Mass transfer impacts in flocculent and granular biomass from SBR systems

D. Gapes*,**, B.-M. Wilén*** and J. Keller*

* Advanced Wastewater Management Centre, University of Queensland, Brisbane, Australia
(E-mail: Daniel.Gapes@forestresearch.co.nz; j.keller@awmc.uq.edu.au)
** Forest Research, Rotorua, New Zealand
*** Water Environment Transport, Chalmers University, Gothenburg, Sweden
(E-mail: britt-marie.wilen@wet.chalmers.se)

Abstract An experimental study was conducted to describe mass transfer impacts within nitrifying aggregates sourced from sequencing batch reactor (SBR) activated sludge systems. Flocculent and granular sludge with high nitrification activity was obtained in two laboratory SBR systems, supplied with a synthetic, ammonium-based feed. The flocculent biomass was fractionated using a sieving procedure, in order to obtain biomass fractions with different particle size distributions. The oxygen uptake rate (OUR) response to changes in dissolved oxygen concentration was measured under highly controlled conditions in a titrimetric and off-gas analysis (TOGA) sensor, and the results used to assess mass transfer effects. As the average particle size of the biomass increased, mass transfer limitations were found to increase significantly. Empirically fitted, apparent $K_{S,O2}$ values were demonstrated to be highly dependent on particle size, and reflect the mass transfer limitations occurring in the aggregates within a given system. Such parameters thus have little to do with the actual biokinetic parameter from which they are derived. The results obtained from the TOGA sensor study were consistent with those obtained from a microelectrode study on the same nitrifying granules. Together, these studies add considerable weight to the conclusion that consideration of external and internal mass transfer limitations is vital to the accurate description of activated sludge treatment processes, particularly those with a high oxygen uptake rate.

Keywords Biological aggregates; flocs; granules; mass transfer; off-gas analysis; particle size distribution

Introduction Activated sludge treatment fundamentally occurs at a microscopic level, whereby microorganisms (bacteria, archaea, fungi and protozoa) within suspended aggregates utilise oxygen for the destruction of target pollutants. The transfer of oxygen from the gas phase, through the bulk liquid to the surface of the aggregate (a floc or granule), through the aggregate to the cell surface, then into the cell where biological reaction occurs, involves a number of mass transfer operations.

The current work focuses on two major mass transfer processes. **External mass transfer** describes the movement of molecules from the bulk liquid to the aggregate surface. This occurs in the region where the convective fluid velocity decreases from that observed in the bulk liquid, down to zero at the solid surface. In this region, often referred to as the external boundary layer, diffusive mass transfer increasingly dominates as the surface is approached, but advection (transport due to fluid motion) can still play a significant role. **Internal mass transfer** describes the diffusive movement of a molecule within the biological aggregate, to the site of the biochemical reaction (at or within the cell).

Either of these transfer steps is capable of exerting significant influence on the overall rate of biological treatment. Their relative importance in providing a process rate limitation is influenced by numerous factors, including the size and shape of the matrix, fluid and substrate properties, intrinsic reaction kinetics of the cells within the matrix, and the hydrodynamics of the bulk phase.
Some mathematical modelling, but very little experimental work has been carried out on investigating mass transfer within activated sludge flocs or granules. Internal mass transfer has been paid far greater consideration than external mass transfer limitation (e.g. Baillod and Boyle, 1970; Matson and Characklis, 1976; Atkinson and Rahman, 1979; Stenstrom and Song, 1991; Bakti and Dick, 1992; Beccari et al., 1992). However, because such scant attention has been paid to the external transfer process, it is possible that the input of internal mass transfer on limiting the aggregate reaction rates has in some cases been overemphasised. Obviously, both transport processes must be considered in order to obtain unbiased results.

The aim of the current work is to experimentally demonstrate external and internal mass transfer impacts in suspended biomass systems from laboratory reactors. The difficulty in quantitatively isolating external from internal mass transfer effects has been demonstrated by Gapes (2003), and therefore their combined impact is the major subject of the work. The work focuses on mass transfer of oxygen in nitrifying biomass sourced from sequencing batch reactors (SBRs), but has applicability to numerous soluble substrates within biological systems.

**Experimental methodology**

**SBR operation for activated sludge floc generation**

Activated sludge flocs were taken from a nitrifying SBR, which was operated under conditions detailed in Table 1. The synthetic feed, used for both the SBR operation and as the feed source in the batch (TOGA sensor) experiments consisted of (per g of NH$_4$-N): Na$_2$-EDTA (6.2 mg); ZnSO$_4$.7H$_2$O (27 mg); FeSO$_4$.7H$_2$O (17.2 mg); MnSO$_4$.H$_2$O (3.8 mg); CuSO$_4$.5H$_2$O (0.97 mg); Co(NO$_3$)$_2$.6H$_2$O (0.61 mg); Na$_2$B$_4$O$_7$.10H$_2$O$^-$ (0.44 mg); MgSO$_4$.7H$_2$O (712 mg); CaCl$_2$.2H$_2$O (164 mg); (NH$_4$)$_6$Mo$_7$O$_24$.4H$_2$O (0.46 mg); KH$_2$PO$_4$ (221 mg); K$_2$HPO$_4$ (271 mg); Peptone (98.6 mg); concentrated H$_2$SO$_4$ (1.44 mL). The reactor pH was controlled via addition of 1M NaHCO$_3$. The reactor temperature was maintained at the ambient temperature within the laboratory (19–24°C). The flocculent reactor was operated for approximately two years, with complete oxidation of ammonia to nitrate throughout the experimental period (data not shown).

**SBR operation for granule generation**

Subsequent to the completion of the flocculent experiments, the granular reactor was initiated by reducing the length of the settling phase from that of the flocculent system. Inoculum was provided by the flocculent reactor, supplemented with some biomass from an SBR treating landfill leachate. The SBR operational conditions are provided in Table 1. This reactor was operated for approximately 9 months.

**Table 1** Reactor conditions for flocculent and granular SBR operation

<table>
<thead>
<tr>
<th>Reactor conditions</th>
<th>Flocculent reactor</th>
<th>Granular reactor</th>
</tr>
</thead>
<tbody>
<tr>
<td>Reactor volume (L)</td>
<td>5.5</td>
<td>3</td>
</tr>
<tr>
<td>DO (feed and react periods, mg.L$^{-1}$)</td>
<td>4–6</td>
<td>5–6</td>
</tr>
<tr>
<td>pH</td>
<td>7.8</td>
<td>7.8</td>
</tr>
<tr>
<td>Operating temperature (°C)</td>
<td>20–25</td>
<td>20–25</td>
</tr>
<tr>
<td>Feed time (min)</td>
<td>25</td>
<td>25</td>
</tr>
<tr>
<td>React time (min)</td>
<td>285</td>
<td>448</td>
</tr>
<tr>
<td>Settle time (min)</td>
<td>45</td>
<td>2</td>
</tr>
<tr>
<td>Decant time (min)</td>
<td>5</td>
<td>5</td>
</tr>
<tr>
<td>NH$_4^+$-N load (g.m$^{-3}$.d$^{-1}$)</td>
<td>1.0–1.3</td>
<td>0.8–0.9</td>
</tr>
<tr>
<td>HRT (d)</td>
<td>0.6</td>
<td>0.7</td>
</tr>
<tr>
<td>SRT (d)</td>
<td>18–20</td>
<td>22–25</td>
</tr>
</tbody>
</table>
Operational performance was not as stable as that of the flocculent reactor. Care had to be taken to maintain the biomass levels within the reactor by minimising shocks to the system (e.g. accidental pH decrease, or starvation due to feed deprivation). However, the biomass retained within the reactor was highly active, as is demonstrated in the results section below.

Shortening the time available for biomass settling was used as the principal means of attaining a granular sludge (as described in Morgenroth et al. (1997), Beun et al. (2000)). Those aggregates with low settling velocity (loose or small aggregates) are unable to be retained, while those with high settling velocity (relatively large and dense) have greater probability of remaining within the reactor.

\[ u_{\text{settle}} = \frac{L_{\text{decant}}}{t_{\text{settle}}} \]  

Beun et al. (2000) described the minimum settling velocity that a particle must have to be effectively retained within the SBR \( (u_{\text{settle}}) \). This can be calculated from the settling time \( (t_{\text{settle}}) \) and the height of reactor liquid that is above the decanting port \( (L_{\text{decant}}) \).

Using Eq. (1), the settling velocity for the nitrifying granules was around 4.5–5 m.h\(^{-1}\). This is significantly lower than the values of 9–40 m.h\(^{-1}\) reported for granular systems by previous authors (Morgenroth et al. (1997), Beun et al. (2000)). Despite this, the desired aim of producing biomass aggregates with significantly greater size than the flocculent system was achieved (as shown below).

These reactors were primarily used to provide biomass for the TOGA sensor analysis (see below), where the nitrification performance was analysed in great detail.

**TOGA sensor experiments**

The TOGA sensor (titrimetric and off-gas analysis) sensor was used as the basis of the experimental method. This sensor is a highly controlled bioreactor, which is capable of pH titration and off-gas mass balancing, as described in detail by Gapes and Keller (2001) and Pratt et al. (2003). In the current work, the primary measurement of nitrification rate in these experiments was the oxygen uptake rate per unit TSS (the specific OUR, \( \text{OUR}_{S} \)).

The SBR biomass was settled and a volume of supernatant decanted and replaced by feed solution (typically 1 L for a 3 L reaction volume in the TOGA). Initial ammonium concentrations within the TOGA ranged from 100–250 mg.L\(^{-1}\) \( \text{NH}_{4}^{+}-\text{N} \), similar to the concentrations experienced in the SBR during the initial period of the react phase. This mixed liquor was then used in the TOGA sensor for the experiments described below.

Ambient temperatures (20–24°C) were used in all of the activated sludge experiments, without temperature control. However, due to the short duration of each experiment (usually 3–4 h to give one set of results) little change in temperature (usually less than 1°C) was observed. The tests on granular sludge were carried out under controlled temperature conditions at 22–23°C. A pH of 7.8 was maintained for all tests. In order to eliminate the possibility of carbon limitation, 30 mL of 1M NaHCO\(_{3}\) was added to the reactor at the beginning of each TOGA experiment.

The response of the different biomass types was measured over a range of DO setpoints in the TOGA, from 0.5 mg.L\(^{-1}\) up to levels at which the oxygen became non-limiting. In each of these DO experiments, the energy dissipation rate (as controlled by gas flow rate) within the TOGA was maintained at a constant level.

**Particle size distribution**

The volumetric-based size distributions of the systems were assessed using a laser diffractometer (Mastersizer/E, Malvern Instruments Limited, UK) for the activated sludge flocs, and image analysis for the granules. The latter method was required due to the large
size of the granules, coupled with the limitation of the Malvern instrument to measurement of particles with dimensions less than 600 µm. Full details of the size analysis procedures are found in Gapes (2003).

To determine the mass transfer effects on various floc sizes, a size separation procedure, using mechanical sieving was performed. This was achieved by gently washing the mixed liquor from the SBR through a particle sieve of defined aperture size (212 µm or 106 µm, Laboratory Sieve, Endacotts Ltd, London, England). Flocs, smaller than the aperture size, passed through with the liquid and the oversize fraction remaining was washed into diluted feed solution, thus obtaining two fractions for use in experiments.

Results and discussion
Size distribution
Activated sludge flocs. The size distribution of biomass from the flocculent SBR is presented as a cumulative volumetric percentage in Table 2. For a given particle diameter, the volume of particles in the sample that are smaller than this particular size can be obtained from the graph. The size distributions observed in the current work are well within the range found in the literature (e.g. Li and Ganczarczyk (1991)). Table 2 also shows the effect of the sieving procedure on the cumulative size distribution of the flocculent system. Clearly, sieving was successful in producing a separation based on floc size.

Granules. Using averaged data for 5 volumetric size distribution profiles, the median granule diameter was found to be 640 µm, with 10th and 90th percentiles of 380 µm and 900 µm, respectively. The granules were thus confirmed as being of significantly larger size than the aggregates obtained from the flocculent system.

Dissolved oxygen effects
Flocculent system. The rate response of the nitrifying activated sludge to changes in bulk dissolved oxygen (DO) concentration is shown in Figure 1. Also plotted in Figure 1 are three curves for reference:
1. the Monod curve for planktonic cells, based on a set maximum oxygen uptake rate of 340 mg.gTSS⁻¹.h⁻¹ and a half saturation coefficient for oxygen in nitrification ($K_{S,O2}$) of 0.3 mg.L⁻¹;
2. the Monod curve generated with a set maximum OUR (340 mg.g.h⁻¹), and apparent $K_{S,O2}$ value which provides the best fit of the curve to the data present (using non-linear least squares regression; regressed $K_{S,O2} = 1.6$ mg.L⁻¹);
3. the Monod curve using values for maximum OUR (520 mg.g.h⁻¹) and apparent $K_{S,O2}$ (3.4 mg.L⁻¹) that provide the best fit to the plotted data (using non-linear least squares regression).

At DO values less than 3–4 mg.L⁻¹, the experimental OUR data is significantly lower than that of curve 1, the Monod kinetics of planktonic cells. This clearly indicates the effect

<table>
<thead>
<tr>
<th>Sample</th>
<th>10th percentile</th>
<th>Median</th>
<th>90th percentile</th>
</tr>
</thead>
<tbody>
<tr>
<td>Unfractionated (11 samples)</td>
<td>25–86</td>
<td>105–224</td>
<td>228–422</td>
</tr>
<tr>
<td>212 µm sieve</td>
<td>unfractionated</td>
<td>36</td>
<td>113</td>
</tr>
<tr>
<td></td>
<td>oversize</td>
<td>72</td>
<td>247</td>
</tr>
<tr>
<td></td>
<td>undersize</td>
<td>34</td>
<td>108</td>
</tr>
<tr>
<td>106 µm sieve</td>
<td>unfractionated</td>
<td>35</td>
<td>116</td>
</tr>
<tr>
<td></td>
<td>oversize</td>
<td>68</td>
<td>149</td>
</tr>
<tr>
<td></td>
<td>undersize</td>
<td>21</td>
<td>69</td>
</tr>
</tbody>
</table>
of mass transfer limitations on the reaction rate of flocs. The plotting of curves 2 and 3 provide insight into the differences between true biological Monod kinetics, and those which can be obtained by fitting Monod-type equations to experimental data from flocculent systems, without accounting for mass transfer effects.

Granular system. The OUR$_S$ vs. DO profile of the granular sludge is shown in Figure 2, along with two sets of data from the activated sludge floc analysis. Also included is the best fit curve for a Monod-type equation, obtained by nonlinear regression for all the given data. Although the maximum OUR of the granular sludge was not attained over the DO range tested, it appears to be similar to that found in the flocculent sludge. This similarity reduces the complexity of comparisons between the flocculent and granular biomass, allowing rate differences to be attributed to size rather than difference in biomass composition or

**Figure 1** Effect of dissolved oxygen on specific OUR of flocculent biomass. Symbols identify different experimental dates. Curved lines are modelled results (see text for details)

**Figure 2** Impact of DO on OUR of granular system (●), including comparison with flocculent system (*)
intrinsic biological kinetics. The saturation DO level was significantly higher than was observed for the flocculent sludge. Indeed, even at a DO of 12 mg.L\(^{-1}\), the maximum rate appears not quite reached, compared with a saturation level of 5–6 mg.L\(^{-1}\) for the flocculent biomass. This finding is consistent with a model-based analysis (Gapes (2003)), indicating an increase in mass transfer effects with increased particle size.

**Size fractioned floc samples.** The impact of aggregate size on the \(OUR_s\) vs. DO profiles, using results from both the unfractionated and size fractioned floc systems, is shown in Figure 3. At DO setpoints less than 5–6 mg.L\(^{-1}\), the OUR of the different samples decreases as the median floc size within the sample increases. Under mass transfer limiting conditions, substrate penetration within the particle is incomplete, and at a given bulk liquid concentration, a particle’s inactive mass fraction increases with its size, thus reducing the mass specific rate.

Figure 3 shows that at DO levels between 5 and 6 mg.L\(^{-1}\) and flocs with median diameter of around 110 \(\mu\)m, there is little difference in measured \(OUR_s\) values. This indicates that flocs in these samples were fully penetrated by oxygen, and therefore no longer diffusion limited. Conversely, the floc sample with the largest median diameter was never fully saturated over the experimental DO range.

**Internal mass transfer limitation in flocs and granules**

In the TOGA sensor experiments completed, it was not possible to isolate internal from external mass transfer effects, if the latter were significant. However, assuming that diffusive mass transfer occurs within the particle, external mass transfer limitation cannot occur in these systems in the absence of internal mass transfer limitation, as transport resistance should only ever be enhanced within the aggregate matrix when compared with transport in the fluid boundary layer. Conversely, internal mass transfer limitation could potentially occur in the absence of a significant external mass transfer limitation. It follows, therefore, that any observable mass transfer limitation includes the impact of internal mass transfer.

As previously mentioned, a particle model (described in Gapes (2003)) predicts that as particle size increases, the importance of internal mass transfer resistance likewise increases. The experimental work that has been presented herein is consistent with this prediction, as summarised graphically by:

![Figure 3](https://iwaponline.com/wst/article-pdf/50/10/203/419235/203.pdf)

**Figure 3** Effect of floc size on oxygen uptake rates (OUR) under different DO conditions
1. Figures 1 and 2, which reveal that the granular sludge has a lower \( OUR_s \) than the flocculent sludge at any given bulk DO;

2. Figure 3, which demonstrates a significant decrease in \( OUR_s \) with increasing median floc size of the fractionated floc samples.

Further, microbiological analysis (described in Gapes, 2003) showed that the bacterial species were similar in the flocculent and granular systems. Therefore, the differences in observed rate responses between the systems are unlikely to have resulted from differences in microbial cell kinetics (particularly \( K_{s,O2} \)).

Hence, internal mass transfer limitation has clearly been demonstrated within these two systems. To obtain some quantitative data for comparison, the apparent Monod kinetic parameters for the various experimental datasets are presented in Table 3.

Excluding the results from the floc fraction with median diameter of 232 \( \mu \text{m} \), the fitted \( OUR_{s,\text{max}} \) values were relatively consistent, covering the range 440–520 \( \text{mg.gTSS}^{-1}.\text{h}^{-1} \). This adds weight to the argument that the overall makeup of the biomass is similar for the two systems. The apparent \( K_{s,O2} \) is greatly different to the biokinetic values, estimated to be (at most) less than 1.3 \( \text{mg.L}^{-1} \) for both steps in the nitrification process (Painter, 1970; Sharma and Ahlert, 1977). Unlike the \( OUR_{s,\text{max}} \), the calculated values differed considerably, and are observed to consistently increase with median floc size.

The results of a literature review of \( K_{s,O2} \) values by Stenstrom and Poduska (1980) are also presented in Table 3 for comparison. These authors suggested that such variation in values reported for this parameter could be due to neglecting (amongst other things) mass transfer impacts in the system being studied. Further, Bakti and Dick (1992) estimated \( K_{s,O2} \) for a nitrifying activated sludge, using both a diffusional floc model and a model in which diffusion was not considered. The differences between \( K_{s,O2} \) values obtained using the two models led to the conclusion that diffusional limitation was highly likely.

The term \( K_s \) has become misused when discussing the kinetics involved in activated sludge systems. Sometimes, such as in models describing diffusion/reaction within biological aggregates, it is used correctly for defining an intrinsic biokinetic parameter. At other times it is a highly system-specific, empirical value. In models like the Activated Sludge Model Nos 1 and 2 of Henze et al. (2000), the biological system is modelled as a whole reactor, with no consideration given to diffusional or structural impacts. The apparent \( K_s \) utilised in these latter models is, as has been clearly shown in the current work, significantly affected by the mass transfer limitations occurring due to particle size and structure. In these cases, care should be taken to define the “\( K_s \)” term as a system specific, empirical value, rather than implying an intrinsic kinetic property of the microbial biomass.

Finally, consideration of the internal mass transfer limitation within the granules revealed an interesting outcome. Both the flocculent and granular biomass were developed within their respective SBR systems at bulk liquid DO concentrations in the range 4–6 \( \text{mg.L}^{-1} \). Therefore, the granular sludge would have been operating under a significant

### Table 3 Fitted apparent Monod parameters: current work and literature sources

<table>
<thead>
<tr>
<th>Median diameter (( \mu \text{m} ))</th>
<th>Apparent ( K_{s,O2} ) (mg.L(^{-1} ))</th>
<th>Fitted ( OUR_{s,\text{max}} ) (mg.gTSS(^{-1}.\text{h}^{-1} ))</th>
<th>( R^2 )</th>
</tr>
</thead>
<tbody>
<tr>
<td>Flocc (from Figure 1)</td>
<td>110–190</td>
<td>3.4</td>
<td>520</td>
</tr>
<tr>
<td>Fractionated floc samples</td>
<td>69</td>
<td>2.05</td>
<td>448</td>
</tr>
<tr>
<td></td>
<td>108</td>
<td>2.62</td>
<td>447</td>
</tr>
<tr>
<td></td>
<td>149</td>
<td>3.44</td>
<td>441</td>
</tr>
<tr>
<td></td>
<td>247</td>
<td>5.03</td>
<td>232</td>
</tr>
<tr>
<td>Granules (from Figure 2)</td>
<td>640</td>
<td>4.9</td>
<td>430</td>
</tr>
<tr>
<td>Nitrifying bacteria, various systems</td>
<td>Stenstrom and Poduska (1980)</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
internal oxygen mass transfer limitation during periods of the react phase where ammonia was non-limiting, as evidenced by the OUR vs. DO curve in Figure 2. Conversely, much of the biomass in the flocculent system would have been operating at, or close to, oxygen saturation conditions. Despite this difference, the $OUR_{S_{max}}$ for the granular system was at least as high as that of the flocculent system (Figure 2 and Table 3). This is a somewhat surprising result, given that biomass that is under no growth conditions could be expected to decay, thus decreasing the overall specific activity. One possibility for this similarity in $OUR_{S}$ is that inorganic material, which would neither have an oxygen demand nor be counted in the VSS measurement, was dominant in the interior of the floc. However, the VSS/TSS ratio, a measure of biomass inorganic content, does not support this hypothesis, as it remained consistent between the two systems (data not shown).

Three alternative suggestions are offered as explanations for observations contrary to the expectation of a decreased maximum rate for the granular sludge. First, the ammonia substrate famine period at the end of the SBR cycle may have allowed just enough access of oxygen to the bacteria within the interior of the large granules to limit endogenous activity. A second possibility is that the granules were constantly undergoing formation/break-up processes, therefore exposing the interior of the biomass to non-limiting oxygen conditions. Finally, it is possible that the absence of oxygen limits the rate of endogenous decay, as found by Siegrist et al. (1999). No experimental work was carried out to further examine this phenomenon, but this may be important for understanding processes occurring within flocs and granules.

The current results support the work by authors such as Baillod and Boyle (1970), Matson and Characklis (1976), Atkinson and Rahman (1979), Stenstrom and Song (1991), Bakti and Dick (1992), and Beccari et al. (1992) in concluding that internal mass transfer limitations are highly significant in suspended biomass systems, and highlights the need to consider this effect when carrying out investigations on such systems.

Comparison of TOGA sensor results with microelectrode study

The results of a microelectrode study carried out on the granules used in the current work provides an extremely useful comparison for the reactor-based (TOGA) work. Wilén et al. (submitted) have conducted intensive experiments on measuring the oxygen profiles in and around the nitrifying granules, within a flowcell device that suspended the granule in an upwelling liquid flow. Granules of diameter greater than 200 $\mu$m were assessed in the flow-cell, under various DO and liquid flowrate conditions. A typical DO profile is presented in Figure 4. The work by these authors demonstrated the presence of a significant external mass transfer boundary layer in the granules. Further, the rapid decrease in DO concentration within the granule structure confirms a large internal mass transfer limitation within the granules, as concluded from the current, reactor-based, work. Mirroring the TOGA sensor work, the microelectrode studies found that maximum reaction rates within the granules (calculated from the DO profiles) were not reached until the bulk liquid DO was greater than 12 mg.L$^{-1}$.

Together, the two studies clearly demonstrate the significance of both external and internal mass transfer limitation in flocculent and granular activated sludge, and the importance of including these resistances within models of such systems.

Conclusions

A number of conclusions can be drawn from the current work.

1. For flocculent or granular systems with high specific substrate uptake rates (such as oxygen consumption in nitrification) internal mass transfer provides a significant rate limitation. As particle size increases, mass transfer limitations increase significantly.
2. Empirically fitted, apparent $K_{S,O2}$ values were demonstrated to be highly dependent on particle size, and reflect the mass transfer limitations occurring in the aggregates within a given system. Such parameters have little to do with the actual biokinetic parameter from which the name is derived. To avoid confusion or generation of erroneous results, care should be taken in defining, measuring and utilising the half saturation coefficient.

3. The results obtained from the TOGA sensor study were consistent with those obtained from a microelectrode study on the same nitrifying granules, which provided direct evidence of an external mass transfer limitation (Wilén et al., submitted). Together, these studies provide considerable weight to the conclusion that consideration of external and internal mass transfer limitations is vital to the accurate description of activated sludge treatment processes.

References


Figure 4: Example of DO profile of granule (size approximately 400 µm, from Wilén et al. (submitted)).

Liquid flowrate 0.75 mm.s⁻¹

2. Empirically fitted, apparent $K_{S,O2}$ values were demonstrated to be highly dependent on particle size, and reflect the mass transfer limitations occurring in the aggregates within a given system. Such parameters have little to do with the actual biokinetic parameter from which the name is derived. To avoid confusion or generation of erroneous results, care should be taken in defining, measuring and utilising the half saturation coefficient.

3. The results obtained from the TOGA sensor study were consistent with those obtained from a microelectrode study on the same nitrifying granules, which provided direct evidence of an external mass transfer limitation (Wilén et al., submitted). Together, these studies provide considerable weight to the conclusion that consideration of external and internal mass transfer limitations is vital to the accurate description of activated sludge treatment processes.

References


