A typical alum dosage might be 150 mg/L to produce an effluent from the controlled-discharge pond that contains less than 1 mg/L of \(P\) and less than 20 mg/L BOD\(_5\) and TSS. The sludge buildup from the additional chemicals is insignificant and would allow years of operation before requiring cleaning. The costs for this method were very reasonable and much less than conventional \(P\) removal methods. It has been applied successfully in several Mid-western states (U.S. EPA, 1992). The procedure does require a long-term management plan that includes calibrating applications to minimize use and monitoring of sludge quality.

6.5.2 Continuous-Overflow Chemical Treatment
Studies of in-pond precipitation of \(P\), BOD\(_5\), and TSS were conducted in Ontario, Canada. The primary objective of the chemical dosing process was to test for the removal of \(P\) with ferric chloride \((FeCl_3)\), alum and lime. Ferric chloride doses of 20 mg/L and alum doses of 225 mg/L, when added continuously to the pond influent, effectively maintained pond effluent \(P\) levels below 1 mg/L over a 2-year period. Hydrated lime at dosages up to 400 mg/L was not effective in consistently reducing \(P\) below 1 mg/L (1-3 mg/L was achieved), and yielded no BOD\(_5\) reduction, while slightly increasing the TSS concentration. Ferric chloride reduced effluent BOD\(_5\) from 17 to 11 mg/L and TSS from 28 to 21 mg/L; alum produced no BOD\(_5\) reduction and a slight TSS reduction (from 43 to 28-34 mg/L). Direct chemical addition appears to be effective only for \(P\) removal.

A six-cell pond system located in Waldorf, Maryland, was modified to operate as two three-cell units in parallel. Alum was added one system, while the other was the control. Each system contained an aerated first cell. Alum addition to the third cell of the system proved to be more efficient in removing total \(P\), BOD\(_5\), and TSS than alum addition to the first cell. Total \(P\) reduction averaged 81 percent when alum was added to the inlet to the third cell and 60 percent when alum was added to the inlet of the first cell. Total \(P\) removal in the control ponds averaged 37 percent. When alum was added to the third cell, the effluent total \(P\) concentration averaged 2.5 mg/L, with the control units averaging 8.3 mg/L. Improvements in BOD\(_5\) and TSS removal by alum addition were more difficult to detect, and at times increases in effluent concentrations were observed.
Thirty-seven pond systems in Michigan and Minnesota using chemical treatment to remove $P$ were studied (U.S. EPA, 1992). In Minnesota liquid alum was added to the secondary cells of eleven facultative ponds using a motorboat. These ponds were designed with a hydraulic residence time of 180 days and discharge in the spring and fall. The system used is essentially the same as that developed in Ontario, Canada to achieve a total $P$ effluent concentration of 1.0 mg/L. Influent concentrations ranged from 1.5 - 6.0 mg/L and averaged approximately 3.3 mg/L. In general, the facilities satisfied the requirement for 1.0 mg/L with several minor excursions outside the limit by 10 percent. In 2010, 38 percent of the treatment ponds in Minnesota were treating the wastewater for $P$ removal (Steve Duerre, pers. comm., 2010). The majority use alum, although other chemicals, such as potassium permanganate, are being evaluated as possible substitutes. All of the ponds are able to meet the $P$ limit. The chemical is applied twice a year, prior to the seasonal spring and fall discharges.

The State of Michigan evaluated 26 ponds that had been in operation ranging from 1 to 20 years. Both facultative and aerated ponds were evaluated. Discharge was seasonal (once or twice a year) or continuous, (varying from 24 hours/day, 7 days/week to 8 hours/day, 5 days/week). The chemicals were added to a clarifier following the pond system. None of the systems applied the chemicals by boat. The influent total $P$ concentration in the Michigan systems ranged from 0.5 to 15 mg/L, with an average of 4.1 mg/L. By 2010, more than 300 ponds in Michigan have or will have a $P$ limit of 1 mg/L (Dan Holmquist, pers. comm.). Phosphorus removal has been successful as long as the chemical flocculant is added at the appropriate rate at the end of the pond system, i.e., the clarifier or the maturation pond.

A description of the facilities and the influent and effluent $P$ concentrations is shown in Table 6-11.


<table>
<thead>
<tr>
<th>Location</th>
<th>Discharge schedule</th>
<th>Chemical treatment</th>
<th>Facility description</th>
<th>Influent</th>
<th>Effluent</th>
</tr>
</thead>
<tbody>
<tr>
<td>Beldinga</td>
<td>Continuous</td>
<td>Alum</td>
<td>5-cell pond</td>
<td>4.0</td>
<td>0.6 - 0.7</td>
</tr>
<tr>
<td>Bessemera</td>
<td>Continuous</td>
<td>Alum polymer</td>
<td>3-cell pond with clarifier</td>
<td>1.8 – 2.0</td>
<td>0.6 – 0.9</td>
</tr>
<tr>
<td>Coopersvillea</td>
<td>Continuous</td>
<td>FeCl$_3$</td>
<td>4-cell pond</td>
<td>5.0</td>
<td>0.3</td>
</tr>
<tr>
<td>Kalamazoo Lakea</td>
<td>Continuous</td>
<td>FeCl$_3$ polymer</td>
<td>3-cell pond</td>
<td>6.0 – 7.0</td>
<td>0.5 – 0.6</td>
</tr>
<tr>
<td>Elk Rapidsa</td>
<td>Continuous</td>
<td>FeCl$_3$</td>
<td>3-cell pond with clarifier</td>
<td>3.2 – 4.3</td>
<td>0.6 – 0.7</td>
</tr>
<tr>
<td>Carson Citya</td>
<td>Seasonal</td>
<td>Alum</td>
<td>5-cell pond</td>
<td>6.0 – 7.0</td>
<td>4.0</td>
</tr>
<tr>
<td>Fowlervillea</td>
<td>Seasonal</td>
<td>FeCl$_3$</td>
<td>6-cell pond</td>
<td>2.5 – 3.5</td>
<td>0.8</td>
</tr>
<tr>
<td>Remusb</td>
<td>Seasonal</td>
<td>FeCl$_3$</td>
<td>4-cell pond</td>
<td>4.7</td>
<td>0.4 – 1.0</td>
</tr>
<tr>
<td>Serpent Lakeb</td>
<td>Seasonal</td>
<td>Alum</td>
<td>3-cell pond</td>
<td>1.8 – 2.8</td>
<td>0.6 – 0.7</td>
</tr>
<tr>
<td>Grand Portageb</td>
<td>Seasonal</td>
<td>Alum</td>
<td>2-cell pond</td>
<td>2.9 – 3.3</td>
<td>0.3 – 1.2</td>
</tr>
</tbody>
</table>

$^a$Michigan, $^b$Minnesota