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Predicting wastewater treatment plant performance during aeration demand shifting with a dual-layer reaction settling model

Matteo Giberti, Recep Kaan Dereli, Damian Flynn and Eoin Casey

ABSTRACT

Demand response (DR) programmes encourage energy end users to adjust their consumption according to energy availability and price. Municipal wastewater treatment plants are suitable candidates for the application of such programmes. Demand shedding through aeration control, subject to maintaining the plant operational limits, could have a large impact on the plant DR potential. Decreasing the aeration intensity may promote the settling of the particulate components present in the reactor mixed liquor. The scope of this study is thus to develop a mathematical model to describe this phenomenon. For this purpose, Benchmark Simulation Model No.1 was extended by implementing a dual-layer settling model in one of the aerated tanks and combining it with biochemical reaction kinetic equations. The performance of this extended model was assessed in both steady-state and dynamic conditions, switching the aeration system off for 1 hour during each day of simulation. This model will have applications in the identification of potential benefits and issues related to DR events, as well as in the simulation of the plant operation where aerated tank settling is implemented.

Key words | activated sludge, benchmark simulation model, demand and response, energy, settling, wastewater treatment plant

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ABBREVIATIONS

a	Logistic function parameter [−]
A_{tank}	Aerated tank surface area $\mathrm{m^2}$
b	Logistic function parameter [−]
b_{suct}	Suction depth positive parameter [−]
d_0	Suction depth positive parameter m
d_{sb}	Sludge blanket depth m
d_{suct}	Suction depth m
d_{tot}	Total depth of aerated tank m
f_{ns}	Non-settleable solids fraction [−]
k_La	Oxygen transfer coefficient $\mathrm{d^{-1}}$
m	Mixing parameter [−]
Q	Flowrate in aerated tank $\mathrm{m^3/d}$
Q_0	Suction depth normalisation constant $\mathrm{m^3/d}$
Q_{eff}	Effluent flowrate $\mathrm{m^3/d}$
Q_{in}	Plant influent flowrate $\mathrm{m^3/d}$
$Q_{internal}$	Internal recycle flowrate $\mathrm{m^3/d}$
Q_w	Sludge wastage flowrate $\mathrm{m^3/d}$
R_i	Reaction rate for $\mathrm{i^{th}}$ ASM1 component $\mathrm{g/m^3 \cdot d^{-1}}$
r_h	Hindered zone settling parameter $\mathrm{m^3/g_{SS}}$
r_p	Flocculant zone settling parameter $\mathrm{m^3/g_{SS}}$
S_i	ASM1 $\mathrm{i^{th}}$ soluble component concentration in aerated tank $\mathrm{g/m^3}$
S_i^{in}	ASM1 $\mathrm{i^{th}}$ soluble component concentration entering aerated tank $\mathrm{g/m^3}$
S_o^{sat}	Oxygen saturation concentration $\mathrm{g/m^3}$
τ_{mix}	Mixing time constant d
TSS^{ave}	Average total suspended solids concentration in aerated tank $\mathrm{g/m^3}$
TSS^B	Total suspended solids concentration in sludge layer $\mathrm{g/m^3}$
TSS^{out}	Total suspended solids concentration exiting aerated tank $\mathrm{g/m^3}$
v_0	Maximum Vesilind settling velocity m/d
v_0'	Maximum settling velocity m/d

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V_B	Sludge layer volume m^3
V_{tank}	Aerated tank volume m^3
v_s	Particulate settling velocity m/d
X	Total suspended solid concentration in wastewater g/m^3
X_i^B	ASM1 ith particulate component concentration in sludge layer g/m^3
X_i^{in}	ASM1 ith particulate component concentration entering aerated tank g/m^3
X_i^{out}	ASM1 ith particulate component concentration exiting aerated tank g/m^3
X_{min}	Minimum attainable suspended solids concentration g/m^3

INTRODUCTION

In Europe, the share of electrical energy generated from renewable sources has increased from around 8.5% in 2004 to 17% in 2016. In particular, wind and solar generation are responsible for 31.8% and 11.6% of the total renewable generation (Eurostat 2018), which is associated with a higher dependence of the energy systems on variable and uncertain generation. The operation of the electric grid requires a continuous balance between the production and consumption of energy, so a higher share of wind and solar power poses a challenge to the maintenance of such equilibrium. Addressing this issue requires greater flexibility of the entire energy system, for instance increasing the available energy storage capacity or managing end-user behaviour (e.g. shifting or shedding their energy consumption) (Bird *et al.* 2016). The concept of demand response (DR) strategies lies in this context. DR can be defined as '… the changes in electricity usage by end-users from their normal consumption patterns in response to changes in the price of electricity over time' (Goldman *et al.* 2010). Practically, this means that energy users are encouraged to adjust their consumption according to energy availability, for example scheduling the usage of redundant devices when the energy price is lower.

Municipal wastewater treatment plants (WWTPs) are responsible for 2–3% of the world's electricity consumption (Emami *et al.* 2018). They can be suitable candidates for the application of DR programmes since they are quite energy intensive, and they tend to have high electrical loads during utility peak demand periods (Goli *et al.* 2013). WWTPs are designed to cope with peak influent concentrations and loads which can be significantly higher than the average values. Another aspect to be considered is that some unit processes in a WWTP (e.g. sludge dewatering) can be operated discontinuously, contributing to the potential for flexible operation. To exploit this potential, adjusting the plant demand (e.g. reducing it when renewable energy is less available) can decrease the costs for energy generation, as less fossil fuel would be required to meet the grid demand. Moreover, in a scenario where dynamic energy tariffs are applied, lower plant consumption during peak demand periods results in lower operating costs.

Conventional activated sludge is the most commonly applied process in WWTPs (Gernaey *et al.* 2004b). In this type of plant, aeration is generally the largest single energy consumer (45–75% of the total consumption) (Rosso *et al.* 2008). Demand shedding through aeration control, subject to maintaining the plant operational limits, could therefore have a large impact on the DR potential of the plant (Aymerich *et al.* 2015). Examples of demand shedding through aeration control applied to full-scale WWTPs have been already reported in the literature. More specifically, studies were conducted on three different plants in California. The potential for the curtailment of 78 kW (6% of the total plant consumption) from the blowers load during peak periods was found in a WWTP with a daily average flow of 36,000 m^3/d (Thompson *et al.* 2010). Another study on a 110,000 m^3/d plant demonstrated a DR potential associated with the blowers of between 33 and 45 kW (12–16% of the total demand), whereas 132 kW (3% of the overall plant consumption) was shown to be available from the aeration trains and mixers in a 320,000 m^3/d WWTP (Aghajanzadeh *et al.* 2015). Similar studies were also conducted in Germany (Schäfer *et al.* 2015). For instance, Schäfer *et al.* (2017) report that it was possible to shut down aeration for 1 hour without significant deterioration of effluent quality in a 58,000 population equivalent (PE) WWTP, with a 150 kW consumption reduction.

However, reduced aeration intensity may have negative effects on the treatment performance and on the plant's effluent quality. In many cases (Åmand *et al.* 2013; Aghajanzadeh *et al.* 2015), aeration is responsible for the mixing of the tanks, so that decreasing the aeration intensity for a while not only affects the dissolved oxygen (DO) concentration but also promotes settling of the biomass flocs and particulate matter present in the reactor mixed liquor. Once aeration is restarted, the settled solids are resuspended, and their concentration in the stream leaving the activated sludge tank is increased. This may exceed the solids loading capacity of the secondary clarifier, leading to sludge washout and to an increase in the effluent turbidity, as occasionally reported in the literature (Thompson *et al.* 2010).

A mathematical model capable of simulating this situation is a valuable tool that could provide useful information to design a DR strategy and evaluate its effectiveness without endangering the operation of existing plants. The Benchmark Simulation Model 1 (BSM1) is one example of WWTP dynamic modelling (Gernaey *et al.* 2014; Saagi *et al.* 2017). It was originally developed as a tool to assess the performance of different control strategies on a generic conventional activated sludge process (Figure S2, Supplementary Material), and since its introduction it has been widely used by different research groups for a number of purposes (Jeppsson *et al.* 2013), such as avoiding effluent violations (Corriou & Pons 2004), finding a trade-off between effluent violations and operational costs (Santín *et al.* 2016) or extending the original model (Daelman *et al.* 2014).

Activated sludge settling models for the final clarifiers, based on discretization of the settling tanks into layers, are well established in the literature (Takács *et al.* 1991). There are also several approaches to combine sludge settling and biological processes in secondary clarifier tanks (Gernaey *et al.* 2006). To include the biological reactions into a clarifier model, one option is to modify a multi-layer settling model (e.g. Takács *et al.* 1991) into a series of continuous stirred-tank reactor (CSTR) bioreactors, implementing a kinetic model in each (Gernaey *et al.* 2006). However, the use of the dual-layer aeration tank settling model proposed by Bechmann *et al.* (2002) can be a suitable approach to limiting the number of additional equations required to describe the reactions and settling phenomena when aeration and mixing are switched off. This model was originally used to simulate the behaviour of a WWTP when the aeration tank settling operational strategy was applied. With this control approach, the activated sludge is allowed to settle in the aeration tanks when the influent flowrate exceeds a certain threshold (e.g. during rainfall events) (Gernaey *et al.* 2004a). This results in a reduction of the suspended solids load to the secondary clarifiers, which determines an increase in their hydraulic capacity (Sharma *et al.* 2013).

The focus of this study is to develop a mathematical model that is capable of representing activated sludge settling when the aeration system is temporarily shut down in DR operation. Scenarios in which the aeration is modulated are not currently considered. Although it is possible to manipulate the aeration intensity in WWTPs equipped with a dedicated control system, to simply switch off the blowers can be more easily implemented in plants that are lacking that level of automation. The model developed has the potential to bridge the knowledge gap between DR strategies and their impact on biochemical and physical processes in WWTPs. For this purpose, BSM1 was extended by implementing a dual-layer settling model to activated sludge tanks. It uses Activated Sludge Model No. 1 (ASM1; Henze *et al.* 2006) to describe the biological phenomena that take place in the plant. To provide a sufficiently good representation of biomass settling, the general mass balance equations based on ASM1 kinetics used in BSM1 were modified. Although the combination of aeration management and other flexibility measures is possible, it was deemed to be beyond the scope of the present paper. Validation and calibration are, however, still required to assess the quality of the model prediction.

MATERIALS AND METHODS

Implementation of the settling model in BSM1

This study implemented the ASM1 kinetic equations into the dual-layer settling model proposed by Bechmann *et al.* (2002). A schematic of this approach is shown in Figure 1. The layer above the sludge blanket level is assumed to be a clear water (no suspended solids) zone, whereas the layer at the bottom of the tank contains all the suspended solids.

The sludge blanket depth d_{sb} is evaluated from Equation (1), where the settling is only considered when the aeration is off using the m parameter, which can be either 1 (aeration ON) or 0 (aeration OFF). Non-binary aeration states were not considered.

$$\frac{d(d_{sb})}{dt} = m\left(-\frac{1}{\tau_{mix}}d_{sb}\right) + (1-m)v_s \tag{1}$$

Two contributions are present in Equation (1): when aeration is active, only the first term is considered, whereas the second term is non-zero only when aeration is off. While aeration is in operation, the sludge blanket depth

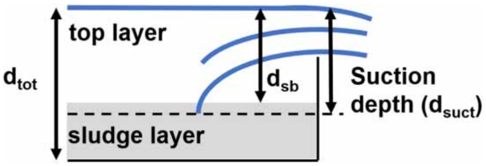

Figure 1 | Dual-layer model of settling in an aeration tank.

approaches zero at a rate that is defined by the mixing constant τ_{mix}. When aeration is switched off, d_{sb} starts to increase at a rate defined by the settling velocity v_s.

The last step of this approach involves the calculation of the suction depth d_{suct} in order to evaluate the total suspended solids (TSS) concentration in the stream that is leaving the tank. In principle, the suction depth value would be a function of the tank geometry and the hydraulic conditions of the system. However, a less complex model based on an empirical relationship was used (Equation (2)), with d_0, Q_0 and b_{suct} being positive parameters.

$$d_{suct} = d_0 \cdot \left(\frac{Q}{Q_0}\right)^{b_{suct}} \tag{2}$$

The TSS concentration leaving the tank is then a function of the suction depth and of the sludge blanket depth.

$$TSS^{out} = \begin{cases} \dfrac{d_{suct} - d_{sb}}{d_{suct}} \cdot TSS^B, & d_{suct} \geq d_{sb} \\ 0, & d_{suct} < d_{sb} \end{cases} \tag{3}$$

Finally, a logistic function l was used to smooth the changes in TSS^{out} when $d_{suct} = d_{sb}$.

$$l(d_{suct} - d_{sb}) = \frac{1}{1 + e^{a - \frac{(d_{suct} - d_{sb})}{b}}} \tag{4}$$

In the present study, some modifications were made to the settling model (Bechmann *et al.* 2002) described above (Equations (1)–(4)) in order to combine it with the ASM1 kinetic equations set within the BSM1 framework. First of all, the sludge settling velocity was modelled using the same relationship used for the BSM1 secondary clarifier (Takács *et al.* 1991), to maintain consistency.

$$v_s = \max\left(0,\ \min\left(v'_0,\ v_0 \cdot \left(e^{-r_h(X - X_{min})} - e^{-r_p(X - X_{min})}\right)\right)\right) \tag{5}$$

The parameter values used in Equation (5) are obtained from the default BSM1, and they are shown in Table 1; X is the TSS concentration in the wastewater, whereas X_{min} is the minimum attainable suspended solids concentration.

Since the sludge blanket depth changes over time, the volumes of the two layers are not constant. This has some implications for the mass balances of the ASM1 components and for their reaction rates. In the original BSM1, the mass balances are written in the form:

Table 1 | Dual-layer settling parameters

Parameter	**Units**	**Value**
v'_0	m/d	250
v_0	m/d	474
r_h	m^3/g_{SS}	0.000576
r_p	m^3/g_{SS}	0.00286
Q_0	m^3/d	24000
d_0	m	1.005
b_{suct}	–	0.164
τ_{mix}	d	0.0210
a	–	0
b	–	0.1
Threshold	m	0.01

$$\frac{dX_i}{dt} = \frac{Q}{V_{tank}}(X_i^{in} - X_i) + R_i \tag{6}$$

where X_i, is the ASM1 ith particulate component concentration in liquid phase (g/m^3).

However, this expression is derived from the general form of the mass balances and is only valid if the volume of the tank does not change. Furthermore, the bacteria are now assumed to be present exclusively in the bottom layer of the tank, which consequently is considered to be the only reactive volume in the system. For these reasons, the mass balance equations are modified as follows.

$$\frac{d(X_i^B \cdot V_B)}{dt} = Q \cdot X_i^{in} - Q \cdot X_i^{out} + R_i \cdot V_B \tag{7}$$

Equation (7) describes the mass conservation of the particulate state variables in the bottom layer. Both the concentration (X_i^B) and the volume of the sludge layer V_B are variables; therefore the product rule is applied to the derivative.

$$\frac{d(X_i^B \cdot V_B)}{dt} = \frac{dX_i^B}{dt} \cdot V_B + \frac{dV_B}{dt} \cdot X_i^B \tag{8}$$

Since the sludge layer volume can also be expressed as the product of the aerated tank surface area A_{tank} (333.25 m^2) and the sludge layer depth, $V_B = A_{tank} \cdot (d_{tot} - d_{sb})$. Hence, its time derivative can be calculated as

$$\frac{dV_B}{dt} = \frac{d(A_{tank} \cdot (d_{tot} - d_{sb}))}{dt} = -A_{tank} \cdot \frac{d(d_{sb})}{dt} \tag{9}$$

The mass balance of particulate components can thus be written in the form

$$\frac{dX_i^B}{dt} = \left(Q\cdot(X_i^{in} - X_i^{out}) + R_i\cdot V^B + A\cdot X_i^B\cdot\frac{d(d_{sb})}{dt}\right)\cdot\frac{1}{V_B} \qquad (10)$$

As the soluble components do not settle, their concentration is assumed to have the same value in both layers of the system. Hence, the soluble components mass balances have the form:

$$\frac{dS_i}{dt} = \frac{Q}{V_{tot}}\cdot(S_i^{in} - S_i) + R_i\cdot\frac{V^B}{V_{tot}} \qquad (11)$$

Concerning the DO concentration, in addition to the terms relating to wastewater flow through the tank and to oxygen consumed by the microorganisms, a term involving the mass transfer from the gas to the liquid phase must be considered to account for the aeration (k_La is the oxygen transfer coefficient and S_o^{sat} is the oxygen saturation concentration).

$$\frac{dS_{oxygen}}{dt} = \frac{Q}{V_{tot}}\cdot(S_o^{in} - S_o) + k_La\cdot(S_o^{sat} - S_o) + R_o\cdot\frac{V^B}{V_{tot}} \qquad (12)$$

where R_o is the oxygen reaction rate (g/m^3 *d^{-1}).

From the particulate components' concentration it is possible to calculate the TSS concentration in the bottom layer (TSS^B), as well as the ratios $\alpha_i = X_i/TSS^B$ and the average TSS concentration in the whole tank. If the sludge blanket depth is lower than a threshold value, the tank is assumed to be fully mixed and the TSS concentration leaving the tank is equal to the average TSS concentration in the tank. When d_{sb} increases above the threshold value, the TSS concentration leaving the tank is evaluated as a function of TSS^B and the suction depth.

$$\begin{cases} TSS^{out} = TSS^{ave}, & d_{sb} < threshold\ value \\ TSS^{out} = l(d_{suct} - d_{sb})\cdot\dfrac{d_{suct} - d_{sb}}{d_{suct}}\cdot TSS^B, & d_{sb} \geq threshold\ value \end{cases} \qquad (13)$$

Finally, α_i ratios between the concentration of the i^{th} particulate component in the sludge layer and TSS^B are assumed to be maintained within the stream that is leaving the tank. Therefore, the concentration of the particulate components is calculated from

$$X_i^{out} = \alpha_i\cdot TSS^{out} \qquad (14)$$

This set of equations (Figure S1, Supplementary Material) was then included in the final aerated tank (fifth tank) of the BSM1 plant layout (Figure S2). The model was also modified so that aeration could be periodically switched off. This modified BSM1 model, which combines the effects of particulate settling with the reaction kinetics, was implemented in MATLAB Simulink, and it will hereafter be referred to as BSM1_RS. The parameters involved in the calculation of suction depth and TSS^{out} are presented in Table 1.

Scenario analysis

The combination of dual-layer settling and biochemical reactions was implemented in the last aerated tank of BSM1. The model's behaviour was assessed using the constant influent file provided by the BSM1 framework and under more dynamic conditions, with dry weather, rainfall event and storm event files as inputs for the WWTP. Although a constant influent does not represent a realistic scenario for WWTP operation, it was initially used to isolate variations in the TSS dynamics associated with particulate settling and the biological reactions in the aerated tank. For the same reason, the first simulations were carried out with the BSM1 open loop configuration, using a fixed default value for the oxygen transfer coefficient in the last aerated tank ($k_La = 84\ \mathrm{d}^{-1}$) and internal recycle flowrate ($Q_{internal} = 55{,}338\ \mathrm{m}^3/\mathrm{d}$). The 14-day constant influent file was used for these simulations, turning the aeration in the last aerated tank off for 1 hour each day. A similar DR programme was then applied to dynamic influent conditions, turning the blower off between 5pm and 6pm to test the behaviour of the BSM1_RS in a more realistic scenario. This particular time slot was chosen as an example of the daily peak electricity demand timeframe. The implementation of DR strategies during peak demand periods can produce the highest savings in the plant operational costs if variable energy tariffs are applied. The DO concentration in the fifth tank and the nitrate concentration in the second tank were controlled according to the closed loop configuration of BSM1. Of course, the DR duration can be extended (or reduced) depending on the plant operation and on the equipment characteristics. However, since the focus of this study is on the presentation of a combined model for settling and reaction, 1 hour was arbitrarily chosen for the DR duration.

RESULTS AND DISCUSSION

Constant influent and open loop configuration

The results of a simulation with constant influent in which aeration was switched off for 1 hour are shown in Figure 2.

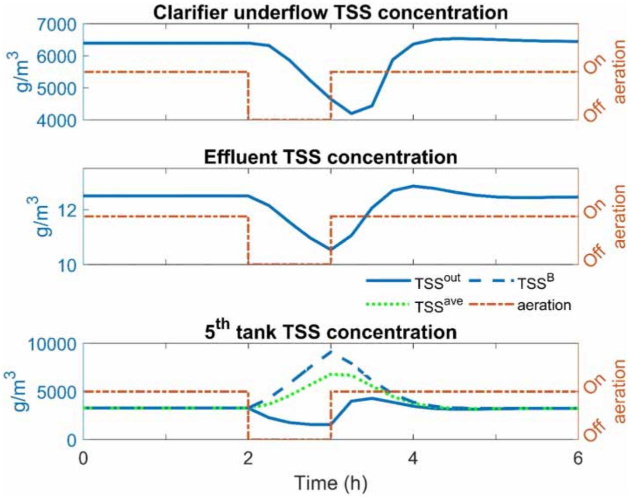

Figure 2 | Total suspended solids profiles in the plant.

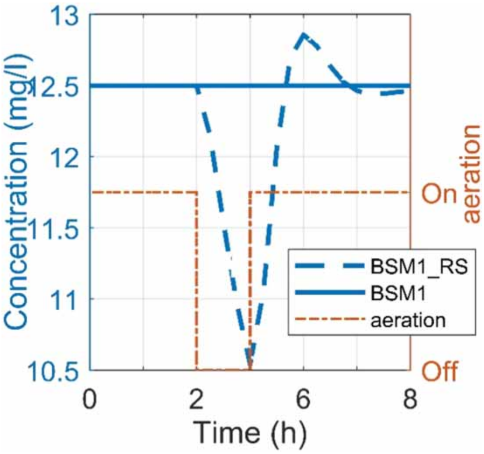

Figure 3 | Effluent total suspended solids concentration profiles. Comparison between BSM1 and BSM_RS models.

When the blower is switched off, the sludge blanket depth in the fifth tank starts to increase due to particulate settling. As greater quantities of solids are retained in the system, the sludge concentration in the bottom layer and the average sludge concentration in the whole tank further increase (up to 9,100 and 6,790 g/m^3, respectively), whereas the TSS concentration in the stream leaving the tank decreases to 1,565 g/m^3. This also results in a decrease in both the clarifier underflow and the plant effluent TSS concentration (from 6,395 to 4,200 g/m^3 and from 12.5 to 10.5 g/m^3, respectively). When aeration is restarted, the trends are reversed. It is also noteworthy that, as expected, the solids concentration leaving the aerated tank reaches a higher value (4,290 g/m^3) than the steady state condition (3,270 g/m^3). If that peak is high enough to overload the secondary clarifier, turbidity issues in the effluent such as those observed in Thompson *et al.* (2010) may occur.

The results obtained from BSM1_RS were also compared with the simulation output of the original BSM1. The profile of the TSS concentration in the effluent of the WWTP is shown in Figure 3. As expected, switching off the blower does not produce any effect on the effluent TSS concentration calculated with the original BSM1. By contrast, when settling of the particulate components is considered in BSM1_RS, the effluent TSS concentration shows a minimum during the DR event, and a peak when the aeration is switched back on and the solids are resuspended.

Figure 4 illustrates the effluent concentration profiles related to total nitrogen and ammonia. Substantial differences between the original BSM1 and BSM1_RS can be observed. In particular, the total nitrogen concentration resulting from the combination of reaction and settling tends to be lower than that predicted by the original BSM1. In the BSM1 plant configuration, the mixed liquor from the fifth reactor is recirculated to the first one to denitrify the nitrate produced in the aerobic zone. A high DO concentration in the fifth tank can result in oxygen intrusion into the anoxic tanks, which has a detrimental effect on the denitrification performance of the plant. During the DR event, the oxygen dissolved in the fifth tank is rapidly consumed, thus limiting the intrusion of oxygen recycled back to the anoxic zone. Additionally, oxygen depletion in the fifth tank promotes denitrification, which further enhances the nitrate removal performance of the plant. Hence, the effluent total nitrogen concentration following the switching

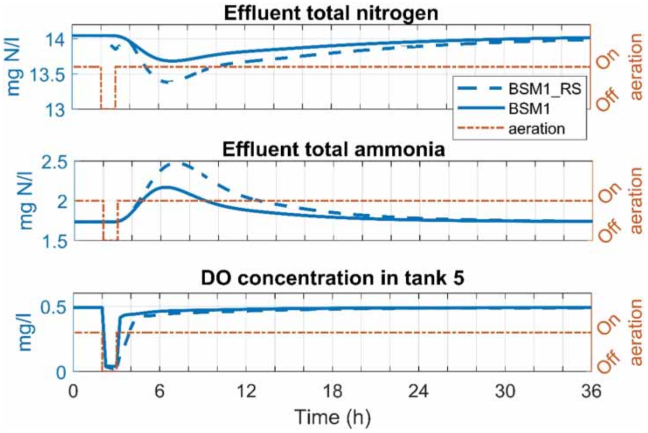

Figure 4 | Total nitrogen and ammonia effluent concentrations, and dissolved oxygen concentration in the fifth tank profiles. Comparison between the BSM1 and BSM1_RS models.

off of aeration is lower than the steady state value. Moreover, including the settling means that an increased amount of biomass is retained in the tank, which acts as a post anoxic zone when the aeration is switched off. Therefore, it is not unexpected that a lower effluent total nitrogen concentration was observed in BSM1_RS compared to the original BSM1. By contrast, turning the blowers off also results in the loss of one third of the plant's aerobic volume, with an associated disruption of the nitrification performance and a higher effluent total ammonia concentration. Compared to the original BSM1, BSM1_RS predicts a higher ammonia concentration peak in the effluent, which could potentially pose a challenge for maintaining this parameter below the discharge limitations. Neglecting the settling can then lead to underestimating the impact of DR on the effluent quality. The combination of reaction and settling also affects the DO dynamics. The DO profiles overlap until aeration is turned back on. At this time, the increased bacteria concentration in the tank leads to a higher oxygen consumption, and this results in a longer time required for the DO concentration to return to the pre-DR event value.

Parameter sensitivity analysis

As discussed in the 'Implementation of the settling model in BSM1' section, the particulate settling is described through several parameters (Bechmann *et al.* 2002). To obtain a better insight into their effect on the simulation, a sensitivity analysis was performed assessing the impact of ±10% changes in the values of Q_0, b_{suct}, d_0 (Equation (3)) and τ_{mix} (Equation (1)) over some critical variables (the maximum sludge blanket depth, the maximum TSS concentration leaving the aerated tank and the maximum TSS concentration in the effluent).

The largest impact on the observed variables is associated with changes in d_0. In more detail, a 10% increase in d_0 results in a 5% increase in the maximum sludge blanket depth, whereas TSS^{out} decreases by roughly 4%. Compared to d_0, Q_0 shows a smaller influence on the maximum sludge blanket depth and on the maximum TSS concentration leaving the aerated tank, its impact being ±1%. Finally, the influence of τ_{mix} and b_{suct} on all the observed variables appears to be negligible (Figure S3, Supplementary Material).

The TSS concentration in the effluent is barely affected by any change in the four parameters studied. This can be an indication that the BSM1 secondary clarifier is capable of buffering the perturbations on TSS^{out} generated by parameter changes. To further investigate this phenomenon, the secondary clarifier overflow rate and solids loading were evaluated for each of the influent files provided by BSM1. The results are shown in Table 2. Comparing them with typical design values ($16-28\ \mathrm{m}^3/(\mathrm{m}^2 \cdot \mathrm{d})$ for the average overflow rate, and $4-6\ \mathrm{kg}/(\mathrm{m}^2 \cdot \mathrm{h})$ for the average solids loading), Tchobanoglous *et al.* (2003) indicate that the BSM1 secondary clarifier is utilised below its capacity. Thus, it can buffer the effects of the parameter variations.

Dynamic influent and closed loop configuration

BSM1_RS was also tested under dynamic conditions. Specifically, simulations were performed using the dry weather (Figure S4, Supplementary Material), rainfall event and storm event influent files in closed loop mode. The same DR strategy used in the constant influent simulations was applied, turning off the aeration for 1 hour every day at 5pm.

A summary of the results related to each weather condition can be found in Table 3. As with BSM1, the results are averaged over the last 7 days of a 2-week simulation, and are described in terms of effluent quality index (EQI) and daily average aeration energy consumption. However, under the current operating conditions, the percentage variations between BSM1 and BSM1_RS are quite small and they may not be significant. BSM1_RS produces an effluent with a slightly better EQI regardless of the weather file used, and it is also noteworthy that the calculated aeration energy consumption is higher compared to the original BSM1. The BSM1 aeration energy consumption is a function of the oxygen transfer coefficient k_La. As previously discussed, the microorganism concentration in the fifth tank increases

Table 2 | Secondary clarifier overflow rate and solids loading

	Overflow rate [$\mathrm{m}^3/(\mathrm{m}^2 \cdot \mathrm{d})$]			Solids loading [$\mathrm{kg}/(\mathrm{m}^2 \cdot \mathrm{h})$]		
	Minimum	**Average**	**Maximum**	**Minimum**	**Average**	**Maximum**
Dry weather	6.67	12.30	21.45	0.13	3.31	5.44
Rainfall events	6.67	16.12	34.75	0.00	3.44	6.59
Storm events	6.67	14.03	40.00	0.24	3.48	7.01

Table 3 | Dynamic simulations results summary

	Parameter	Units	BSM1	BSM1_RS
Dry weather	EQI	kg poll. units/d	6,118	6,104
	Daily average aeration energy	kWh/d	3,675	3,702
Rainfall event	EQI	kg poll. units/d	8,194	8,151
	Daily average aeration energy	kWh/d	3,646	3,675
Storm event	EQI	kg poll. units/d	7,222	7,180
	Daily average aeration energy	kWh/d	3,696	3,722

if their settling is taken into account. This results in a higher oxygen consumption when the aeration is switched on again, which forces the DO control loop to increase the aeration intensity in the tank, leading to the higher energy consumption observed. Further evaluations are then required to determine whether there is a trade-off between the benefits of DR and the consequent increased energy consumption.

It is important to consider that discontinuous operation of the aeration system may also be associated with other effects that can have a negative impact on the plant performance. For instance, increased microorganism concentration in the tank during the settling periods may promote biofilm growth on the diffusers, increasing fouling over the long term and leading to more frequent cleaning (Garrido-Baserba *et al.* 2018). High suspended solids concentration can also impair the oxygen transfer efficiency (Henkel *et al.* 2011), which can increase the amount of energy required when aeration is restarted. Issues related to tank geometry and hydraulics can also become relevant, as the diffuser layout may not guarantee complete resuspension of the settled solids. The formation of persistent anoxic zones in the aerated tank may also promote the growth of filamentous bacteria, which can impair the sludge settling characteristics (Tchobanoglous *et al.* 2003; Rosso *et al.* 2008). However, these factors were outside the scope of the present study, which is primarily designed to describe a structured way of accounting for the effects of aeration shut-down periods on the biochemical reaction kinetics.

The extended version of BSM1 with aeration tank settling presented in this study (BSM1_RS) can be part of a decision-support tool for the application of DR programmes on WWTPs, offering more insight into the effects of particulate settling on the pollutant removal performances. The model can be used to identify potential benefits and issues related to DR events (e.g. effluent turbidity; Thompson *et al.* 2010) without jeopardising plant performance. Other potential applications may be found in WWTPs where aerated tank settling is already implemented for the management of high hydraulic loading periods (Nielsen *et al.* 2000; Gernaey *et al.* 2004a), or where the DO concentration is controlled using intermittent aeration (Sánchez *et al.* 2018).

CONCLUSIONS

Mathematical modelling of DR programmes applied to the aeration system of conventional activated sludge WWTP can help to assess their effects on the effluent quality without jeopardising the operation of a real plant. However, phenomena associated with turning off the blowers, such as particulate settling and its interaction with the biochemical reactions, are not taken into account in the currently available models. The present paper addresses this issue, combining a dual-layer settling model and the ASM1 kinetic model within the BSM1 framework. The modified BSM1 (BSM1_RS) behaviour was tested through simulations under different load conditions. The results obtained show realistic trends for the biological processes kinetics as well as for the TSS concentration in the aeration tank, in the secondary clarifier and in the effluent. Simulations that neglect particulate settling can underestimate DR impact on the effluent quality, predicting lower ammonia concentrations. A measurement campaign is, however, still required to calibrate and validate the model against real data. For this purpose, sensors for the sludge blanket depth and the TSS concentration in the stream that leaves the aerated tank would be necessary, together with the possibility of operating the aeration system intermittently. The use of this approach to combine reaction and settling in BSM1 may extend its capabilities, so that it can provide improved understanding of the impact of various DR programmes on WWTP operation.

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SUPPLEMENTARY MATERIAL

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