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Model development and application for surfactant biodegradation in an acclimatising activated sludge system

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Abstract

A model for the biodegradation of non-ionic surfactants in an activated sludge system during acclimatisation was developed, based on respirometric and titrimetric experimental data. The data were obtained in a sequencing batch reactor (SBR) using a non-ionic surfactant as sole carbon source and sludge previously acclimatised to a different surfactant. The model was successfully applied to successive SBR-cycles of sludge acclimatisation processes subjected to two ethoxylated surfactants. The model was validated using the corresponding total organic carbon data. The evolution of the model parameters along the acclimatisation process was examined. An acclimatisation model was developed using the evolution of two of these parameters (affinity constant and inhibition constant), supported by two independently calculated acclimatisation indicators. This acclimatisation model was then applied to determine an optimal surfactant concentration sequence to feed non-acclimatised sludge during a period of 41 days, in order to induce pre-acclimatisation to this surfactant before it replaces another one in the wastewater. The model was also useful in the economical assessment of this and alternative procedures to cope with frequent changes in activated sludge feed composition.

Keywords: Acclimatisation; Model development; Non-ionic surfactants; Optimisation; Textile industry wastewater; Transient state

1. Introduction

Modelling a biological sludge acclimatisation process can be a useful tool in the effort to understand the biological processes underlying it and to obtain important data for adequate wastewater treatment plant design and control. Although no activated sludge system can truly be considered to run at steady state, these systems are usually assumed to be "steady" enough in average terms to be studied and modelled as such (Olsson and Newell, 1999). However, only focusing on steady state modelling leads to descriptions which are often not flexible enough for control purposes. Therefore, it is important to model the transient states whenever they significantly affect the system performance due to their frequency or extent. The study reported here was performed within the context of wastewater treatment in the textile wet processing industry, a sector characterised by campaign-wise manufacturing schedules. Thus, the textile wastewater composition varies frequently and markedly. Furthermore, it has been observed that even a slight change in the surfactant content of such wastewater can affect the COD removal efficiency (Carvalho et al., 2000). The aim of the work presented in this paper was to develop a model which could be used to predict activated sludge behaviour when subjected to realistic variations in the surfactant fraction of an incoming textile effluent. The developed model should be applicable to wastewater treatment control and to optimise the operational methodology under transient conditions. An example of this application is given, and the

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Nomenclature

a	acclimatisation rate constant for $K_{S1,\mathrm{acc}}$ (mgCOD min/l)$^{-1}$
b	affinity constant for $K_{S_1,\mathrm{acc}}$ (mgCOD/l)
b_{H}	decay rate coefficient of heterotrophic biomass (min^{-1})
c	acclimatisation rate constant for $K_{\mathrm{I,acc}}$ (min^{-1})
d	affinity constant for $K_{\mathrm{I,acc}}$ (mgCOD/l)
k_3	first-order kinetic constant for heterotrophic growth on S_3 (mgCOD/l min)
k_{h}	hydrolysis rate constant for S_1 (min^{-1})
K_{I}	inhibition constant for S_2 (mgCOD/l)
$K_{\mathrm{I},0}$	constitutive inhibition constant exhibited by non-acclimatised sludge (mgCOD/l)
$K_{\mathrm{I,acc}}$	inhibition constant increment resulting from acclimatisation (mgCOD/l)
$K_{\mathrm{I,acc,max}}$	maximum value for $K_{\mathrm{I,acc}}$ (mgCOD/l)
K_{S_1}	hydrolysis affinity constant for S_1 (mgCOD/l)
$K_{S_{1,0}}$	constitutive S_1 hydrolysis affinity constant exhibited by non-acclimatised sludge (mgCOD/l)
$K_{S_1,\mathrm{acc}}$	S_1 hydrolysis affinity constant reduction resulting from acclimatisation (mgCOD/l)
$K_{S_1,\mathrm{acc,max}}$	maximum value for $K_{S_1,\mathrm{acc}}$ (mgCOD/l)
K_{S_2}	heterotrophic affinity constant for S_2 (mgCOD/l)
r_{O}	exogenous respiration rate (mg/l min)
S_1	concentration of surfactant in its initial form (first COD fraction) (mgCOD/l)
S_2	concentration of surfactant's metabolites (second COD fraction) (mgCOD/l)
S_3	concentration of the surfactant's secondary metabolite (third COD fraction) (mgCOD/l)
t	assay time after surfactant feed (min)
X_{H}	active heterotrophic biomass concentration (mgCOD/l)
Y_{H_2}	yield of heterotrophic biomass for S_2 (mgCOD/mgCOD)
Y_{H_3}	yield of heterotrophic biomass for S_3 (mgCOD/mgCOD)
β	transition parameter
μ_{H_2} and μ_{H_3}	specific growth rates of heterotrophic biomass on S_2 and S_3, respectively (min^{-1})
μ_{maxH_2}	maximum specific growth rate on S_2 (min^{-1})
v_{S_3}	yield of conversion of S_2 into S_3 (mgCOD/mgCOD)
τ	transition time constant (min)

suitability of the model to support an economic evaluation of alternatives for wastewater treatment plant (WWTP) design and operation is also evidenced.

Surfactants were selected as the sole carbon source fed to the studied activated sludge system as they represent one of the highest COD fractions in textile wastewaters (Knutzen and Wenzel, 1996). The ethoxylated non-ionic type is the most commonly employed surfactant class in textile wet processing (Patoczka and Pulliam, 1990). The biodegradation of polyethoxylated surfactants is known to begin either by the stepwise oxidation of the ethoxylated chain (Kravetz, 1981; Brunner et al., 1988; Ahel et al., 1994; Maki et al., 1994; Tidswell et al., 1996) or by the alkanol– or alkylphenol–ether bond cleavage. The latter, more common in linear alkyl ethoxylates (LAE), is followed by rapid degradation of the hydrophobic moiety and slower degradation of the ethoxylate chain (Swisher, 1987).

2. Materials and methods

The experimental data were obtained with an activated sludge sequencing batch reactor (SBR) of 4.8 l capacity, operated on a 24 h-cycle, connected to a 0.8 l, closed respirometric cell. A basic nutrient solution excluding the carbon source was fed at the beginning of each cycle. The surfactant was added to the medium at the end of the fill phase and it consisted of a polyethoxylate 4- or 10-lauryl ether (identified as Brij 30 and POE 10-LE, respectively), both laboratory-grade products from Sigma, USA. A hydraulic retention time of 26 h was imposed and the sludge retention time was set at 10 and 15 days, respectively for the Brij 30 and the POE 10-LE assays.

Sludge activity during the aerated phase was monitored by closed respirometry. The uptake of surfactants was followed by measurements of non-ionic surfactant concentration (NIO) using a titrimetric method with a specific electrode and a reference Ag/AgCl electrode (Metrohm, Switzerland, Application Bulletin No. 230/1e). COD and total suspended solids (TSS) were determined by standard methods (American Public Health Association, 1995). Total organic carbon (TOC) was measured in a Dhormann (USA) apparatus, model DC-190, according to the manufacturer's instructions. Dissolved oxygen was measured in the respirometric cell with a YSI (USA) Biological Oxygen Monitor, model 5300. The SBR was inoculated with sludge from an urban WWTP (Beirolas, Loures, Portugal) receiving a mixed domestic–industrial wastewater. Other details concerning the feed composition and maintenance, and employed monitoring techniques are described elsewhere (Carvalho et al., 2001, 2002).

Sludge acclimatisation to Brij 30 was followed along five 24-h SBR cycles. The process was monitored by respirometry and COD, TOC and NIO measurements during the first, second, fourth and fifth cycles. The acclimatisation to POE 10-LE was followed for up to 21 SBR cycles, all monitored except cycles 4, 10, 11, 16–18 and 20. The tested models were fitted to the experimental, respirometric and NIO data automatically, by means of the WEST® (Hemmis NV, Kortrijk, Belgium) modelling and simulation software environment (Vanhooren et al., 2003). The automatic parameter estimation procedure consisted of minimising the weighted sum of squared errors (SSE) for a set of parameters using the Simplex method. The weights used in the fitting multivariable criterion were 1000 and 0.0001 for respirometric and NIO data, respectively, so as to correct the different orders of magnitude and accuracies of the two types of data.

3. Results and discussion

3.1. Development of a dynamic model for surfactant biodegradation

3.1.1. Model development

A preliminary, qualitative analysis of the results (Carvalho et al., 2000, 2002) suggested that saturation and inhibition effects were involved in this surfactant biodegradation system. However, simple Monod and Haldane models based on a single carbon source were not able to predict the wide variety of respirometric profiles (see Fig. 1) obtained from the acclimatisation process, such as multiple-peak respirograms, initial ascending slopes or "shoulders" in the descending part of the respirogram. Respirograms obtained with non-acclimatised sludge indicated that the degradation of non-ionic surfactants might be occurring in multiple, successive steps (Carvalho et al., 2000). Indeed, the original surfactant molecule and its successive metabolites can be regarded as different substrates which are degraded by different enzymes or bacterial consortia with different kinetic character. This is supported by the fact that the metabolic pathways proposed for the degradation of ethoxylate surfactants both include the sequential degradation, at different rates, of different parts of the initial molecule. Some of the intermediates can eventually be resistant to biodegradation by non-acclimatised sludge.

In view of this, a dynamic model was developed (Eqs. (1)–(8) and (10)) in which primary degradation of the intact surfactant molecule (a COD fraction identified as S_1) was considered to be an enzymatic conversion, with no growth associated to it (Grady, personal

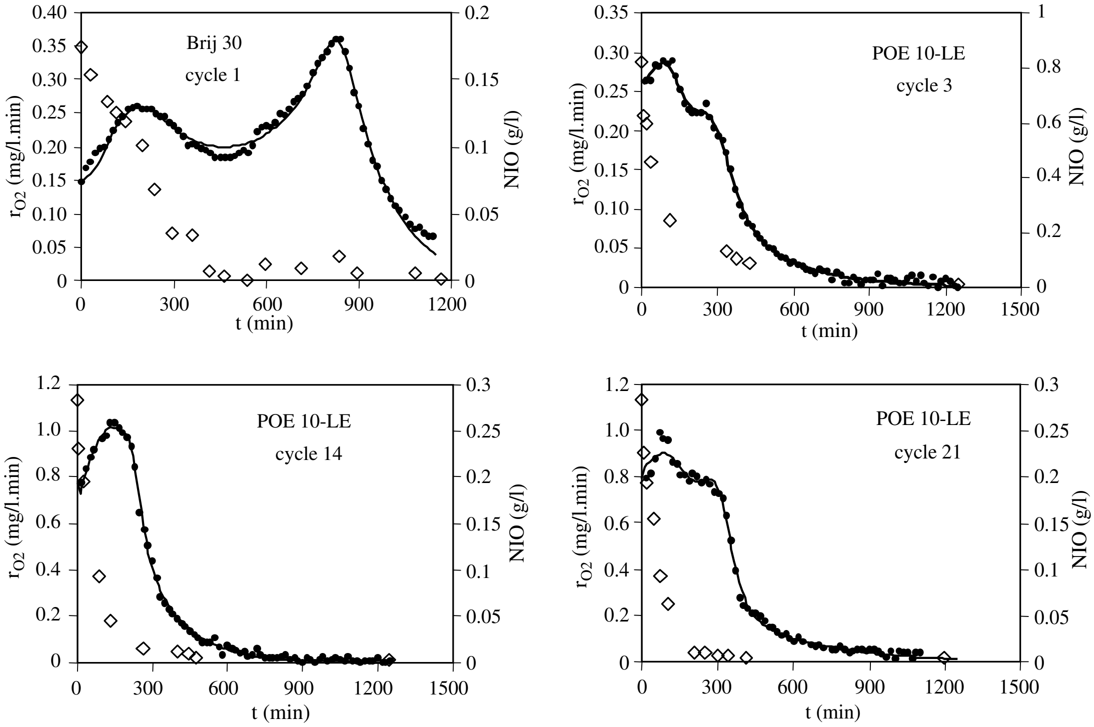

Fig. 1. Experimental respirometric data (●), r_O values predicted by the model (—) and NIO data (◇) for the first SBR cycle of the acclimatisation process to Brij 30 (weighted SSE = 8.3) and for the third, 14th and 21st cycles of the acclimatisation process to POE 10-LE (weighted SSE = 25, 20 and 62, respectively).

communication). Therefore, a kinetic equation (Eq. (1)) of the Michaelis–Menten type was used to describe the first degradation step.

$$\frac{dS_1}{dt} = -k_h \cdot \frac{S_1 \cdot X_H}{K_{S_1} + S_1} \tag{1}$$

The apparent inhibition effect observed in previous experiments, which was assumed to cause the multiplicity of peaks in a single respirogram, was admitted to be associated to the degradation of one of the metabolites and not to the initial molecule, as no apparent inhibition was observed in the initial respiration rate in the studied range of concentrations (Carvalho et al., 2001). Therefore, it was assumed that the biomass growth due to an intermediate metabolite (a second COD fraction identified as S_2) followed Haldane inhibition kinetics, represented by Eqs. (2) and (3).

$$\mu_{H_2} = \mathrm{Trans} \cdot \mu_{\max H_2} \cdot \frac{S_2}{K_{S_2} + S_2 + S_2^2/K_I} \tag{2}$$

$$\frac{dS_2}{dt} = -\frac{\mu_{H_2} \cdot X_H}{Y_{H_2}} + k_h \cdot \frac{S_1 \cdot X_H}{K_{S_1} + S_1} \tag{3}$$

The remaining metabolites (COD fraction S_3) were assumed to follow first-order degradation kinetics (Eqs. (4) and (5)), thus giving the low tail observed in the respirograms.

$$\mu_{H_3} = k_3 \cdot S_3 \tag{4}$$

$$\frac{dS_3}{dt} = -\frac{\mu_{H_3} \cdot X_H}{Y_{H_3}} + v_{S_3} \cdot \frac{\mu_{H_2} \cdot X_H}{Y_{H_2}} \tag{5}$$

The parameter Trans (Eq. (6)) takes into account the initial start-up time required for solution homogenisation after substrate addition (Vanrolleghem et al., 1998).

$$\mathrm{Trans} = 1 - \beta \cdot e^{-t/\tau} \tag{6}$$

The final proposed model is thus based on the sequential conversion of S_1 to S_2, which is then oxidised to a third intermediate, S_3, and finally mineralised (Carvalho et al., 2001). The exogenous respiration rate (r_O) and the biomass growth (Eqs. (7) and (8), respectively) were considered to be due to the consumption of S_2 and S_3 only.

$$r_O = \left[\frac{1 - Y_{H_2} - v_{S_3}}{Y_{H_2}} \cdot \mu_{H_2} + \frac{1 - Y_{H_3}}{Y_{H_3}} \cdot \mu_{H_3}\right] \cdot X_H \tag{7}$$

$$\frac{dX_H}{dt} = \left(\mu_{H_2} + \mu_{H_3} - b_H\right) \cdot X_H \tag{8}$$

The initial conditions (biomass concentration X_H (0), surfactant concentration S_2 (0) and respiration rate r_O (0)) were applied to Eq. (7) to calculate the values of the mathematical parameter β, introduced in Eq. (6). At $t = 0$, combining Eqs. (7), (6) and (2), and assuming that metabolite S_3 is in negligible concentration,

$$r_O(0) = X_H(0) \cdot \left(1 - \beta \cdot e^{-t/\tau}\right) \cdot \mu_{\max H_2} \cdot \frac{S_2(0)}{K_{S_2} + S_2(0) + S_2(0)^2/K_I} \cdot \frac{1 - Y_{H_2} - v_{S_3}}{Y_{H_2}} \tag{9}$$

which can be manipulated to obtain Eq. (10), below.

The dynamic model proposed is given by Eqs. (1)–(9).

$$\beta = 1 - \frac{K_{S_2} + S_2(0) + S_2(0)^2/K_I}{S_2(0) \cdot \mu_{\max H_2}} \cdot \frac{r_O(0) \cdot Y_{H_2}}{X_H(0) \cdot (1 - Y_{H_2} - v_{S_3})} \tag{10}$$

3.1.2. Parameter calibration process

A structural identifiability analysis was carried out, i.e., an attempt to prove that the model parameters are independent in such a way that there is, theoretically, a unique set of parameter values which correctly fits the experimental data (Dochain and Vanrolleghem, 2001). The employed methodology consisted of transforming the non-linear model to a linear form by the generating series approach. Due to the high number of equations included in this model, an overall analysis appeared impossible. Therefore, a simplification was attempted considering sequential time periods where the three COD fractions were not all present simultaneously. However, the complexity of the calculations required to solve the resulting equations was still so high that the employed symbolic software was not able to complete the computation procedure (results not shown, see Carvalho, 2001). Therefore, this approach failed to produce any conclusion about the structure identifiability of the model. This limitation of current symbolic manipulation packages has been reported before (Dochain et al., 1995; Petersen et al., 2002). Another way of analysing the identifiability of a model is to examine the change of the system results following given changes in the parameters (Olsson and Newell, 1999). In this work, such sensitivity analysis was carried out by imposing 50% increases and reductions to each parameter value and running model simulations (Spérandio, 1998). The resulting outputs were compared to the model values obtained with all the parameters at their optimised values and the SSE were calculated. The parameters were estimated by trial and error, aiming at the minimisation of the SSE.

3.1.3. Application to different sets of data

The model was successfully tested with several SBR-cycles of an acclimatisation process to Brij 30, with different food/microorganism ratios (Carvalho et al., 2001). It was then validated with data from a different acclimatisation experiment, carried out with POE 10-LE. Good fits could be achieved in all assays, as illustrated in

Fig. 1. From the analysis of the evolution of the parameter values (Carvalho, 2001), it seemed that the main causes for the acclimatisation effect in the studied cases were the loss of inhibition of the secondary degradation step (increase of K_I values in the calibration to Brij 30 data) and an increase in affinity for the original surfactant molecule (decrease of K_{S_1} values in the calibration to POE 10-LE data). It was also observed that the parameters to which the model was most sensitive were K_{S_1} and b_H, and also K_I in the assays performed with POE 10-LE.

3.1.4. Validation of the model with TOC data

In parallel with the NIO titrimetric and respirometric data used in the development of the model, TOC data were collected during the experimental assays. Since they were not employed in the calibration process, these TOC data could be used to validate the model by confronting its predictions with the experimental values. The theoretical total COD was calculated from the sum of S_1, S_2 and S_3 values given by the model at each time instant. In order to be able to compare the theoretical COD with the measured TOC, the latter had to be converted into COD units. A range of probable successive metabolites of Brij 30 and POE 10-LE was deduced, considering initial central ether bond fission. A preferential ω-oxidation of the hydrophobic chain metabolite (fatty alcohol) was assumed, followed by degradation of the ethoxylated chain when only two C-atoms remained in the hydrophobic chain (Swisher, 1987). By calculating the theoretical COD and TOC values of these metabolites, average COD/TOC ratios (3.7 ± 0.1 and 3.6 ± 0.1 for Brij 30 and POE 10-LE, respectively) could be determined, which were the factors used to convert the measured TOC into COD units. Previously determined adsorption yields (17% for Brij 30 and 5% for POE 10-LE, Carvalho, 2001) showed that this was not a major surfactant removal process in this study. Furthermore, any eventual surfactant adsorbed to the biomass at the beginning of the SBR-cycle is likely to remain constant along the 24 h-study and not interfere with the obtained removal rates.

The results show that there is a very good agreement between the values predicted by the model and the experimental data, as illustrated by the four assays represented in Fig. 2. The differences observed for the final COD values of each assay could be due to accumulation of soluble microbial products (SMP), that were not taken into account in the model. The average percentage of residual non-degraded TOC at the end of the process was 13%, which is in agreement with the 10% typically attributed to SMP (Chudoba, 1985).

3.2. Modelling acclimatisation

3.2.1. Model development

As discussed above, the model parameter values obtained along the acclimatisation processes seemed to change consistently. The parameter value evolution observed for the experiment performed with POE 10-LE, which was followed for 21 SBR cycles, suggested that an acclimatisation model could be developed by converting the parameters of the surfactant biodegradation model into time varying state variables. Although all parameters varied along the SBR cycle series, two of them (K_{S_1} and K_I) showed a wider and monotonous evolution

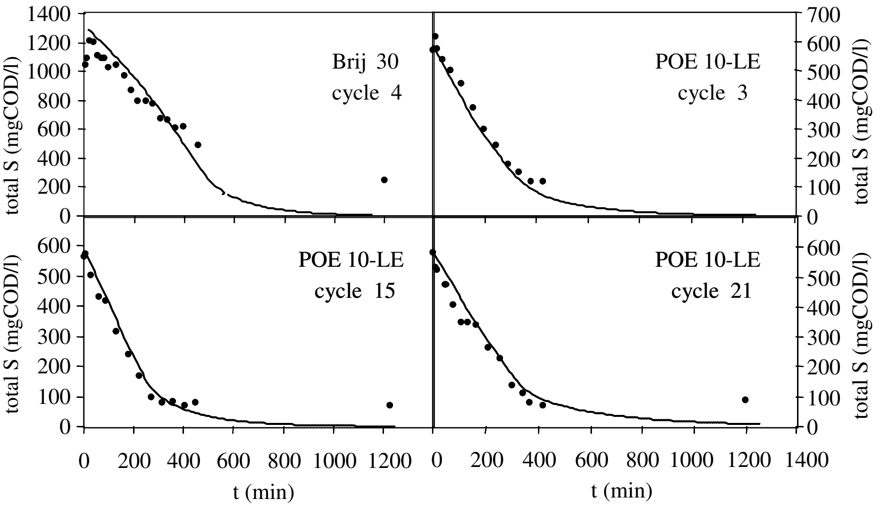

Fig. 2. Variation of the soluble carbonaceous material concentration in assays of the acclimatisation process to Brij 30 (cycle 4) and to POE 10-LE (cycles 3, 15 and 21). Model predictions (—) and measured TOC values converted to COD (●) (see text).

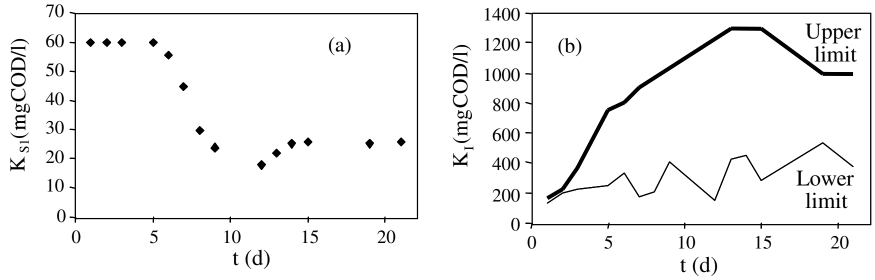

Fig. 3. (a) Variation of the model parameter K_{S_1} along the 21 days of the acclimatisation study to POE 10-LE. (b) Range of $K_{\rm I}$ values corresponding to the minimum weighted SSE values for the cycles of the acclimatisation experiment to POE 10-LE.

(Fig. 3), and were therefore selected for this model transformation. All the other parameters were considered as having constant values in the acclimatisation process, set as the averages resulting from calibration of the biodegradation model. The proposed model for K_{S_1} is given by the following equations:

$$\frac{dK_{S_1,\rm acc}}{dt} = a \cdot \frac{S_1}{b + S_1} \cdot (K_{S_1,\rm acc,max} - K_{S_1,\rm acc}) \cdot K_{S_1,\rm acc} \qquad (11)$$

$$K_{S_1} = K_{S_{1,0}} - K_{S_1,\rm acc} \qquad (12)$$

$K_{\rm I}$ is one of the parameters which is affected by the acclimatisation process, as could be observed from the results obtained with Brij 30 (Carvalho et al., 2001). However, the low sensitivity of the model to $K_{\rm I}$ observed in the model calibrations with POE 10-LE data made it difficult to find the optimal values for this parameter. A range of acceptable $K_{\rm I}$ values around the minimum weighted SSE (Fig. 3) could be set by trial and error for these assays. Due to this identification problem, it was necessary to use auxiliary acclimatisation indicators and their evolutions as clues for model development. These indicators were k_1 and k_2, two apparent first-order kinetic constants introduced in a preliminary quantification of the acclimatisation processes (Carvalho et al., 2000; Carvalho et al., 2003). These were calculated from the linear slope resulting from the plot against time of the logarithm of NIO (k_1) and TOC (k_2) data obtained in each SBR cycle. Therefore, k_1 can be taken to express the decrease in surfactant properties (fraction S_1) or primary degradation, whereas k_2 is associated to the secondary or ultimate degradation (all fractions). Thus, the two model parameters selected to translate acclimatisation (K_{S_1} and $K_{\rm I}$) can be related to k_1 and k_2, respectively, and the corresponding value evolutions can be compared. In Carvalho (2001), a distinct similarity was shown between the K_{S_1} model and a model which adequately describes k_1 evolution, although inverted. Similarly, the model for $K_{\rm I}$ evolution with acclimatisation could be inspired by the model structure set for k_2. Both acclimatisation indicators were also helpful in the calibration of the $K_{\rm I}$ model. A unique set of $K_{\rm I}$ values could be found which predicted the time required for acclimatisation which was in agreement with the other indicating parameters (k_1, k_2 and K_{S_1}). This time value was determined on the selected criterion for acclimatisation, which was each parameter reaching a value differing not more than 10% from the new pseudo steady-state value. The proposed $K_{\rm I}$ model is given by Eqs. (13) and (14).

$$\frac{dK_{\rm I,acc}}{dt} = c \cdot \frac{S_2}{d + S_2} \cdot (K_{\rm I,acc,max} - K_{\rm I,acc}) \qquad (13)$$

$$K_{\rm I} = K_{\rm I,0} + K_{\rm I,acc} \qquad (14)$$

Thus, the acclimatisation model is composed by Eqs. (1)–(9), with K_{S_1} and $K_{\rm I}$ being given by Eqs. (11)–(14) and the other parameters set as constants and equal to the average of their values obtained along the acclimatisation process.

3.2.2. Application example of the acclimatisation model for optimal operation in transient state

Overall, the work presented in this contribution was performed within the context of the biotreatment of textile industry wastewater, which is characterised by a wide variability in flow and composition. In previous studies, a temporary efficiency loss was observed in the activated sludge SBR due to lack of acclimatisation when, for example, a surfactant product was replaced by another (Carvalho et al., 2000). In a conventional WWTP, this problem can be addressed by designing the reactor capacity to cover for these low efficiency episodes. However, the metabolic changes or specific bacterial growth required for an effective acclimatisation process could be enhanced by feeding the bioreactor with small amounts of the new surfactant soon to be discharged in the wastewater, before the surfactant replacement actually takes places in the textile process. This procedure is expected to induce sludge pre-acclimatisation. By making use of the acclimatisation model presented above, it was possible to schedule and quantify the additions of the new surfactant (POE 10-LE, in this simulation) to the fed wastewater, so as to pre-acclimatise the bioreactor and ensure no loss of performance upon its introduction in the textile process. The best pre-acclimatisation scheme was found by trial and error, testing various day-to-day combinations of added POE 10-LE concentrations so as to induce sludge acclimatisation with simultaneous control of biomass growth (Table 1). From the several tested sequences, procedure 7 was selected as the one which best balanced the treated effluent quality and the surfactant requirements, avoiding excessive additional sludge production.

An economic analysis was carried out comparing the scenario of a WWTP designed and operated according to this pre-acclimatisation procedure with other scenarios where pre-acclimatisation was not considered in system design, in its operation or in both. This was possible by running simulations of the acclimatisation model in order to calculate, for each scenario, the sludge degradation capacity, the amount of surfactant needed for pre-acclimatisation, the sludge production, the pollution discharged, the energy requirements and the investment costs for tank volume and installed aeration capacity. This economic comparison, described in more detail elsewhere (Carvalho, 2001, 2002), led to the conclusion that the pre-acclimatisation strategy resulted in higher operation costs and lower investment costs. Moreover, by running simulations of the model, it could be concluded that pre-acclimatisation could be an overall cost-saving option when higher surfactant loads are expected and if a cheap product is employed for the pre-acclimatisation period. Indeed, a 5-fold increase in the organic load content of the textile wastewater would result in 20-fold higher fines due to bad quality effluent for a not prepared system such as scenario B (Table 2). This fact would lead to a 19% rise of the total costs in comparison to the base case. Furthermore, it was shown that using a 33% cheaper surfactant-containing product for pre-acclimatisation (Table 2) would make this methodology advantageous as compared to the conventional WWTP.

Table 1
Simulated values of surfactant required for pre-acclimatisation, pollution discharged in the treated effluent and excess sludge produced per acclimatisation period, and time required for acclimatisation of each simulated surfactant feeding sequence for pre-acclimatisation

Procedure no.	Organic load sequence [surfactant load (kgCOD/kgTSS d)]	Surfactant for pre-acclimatisation (kgCOD/m^3)	Pollution discharged (kgCOD/m^3)	Excess sludge (kgCOD/m^3)	Time required for acclim (days)
1	0.25 (6 days) + 0.4 (6 days) + 0.8 (23 days)	22.3	0.013	37.9	35
2	5 times [0.3 (2 days) + 0.6 (2 days)] + 3 times [0.8 (2 days) + 0.3 (2 days)] + 0.8 (1 day)	16.4	0.016	31.0	33
3	0.2 (1 day) + 3 times [0.2 (1 day) + 0.5 (1 day)] + 6 times [0.2 (1 day) + 0.6 (1 day)] + 5 times [0.2 (1 day) + 0.7 (1 day)] + 4 times [0.2 (1 day) + 0.8 (1 day)]	15.6	0.017	33.5	37
4	0.2 (1 day) + 0.3 (1 day) + 10 times [0.6 (2 days) + 0.2 (2 days)] + 0.6 (3 days)	18.3	0.02	40.2	45
5	0.2 (1 day) + 0.3 (1 day) + 7 times [0.6 (2 days) + 0.2 (2 days)] + 2 times [0.7 (2 days) + 0.2 (2 days)] + 0.7 (2 days)	16.7	0.018	36.1	40
6	6 times [0.5 (1 day) + 0.05 (1 day)] + 4 times [0.2 (2 days) + 0.6 (2 days)] + 3 times [0.2 (2 days) + 0.7 (2 days)]	15.1	0.018	36.1	40
7	10 times [0.6 (1 day) + 0.05 (3 days)] + 0.6 (1 day)	8.1	0.033	29.5	41

The total values are given per m^3 of SBR tank working volume. On column 2, surfactant load values, followed by the number of consecutive days during which each value was applied, are given for each simulated acclimatisation sequence.

Table 2
Cost analysis of different scenarios considering the application of the pre-acclimatisation methodology at design stage (scenarios A and B) and at operation stage (scenarios A and C), compared to a conventionally designed (scenarios C and D) and/or conventionally operated (scenario B and D) WWTP

Case	Scenario			
	A	B	C	D
Base case	100	86	106	91
5-fold load increase	102	119	106	93
66% reduction in surfactant price	90	86	96	91

The relative total cost values are presented, with respect to the base case of scenario A (the value 100 corresponds to 48630 Euro/year), for the base case of the four scenarios, for the case of a 5-fold load increase and for a situation considering a 66% reduction in the costs of surfactant for pre-acclimatisation.

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