Continuous aerobic granular sludge plants: Better settling versus diffusion limitation

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ABSTRACT

The application of aerobic granular sludge in continuous wastewater treatment plants is receiving increased attention. The introduction of better settling sludge in existing installations is expected to increase the plant’s treatment capacity by allowing a higher biomass concentration in the reactor. A lot of recent research has therefore focused on how to get stable granules in a continuous flow plant, which make up the majority of current infrastructure. Still, the effect of aerobic granular sludge on the treatment capacity and energy requirements has not thoroughly been studied so far and is not that straightforward. While it is clear that the introduction of aerobic granular sludge will result in better settling, it will also bring about a higher diffusion limitation – which is often overlooked. This study scrutinized the effect of better settling versus diffusion limitation in a typical continuous activated sludge plant (predenitrification type) through a simulation study. When only considering the improved settling velocity, the treatment capacity increased by 40% compared to conventional activated sludge. However, diffusion limitation could almost totally counteract the positive effect of better settling for a non-improved continuous system. The continuous system could be improved for aerobic granular sludge by increasing the aerobic reactor volume fraction, while lowering the oxygen set-point to benefit from simultaneous nitrification-denitrification. The optimization led to a 20% improvement in treatment capacity and a 10% reduced energy consumption compared to a conventional activated sludge system. Overall, the performance of continuous wastewater treatment plants could indeed benefit from the excellent settling properties of aerobic granular sludge, as long as process operation was improved in order to take into account diffusion limitation.

1. Introduction

The aerobic granular sludge technology is increasingly being applied for wastewater treatment all over the world. Over 80 full-scale applications have proven a significant reduction of surface area, energy consumption and operational costs compared to conventional activated sludge wastewater treatment plants [1–3]. The reason for this success is the growth of bacteria in dense, well-settling granules instead of in dispersed flocs [4–6]. The aerobic granular sludge technology is typically applied in batch configurations [7,8]. At the same time, there is also a growing interest in applying aerobic granular sludge in continuous installations [9,10]. More specifically, this could be an option for existing wastewater treatment plants of which the capacity is no longer sufficient.

Existing wastewater treatment plants are confronted with increasing loading rates due to higher living standards and increasing amount of households connected to sewage systems [11]. Moreover, the effluent standards will become more stringent in the future [12]. Replacement or extension of existing plants is very expensive and sometimes impossible due to lack of space. A bottleneck in increasing the capacity of existing continuous activated sludge plants is the poor settling characteristics of the activated sludge flocs, resulting in large settlers to separate the activated sludge from the treated effluent [13–15]. The introduction of aerobic granular sludge, having excellent settling properties, is expected to solve this problem. For this reason, recent research has focused on the experimental (lab-scale) work cultivation of aerobic granules with long-term stability into continuous systems [9,10]. The specific conditions to ensure granule formation and long-term stabilization are well-known, namely feast/famine conditions, an additional selection pressure to ensure wash-out of flocs (e.g. based on settling velocity) and high shear rates [16–19].

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Still, the beneficial effect of introducing aerobic granular sludge in continuous reactors as such has not yet been demonstrated. It is expected that the higher settling velocity of aerobic granular sludge compared to activated sludge would increase the biomass retention capacity of the reactor because the separation of sludge from the clean effluent would be enhanced [20]. A higher biomass concentration would increase the volumetric removal rates and thus the maximal treatment capacity of the reactor. However, the increased diffusion limitations associated with granular sludge compared to activated sludge are typically neglected [9]. Diffusion limitation could decrease the metabolic activity of the granule [21,22]. Increased diffusion limitation will lead to decreased conversion rates and thus lower the treatment capacity. The latter effect is expected to be more pronounced for the relatively low substrate concentrations typically present in continuous systems compared to batch configurations. Therefore, it is not yet known whether or not continuous aerobic granular sludge plants are more advantageous than continuous activated sludge plants.

This study investigates to which extent the treatment capacity of a continuous wastewater treatment plant can be increased through the introduction of aerobic granular sludge. Assuming that the cultivation of aerobic granules with long-term stability in continuous plants is possible \textit{a priori}, the focus of this study was on the effect of better settling sludge and increased diffusion limitation on wastewater treatment plant capacity and energy consumption. The fundamentals regarding the impact of increased biomass concentrations and diffusion limitation on the volumetric conversion rates were scrutinized first. The individual and combined effects of better settling sludge and increased diffusion limitation were then assessed through simulation for a typical continuous aerobic-aerobic-oxic (A²O) configuration (Fig. 1). This typical configuration is shown in Fig. 1. The reference case of a continuous activated sludge plant in an A²O configuration with a 10 layered settler consists of a preanaerobic, anoxic, and aerobic reactor. The influent flow rate is distributed to the preanaerobic, anoxic, and aerobic reactor in the ratio of 1:4:5, respectively. The effluent flow rate is calculated as

\[ Q_{\text{eff}} = Q_{\text{in}} - Q_{\text{r}} - Q_{\text{u}} \]

The influent ammonium concentration is set at 2 g N m⁻³, and the influent nitrate concentration is set at 4 kg TSS m⁻³. The settler underflow rate is set at 0.5 Q_{\text{in}}. The internal recycle flow rate is set at 0.5 Q_{\text{in}}. The recycle sludge flow rate is set at 2 g O₃ m⁻³. The oxygen concentration set-point in the bioreactor is set at 5 Q_{\text{in}}.

2. Materials and methods

2.1. Continuous wastewater treatment plant under study

The continuous wastewater treatment plant under study consisted of an anaerobic-anoxic-aerobic (A²O) configuration (Fig. 1). This typical

![Fig. 1. Reference case of a continuous activated sludge plant in an A²O configuration with a 10 layered settler. Q_{\text{in}}: influent flow rate (m³ d⁻¹), Q_{\text{eff}}: effluent flow rate (m³ d⁻¹), Q_{\text{r}}: recycle sludge flow rate (m³ d⁻¹), Q_{\text{u}}: underflow rate (m³ d⁻¹).]
activated sludge configuration was modelled as in Solon et al. (2017) [23], on its turn based on the widely accepted Benchmark Simulation Model no. 2 (BSM2) [24]. The bioconversion reactions were described according to the ASM2d model [25] which have been widely calibrated and validated with data from full-scale activated sludge plants [26,27]. Dynamic settling of particles taking place in the settler compartment was described by the 10-layered Takacs model [28] neglecting biological reactions occurring in the settler (i.e., non-reactive settling). An overview of the models used for the various process units is given in Table A1. The plant dimensions and influent characteristics are given in Table A2 and Table A3, respectively. Constant influent concentrations were assumed, corresponding with average values of the dynamic influent conditions encountered in reality. This allows for performing steady state simulations, which is common practice for (preliminary) process design and for process comparison [29], as is the aim in this study. The model was implemented in the Matlab-Simulink (The Math Works, Inc. MATLAB. Version R2020a) simulation environment.

The focus of this study was on nitrogen removal, which is typically the limiting factor in practice, rather than organic carbon or phosphorus removal. Phosphorus removal was considered not limiting, since practice has proven that nitrogen removal is mostly limiting over phosphorous removal (personal communication, as can be considered standard operation, see in Pronk et al. (2015) [1]) and excess phosphorus can still be removed using chemical precipitation [30]. As such, the analysis was concentrated on the anoxic and aerobic bioreactors as well as the settler, while the anaerobic reactors were not explicitly considered. In practice, anaerobic bioreactors will be indispensable in refurbishment projects of continuous activated sludge plants towards an aerobic granular sludge plant, for the selection of better settling sludge via the feast-famine regime [4]. However, the cultivation of aerobic granules with long-term stability in continuous plants was assumed possible a priori in this study, which made it unnecessary to take up the anaerobic bioreactors explicitly in the simulation study. This allowed for a more straightforward analysis, without limiting the applicability of the results.

The oxygen supply to the three aerobic bioreactors was controlled according to the default control strategy applied in BSM2 [24]. The oxygen concentration in the second aerobic reactor (bioreactor 6 in Fig. 1) was controlled at a set-point of 2 g O₂ m⁻³ by manipulating the oxygen transfer coefficient KLa6 (d⁻¹), varying between 0 and 360 d⁻¹ [31]. The oxygen transfer coefficients in the preceding and subsequent aerobic reactor were adjusted accordingly: KLa5 (in bioreactor 5) was set to the same value as KLa6 and KLa7 (in bioreactor 7) was set to half the value KLa6. The reactor total suspended solids concentration (TSS) was controlled as in Solon et al. (2017) [23]. The TSS concentration in bioreactor 7, XTSS,7 (kg TSS m⁻³) was controlled at a set-point of 4 kg TSS m⁻³ by manipulating the waste sludge flow rate Qw, up to 8584 m³ d⁻¹. The controller parameters used for the oxygen and TSS control strategies are summarized in Table A4.

The sludge recycle ratio was fixed at Qr/Qm = 1.5 and the internal recycle ratio at Qr/Qm = 5. A (relatively high) sludge recycle ratio of 1.5 is typically applied to prevent denitrification taking place in the settler [32]. The internal recycle ratio of 5 corresponded with the optimal value according to the method by Ekama and Wentzel (2008) [33] for a pre-denitrification configuration (Table A5 and Table A6).

2.2. Scenario analysis

The individual and combined effects of the better settling characteristics and increased diffusion limitation associated with granular sludge were assessed through simulation, for the given continuous wastewater treatment plant configuration and operation strategies. Three different scenarios were considered (Table 1). Activated sludge served as a reference case. Better settling sludge was modelled with a five times higher settling velocity than activated sludge. This value was set based on a literature review of the settling velocity of aerobic granular sludge, considering experimental results from considering 15 different studies covering a variety in granule size distributions (Table A7). The settling velocity of aerobic granular sludge was compared to the settling velocity of activated sludge, which was varied between 2 and 10 m h⁻¹ [34]. Note that the better settling sludge scenario only simulates the effect of the better settling characteristics, which will never exist in practice as this characteristic will also influence diffusion. Diffusion limitation was modelled by increasing the apparent half-saturation coefficients in the ASM2d model [35–37]. The apparent half-saturation coefficients are higher than the intrinsic half-saturation coefficients, which are the substrate concentrations at half of maximum specific growth rate (umax) in Monod kinetics [38] as they account for the rate limiting effect of diffusion. The adjustment of the half-saturation constants in this study was taken from Baeten et al. (2018) [37]; these parameters had been calibrated for a full-scale (batch) aerobic granular sludge plant (Table A8). Note that some authors denote the ‘half-saturation coefficient’ as ‘half-saturation index’ to indicate that it is not a constant as it varies in time and space due to substrate transport resistance (advection and diffusion) and the competition between the different microbial groups [38]. In this article, these different contributing factors are lumped in one coefficient which is here called, the apparent half-saturation coefficient.

The oxygen set-point of 2 g O₂ m⁻³ was increased to 4 g O₂ m⁻³ for aerobic granular sludge to ensure that nitrification could take place. The doubled oxygen set-point is proportional to the increase of the half saturation constant for SO₄ of ANO from 0.5 to 1 g O₂ m⁻³ (Table A8).

2.3. Process performance evaluation

The process performance of the three scenarios under study (Table 1) was evaluated by determining the maximal treatment capacity (kg COD m⁻³ d⁻¹) which could be applied while still meeting the prevailing effluent criteria. The corresponding nutrient removal efficiencies and energy requirements were calculated as well. The effluent criteria were defined according to EEC Council (1991) [40] for more than 100000 P.E., comprising the following upper limits CODₜₚₚₚ = 125 g COD m⁻³, BOĐₜₚₚₚ = 25 g BOĐ m⁻³, Nₜₚₚₚ = 10 g N m⁻³ and TSSₜₚₚₚ = 35 g TSS m⁻³. The effluent criterion for phosphorus (1 g P m⁻³) was not considered, given that nitrogen removal is mostly

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**Table 1** Simulation set-up: scenarios under study, characterized by reactor TSS concentration, settling velocity and the apparent half-saturation coefficient.

<table>
<thead>
<tr>
<th>Scenario</th>
<th>Activated sludge (reference case)</th>
<th>Better settling</th>
<th>Aerobic granular sludge</th>
</tr>
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<tbody>
<tr>
<td>Fig. 2</td>
<td>4</td>
<td>not applicable</td>
<td>5*Vesilind</td>
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<td>Fig. 3</td>
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<td>not applicable</td>
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<td>Fig. 4</td>
<td>4</td>
<td>Vesilind</td>
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<td>Fig. 5</td>
<td>2–6</td>
<td>Vesilind</td>
<td>5*Vesilind</td>
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<td>Fig. 6</td>
<td>2–6</td>
<td>Vesilind</td>
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<td>Fig. 7</td>
<td>5(optimal)</td>
<td>Vesilind</td>
<td>5*Vesilind</td>
</tr>
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</table>

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1. Vesilind settling velocity [28].
2. ASM2d apparent half-saturation coefficients [39].
3. The reactor TSS concentration was determined as optimal when the treatment capacity was maximal.
4. 5 times the Vesilind settling velocity (5 is an average value based on a literature review (Table A7) compared to the range of settling velocities of activated sludge varying between 2 and 10 m h⁻¹ [34]).
5. Increased apparent half-saturation coefficients [37] (Table A8).
limiting over phosphorus removal (personal communication, as can be considered standard operation, see Fig. 6 in Pronk et al. (2015) (1)). Chemical phosphorus removal was assumed to be applied in case biological phosphorus removal would not be sufficient.

Steady state simulations were run for the three scenarios under study (Table 1), for the influent conditions given in Table A3 and a fixed TSS concentration in the reactor (kg TSS m⁻³). The reactor TSS concentration ranged from 2 to 6 kg TSS m⁻³ in the case of activated sludge and from 4 to 8 kg TSS m⁻³ for better settling sludge and aerobic granular sludge. Steady state was reached after 200 days of simulation.

The maximum treatment capacity corresponding with each scenario and every reactor TSS concentration was found by step-wisely increasing the organic carbon loading rate (kg COD m⁻³ d⁻¹) by increasing the influent flow rate (keeping the influent concentrations constant). The internal recycle ratio was kept constant at 5. Given that the COD:N influent ratio remained unchanged, the system was thus always operated closed to the optimal value, ensuring that denitrification was not limited. The maximum treatment capacity (kg COD m⁻³ d⁻¹) was determined as the one for which all effluent concentration limits (TSS, COD, COD₄, BOD₅, or N₄) were still met in a range of ±0.5 g m⁻³.

The process performance was further defined by calculating the nitrate removal efficiency. The nitrate removal efficiency (η) was calculated as Eq. (1)

$$\eta_{\text{NO}_3^-} = \frac{S_{\text{NO}_3^-_{\text{in}}}}{S_{\text{NO}_3^-_{\text{out}}}}$$

where $S_{\text{NO}_3^-_{\text{in}}}$ is the influent nitrate concentration (g N m⁻³), and $S_{\text{NO}_3^-_{\text{out}}}$ is the effluent nitrate concentration (g N m⁻³).

The nitrate (+nitrite) conversion efficiency (η) was calculated as Eq. (2)

$$\eta_{\text{NO}} = \frac{S_{\text{NO}_3^-_{\text{in}}} + S_{\text{NO}_2^-_{\text{produced}}} - S_{\text{NO}_3^-_{\text{out}}}}{S_{\text{NO}_3^-_{\text{in}}} + S_{\text{NO}_2^-_{\text{produced}}}}$$

3. Results and discussion

3.1. Contrasting effects of aerobic granular sludge on ammonium removal rate

Granular sludge has better settling characteristics but also higher diffusion limitations compared to activated sludge. Both these effects impact the volumetric ammonium removal rate, $q_{\text{NH}_4}$ (g N m⁻³ d⁻¹), which can be expressed via Eq. (3).

$$q_{\text{NH}_4} = \mu_{\text{ANO}, \text{MAX}} \frac{S_{\text{NH}_4} X_{\text{ANO}}}{K_{\text{O}_2, \text{ANO}} + S_{\text{O}_2} Y_{\text{ANO}}}$$

where $\mu_{\text{ANO}, \text{MAX}}$ is the maximal specific growth rate of autotrophic nitrifying organisms (ANO) (d⁻¹), $S_{\text{O}_2}$ the oxygen concentration in the bulk liquid (g O₂ m⁻³), $K_{\text{O}_2, \text{ANO}},$ the apparent half-saturation coefficient of $S_{\text{O}_2}$ (g O₂ m⁻³), $Y_{\text{ANO}}$, the biomass yield of ANO (g COD g⁻¹ N). To order one’s thoughts, oxygen was considered as the only limiting substrate for nitrification, while ammonium was considered non-limiting. The ANO biomass concentration (g COD m⁻³) was set to $X_{\text{ANO}} = 0.04 \cdot X_{\text{TSS}}$ [44].

The individual effect of better settling of aerobic granular sludge compared to activated sludge allow a higher biomass concentration in the bioreactors ($X_{\text{ANO}}$) for a given settler surface area [45]. A higher biomass concentration in the bioreactors ($X_{\text{ANO}}$) results in a higher volumetric ammonium removal rate $q_{\text{NH}_4}$ (Fig. 2a). On the other hand, the individual effect of increased diffusion limitation, which can be described through an increased apparent half-saturation coefficient $K_{\text{O}_2, \text{ANO}}$ [37,38] will result in a lower volumetric ammonium removal rate $q_{\text{NH}_4}$ (Fig. 2b). The effect of diffusion limitation was most pronounced at low oxygen concentrations in the bulk liquid.

The combined effect of biomass concentration and diffusion limitation on the volumetric ammonium removal is given in Fig. 3. The results clearly show that the increased ammonium removal rate caused by higher biomass concentrations was counteracted by increased diffusion limitation. This effect could partly be compensated by operating at a higher bulk oxygen concentration, which not only increased the ammonium removal rate as such but also weakened the effect of diffusion limitation (reflected by a higher slope in Fig. 3b compared to Fig. 3a).

3.2. Maximum treatment capacity of activated versus granular sludge plants

3.2.1. Activated sludge (reference case)

The continuous activated sludge plant controlled at fixed reactor TSS concentration of 4 kg TSS m⁻³ was subjected to an increasing organic carbon load (Fig. 4). To keep the TSS concentration in the reactor at a constant value of 4 kg TSS m⁻³ at increasing organic carbon load, the amount of sludge wasted per day increased, resulting in a decreasing sludge retention time (SRT). Once the organic carbon load exceeded 1.36 kg COD m⁻³ d⁻¹, the SRT became too low to sustain nitrification, resulting in a violation of the effluent nitrogen concentration (N₄ < N₄ target). The ammonium removal efficiency was 59%, while the nitrate removal efficiency was 91%, meaning that nitrification was the primary inducer of failure of the system. Note that nitrification is not oxygen limited due to the presence of the DO controller. When further increasing the organic carbon load to 2.5 kg COD m⁻³ d⁻¹, the TSS load in the settler became too high, leading to settling failure and an associated violation of the TSS effluent concentration. The effluent COD and BOD concentration limits were met for all organic carbon loads considered (Fig. A2). As the effluent nitrogen criterion was the first one that was exceeded with increasing organic carbon load, it was the determining factor for the maximum treatment capacity of the reference activated sludge case operated at 4 kg TSS m⁻³, namely 1.36 kg COD m⁻³ d⁻¹.

The maximum treatment capacity of the reference activated sludge plant before nitrification and/or settler failure, at various controlled TSS
concentration (Fig. 5), because the higher TSS concentration entering a higher ammonium removal rate (Fig. 2a). In contrast, the treatment concentrations in the reactor, is displayed in Fig. 5 Fig. 6. The maximum Fig. 3.

Fig. 2.

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effect of (A) biomass concentration \( X_{\text{ANO}} = 0.04 \times X_{\text{TSS}} \) for a constant apparent half-saturation coefficient \( K_{\text{O2,ANO}} = 0.5 \text{ g O}_2 \text{ m}^{-3} \) and (B) diffusion limitation \( K_{\text{O2,ANO}} \) for a constant biomass concentration \( X_{\text{TSS}} = 4 \text{ kg TSS m}^{-3} \) on the volumetric ammonium removal rate \( q_{\text{NH4}} \), considering oxygen as the limiting substrate according to Eq. (3) \( \mu_{\text{ANO,Max}} = 0.61 \text{ d}^{-1}, Y_{\text{ANO}} = 0.24 \text{ g COD} \text{ g}^{-1} \text{ N} \).

Fig. 3. Combined effect of biomass concentration \( X_{\text{ANO}} = 0.04 \times X_{\text{TSS}} \) and diffusion limitation on the volumetric ammonium removal rate for (A) 2 g O\(_2\) m\(^{-3}\) and (B) 4 g O\(_2\) m\(^{-3}\) according to Eq. (3) \( \mu_{\text{ANO,Max}} = 0.61 \text{ d}^{-1}, Y_{\text{ANO}} = 0.24 \text{ g COD} \text{ g}^{-1} \text{ N} \).

concentrations in the reactor, is displayed in Fig. 5 Fig. 6. The maximum treatment capacity before failure of nitrification increased with increasing reactor TSS concentration (Fig. 5), which can be explained by a higher ammonium removal rate (Fig. 2a). In contrast, the treatment capacity before settler failure decreased with increasing reactor TSS concentration (Fig. 5), because the higher TSS concentration entering the settler had to be compensated by a lower hydraulic load. Settler failure resulted in extensive wash-out of suspended sludge, which made that also the effluent nitrogen concentration exceeded its standard, due to the concentrations of particulate-bound nitrogen in the effluent.

For reactor TSS concentrations between 2 and 4 kg TSS m\(^{-3}\), nitrification failed before the settler did whereas at higher reactor TSS concentrations (5–6 kg TSS m\(^{-3}\)) the opposite happened. The effluent COD and BOD concentration limits were never the limiting factor (results not shown). The maximum treatment capacity reached an optimum of 1.46 kg COD d\(^{-1}\) m\(^{-3}\) at a concentration of 5 kg TSS m\(^{-3}\), which was thus the optimal reactor TSS concentration for the reference continuous activated sludge system.

3.2.2. Better settling (granular) sludge

For better settling sludge, where the settling velocity of activated sludge is hypothetically increased with a factor five (Table 1), the maximal treatment capacity reached an optimum of 2 kg COD d\(^{-1}\) m\(^{-3}\), which was 39% higher than for activated sludge (Fig. 6). This optimum was reached at a reactor TSS concentration of 6 kg TSS m\(^{-3}\). At lower reactor TSS concentrations, nitrification was the limiting factor whereas at higher reactor TSS concentrations the settler failed before nitrification did. While this trend is the same as for activated sludge, in case of better settling sludge the settler could handle a higher incoming TSS load, allowing a higher reactor TSS concentration (6 kg TSS m\(^{-3}\)) instead of 5 kg TSS m\(^{-3}\) as well as a higher hydraulic load (+39%).

Besides a higher settling velocity, aerobic granular sludge is also characterized by diffusion limitation, which was subsequently considered as well (Table 1) to quantify the maximal treatment capacity of the continuous aerobic granular sludge system (Fig. 7). A higher oxygen set-point of 4 g O\(_2\) m\(^{-3}\) was needed to counteract the severe nitrification reduction because of diffusion limitation. However, the diffusion limitation of ammonium still limited nitrification and thus the treatment capacity decreased for a reactor TSS concentration between 4 and 7 kg TSS m\(^{-3}\). The maximal treatment capacity of the resulting aerobic granular sludge system amounted to 1.56 kg COD d\(^{-1}\) m\(^{-3}\), representing only a 7% increase compared to the activated sludge case. This optimum was reached at a reactor TSS concentration of 7 kg TSS m\(^{-3}\). For higher reactor TSS concentrations, the settler failed at the same treatment capacity as better settling sludge because of the same higher settling velocity (Fig. 6).

3.2.3. Improved aerobic granular sludge plant – Simultaneous nitrification-denitrification

The maximal treatment capacity of the improved aerobic granular
Sludge plant increased to 1.77 kg COD⋅d⁻¹⋅m⁻³ which is 21% higher compared to activated sludge (Fig. 6). This was reached at a reactor TSS concentration of 6 kg TSS⋅m⁻³. At a reactor TSS concentration of 7 kg TSS⋅m⁻³, the settler failed whereby only a 7% higher load is possible. The exact optimum would be between 6 and 7 kg TSS⋅m⁻³. Failure of the settler could not be influenced by an optimization strategy but only by further increasing the settling velocity of the sludge. Optimisation of the configuration and oxygen control strategy of the continuous plant improved nitrogen removal at lower reactor TSS concentrations. Anoxic reactor volume was not necessary anymore as this volume was created in the granules itself due to diffusion limitation. An oxygen set-point of 1.5 was found to optimally balance nitrification and denitrification in this case. By applying simultaneous nitrification–denitrification instead of using a predenitrification configuration, higher COD loads were possible.

3.3. Energy requirements

The impact of introducing aerobic granular sludge in a continuous wastewater treatment plant was assessed not only on the maximum treatment capacity (summarized in Fig. 7A) but also on the energy consumption (Fig. 7B and Table A9).
For all scenarios, the vast majority of energy (80–90%) was consumed for aeration. The specific aeration energy consumption was 50% higher for the non-improved aerobic granular sludge system compared to the activated sludge reference case. This was due to the higher imposed oxygen set-point (4 g O₂ m⁻³ compared to the 2 g O₂ m⁻³), requiring a higher K_La. By applying aerobic granular sludge in a simultaneous nitrification–denitrification configuration (improved system), the aeration energy was decreased significantly because of the lower oxygen set-point and was almost the same as for the activated sludge reference case.

Pumping energy represented 10–15% of the total energy requirements for the activated sludge and non-improved aerobic granular sludge systems, in predenitrification configurations. It was reduced by almost two thirds in the improved aerobic granular sludge system, since the simultaneous nitrification–denitrification system did not require a nitrate recycle.

Mixing energy represented an even smaller fraction (6% or less) of the total energy consumption. For the scenarios with predenitrification, the absolute amount of mixing energy was the same so the specific mixing energy only depended on the treatment capacity. For the improved aerobic granular sludge scenario, involving simultaneous nitrification–denitrification, mixing in the formerly anoxic bioreactors was established through aeration. As a result, mixers were only required in the anaerobic bioreactors, resulting in a significant reduction in the required mixing energy.

Overall, replacing a continuous activated sludge plant (predenitrification type) with an aerobic granular sludge plant with simultaneous nitrification–denitrification resulted in a significant increase in treatment capacity (+20%) while the total specific energy consumption was reduced by 10%. In contrast, the treatment capacity of the non-improved aerobic granular sludge system (keeping the predenitrification configuration) was only 7% higher and the total specific energy consumption was almost 40% higher than the activated sludge reference case. It is clear that the operational strategy of a continuous wastewater treatment plant needs to be improved along with the introduction of granular sludge, in order to fully profit from an increased treatment capacity and lower energy consumption.

3.4. Limitations and implications

The results in this study indicated the possible beneficial effects of replacing activated sludge with aerobic granular sludge in continuous wastewater treatment plants, considering differences in diffusion limitation and settling characteristics. Only two sludge categories with specific characteristics were considered, even though it is clear that in practice sludge diffusion and settling parameters may vary within a
wider range and sludge types with different characteristics are even likely to be present simultaneously. Besides, the plant performance was assessed based on relatively simple models. While this way of working allowed straightforward interpretation and good understanding of the fundamentals, it also implies that the results should not be interpreted in a fully quantitative way. The implications of these limitations are discussed in what follows. Indications are given for further research, given the positive indications for introducing aerobic granular sludge in continuous wastewater treatment plants.

Granular sludge experiences higher diffusion limitations than activated sludge flocs. The diffusion resistance will increase with increasing granule size. Granules formed in continuous flow reactors are typically smaller with a relatively more loose structure compared to those in batch reactors [46–48]. These smaller granules could be the result of low substrate concentrations [49–52] and a higher shear stress [53,54] possibly caused by pumping, mixing and/or continuous aeration. In this study, a 0D-model (neglecting spatial gradients within granules) with apparent kinetics was used to describe the bioconversion processes inside the granules. The applied values of the apparent half-saturation coefficients were calibrated based on experimental data of a full-scale (batch) aerobic granular sludge reactor [37]. These values served as the best available parameter estimates. In reality, apparent half-saturation coefficients will be influenced by various factors, such as granule size and density, substrate gradients and competition between microbial groups [9,35–37], which on their turn may be affected by the flow regime (continuous versus batch). For instance, the relatively low diffusion limitation in the small granules expected in continuous flow reactors would be described with relatively low apparent half-saturation coefficients. However, this effect would be counteracted by the lower substrate concentrations in continuous installations compared to batch reactors. One-dimensional biofilm models allow describing diffusion in more detail. Still, uncertainties associated with granule size and density in continuous flow reactors remain, as well as the effect of spatial substrate concentration variations within the reactor caused by e.g., dynamic influent conditions. An easy-to-interpret 0D-model was therefore preferred for the current proof-of-principle study.

Granular sludge has better settling properties than activated sludge. The relation between granule size and settling velocity is determined by Stokes law, expressing that the settling velocity is proportional to the square of the granule radius \(v_s \sim R^2\). In this study, a five times higher settling velocity was applied for aerobic granular sludge compared to activated sludge, which can be seen as a rather conservative value. A broad range of settling velocities for aerobic granular sludge is reported in literature (10–90 m.h\(^{-1}\)), resulting from diverse studies with laboratory batch reactors with a variety in granule size distributions (Table A7). Peak values of 70–90 m.h\(^{-1}\) \([55,56]\) would result in a factor 10 (or even higher) increase compared to the factor 5 used in this study, which would turn the result of the analysis even more favourable in terms of applying aerobic granular sludge in continuous systems.

As both diffusion limitation and settling velocity are influenced by the granule size, it may be expected that there exists an optimal granule size for aerobic granular sludge in continuous wastewater treatment plants, corresponding with a maximal treatment capacity at minimal energy requirements. In general, the granules need to be small enough to minimize diffusion limitation and still large enough to maintain good settling properties. However, even if such optimal granule size could be identified theoretically (e.g., based on simulations), the results would be most sensitive to the abovementioned biomass characteristics and operating conditions. Moreover, it would be most doubtful if the optimal granule size could be established in practice. For this reason, no attempts in this respect have been made. Sludge in practice will always be a mixture of flocs and granules with different sizes. In a full-scale aerobic granular sludge reactor, up to 20% of the total biomass consists of flocculent particles \([1,57]\). Moreover, in contrast to the smooth spherical granules obtained in lab-scale studies, granules with outgrowth on the sphere surface are typically present in practice \([58]\). These phenomena will influence both the diffusion resistance and the settling behaviour. It is worth noting that the settling behaviour of flocs and granules will not only differ in terms of the settling velocity, but also in terms of the settling mechanisms. Granular sludge experiences less hindered settling \([59]\) and less compression of the sludge in the lower layers of the settlers \([58]\). As a result, the Takács model \([28]\) for activated sludge may be no longer valid for aerobic granular sludge. A dedicated settling model for full-scale aerobic granular sludge, to describe the segregation of granules based on size during the settling process in a batch reactor, has been proposed by van Dijk et al. (2020) \([58]\) and is likely more suitable. Further refinements to the settler model could also include the consideration of specific flow patterns in the secondary settling tank. While such detailed settler model may be relevant in view of the cultivation of aerobic granules with long-term stability in continuous systems, it was not needed to see the overall effect of better settling on the maximal treatment capacity, as was done in this study.

4. Conclusions

The effect of introducing aerobic granular sludge in continuous wastewater treatment plants was scrutinized based on modelling and simulation.

- Aerobic granular sludge had contrasting effects on the volumetric bioconversion rates, which were positively influenced by the better settling properties - allowing a higher reactor biomass concentration - but negatively affected by increased diffusion limitation compared to activated sludge. This was exemplified for the ammonium removal rate, which is typically the most limiting.
- The maximal treatment capacity of a continuous wastewater treatment plant could be significantly increased by the introduction of aerobic granular sludge instead of activated sludge. However, this effect was strongly overestimated when only considering the improved settling properties and neglecting the increased diffusion limitation - which is more pronounced for continuous plants than for batch reactors.
- Diffusion limitation almost totally counteracted the positive effect of better settling properties when introducing aerobic granular sludge in continuous wastewater treatment plants without optimizing the plant operation accordingly. Not only did diffusion limitation lead to a lower maximal treatment capacity, it also caused an excessive additional energy consumption.
- A continuous wastewater treatment system could be improved for aerobic granular sludge by applying simultaneous nitrification-denitrification instead of using a pre-denitrification configuration. This resulted in a higher (+20%) treatment capacity and lower energy consumption (-10%) compared to a conventional activated sludge system.
- As both settling properties and diffusion limitation are influenced by the granule size, it was recommended to grow relatively small granules in view of introducing aerobic granular sludge in continuous wastewater plants.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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