Heterotrophic Kinetic Parameter Estimation for Enhanced Biological Phosphorus Removal Processes Operated in Conventional and Membrane-Assisted Modes

Zhe Zhang and Eric R. Hall*

Department of Civil Engineering, University of British Columbia, 6250 Applied Science Lane, Vancouver, British Columbia V6T 1Z4

Parameter estimation and wastewater characterization are crucial for modelling of the membrane enhanced biological phosphorus removal (MEBPR) process. Prior to determining the values of a subset of kinetic and stoichiometric parameters used in ASM No. 2 (ASM2), the carbon, nitrogen and phosphorus fractions of influent wastewater at the University of British Columbia (UBC) pilot plant were characterized. It was found that the UBC wastewater contained fractions of volatile acids (SA), readily fermentable biodegradable COD (SF) and slowly biodegradable COD (XS) that fell within the ASM2 default value ranges. The contents of soluble inert COD (SI) and particulate inert COD (XI) were somewhat higher than ASM2 default values. Mixed liquor samples from pilot-scale MEBPR and conventional enhanced biological phosphorus removal (CEBPR) processes operated under parallel conditions, were then analyzed experimentally to assess the impact of operation in a membrane-assisted mode on the growth yield (YH), decay coefficient (bH) and maximum specific growth rate of heterotrophic biomass (µH). The resulting values for YH, bH and µH were slightly lower for the MEBPR train than for the CEBPR train, but the differences were not statistically significant. It is suggested that MEBPR simulation using ASM2 could be accomplished satisfactorily using parameter values determined for a conventional biological phosphorus removal process, if MEBPR parameter values are not available.

Key words: ASM2, enhanced biological phosphorus removal, estimation, kinetics, membrane assisted, modelling, parameters, simulation, stoichiometric, UCT process, wastewater characterization

Introduction

The membrane enhanced biological phosphorus removal (MEBPR) process is a combination of membrane filtration and enhanced biological phosphorus removal (EBPR) technologies. In such a system, the membrane unit replaces a secondary clarifier for separating biomass from the treated effluent. This approach avoids problems associated with secondary clarification such as denitrification-induced sludge flotation or sludge settleability problems associated with filamentous organism sludge bulking or dispersed growth.

Optimized process configurations and operating conditions for the MEBPR system are required to achieve the desired carbon and nutrient removal efficiencies while also delivering the promised technical and economic benefits of a membrane-assisted process. Computer-based simulation offers the best systematic approach for optimizing the process design of complex biological treatment systems such as MEBPR. The International Water Association has developed the Activated Sludge Model (ASM) series of models for dynamic simulation of suspended growth wastewater treatment processes. In particular, Activated Sludge Model No. 2 (ASM2) has been proposed (Henze et al. 1995, 2000) for simulation of the behaviour of activated sludge systems incorporating biological processes for carbon, nitrogen and phosphorus removal. ASM2 can also be used as a conceptual platform for further model development.

Prior to using ASM2 for process simulation, it is necessary to specify or estimate the values of the many model parameters that are associated with the biochemical model. Further, the ASM models require a rigorous specification of the wastewater characteristics for realistic simulation. At this time it is not clear what fundamental differences exist between the microbial biomasses in the MEBPR and the conventional EBPR processes. It could be anticipated that differences in average shear in the two processes may lead to significant differences in microbial community composition or floc structure that may affect the measured biokinetic parameter values. It is unclear whether commonly used model parameter values derived for simulation of a conventional EBPR process can also be used for simulation of a membrane-assisted process. Further study of the microbial activities in each alternative will provide a better understanding of the differences.

The purpose of the present research was to experimentally estimate a subset of the kinetic and stoichiometric parameters used in ASM2, for the biomass from two parallel pilot-scale biological phosphorus removal
processes. One unit was operated as a conventional UCT (University of Cape Town) enhanced biological phosphorus removal (CEBPR) system with gravity clarification for solids-liquid separation. A parallel train was operated with a submerged membrane filtration system in the aerobic zone (MEBPR). The ASM2 model parameters estimated for both trains included growth yield, decay rate and maximum specific growth rate of the heterotrophic biomass. However, first the influent wastewater was characterized using the usual ASM-type approach. This was done to determine whether the wastewater treated at the University of British Columbia (UBC) pilot plant site could be considered typical from the ASM modelling perspective, or conversely, whether there was reason to believe that the wastewater characteristics could have significantly influenced the estimated model parameter values.

Materials and Methods

Configuration and Operating Conditions of the UBC Pilot Plant

The UBC wastewater treatment pilot plant receives domestic wastewater from the UBC campus. Raw wastewater is pumped first into large storage tanks and then into a circular primary clarifier. For the present study, primary effluent was pumped into each train (train A and B) at a constant flow rate of 2 L/min. For the experimental work described here, the two trains were operated in parallel for comparison of the different process designs (Fig. 1). Train A of the plant was a membrane enhanced biological phosphorus removal (MEBPR) process. The MEBPR train consisted of a UCT-type bioreactor design with three completely mixed zones in series, separated by moveable baffles. A total of six Zee-Weed ZW-10 membrane modules were installed in the aerobic zone to replace the secondary clarifier for suspended solids removal from the final effluent. The membrane filtration equipment occupied just under 5% of the total bioreactor volume. Mixed liquor was pumped from the aerobic zone to the anoxic zone at a recycle ratio of 1:1 relative to the influent flow rate. The anoxic zone to anaerobic zone recycle ratio was also held constant at approximately 1:1.

Train B was a simplified UCT conventional enhanced biological phosphorus removal (CEBPR) process with a traditional secondary clarifier, from which the return activated sludge (RAS) was recycled back to the anoxic zone. Aerobic and anoxic zone mixed liquors were recycled as noted above at ratios of about 1:1 relative to the influent flow rate.

The operating conditions of both units were set to similar values (Table 1) in order to compare the estimated model parameter values associated with the two processes. Process volumes were fixed during the testing; process flow rates and aerobic zone DO (dissolved oxygen) concentrations were measured and adjusted at least twice per week. Aerobic sludge was wasted daily and waste volumes were varied depending on mixed liquor (MLSS) and effluent suspended solids concentrations. Waste sludge flow rates were adjusted weekly according to the most recent weekly aggregated solids data to achieve the target SRTs. Mixed liquor temperatures were not controlled and as a result, the process temperatures varied between 13 and 20.5°C during the course of the study, which was carried out between September and May.

Laboratory Experimental Apparatus

Two sets of equipment (Fig. 2 and 3) were utilized for the experimental estimation of kinetic and stoichiometric parameters, as well as the estimation of the concentrations of various wastewater components. The oxygen uptake rate (OUR) test system (Fig. 2) was used to carry out batch tests for the determination of heterotrophic decay rate ($b_H$), heterotrophic maximum specific growth rate ($\mu_H$) and readily biodegradable COD concentration ($S_f$) by respirometry. These respirometers were constructed from acrylic pipe sealed to an acrylic base. The respirometer was topped with a vented conical lid that minimized the gas-liquid surface area at the top of the respirometer. A YSI Model 54 dissolved oxygen probe was inserted into each respirometer at the top and a fine bubble air diffuser was placed at the bottom. A laboratory stir plate and magnetic stir bars were used to maintain the respirometer contents in suspension. Respirometers were placed in a temperature-controlled chamber to maintain the contents at the prevailing temperature of the pilot plant reactors.

Batch reactors were also used to estimate the growth yield of heterotrophic biomass ($Y_H$), soluble inert COD ($S_I$) and particulate inert COD ($X_I$). As illustrated in

Fig. 1. Schematic layout of the University of British Columbia pilot plant.
Zhang and Hall

Fig. 3, simple 1-L batch reactors were constructed from acrylic tubing sealed to an acrylic base. An air diffuser was placed in the bottom of each reactor and the air delivery rate was adjusted manually to maintain the dissolved oxygen at 4 to 6 mg O₂/L during the tests. A magnetic stir plate and stir bars were used to keep the contents of the reactors in suspension. The water loss due to evaporation was measured and was replaced with distilled water prior to each sampling.

Sample Collection and Preservation

The test media were influent wastewater (primary effluent), final effluents and mixed liquors from the MEBPR and the CEBPR processes at the UBC pilot plant. Influent wastewater was collected weekly between September and April for the determination of ammonium-N, nitrate plus nitrite-N, ortho-phosphate, TKN (total Kjeldahl nitrogen), TP (total phosphorus) and VFA (volatile fatty acid) concentrations, using the sample preservation and analytical methods described in Standard Methods (APHA et al. 1998). The concentrations of soluble constituents were measured in filtrates collected after filtration through 0.45-µm filters. Samples were preserved immediately after collection and/or filtration and were then analyzed in triplicate. The experimental tests for heterotrophic growth yield (\( Y_H \)), decay rate (\( b_H \)), maximum growth rate (\( \mu_H \)) and readily biodegradable COD (\( S_F \)), were carried out immediately after sampling and these were run in parallel with biomass from the MEBPR and CEBPR processes.

### Experimental Methods

**Determination of heterotrophic growth yield, \( Y_H \).** The heterotrophic growth yield tests were based on a method described by several researchers (Henze et al. 1987; Slade et al. 1991; Grady et al. 1999). A 0.5-L sample of influent wastewater was pre-filtered through a Gf GF/C glass fibre filter, then through a 0.45-µm membrane fil-

---

### Table 1. Operating conditions and configuration of UBC pilot plant

<table>
<thead>
<tr>
<th></th>
<th>CEBPR process</th>
<th>BPR process</th>
</tr>
</thead>
<tbody>
<tr>
<td>Process flow rate (L/min)</td>
<td>2.0</td>
<td>2.0</td>
</tr>
<tr>
<td>Influent, Q</td>
<td>2.0</td>
<td>2.0</td>
</tr>
<tr>
<td>Return sludge, Qs</td>
<td>2.0</td>
<td>2.0</td>
</tr>
<tr>
<td>Anaerobic recycle, Qa</td>
<td>2.0</td>
<td>2.0</td>
</tr>
<tr>
<td>Anoxic recycle, Qr</td>
<td>2.0</td>
<td>2.0</td>
</tr>
<tr>
<td>Total volume of bioreactor (L)</td>
<td>1386</td>
<td>1386</td>
</tr>
<tr>
<td>Anaerobic zone</td>
<td>252</td>
<td>252</td>
</tr>
<tr>
<td>Anoxic zone</td>
<td>378</td>
<td>378</td>
</tr>
<tr>
<td>Aerobic zone</td>
<td>736</td>
<td>736</td>
</tr>
<tr>
<td>HRT (h) (based on Q)</td>
<td>11.6</td>
<td>11.6</td>
</tr>
<tr>
<td>System HRT (h)</td>
<td>11.6</td>
<td>11.6</td>
</tr>
<tr>
<td>Anaerobic zone</td>
<td>2.1</td>
<td>2.1</td>
</tr>
<tr>
<td>Anoxic zone</td>
<td>3.2</td>
<td>3.2</td>
</tr>
<tr>
<td>Aerobic zone</td>
<td>6.3</td>
<td>6.3</td>
</tr>
<tr>
<td>SRT (d)</td>
<td>17–25</td>
<td>17–25</td>
</tr>
<tr>
<td>Process sludge wastage (L/d)</td>
<td>50 (approx.)</td>
<td>40 (approx.)</td>
</tr>
<tr>
<td>Temperature (ºC)</td>
<td>13–20.5</td>
<td>13–20.5</td>
</tr>
<tr>
<td>Aerobic DO concentration (mg O₂/L)</td>
<td>2–3</td>
<td>2–3</td>
</tr>
</tbody>
</table>

*Clarifier solids were not included in the calculation of SRT.*

---

![Fig. 2. Sketch of respirometers used for oxygen uptake rate (OUR) testing.](image1)

![Fig. 3. Sketch of batch test reactors.](image2)
ter. The filtered wastewater was placed into a 1-L batch reactor and the initial soluble COD (chemical oxygen demand) of the reactor contents was determined in triplicate samples. A predetermined amount of process biomass was then added to the reactor to achieve a target soluble COD to biomass (VSS) ratio of 75:1. The initial total COD concentration of the reactor was then measured to determine the actual initial VSS to COD ratio.

Heterotrophic growth yield tests were run in duplicate. Reactors with the mixture of wastewater and biomass from the CEBPR or MEBPR processes were operated in parallel in a temperature-controlled chamber set at the prevailing temperature of the pilot plant bioreactors. The batch reactor contents were mixed and aerated continuously and care was taken to regularly remove the wall growth from the side of the reactor and aeration tubing to keep microbial growth in suspension. After making up water loss due to evaporation, duplicate 16-mL samples were removed from the reactor twice daily for several days and analyzed for soluble and total COD, until the COD concentrations stabilized (usually 5–8 d).

The heterotrophic biomass yield, \( Y_{H} \), was estimated by observing the COD of insoluble or particulate material formed during the removal of soluble substrate. The biomass COD was calculated as the difference between the total COD and the soluble COD, and the yield was determined from equations 1 and 2.

\[
\Delta \text{Biomass COD} = \text{total COD} - \text{soluble COD} \tag{1}
\]

\[
Y_{H} = \frac{\Delta \text{Biomass COD}}{\Delta \text{Soluble COD}} \tag{2}
\]

The \( Y_{H} \) was determined by plotting the biomass COD as a function of the soluble COD removed and taking the slope of the resulting line (Grady et al. 1999).

Decay of heterotrophic microorganisms, \( b_{H} \). The batch respirometric method of Ekama et al. (1986) outlined in the ASM1 documentation (Henze et al. 1987) was used to determine the decay coefficient. Process biomass removed from the pilot plant aerobic zones was placed into two 2-L respirometers in a temperature-controlled chamber. Samples from the two processes were tested in duplicate. The reactor contents were continuously stirred and aerated, and periodically the aeration was stopped and the decreasing dissolved oxygen concentration was monitored. The OUR was measured in this manner a number of times over a period of 11 to 15 d. The respirometer pH was maintained at 7.5 by adding alkalinity (sodium bicarbonate) and nitrification was inhibited by the addition of 0.16 g nitration inhibitor formula 2533 (supplied by HACH Company) per 300 mL of sample. The respirometer contents were topped up daily with distilled water to compensate for evaporative losses.

The slope of a plot of the natural logarithm of the oxygen uptake rate versus time is the traditional decay coefficient, \( b_{H}^{'} \). The model decay coefficient, \( b_{H} \), was calculated from the traditional decay coefficient using equation 3:

\[
b_{H} = \frac{b_{H}^{'}}{[1 - Y_{H} (1 - f_{XI})]} \tag{3}
\]

where \( f_{XI} \) is the fraction of inert COD generated in biomass decay, assumed to be equal to 0.08 for ASM2 (Henze et al. 2000). The temperature was compensated for by correction to a reference temperature of 20°C by equation 4:

\[
b_{H} = b_{H}^{20 \circ} \cdot \theta (T - 20) \tag{4}
\]

where \( \theta \) is the temperature coefficient of 1.029 for biomass decay (Grady et al. 1999).

Maximum aerobic growth rate of heterotrophic microorganisms, \( \mu_{H} \). The maximum growth rate of heterotrophic biomass (\( \mu_{H} \)) was determined using the approach developed by Kappeler and Gujer (1992). A 2-L sample of influent wastewater was centrifuged at 1560 \( \times g \) and the COD of the centrate was measured. Mixed liquor samples from the CEBPR and MEBPR processes were pre-aerated for measuring the endogenous OUR. Wastewater and biomass were mixed into a 2-L respirometer in a COD to VSS ratio of 4:1 as discussed by Ekama et al. (1986). Nitrification was inhibited as above. The reactors were placed into a temperature-controlled chamber and the reactor contents were continuously stirred and aerated. The OUR was measured immediately after mixing until a sharp decrease in rate was observed. The maximum specific growth rate was calculated from the slope (\( \mu_{H} - b_{H} \)) of the relative OUR versus time plot according to equation 5, using the \( b_{H} \) value determined previously:

\[
\ln\left[\frac{r_{O2}(t)}{r_{O2}(0)}\right] = (\mu_{H} - b_{H}) \cdot t \tag{5}
\]

where \( r_{O2} \) is the oxygen utilization rate (mg O/L h).

The measured rates were corrected to a reference temperature of 20°C by equation 4, using a \( \theta \) of 1.094 for aerobic growth of heterotrophic biomass (Grady et al. 1999).

Determination of readily fermentable substrate, \( S_{F} \). Two methods were used to determine \( S_{F} \) in untreated wastewater. One was the aerobic batch method described by Ekama et al. (1986) and Kappeler and Gujer (1992); the other one was a rapid physical-chemical assay developed by Mamais et al. (1993).

Aerobic batch method. A predetermined amount of wastewater was mixed with a known amount of biomass to a target initial COD to VSS (mg/mg) ratio of 0.1 to 0.2. The mixture was placed into a respirometer in a temperature-controlled chamber and the contents were aerated. OUR monitoring was started immediately and
Aliquots of 100 mL of Ammonium (NH$_4^+$), nitrite plus nitrate (NO$_3^-$ and TKN) and phosphorus (PO$_4^{3-}$ and TP). Ammonium (NH$_4^+$), nitrite plus nitrate (NO$_3^-$) and phosphorus (PO$_4^{3-}$) in the filtrate of samples were analyzed using a Lachat Quickchem 8000 Automated Ion Analyzer with an automated sampler, according to Standard Methods (APHA et al. 1998). Samples for TKN and TP (total phosphorus) were digested and preserved before analysis as NH$_4^+$ and PO$_4^{3-}$, respectively.

**Results and Discussion**

The wastewater treated at the UBC pilot plant originates from a 4500 population equivalent residential area on the campus of UBC and from UBC institutional buildings. Prior to the estimation of values for the selected ASM2 kinetic and stoichiometric parameters, the wastewater was characterized to ensure that its general char-

Physical-chemical method. Aliquots of 100 mL of process influent or effluent were flocculated by the addition of 1 mL of a 100 g/L zinc sulfate solution, followed by vigorous mixing for approximately 1 min. The pH of each sample was then adjusted to 10.5 with a 6 M sodium hydroxide solution. Samples were settled for a few minutes, after which the supernatant was withdrawn with a syringe and filtered through a 0.45-µm membrane filter. The soluble COD in the filtrate was then measured. The value of soluble biodegradable COD was estimated based on the difference between the test results for the influent and effluent samples. The concentration of readily biodegradable COD, $S_F$, was calculated according to equation 7:

$$S_F = S - S_{ML} - S_A$$

where $S_A$ is the concentration of fermentation products expressed as acetate (mg COD/L), $S_{ML}$ is the inert soluble organic matter (mg COD/L) in the filtrate, and $S$ is the measured soluble COD concentration in the influent.

**Determination of inert soluble organic matter, $S_I$.** The soluble inert COD was determined as described by Lesouef et al. (1992). An aliquot of 500 mL unfiltered wastewater was placed in a 1-L reactor. The reactor contents were stirred and aerated continuously. An initial soluble COD was determined as described for the heterotrophic growth yield test. A predetermined amount of biomass from the aerobic zone of the MEBPR process was seeded to a target VSS to COD ratio of 1:75 in each reactor, and the initial total COD was measured to determine the exact value of the initial VSS to COD ratio. Tests were carried out in parallel and samples from the reactors were taken periodically for measurement of soluble COD, until a constant final soluble COD was observed. This generally required 8 to 11 d (Lesouef et al. 1992). The soluble inert COD of the influent was thus determined as the final soluble inert COD after long-term aeration.

**Determination of inert particulate matter, $X_I$.** Although accurate estimates of the wastewater inert particulate matter ($X_I$) are best made by comparing measured and predicted sludge production rates in a treatment process (Henze et al. 2000), laboratory-scale assay techniques have been proposed as alternatives for estimation of $X_I$. The procedure of Lesouef et al. (1992) was used in the present study. A sample of influent wastewater was filtered through a 0.45-µm membrane filter and 500 mL of filtrate was then stirred and aerated in a 1-L reactor. An initial soluble COD was determined as described for the heterotrophic growth yield test. Biomass from the aerobic zone of either pilot process was seeded to achieve a target initial VSS to COD ratio of 1:75 in each reactor, and the actual initial total COD and VSS to COD ratio was then determined by measurement. At the same time, an assay with a non-filtered wastewater sample was undertaken in parallel. Samples from the two reactors were analyzed for soluble and total COD periodically, until a constant final soluble COD was observed. The inert particulate COD of the influent was then determined by calculation, using the detailed procedure described by Lesouef et al. (1992).

**Determination of slowly biodegradable COD, $X_S$.** The slowly biodegradable COD, $X_S$, was determined by a COD mass balance following the approach of Henze et al. (1987). $X_S$ was estimated by calculation from the difference between the total COD and the other fractions in the influent, as noted in equation 8:

$$X_S = COD_T - S_A - S_F - S_I - X_I$$

where $COD_T$ is the total influent COD concentration (mg/L).

**Volatile fatty acids ($S_A$).** Volatile fatty acids (VFAs) were directly measured by gas chromatography using the procedures in GC Bulletin 751 provided by Supelco Inc. Samples were analyzed in triplicate.

was continued until a distinct slope, followed by a flatter slope in the OUR curve was observed. The area under the OUR curve corresponding to the soluble readily biodegradable COD concentration of the influent wastewater was calculated according to equation 6.

$$S_F = \left[ r_{O2, \text{tot}} - r_{O2, \text{baseline - respiration}} \right] (1 - Y_H) = \frac{\Delta O_2}{V_{ww} + V_{ML}} V_{ww} (1 - Y_H)$$

where $S_F$ is readily fermentable (biodegradable) substrate, $r_{O2}$ is respiration rate (mg O/L·h), $\Delta O_2$ is mass of oxygen consumed by $S_F$ (area under OUR curve), $V_{ww}$ is volume of wastewater in the mixture, and $V_{ML}$ is volume of mixed liquor in the mixture.
characteristics were comparable to those of the municipal wastewaters for which ASM2 was developed.

Characteristics of Influent Wastewater

**Readily biodegradable COD,** $S_f$, **and volatile acids,** $S_a$. Samples taken from the influents to both process trains over the course of a seven-month period between September and April were analyzed for $S_f$ and $S_a$. The readily biodegradable COD ($S_f$) was estimated using both the physical-chemical method of Mamais et al. (1993) and a respirometric method. Table 2 indicates that the results of the physical-chemical method compared well to those from the respirometric method. Both methods indicated that the readily biodegradable COD ($S_f$) comprised 10 to 11% of the total influent COD. As there was no significant statistical difference between the two sets of data, the results were pooled for further analysis and presentation in Table 3.

The volatile acids concentration ($S_a$) and the $S_a$ fraction in the total influent COD are shown in Table 3. The $S_f$ and $S_a$ results obtained in this study are similar to municipal wastewater values reported in the literature. The reported default range for volatile acids is 2 to 10% of the total COD (Henze et al. 2000). Henze and Harremoes (1992) reported that the fermentable readily biodegradable fraction, $S_f$, may comprise up to 20% of the total COD. Kappeler and Gujer (1992) reported the $S_f$ fraction to be 11% for municipal wastewater. Readily biodegradable COD from the test of Bjerre et al. (1995) was 25 mg/L. Kristensen et al. (1998) reported a range of $S_f$ between 9 and 34% of total COD. The typical range for readily biodegradable substrate recommended for use with ASM2 is 10 to 20% of the total COD for municipal wastewater, in good agreement with the results from the present study. In general, the concentrations and fractions of readily biodegradable COD and VFA in the UBC wastewater were typical compared to the range of values reported in association with ASM2.

**Inert soluble COD,** $S_i$. The range of measured $S_i$ concentrations of 10 samples was 45 to 71 mg/L, which accounted for 13 to 22% of the total COD (Table 3). These results compare well to the municipal wastewater characteristics reported by Kappeler and Gujer (1992) who reported the range of $S_i$ to be 10 to 20% of total COD. Henze and Harremoes (1992) suggested that $S_i$ should be in the range of 20 to 25% of total COD for raw municipal wastewater. Bjerre et al. (1995) reported inert soluble COD in their research as 55 mg/L. However, lower values of $S_i$ have also been reported in the literature. Ekama et al. (1986) estimated that $S_i$ was 5% of total COD for raw wastewater in South Africa. Henze and Harremoes (1992) reported $S_i$ to be 2% of total COD in Denmark, and 8 to 11% of total COD in raw wastewater in Switzerland. The typical range of $S_i$ recommended as a default by ASM2 is 5 to 10% of total primary effluent COD.

It should be mentioned that the reported literature values for this fraction have been measured with many different methods and that most of the methods included elements of estimation. Therefore, care must be taken in the comparison and interpretation of the results. In the present study, the inert soluble COD was determined from the inert particulate COD tests in order to obtain comparable results to those for $X_i$. Since these tests were usually run over 8- to 11-d periods, the resulting values of $S_i$ might be higher than those estimated using the method recommended by ASM2, for which the soluble COD remaining after 20 d of oxidation is regarded as equivalent to $S_i$ (Ekama et al. 1986).

**Inert particulate COD,** $X_i$. Six tests were conducted to estimate $X_i$. The average $X_i$ concentration was found to be 103 mg/L with 95% confidence limits of ±44 mg/L, which accounted for 23% (±7.7%) of the total influent COD (Table 3). Many researchers have characterized this fraction either in raw or primary wastewaters. Ekama et al. (1986) reported the $X_i$ fraction in raw wastewater to be 13% of the total COD, while Henze et al. (1987) reported a range of 11 to 20%. Similarly, Kappeler and Gujer (1992) estimated the $X_i$ fraction in raw wastewater to be 8 to 10% of the total COD. Some researchers have evaluated this fraction in primary effluent, and the resulting $X_i$ fractions were reported to lie between 4 and 13% of the total COD (Ekama et al. 1986; Henze and Harremoes 1992; Lesouef et al. 1992). The typical values for this fraction recommended by ASM2 lie in the range of 10 to 15% of the total COD in primary effluent. Bjerre et al. (1995) estimated the $X_i$ concentration to be 33 mg/L. Generally, the fraction of inert particulate COD in the present study was somewhat higher than the values reported in the literature for other municipal wastewater sources.

**Slowly biodegradable COD,** $X_s$. The concentration of slowly biodegradable COD was measured four times (Table 3) and the average value was found to be 226 (±34.3) mg/L, which accounted for 56% (±8.4%) of the total COD (at the 95% of confidence level). The slowly biodegradable substrate in this study was comparable to,
or marginally higher than, literature data. Ekama et al. (1986) reported a value of 62% of the total COD. Henze et al. (1987) reported the range of XS to be 40 to 49% of the wastewater COD. Kappeler and Gujer (1992) found the range of XS to lie between 53 and 60% of the total COD. Bjerre et al. (1995) estimated the slowly biodegradable COD in their research as 115 mg/L. The slowly biodegradable COD fraction in primary effluent is relatively low compared with the fraction in raw wastewater. Lesouef et al. (1992) and Henze and Harremoes (1992) reported this fraction to be between 41 to 43% of the total COD, and Ekama et al. (1986) reported a value of 60% in primary effluent. The ASM2 typical value for slowly biodegradable COD is 30 to 60% of the total COD in primary effluent (Henze et al. 2000).

**Wastewater nitrogen fractions.** The concentrations and fractions of nitrogen components are shown in Table 3. The ammonium nitrogen in this study was about 58% of the total nitrogen, only slightly lower than the ASM2 default range of 60 to 70% (Henze et al. 1995). The nitrite plus nitrate-N concentration has been reported to be in the range of 0 to 1 mg/L in the literature (Henze et al. 1995), and this is consistent with the results from the present study. The average COD/TKN ratio measured during the present study was 9.3, and this value should allow efficient biological nitrogen removal (Grady et al. 1999).

**Wastewater phosphorus fractions.** The ortho-phosphate and total phosphorus concentrations were found to be 3.4 (±0.2) and 4.5 (±0.3) mg/L at the 95% confidence level (Table 3). The ortho-phosphate fraction comprised on average, about 75% of the total phosphorus, and in some cases this fraction was as high as 91 to 96% of the total phosphorus. These results are higher than the typical value of 60% associated with ASM2. Therefore, it is clear that characterization of the influent wastewater is very important for a particular modelling application. Considering that the average influent total COD concentration was 418 mg/L, the average calculated COD/P ratio of 93 for the influent of the present study, seems to be highly suitable for efficient biological phosphorus removal (Grady et al. 1999).

### Biokinetic and stoichiometric parameter estimation

Table 3 and the foregoing discussion indicate that the UBC wastewater is similar in general character to the typical municipal wastewaters for which ASM2 was developed. The exception may be the presence of somewhat higher fractions of inert soluble and particulate COD in the UBC wastewater. In any case, there seems to be little evidence that the wastewater characteristics would materially influence the estimated biokinetic and stoichiometric parameter estimates.

**Heterotrophic growth yield, \( Y_{H} \).** Figure 4 presents a typical set of results obtained during the heterotrophic growth yield estimation tests. During the first 24 h, substrate removal and particulate COD accumulation were relatively rapid and linear. Thereafter, when substrate was depleted, a decrease in biomass concentration was often observed that was attributable to decay for the production of maintenance energy. After about 72 h, the soluble COD stabilized to reflect non-biodegradable residual COD. At longer times, that is, beyond 92 h, slight increases in particulate COD and soluble COD were observed. This was likely due to the release of soluble COD due to biomass decay, followed by biomass regrowth on the released soluble COD.

It was shown that only when the testing time was very short and the biomass was growing very rapidly would most substrate be used for growth, allowing the observed yield to approach the true growth yield. These conditions produced the most accurate estimate of \( Y_{H} \).

<table>
<thead>
<tr>
<th>Parameter</th>
<th>No. of samples</th>
<th>Concentration mean ±95% conf. limits (mg/L)</th>
<th>Organic fraction mean ±95% conf. limits (% of COD)</th>
<th>ASM2 typical organic fraction ranges^*</th>
</tr>
</thead>
<tbody>
<tr>
<td>COD(_{T})</td>
<td>32</td>
<td>418 ± 23.6</td>
<td>4.7 ± 0.9</td>
<td>2–10</td>
</tr>
<tr>
<td>S(_{A})</td>
<td>23</td>
<td>19.5 ± 3.7</td>
<td>10.2 ± 1.6</td>
<td>10–20</td>
</tr>
<tr>
<td>S(_{I})</td>
<td>21</td>
<td>44 ± 7.6</td>
<td>23 ± 7.7</td>
<td>10–15</td>
</tr>
<tr>
<td>X(_{S})</td>
<td>6</td>
<td>103 ± 44</td>
<td>16 ± 3</td>
<td>5–10</td>
</tr>
<tr>
<td>S(_{I})</td>
<td>10</td>
<td>60 ± 10</td>
<td>56 ± 8.4</td>
<td>30–60</td>
</tr>
<tr>
<td>X(_{I})</td>
<td>4</td>
<td>226 ± 34.3</td>
<td></td>
<td></td>
</tr>
<tr>
<td>NH(_{4})(^+)-N</td>
<td>17</td>
<td>25.5 ± 2.2</td>
<td></td>
<td></td>
</tr>
<tr>
<td>NO(<em>{2}) and NO(</em>{3})(^-)-N</td>
<td>17</td>
<td>0.1 ± 0.05</td>
<td></td>
<td></td>
</tr>
<tr>
<td>TKN</td>
<td>18</td>
<td>44.8 ± 2.4</td>
<td></td>
<td></td>
</tr>
<tr>
<td>TN</td>
<td>18</td>
<td>44.2 ± 2.7</td>
<td></td>
<td></td>
</tr>
<tr>
<td>PO(_{4})(^3)-P</td>
<td>18</td>
<td>3.4 ± 0.2</td>
<td></td>
<td></td>
</tr>
<tr>
<td>TP</td>
<td>18</td>
<td>4.5 ± 0.3</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

^*From Henze et al. (2000).
To achieve this result, care was taken to set the ratio of the initial soluble COD in the sample to the seeded biomass concentration to a high value (75:1 or greater) as recommended by Grady et al. (1999). For all other situations, a significant amount of energy must be expended for maintenance through decay, thereby lowering the observed yield. The linear portion of data in Fig. 4 (up to 48 h) was plotted in Fig. 5 to determine the value of the heterotrophic growth yield. The overall results from ten \( \text{Y}_H \) batch tests with CEBPR biomass and nine tests with MEBPR biomass are summarized in Table 4.

As shown in Table 4, a slightly smaller average growth yield was observed for the MEBPR process compared to the CEBPR process. Stephenson et al. (2000) indicated that the overall growth yields for membrane bioreactors (MBRs) are similar to those from conventional municipal wastewater treatment systems, typically between 0.14 to 0.50 on a kg COD/kg COD removed basis. However, in some cases the MBR growth yield has been reported to be close to, or greater than, those of conventional activated sludge processes (Chaize and Huyard 1991; Murakami et al. 2000; Ng and Hermanowicz 2005). The growth yield results from the present study agree well with reported values in the literature (Table 5).

The slightly lower growth yield in the MEBPR process versus the CEBPR process might be attributable to differences in the microbial species present in the two processes, which can affect the growth yield (Stephenson et al. 2000). Higher forms of microorganisms have been found in MBRs (Cicek et al. 1999; Ghyoot and Verstraete 1999) and such grazing microorganisms consume bacteria as food, leading to a decrease in observed yield. Ghyoot and Verstraete (1999) reported higher concentrations of flagellates and free ciliates in a submerged membrane MBR than in an activated sludge system operating at the same sludge age. It is known that protozoa and rotifers consume particulate organics, including bacteria. Other larger biological species such as nematode worms and insect larvae may contribute to the consumption of particulate organic matter (Grady et al. 1999).

Other environmental conditions related to membrane bioreactor operation, such as the high shear conditions, may cause breakage of cells and reduced floc size that may be related to increased energy consumption for maintenance (Meijer et al. 1995; Shimizu et al. 1994; Wisniewski and Grasmick 1997; Kim et al. 2001).

### Decay coefficient of heterotrophic microorganisms, \( b_H \)

Figures 6 and 7 show typical results from batch decay tests using biomass from the MEBPR and CEBPR processes. During the endogenous phase testing, there was no external input of substrate and therefore all substrate was initially generated through the processes of decay and hydrolysis. The biomass concentration and respiration rate decreased slowly and continuously, while inert particulate matter and biomass debris accumulated. The model decay coefficients estimated for the heterotrophic biomass from both processes are summa-
It was found that the $b_{M}$ values obtained for the two processes were statistically similar, but as for $Y_{M}$, the estimated value was lowest for the MEBPR train. The model decay coefficient was 0.17 to 0.33 1/d with a mean of 0.24 1/d for the MEBPR process, and 0.2 to 0.36 1/d with a mean value of 0.31 1/d for the CEBPR process, all at 20°C.

From equation 3, the average value for the traditional decay coefficient $b_{H}$ was calculated to be 0.14 1/d for the CEBPR process and 0.13 1/d for the MEBPR process at 20°C. Similar results were reported by Dold et al. (1986) who reviewed the literature concerning $b_{H}$ and concluded that in aerobic and anoxic wastewater treatment systems a typical value for heterotrophic biomass decay is 0.24 1/d. Reported values of $b_{H}$ vary widely, ranging from 0.05 1/d for domestic sewage in the U.S.A. to 1.6 1/d for some food-processing wastes (Henze et al. 1987). Henze et al. (1987) reported a typical model decay coefficient value for municipal wastewater of 0.62 1/d in ASM1. Metcalf & Eddy Inc. (1991) suggested a typical decay coefficient of 0.25 to 0.06 1/d at 20°C for conventional activated sludge processes and aerobic processes. Kappeler and Gujer (1992) suggested a value of 0.4 1/d. Wen et al. (1999) reported $b_{H}$ to be 0.08 1/d at 30°C in a ceramic membrane side-stream aerobic bioreactor treating raw wastewater. Fan et al. (1996) reported 0.05 1/d for an MBR at 30°C.

The model decay coefficient recommended in ASM2 is 0.4 1/d at 20°C. It is this wide range that leads to the recommendation that the decay coefficient be measured for each wastewater treatment situation under consideration (Henze et al. 1987).

### TABLE 5. Reported growth yield values for MBRs and conventional activated sludge (CAS) processes

<table>
<thead>
<tr>
<th>Wastewater</th>
<th>SRT (d)</th>
<th>Process</th>
<th>$Y_{M}$ (kg/kg)</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Municipal</td>
<td>50</td>
<td>MBR</td>
<td>0.2</td>
<td>Buisson et al. (1998)</td>
</tr>
<tr>
<td>Municipal</td>
<td>50</td>
<td>MBR</td>
<td>0.26</td>
<td>Trouve et al. (1994)</td>
</tr>
<tr>
<td>Municipal</td>
<td>50</td>
<td>MBR</td>
<td>0.2</td>
<td>Buisson et al. (1998)</td>
</tr>
<tr>
<td>Urban</td>
<td>10</td>
<td>MBR</td>
<td>0.34</td>
<td>Bouhabila et al. (1998)</td>
</tr>
<tr>
<td>Urban</td>
<td>20</td>
<td>MBR</td>
<td>0.2</td>
<td>Bouhabila et al. (1998)</td>
</tr>
<tr>
<td>Municipal</td>
<td>20</td>
<td>MBR</td>
<td>0.1</td>
<td>Fan et al. (1996)</td>
</tr>
<tr>
<td>Municipal</td>
<td>50</td>
<td>MBR</td>
<td>0.23</td>
<td>Côté et al. (1997)</td>
</tr>
<tr>
<td>Domestic</td>
<td>—</td>
<td>MBR</td>
<td>0.6</td>
<td>Morakami et al. (2000)</td>
</tr>
<tr>
<td>Domestic</td>
<td>100</td>
<td>MBR</td>
<td>0.56</td>
<td>Chaize and Huyard (1991)</td>
</tr>
<tr>
<td>Municipal</td>
<td>—</td>
<td>MBR</td>
<td>0.2</td>
<td>Côté et al. (1997)</td>
</tr>
<tr>
<td>Domestic</td>
<td>20</td>
<td>MBR</td>
<td>0.72</td>
<td>Jiang et al. (2004)</td>
</tr>
<tr>
<td>Synthetic</td>
<td>0.25-5</td>
<td>MBR</td>
<td>0.59</td>
<td>Ng and Hermanowicz (2005)</td>
</tr>
<tr>
<td>Synthetic</td>
<td>2-30</td>
<td>MBR</td>
<td>0.57</td>
<td>Macomber et al. (2005)</td>
</tr>
<tr>
<td>Municipal</td>
<td>17-25</td>
<td>MEBPR</td>
<td>0.50</td>
<td>Present study</td>
</tr>
<tr>
<td>Domestic</td>
<td>2</td>
<td>CAS</td>
<td>0.48-0.72</td>
<td>Ramanathan and Gaudy (1971)</td>
</tr>
<tr>
<td>Municipal</td>
<td>—</td>
<td>CEBPR</td>
<td>0.59</td>
<td>Present study</td>
</tr>
<tr>
<td>Domestic</td>
<td>2</td>
<td>Batch test</td>
<td>0.67</td>
<td>Kappeler and Gujer (1992)</td>
</tr>
<tr>
<td>Municipal</td>
<td>—</td>
<td>CAS</td>
<td>0.7-0.87</td>
<td>Strommann et al. (1999)</td>
</tr>
<tr>
<td>Municipal</td>
<td>—</td>
<td>CAS</td>
<td>0.46-0.69</td>
<td>Henze et al. (2000)</td>
</tr>
<tr>
<td>Synthetic</td>
<td>0.25-5</td>
<td>CAS</td>
<td>0.50</td>
<td>Ng and Hermanowicz (2005)</td>
</tr>
<tr>
<td>Municipal</td>
<td>17-25</td>
<td>CEBPR</td>
<td>0.59</td>
<td>Present study</td>
</tr>
</tbody>
</table>

*Table extended from Stephenson et al. (2000).*

---

**Fig. 6.** Example oxygen uptake rate (OUR) results for determination of heterotrophic decay coefficient using MEBPR process biomass.

**Fig. 7.** Example oxygen uptake rate (OUR) results for determination of heterotrophic decay coefficient using CEBPR process biomass.
Maximum specific growth rate of heterotrophic microorganisms, $\mu_H$.

As shown in Fig. 8 and 10, oxygen respiration increased due to the unlimited growth of heterotrophic biomass during the first two to three hours of the batch $\mu_H$ estimation procedure, followed by a sharp decrease to a low level when the readily biodegradable substrates were exhausted. The oxygen respiration rate at this stage was dominated by growth on substrate released by hydrolysis. The OUR data were then used to estimate the term $(\mu_H - b_H)$ from the slopes in Fig. 9 and 11 according to equation 5. The resulting value of $\mu_H$ was then adjusted to 20°C as explained above. The maximum specific growth rate results are summarized in Table 4.

Some of the $\mu_H$ results from the present study compare well with literature values, but others are markedly higher than the $\mu_H$ range of 0.12 to 0.55 1/h (2.88–13.2 1/d) reported by Grady et al. (1999). Kappeler and Gujer (1992) reported that $\mu_H$ varied in the range of 1 to 8 1/d for typical settled domestic sewage in Switzerland. Henze et al. (1995) reported a typical value of $\mu_H$ to be 6 1/d for primary effluent at 20°C in ASM2. Sözen et al. (1998) reported a $\mu_H$ range of 3.4 to 6.5 1/d for synthetic sewage and different wastewater mixtures under aerobic and anoxic conditions. Biokinetic parameter values, including $\mu_H$, depend strongly on the species of microorganisms and the growth substrates present. The value of $\mu_H$ has also been reported to be influenced by the configuration of the reactor (Grady et al. 1999). It should be noted that batch reactors would favour the fast-growing microorganisms in the biomass, perhaps generating a somewhat higher $\mu_H$ value.

The parameter values summarized in Table 4 indicate that, although all three parameter estimates were smaller for the MEBPR train than for the CEBPR train operated under parallel conditions, the differences between these were relatively small and statistically insignificant. The somewhat higher inert COD content of the UBC wastewater, relative to ASM2 typical values, could be a contributing factor to these differences. The MEBPR process undoubtedly retains a greater proportion of the inert COD in both the particulate and soluble forms. If so, the MEBPR biomass probably contained a higher proportion of nonviable, inert material than did the biomass in the CEBPR train. One effect of higher levels of inerts in the mixed liquor would be a reduction of the value of kinetic parameters such as $b_H$ and $\mu_H$. Even so, the differences in parameter values observed in this study were surprisingly small, such that it is reasonable to conclude that parameter values derived from conventional activated sludge treatment systems could
be applied to membrane bioreactor simulation without undue concern.

Conclusions

Wastewater characteristics determined using an ASM2-type approach indicated that the wastewater treated in the UBC pilot plant was similar to the “typical” municipal wastewater for which ASM2 was developed. Of the five organic substrate fractions considered by ASM2, it was found that the UBC wastewater contained fractions of volatile acids ($S_V$), readily fermentable biodegradable COD ($S_F$) and slowly biodegradable COD ($S_L$) that fell within the ASM2 typical ranges. The content of soluble inert COD ($S_I$) and particulate inert COD ($X_I$) were somewhat higher than ASM2 typical values. In any case, there seems to be little evidence that the wastewater characteristic of UBC would materially influence the estimated biokinetic and stoichiometric parameter estimates.

A selected subset of the ASM2 biokinetic and stoichiometric general parameter set was then estimated using procedures described in the literature, for mixed liquor samples taken from pilot-scale MEBPR and CEBPR processes operated in parallel. The results indicated that estimated values for $Y_{MN}$, $b_M$ and $\mu_M$ were slightly lower for the MEBPR train than for the CEBPR train, but the differences were not statistically significant. It was thought that the MEBPR process retained a higher proportion of the wastewater soluble and particulate inert COD than did the CEBPR process, leading to a higher nonviable or inert fraction of process biomass. This would have the effect of reducing the apparent values of $b_M$ and $\mu_M$. In any case, the differences between the MEBPR and CEBPR parameter values were small, suggesting that MEBPR simulation using ASM2 could be accomplished satisfactorily using parameter values determined for a conventional biological phosphorus removal process, even if MEBPR parameter values are not available.

Acknowledgements

We wish to thank Fred Koch, Susan Harper and Paula Parkinson for their valuable assistance with this study. The financial and material support provided by the Natural Sciences and Engineering Research Council of Canada (NSERC), Zenon Environmental Inc., Stantec Consultants and Dayton & Knight Ltd. are gratefully acknowledged.

References


Received: November 16, 2004; accepted: November 14, 2005.