

# Sequencing batch reactor process for the removal of nitrogen from anaerobically treated domestic wastewater

L. Pelaz, A. Gómez, A. Letona, G. Garralón and M. Fdz-Polanco

## ABSTRACT

This work presents the performance of a sequencing batch reactor (SBR) system used as a means of removing nitrogen from domestic wastewater containing a low chemical oxygen demand (COD) to nitrogen ratio due to pre-treatment with an anaerobic reactor. The aim of the work was to determine the feasibility of this system for the removal of nitrogen from the domestic wastewater. An SBR with a working volume of 5 L was investigated at different cycle times of 12, 8 and 6 h, at 18 °C.

The efficiency of the SBR varied together with the duration of the cycle, where the optimum performance was seen in the 6 h cycle with the anoxic–aerobic–anoxic sequence. Due to the low quantity of organic matter present in the domestic wastewater after the anaerobic treatment, an additional supply of external carbon was necessary before the second anoxic stage. The removal efficiencies obtained were: 98% for total Kjeldahl nitrogen, 84% for total nitrogen and 77% for soluble COD. The reactor was thus shown to be viable, and it was concluded that this process may be successfully applied as a post-treatment for the removal of nitrogen from anaerobically treated domestic wastewater.

**Key words** | denitrification, nitrification, nitrogen removal, organic matter, sequencing batch reactor

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## INTRODUCTION

Conventional activated-sludge treatment, commonly used to treat domestic wastewater, causes problems such as the excessive generation of sludge and involves the consumption of a large amount of energy (Ranade & Bhandari 2014; Hatamoto *et al.* 2016). In contrast, recently anaerobic biological treatment has attracted more attention for domestic wastewater treatment due to its advantages over conventional activated-sludge treatment (Wan *et al.* 2016). These include the fact that energy balances are quite favorable due to the recovery of energy as biogas, the lack of energy required for aeration and the reduction of excess sludge production (Moawad *et al.* 2009; Delgado Vela *et al.* 2015; Hatamoto *et al.* 2016).

The anaerobic reactors treating domestic wastewater can produce two main valuable products, which can be recovered and utilized: methane and sulfide. The methane gas, which is produced during the chemical oxygen demand (COD) removal can be recovered and transformed into energy (Khan *et al.* 2013). The effluent contains solubilized organic matter, high ammonia-nitrogen and organic-nitrogen concentrations. Therefore, the application of a

post-treatment process is necessary in order to remove nutrients from the wastewater and achieve the desired effluent quality (Chan *et al.* 2009; Hatamoto *et al.* 2016; Liu *et al.* 2016). Advancing treatment of domestic wastewater requires the implementation of energy efficient nitrogen removal technologies which preserve the energy savings achieved in the anaerobic process. This process also mitigates greenhouse gas emissions and maintains or reduces the footprint (Khan *et al.* 2013; Delgado Vela *et al.* 2015).

Biological nutrient removal constitutes the most economical and sustainable technique for removing organic carbon and nitrogen, while also meeting the rigorous discharge requirements (Liu *et al.* 2013; Sun *et al.* 2015; Alzate Marin *et al.* 2016). The biological nitrogen (N) removal involves two processes: nitrification and denitrification (Pelaz *et al.* 2017a). Nitrification is an aerobic process performed by autotrophic bacteria, in which ammonium ( $\text{NH}_4^+$ ) is oxidized to nitrite ( $\text{NO}_2^-$ ), by means of ammonium oxidizing bacteria (AOB). Next, nitrite is oxidized to nitrate ( $\text{NO}_3^-$ ) by nitrite oxidizing bacteria (NOB) (Hatamoto *et al.* 2016). Denitrification is an anoxic process performed by a

functional group of bacteria that use oxidized nitrogen as an electron acceptor in respiration. In this process,  $\text{NO}_3^-$  is reduced to  $\text{NO}_2^-$  and then to nitric oxide (NO), nitrous oxide ( $\text{N}_2\text{O}$ ) and finally to molecular nitrogen ( $\text{N}_2$ ) (Liu *et al.* 2013; Alzate Marin *et al.* 2016).

Both nitrification and denitrification possess  $\text{NO}_2^-$  as an intermediate. Hence, if nitrification is stopped at nitrite (nitritation), then complete denitrification from nitrite to nitrogen gas can be achieved. The nitritation/denitrification process, instead of complete nitrification/denitrification, may save 25% of aeration consumption and 40% of COD (Pollice *et al.* 2002; Pelaz *et al.* 2017b). Moreover, the denitrification rates for  $\text{NO}_2^-$  are 1.5–2 times faster than for  $\text{NO}_3^-$  thus enabling higher removal capacities. Furthermore, sludge production is reduced by 40% in comparison with nitrification/denitrification (Ruscalleda Beylier *et al.* 2011). However, the difficulty in utilizing nitrogen removal via nitrite lies in achieving specific inhibition of the NOB (bacteria that oxidize  $\text{NO}_2^-$  to  $\text{NO}_3^-$ ) while still retaining the AOB (bacteria that oxidize  $\text{NH}_4^+$  to  $\text{NO}_2^-$ ), thereby attaining nitritation (Blackburne *et al.* 2008; Mohammed *et al.* 2014; Al-Omari *et al.* 2015).

The biological process with an anaerobic–aerobic–anoxic system is a feasible and sustainable type of technology for removing both nitrogen and organic matter from domestic wastewater. High organic and ammonium removal efficiencies are achieved by using these systems, but the total nitrogen (TN) removal efficiency is not high due to the shortage of the carbon source available for denitrification (Sun *et al.* 2015). Organic substrates such as methanol can be used as the carbon and electron source for biological denitrification (Fulazzaky *et al.* 2015; Mohan *et al.* 2016). The main disadvantage of using methanol lies in the safety issues associated with its transportation, handling, and storage (Cherchi *et al.* 2009). One of the most effective methods to increase the organic matter concentration of the influent without the addition of external organic substrates is achieved by mixing a fraction of the anaerobic reactor's influent with the effluent of the same reactor. In this case, the anaerobic reactor should be used initially to treat only a part of the influent raw sewage (possibly no more than 50–70%), and the remaining part (30–50%) should be directed to the complementary biological treatment facility. The use of this 'bypass' will increase the COD of the reactor effluent, making it more adequate for use in the next denitrification stage (Chernicharo 2006; Chernicharo *et al.* 2015).

The sequencing batch reactor (SBR) is a flexible system that has been used successfully for developing the classic nitrification and denitrification processes (She *et al.* 2016).

The SBR is a 'fill and draw' type modified activated-sludge process that operates over a series of periods that constitute a cycle (Vijayan *et al.* 2016). Four basic steps of filling, reaction, settling and discharge phases take place sequentially in a single batch reactor. The SBR process offers minimum operator interaction, good oxygen contact with both the microorganisms and the substrate, small floor space and good removal efficiency. Moreover, the operation can be adjusted to obtain aerobic and anoxic conditions in the same tank (Chan *et al.* 2009; Khan *et al.* 2013). In contrast with continuous systems, SBRs have become quite a common practice for obtaining high nitrite accumulation, due to the flexibility of the process control (She *et al.* 2016).

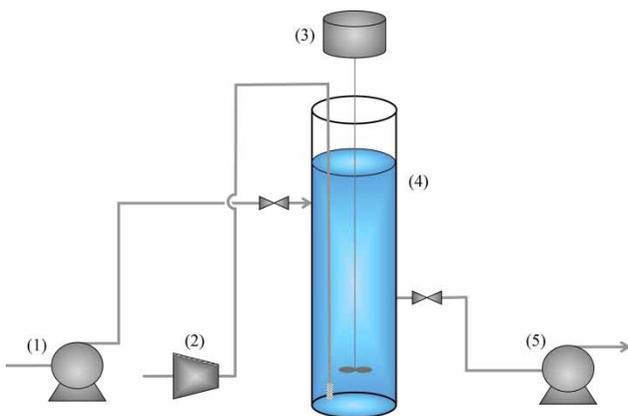
The main objective of the present work was to study the design and the feasibility of the SBR process for removing nitrogen from domestic wastewater that had been previously treated in an anaerobic reactor, at 18 °C of temperature operation.

## METHODS

### Experimental setup

The first part of the study was the optimization of the aeration period. The laboratory-scale system developed for this study consisted of two bioreactors with a total volume of 1 L. Air was supplied through porous diffusers at the bottom of each reactor to promote mixing and allow good diffusion of oxygen in the wastewater.

Once the aeration period was optimized, a second setup, used to develop the research, consisted of an SBR bioreactor and two tanks: feeding and effluent tanks. The reactor, with a total volume of 6 L and a working volume of 5 L, was completely mixed with a mechanical stirrer. It also contained a fine bubble air diffuser, which was part of the aeration system. Two peristaltic pumps were used for the reactor filling and the effluent discharge. The two pumps, the stirrer and the aeration system were connected to an electric timer system. A schematic diagram of the SBR plant is given in Figure 1. The reactor was kept in a room under a controlled temperature of around 18 °C ± 1 °C (Gouveia *et al.* 2015). This was the working temperature of the previous anaerobic reactor producing the effluent that was to be treated here. For the denitrification step, it was necessary to add methanol (1% wt.), which was supplied by another peristaltic pump. The addition of methanol was by means of a model made in order to simulate a bypass of part of the feed-stream from a point before the anaerobic reactor, to another



**Figure 1** | Scheme of the SBR plant. (1) Filling pump, which supplies the wastewater from the anaerobic treatment to the reactor, (2) compressor, responsible for supplying the air for the aeration step, (3) mechanical stirrer, (4) SBR reactor, (5) pump that drains the water after treatment.

point just at the end of it. The reactor was operated for 730 days.

In the two parts of the study, the aeration rate was controlled through a flow meter, maintaining the dissolved oxygen (DO) concentration at 2.0–2.5 mg O<sub>2</sub>/L.

### Inoculum and feeding characteristics

The inoculum of the three reactors studied was secondary aerobic sludge from the wastewater treatment plant (WWTP) of Valladolid (Spain).

The reactors were fed with the effluent from an anaerobic membrane bioreactor (AnMBR) fed with raw domestic wastewater from the city of Valladolid (Spain). The AnMBR is an upflow anaerobic sludge blanket reactor with a submerged ultrafiltration membrane placed on top of the same reactor. The AnMBR pilot plant is explained in detail in a previous work (Gouveia et al. 2015). The mean concentration of the main parameters of the influent feeding the denitrification/nitrification plant are given in Table 1. It is residual water with a high ammoniacal nitrogen content and a low concentration of organic matter, leading to a COD/TN ratio as low as 1.2. In the inlet stream (proceeding from the anaerobic treatment), sulfide is represented by the corresponding amount of sulfide

oxidation, without quantifying the oversaturation, so the real value for sulfide is higher.

### Analytical methods

Samples of wastewater were taken before and at the end of each treatment cycle. Nitrate nitrogen (NO<sub>3</sub><sup>-</sup>-N), nitrite nitrogen (NO<sub>2</sub><sup>-</sup>-N), sulfate (SO<sub>4</sub><sup>2-</sup>), soluble phosphorus (P-PO<sub>4</sub><sup>3-</sup>) and chloride (Cl<sup>-</sup>) concentrations were analyzed by high-performance liquid chromatography using a Waters 515 HPLC pump (Waters, Milford, USA) coupled with an ion conductivity detector (Waters 432, Milford, USA) and equipped with an IC-Pak Anion Guard-Pak column (Waters, Milford, USA) and an IC-Pak Anion HC (150 mm × 4.6 mm) column (Waters, Milford, USA). Ammonium concentration was determined using an ammonia-selective electrode: Orion, model 9512HPBNWP. The analyses of COD, and total Kjeldahl nitrogen (TKN) as well as of total and volatile suspended solids were determined according to standard methods suggested by the *Standard Methods for the Examination of Water and Wastewater* manual (APHA et al. 2005). The measurement of DO concentration was determined with a WTW oximeter, model oxi 330/SET, and a DO probe CeliOx 325. Temperature was measured using a temperature probe.

### Operation strategy

In the first part of the study, before starting with the SBR system, the aeration period was optimized to ensure the nitrification process. To do so, two reactors with a total volume of 1 L were used. One reactor was aerated over a period of 12 h and the other over 7 h. This amount of time was considered more than enough to oxidize the ammonium present in the wastewater.

For the rest of the work, the denitrifying/nitrifying SBR was used and operated with successive cycles. Each cycle consisted of a 15 min feeding stage and a reaction period, and finally during the last 15 min the supernatant draw was discharged, after 30 min of biomass settling. In order to determine the operation cycles, these were initiated (after the filling) with an aerated stage and continued with an anoxic one (Case 1–4). During the aeration phase, the average DO was 2–2.5 mg O<sub>2</sub>/L. During the anoxic phase aeration was not supplied. In the discharge stage, 3.5 L of the total working volume was discharged, leaving 1.5 L in the reactor to be treated in the following cycle. The different cycles studied in the SBR are presented in Table 2.

**Table 1** | Average composition of the wastewater after anaerobic treatment

Soluble COD (mg O <sub>2</sub> /L)	TKN (mg N/L)	NH <sub>4</sub> <sup>+</sup> (mg N/L)	NO <sub>2</sub> <sup>-</sup> (mg N/L)	NO <sub>3</sub> <sup>-</sup> (mg N/L)	SO <sub>4</sub> <sup>2-</sup> (mg S/L)	Soluble P (mg P/L)
100.6	81.9	77.7	0.0	0.0	8.5	9.2

**Table 2** | Time distribution of the stages of the different cycles in the SBR

Stage	Case 1	Case 2	Case 3	Case 4	Case 5
Filling	15 min	15 min	15 min	15 min	15 min
Pre-anoxic	–	–	–	–	30 min
Aeration	7 h	3 h	5 h 30 min	4 h	3 h 45 min
Anoxic	4 h	4 h	1 h 30	1 h	45 min
Sedimentation	30 min	30 min	30 min	30 min	30 min
Discharge	15 min	15 min	15 min	15 min	15 min
Cycle time:	12 h	8 h	8 h	6 h	6 h

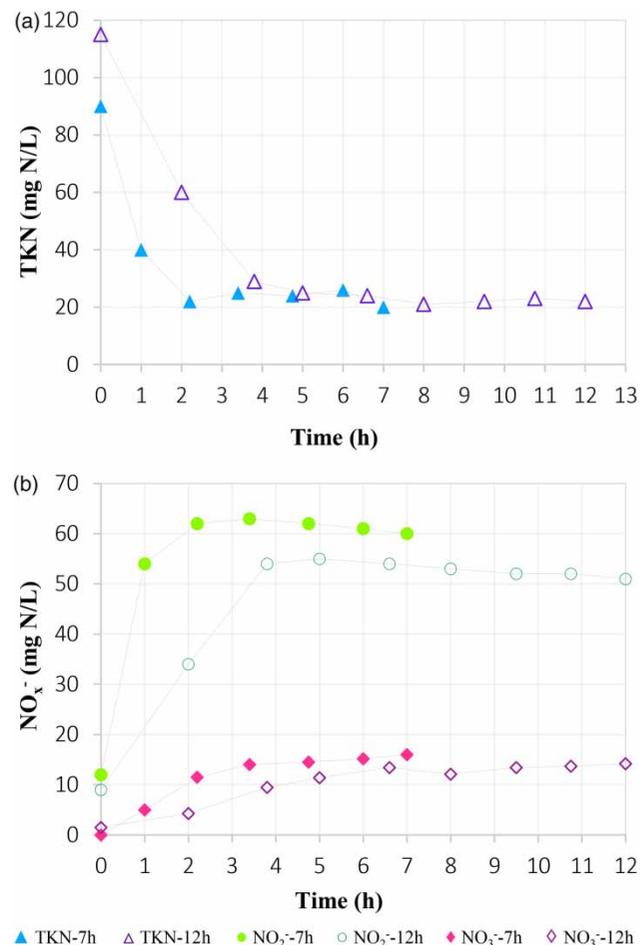
Modifications of the cycle took place to enhance the organic matter and nitrogen removal efficiencies. In Case 5, a pre-anoxic stage was added and the cycles consisted of an anoxic stage, aeration stage and another anoxic stage. Finally, a new cycle was achieved by adding methanol before the last anoxic stage, in order to provide more organic matter to the denitrification step and thus enhance the reactions in this stage.

## RESULTS AND DISCUSSION

### Optimization of the aeration period

The length of time required to ensure the nitrification process was studied. In Figure 2, the evolution of the nitrogen species in the reactors with 1 L of total volume (first setup) during the aeration period of 7 and 12 hours can be observed. In Figure 2(a), representing TKN concentration over time, average values in the feed of 90 and 115 mg N/L can be observed, while in the effluent these were in the range of 20 mg N/L. During the first 2 hours of aeration, around 60% and 70% of the TKN concentration was decreased in the two reactors studied, and the final TKN removal efficiency was about 79.4%. In aerobic conditions ammoniacal nitrogen was nitrified, i.e. it was used as the energy source by nitrifying bacteria leading to the formation of nitrite and nitrate. Residual ammonium was assimilated as the nitrogen source for the biomass synthesis by the bacteria. At the same time that ammonium was oxidized, nitrite and nitrate concentrations increased, although the latter took place more slowly, 3.5 hours vs 5 hours (Figure 2(b)). Nitrate and nitrite co-existed in the reactors, but the accumulation of nitrite was almost four times higher than the nitrate. Nitrite was the primary product of nitrification during the aeration experiment, accumulating up to 63 mg

$\text{NO}_2^-$ -N/L, while the nitrate concentration was always below 15 mg  $\text{NO}_3^-$ -N/L. A higher level of nitrite accumulation indicates a high activity of AOB, suggesting that the partial nitrification performance of the aeration process was good. By contrast, the activity of NOB was limited in the aerobic phase. High nitrite accumulation has been

**Figure 2** | (a) Profile of TKN concentration during aeration, (b) profile of nitrite and nitrate concentration during aeration.

reported by various authors (Aslan *et al.* 2009; Wei *et al.* 2014; She *et al.* 2016) in SBR systems.

The optimum time considered for the aeration process was 4 hours. After 4 hours of aeration, the effluent showed a mean concentration of 15.0 mg  $\text{NH}_4^+$ /L, 58.5 mg  $\text{NO}_2^-$ -N/L and 12.3 mg  $\text{NO}_3^-$ -N/L. After this time, there were variations of less than 5% in the parameters.

### Definition of the reactor operation cycles

Next, the SBR was used (second setup). Different times for the aerobic and anoxic stages were considered with the aim of determining the influence of the duration of these stages on the nitrogen removal. Four cycles of 12 h (Case 1), 8 h (Cases 2–3) and 6 h (Case 4) were studied, as shown in Table 2.

Figure 3(a) and 3(b) depict the graphic comparison in the TKN and  $\text{NO}_2^-$ , respectively, for the different cycles.

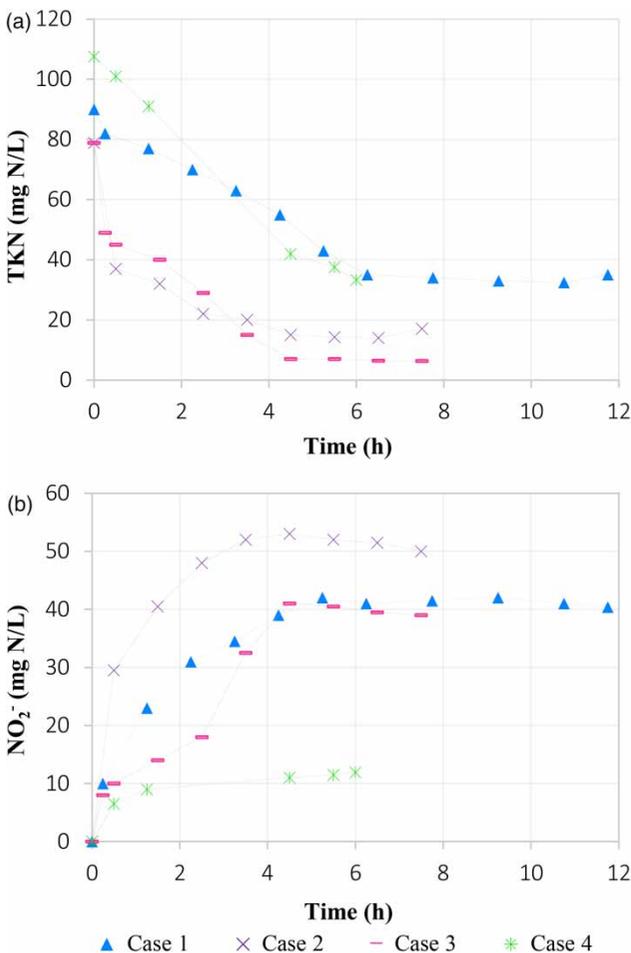


Figure 3 | Profiles of TKN (a) and  $\text{NO}_2^-$  (b) concentrations over time for the different cycles.

Table 3 | Nitrogen removal efficiencies of the different cycles

Parameter	Case 1	Case 2	Case 3	Case 4
% TN removed	16.8%	14.3%	44.0%	54.0%

Nitrate is not represented because it was only detected in the 12 h and 6 h cycles, and in very low amounts, not exceeding the concentration of 4 mg  $\text{NO}_3^-$ -N/L.

In Figure 3(a), it can be observed that the average TKN concentration declined sharply during the first 6 hours.  $\text{NH}_4^+$  was almost completely oxidized within 6 hours. In cycles longer than 6 hours, TKN concentration decreased by less than 4% from this time onwards. Meanwhile, in Figure 3(b), there was an increase in  $\text{NO}_2^-$  concentration, achieving its maximum at around 4 hours of the cycle and then remaining constant or suffering a slight decline in its concentration.

Table 3 shows the nitrogen removal efficiencies for each cycle, achieving the highest value, 54%, in the last cycle. Therefore, it was considered that the optimum cycle for nitrogen removal should have a duration of 6 h.

### Study of a 6 hour cycle

The SBR process with a cycle duration of 6 hours (Case 4) was studied over a 70 day period.

Figures 4 and 5 show the evolution of the soluble COD, TKN,  $\text{NO}_2^-$ -N and  $\text{NO}_3^-$ -N concentrations during the aerobic/anoxic cycle of the operational period in the SBR. In the cases of COD and TKN it can be observed that the influent and effluent followed the same trend (Figure 4), because the effluent concentration increased when the inlet stream had a higher load, and decreased when the influent concentration was lowered. The mean removal efficiencies of COD and TN were 30.3% and 54.0%, respectively. These efficiencies were very low so it was necessary to consider a modification of the cycle.

The majority of the time, the final effluent exhibited a high nitrite concentration which proceeded from a low-yield denitrification process (Figure 5(a)). As stated previously, during the aeration stage, there was nitrite accumulation in the tank, mainly because partial nitrification took place. Moreover, during the aeration phase a high amount of the readily biodegradable organic matter was oxidized. As consequence, when the anoxic stage took place, with no aeration supply, a low denitrification yield was observed due to the small quantity of organic matter available. The COD/N ratio of 1.2 after anaerobic treatment

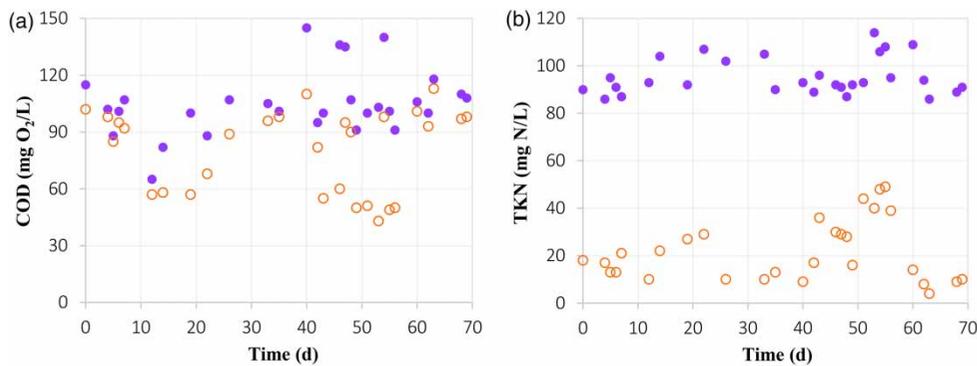


Figure 4 | COD (a) and TKN (b) concentration in the influent (●) and effluent (○) over time, for the 6 h aerobic/anoxic cycle.

was too low to establish an efficient denitrification process, but the COD available after the aerobic phase was even lower. Concerning the nitrate, throughout the study period there was generally an absence of this compound, except in very few specific cases (Figure 5(b)).

Soluble phosphorus concentration did not suffer considerable changes with the SBR process. A slight removal of about 12% was observed.

### Addition of a pre-anoxic stage

In order to improve the denitrification process, a pre-denitrification stage was added to the cycle, prior to the aeration step. Thus, during the first anoxic stage of 30 minutes, residual nitrite and nitrate that had not been eliminated in the previous cycle, and which therefore remained in the 1.5 L that had not been discharged, could be reduced to nitrogen ( $N_2$ ).

Having two anoxic stages allows the TKN effluent concentrations to be lowered. Thus, most of the nitrites and nitrates produced after nitrification in the aeration stage can be treated by flowing them through the second anoxic stage. According to Lu et al. (2016), a pre-anoxic stage has

been used with beneficial results in order to accomplish the removal of organic matter and nitrogen.

To adapt the cycle to this new change, it was necessary to modify the times of the different periods, so that the 6 hour cycle was carried out as indicated by Case 5 in Table 2. The parameters' concentration values before and after SBR treatment for the modified cycle are shown in Table 4.

Almost 80% of TKN was removed in the overall process, resulting in a final effluent with a concentration of around 20.2 mg N/L. During the first anoxic stage,  $NO_2^-$  and  $NO_3^-$  from the previous cycle were removed. During the aeration stage,  $NH_4^+$  was decreased from 88.4 to 18.1 mg N/L under the supply of DO, whereas there was a corresponding increase of  $NO_2^-$  concentration. In the following anoxic phase,  $NO_2^-$  concentration decreased to 25.3 mg  $NO_2^-$ -N/L. Nitrite was the primary product of the process, showing a clear accumulation, while the nitrate concentration was always very low, at about 1.5 mg  $NO_3^-$ -N/L. During the anoxic stages (with a low DO concentration), the organic matter present in the wastewater was used as electron donor by the denitrifying bacteria, while the oxidized nitrogen compounds,  $NO_2^-$  and  $NO_3^-$ , were reduced to  $N_2$ . The value of COD removed corresponded to the biological oxygen demand available in the feeding wastewater.

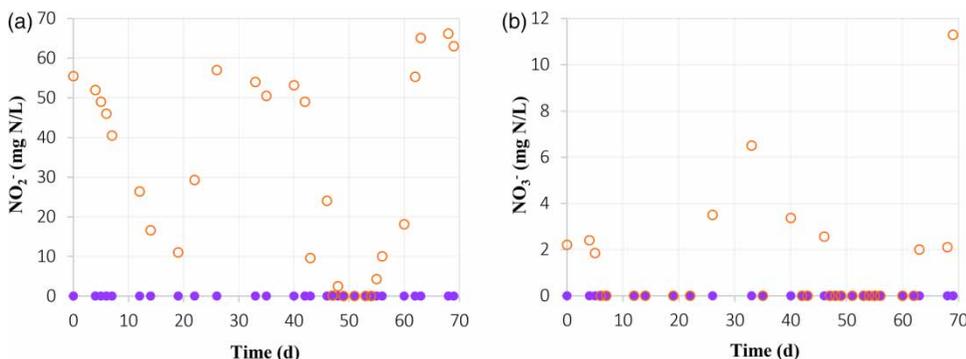


Figure 5 | Nitrite (a) and nitrate (b) concentration in the influent (●) and effluent (○) over time, for the 6 h aerobic/anoxic cycle.

**Table 4** | Parameters' concentration before and after 6 hours' modified cycle treatment

	Soluble COD (mg O <sub>2</sub> /L)	TKN (mg N/L)	NH <sub>4</sub> <sup>+</sup> (mg N/L)	NO <sub>2</sub> <sup>-</sup> (mg N/L)	NO <sub>3</sub> <sup>-</sup> (mg N/L)	Soluble P (mg P/L)
Influent	111.3	99.5	88.4	0.0	0.0	12.3
Effluent	67.9	20.2	18.1	25.3	1.5	12.6

The removal efficiencies of COD and TN were 39.0% and 52.8%, respectively.

### Addition of methanol

The anoxic-aerobic-anoxic process requires sufficient degradable carbon substrate to provide the energy source needed for the denitrification reactions that occur after the aeration stage. Due to the low amount of easily biodegradable organic matter available in the wastewater to be treated in the SBR process, methanol was added as an external carbon source before the denitrification stage (Wu et al. 2007; Ginige et al. 2009; Hwang et al. 2016). This compound could be used as an electron donor by the denitrifying bacteria, responsible for the reduction of the nitrate and nitrite to gaseous nitrogen. A solution of methanol (1% wt.) was added to the cycle before the second anoxic stage, just after the end of the aeration period.

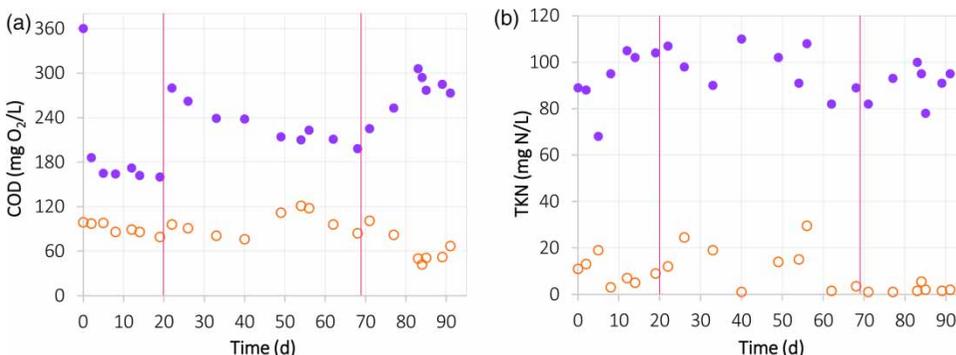
It is worth mentioning that instead of adding methanol, the increase in organic matter within WWTPs can be made by bypassing part of the feed stream from a point before the anaerobic treatment to another point at the end of this reactor. Using this course of action, it is possible to increase the soluble COD available in the liquid stream feeding the denitrification reactor, without adding an external carbon source. In this work, in order to simulate this behavior, the addition of methanol was employed as an extra carbon source.

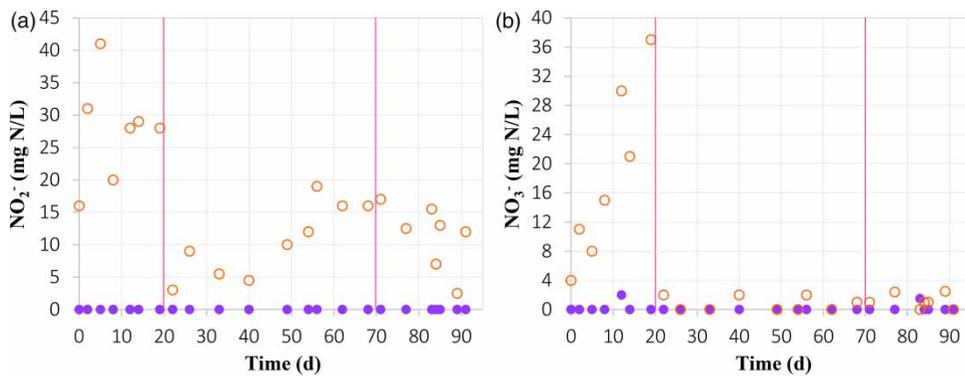
The COD, TKN, NO<sub>2</sub><sup>-</sup> and NO<sub>3</sub><sup>-</sup> concentrations are shown in Figures 6 and 7, respectively. This part of the study had a duration of 3 months and has been divided into three stages as indicated in Table 5.

Up until day 20, the COD in the wastewater from the anaerobic treatment was approximately 196 mg O<sub>2</sub>/L and methanol was not added to evaluate the reactor's reaction. In this period, the COD/N ratio was 2. TKN was removed by almost 90%, obtaining a mean TKN concentration of 9.6 mg N/L in the effluent. The evolution of TKN over time is shown in Figure 6(b). The nitrite and nitrate were increased up to concentrations of 41 mg NO<sub>2</sub><sup>-</sup>-N/L and 37 mg NO<sub>3</sub><sup>-</sup>-N/L. The evolution of NO<sub>2</sub><sup>-</sup> and NO<sub>3</sub><sup>-</sup> over time can be seen in Figure 7. The COD and TN efficiencies decreased to 53.7% and 40.7%, respectively, because of the lower COD/N ratio.

From day 21 to 70, methanol was added at up to 231 mg O<sub>2</sub>/L. The COD/N ratio was 2.4. Approximately, 86% of TKN was removed, obtaining an effluent with a mean TKN concentration of 13.3 mg N/L (Figure 6(b)). Nitrite was found in the effluent during this period, with a concentration of about 10.6 mg NO<sub>2</sub><sup>-</sup>-N/L, achieving 19 mg NO<sub>2</sub><sup>-</sup>-N /L at one point. It is worth noting the effect of the addition of methanol on the nitrate, but mainly on the nitrite (Figure 7). These compounds suffered a significant decrease in their concentrations when the COD was increased. The removal efficiencies of COD and TN rose up to 57.8% and 74.7%, respectively.

Finally, from day 71 to 91, a COD of about 273 mg O<sub>2</sub>/L was achieved with the addition of methanol. The COD/N ratio was 3. Up to 97.7% of TKN was removed in this section, obtaining a mean TKN concentration of 2.1 mg N/L in the effluent (Figure 6(b)). The nitrite concentration in the effluent was about 11.4 mg NO<sub>2</sub><sup>-</sup>-N/L (Figure 7(a)). Removal efficiencies of COD and TN of 76.7% and 83.9%, respectively, were obtained.

**Figure 6** | COD (a) and TKN (b) concentration in the influent (●) and effluent (○) over time, for the 6 h anoxic-aerobic-anoxic cycle with addition of methanol.



**Figure 7** | Nitrite (a) and nitrate (b) concentration in the influent (●) and effluent (○) over time, for the 6 h anoxic–aerobic–anoxic cycle with addition of methanol.

**Table 5** | Summary of the stages

Stage	Days	COD AnMBR	COD methanol	COD inlet	TN	COD/N	%TN removed	%COD removed
1	0–20	195.6	0.0	195.6	93.0	2.1	40.7%	53.7%
2	21–70	148.6	81.9	230.6	97.4	2.4	74.7%	57.8%
3	71–91	176.2	97.1	273.3	90.6	3.0	83.9%	76.7%

COD AnMBR: COD of the AnMBR effluent; COD methanol: COD supplied with methanol; COD inlet: COD of the SBR feed.

Instead of adding methanol, by using the alternative solution of bypassing a volumetric flow of 30% of the anaerobic feed from a point before the AnMBR to another point at the end of this reactor, it is possible to increase the available soluble COD by 55% in order to reach 273 mg O<sub>2</sub>/L. This estimation is based on the average of 501 mg O<sub>2</sub>/L of soluble COD contained in the wastewater before the anaerobic treatment, after the sedimentation stage (Gouveia *et al.* 2015).

About 98% of TKN was removed in the last section of the study. At the end of the cycle, a mean TKN concentration of 2.1 mg N/L was obtained. About 91% of the ammoniacal nitrogen after the aerobic stage was nitrified and 7% was assimilated by heterotrophic bacteria. The effluent after the cycle contained around 11.4 mg NO<sub>2</sub>-N/L while nitrates were occasionally found with a concentration of 2.5 mg NO<sub>3</sub>-N /L. Therefore, nitrite was the main compound accumulated in the reactor. After the nitrification was completed, about 79.7% and 92.2% of the generated nitrite and nitrate, respectively, were removed by denitrification.

The removal efficiencies of COD and TN with this cycle configuration were 76.7% and 83.9%, respectively. After the SBR treatment, the effluent contained a mean TN concentration of 14.6 mg N/L. High removal percentages were

observed. If these efficiencies are compared with the ones without methanol addition, there was an improvement of about 43% in COD removal and the TN removal yield was doubled. These increases in the organic matter and nitrogen removal were attributed to a higher denitrification activity in this cycle configuration.

In comparison with the literature, Hwang *et al.* (2016) reached 80% of nitrogen removal efficiency by using a sequencing batch biofilm reactor treating the rejected water from sludge. The reactor temperature was 15–35 °C. The reactor performed an aerobic–anoxic–aerobic–anoxic sequence and added methanol at the beginning of each anoxic step. The total cycle time used was 8 h, versus 6 h in the present study. Therefore, that process needs two more hours per cycle in order to achieve the same value for nitrogen removal as in the process developed here. In the same research, Hwang *et al.* enhanced the nitrogen removal efficiency up to 91% in an anoxic–aerobic–anoxic sequence when adding methanol as an external carbon source and NaHCO<sub>3</sub> for alkalinity, but with the same total cycle time of 8 h (Hwang *et al.* 2016).

On the other hand, Fernandes *et al.* (2013) used an SBR treating domestic wastewater with a COD/N ratio of 3. Although it operated with a cycle time of 8 h (versus 6 h in this study), its COD removal efficiency was higher: 83%

versus 77% obtained in the present study. In contrast, that process only achieved a mean of 50% TN removal, versus 84% in the present study.

An SBR studied by *Chen et al.* (2015) was used for the bioaugmented treatment of municipal wastewater with a COD/N ratio of 8, achieved with external carbon dosages. Comparing with the present study, *Chen et al.* achieved a better COD removal efficiency, 85.2% versus 76.7%, and 80.5% of N removal versus 84% in this study. Moreover, their system required a much greater supply of external carbon source in order to reach a COD/N ratio almost three times higher than that presented in this work (COD/N = 8 versus COD/N = 3).

## CONCLUSIONS

An SBR process was applied to domestic wastewater proceeding from an anaerobic treatment, which therefore contained a low concentration of organic matter. An experimental study on a pilot plant scale was carried out to ascertain its suitability for simultaneous nitrification and denitrification. Cycle times of 12, 8 and 6 h in the SBR were considered in the study, and the 6 h cycle time was selected as being optimal for the treatment. Results from nitrification and denitrification of domestic wastewater in the SBR showed nitrogen and COD removal efficiencies of about 84% and 77%, respectively. The process was successful in an anoxic-aerobic-anoxic cycle sequence with the addition of methanol just before the second anoxic stage. Thus, it has been demonstrated that the SBR process in a single reactor at a low temperature is a suitable process for the simultaneous removal of nitrogen and organic matter from domestic wastewater with a low COD content, with only the addition of an external carbon source. The operation method and conditions shown in this paper allowed reduction of the required residence time of the SBR for nitrogen elimination compared to available open literature.

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