Technical and economic analysis of real anaerobic digester centrate by means of partial nitrification and sustainable heterotrophic denitrification


ABSTRACT

The reliability of partial nitrification coupled with heterotrophic denitrification for the treatment of real anaerobic digester centrate produced in a wastewater treatment plant was technically and economically assessed in two sequencing batch reactors. Removal efficiencies above 90% were consistently achieved at N-ammonium loads above 1.2 g N L⁻¹ d⁻¹. Ethanol, affluent from a wastewater treatment plant (biological treatment inlet) and a zero-cost liquid residue from a chemical industry containing polyethylene glycol and sorbitol were employed as carbon source for denitrification. In this last case, a total organic carbon (TOC) requirement of 4.5 g TOC g⁻¹ NO₂⁻N was calculated. The denitrification rate was 0.26 g NO₂⁻N g VSS⁻¹ d⁻¹ (VSS: volatile suspended solids).

These results show that a carbon-rich waste can serve as a no-cost feed for denitrifying bioreactors. An in-depth economic analysis considering the main investment and operating costs of the process was developed, showing that it can suppose yearly savings above 50% with respect to the most widely used alternative of returning anaerobic digester centrate untreated to the head of the facility.

Key words | ammonium oxidizing biomass, ammonium removal, anaerobic digester centrate, denitrification, partial nitrification, sequencing batch reactor

INTRODUCTION

During the past two decades, the degree of implementation of innovative technologies for wastewater treatment in Europe has undergone a significant evolution, mostly as a consequence of the enactment of more restrictive regulations, as Directive 91/271/EEC. Thus, the concentrations of biochemical oxygen demand (BOD₃) in European rivers have decreased between 20 and 30% since the early 1990s, while phosphorus and ammonium concentrations are nowadays around 30–40% lower than then (European Environmental Agency 2002).

Still, nitrogen levels in European rivers have remained high. The problem, in most cases, arises from the fact that most wastewater treatment plants (WWTPs) were designed in past times with the mere purpose of removing excess chemical oxygen demand (COD) and, hence, cannot achieve the required nitrogen removal in their original form. In this sense, the depuration of anaerobic digester centrate (sometimes also known as ‘reject water’) supposes an especially dramatic problem. Anaerobic digester centrate, which is mostly produced in the sludge minimization stage of urban and industrial WWTPs, is characterized by its remarkable NH₄⁺-N load (commonly 10–20% of the total nitrogen load in the WWTP), even though its flow rate is only around 1–2% of the total flow rate treated at the WWTP.

Previous works have shown that treating this stream separately is often a more efficient and economical alternative than returning it to the head of the WWTP (Janus & Van der Roest 1997; Wett et al. 1998; Volcke et al. 2005). Among the different existing options for the treatment of these streams at real-scale, partial nitrification (PN) represents one of the most promising. In PN, N-ammonium is biologically oxidized to N-nitrite instead of N-nitrate, which can be subsequently denitrified. Hence, organic matter and O₂ consumption are around 40% and up to 25% lower than in conventional nitrification–denitrification, respectively, and the denitrification rate is around 63% higher (Turk & Mavinic 1987).
The aim of the present work was to technically and economically assess the reliability of the biological treatment of anaerobic digester centrate by means of PN followed by heterotrophic denitrification, considering different strategies to improve the sustainability of conventional denitrification.

### MATERIAL AND METHODS

#### Biological reactors

A fully automated 1 m³ sequencing batch reactor (SBR A) and a 20 L SBR (SBR B) were built and applied for PN and subsequent heterotrophic denitrification on a Spanish WWTP. SBR A incorporates a 5740SC dissolved oxygen sensor (Hach Lange S.L.U., Spain), a PC1R1N pH sensor (Hach Lange S.L.U., Spain) and a NH4D ammonium sensor (Hach Lange S.L.U., Spain) connected to a Programmable Logic Controller (PLC) for measurement, registration and control purposes. pH is continuously controlled via PLC by means of Na₂CO₃ addition. Operation pH was set at 8.0 ± 0.1. Temperature was set at 25 ± 1.0 °C. The nitrite-rich water produced in SBR A was used as affluent to SBR B. On the other hand, SBR B did not have any analytical sensors. Samples were periodically extracted and analyzed externally instead. An AOB (ammonium-oxidizing organisms)-rich inoculum extracted from the WWTP of a textile industry was used for the inoculation of SBR A. SBR B was inoculated with a denitrifying culture extracted from a pilot-scale SBR employed for carbon, nitrogen and phosphorus removal at the Autonomous University of Barcelona (Spain). Table 1 summarizes the main operating conditions during each phase. All the trials were carried out by feeding the SBRs with real anaerobic digester centrate from a WWTP.

#### Analytical methods

Aside from the analytical sensors present in SBR A, samples were periodically extracted from both SBR and analyzed in the laboratory. NH₄-N concentration was measured by means of a Continuous Flow Analyzer (Baeza et al. 2002). NO₂-N and NO₃-N concentrations were determined by means of an ICS-1000 Ion Chromatograph (Dionex Corp., USA), equipped with an IonPac AS9-HC. Volatile and total suspended solids and COD were determined according to Standard Methods (APHA 1989). The concentrations of organic carbon were determined by means of a TC/TIC/
TOC (total carbon/total inorganic carbon/total organic carbon) 1020A total carbon analyzer (O.I. Analytical, USA).

**Economic analysis considerations**

Three different alternatives for the treatment of anaerobic digester centrate were economically assessed in the present study. Case 1 represents the most commonly employed option, in which anaerobic digester centrate is sent back untreated to the head of the WWTP. This alternative implies an increase in the volume of the biological reactors downstream, both for nitrification (aerobic) and for denitrification (anoxic). The design criteria of these reactors (Carrera et al. 2004) are: (i) nitrification rate of 0.03 kg N kg$^{-1}$ VSS d$^{-1}$, (ii) biomass concentration in the IFAS (Integrated Fixed-Film Activated Sludge) aerobic reactor of 7.0 kg TSS m$^{-3}$, with a VSS:TSS ratio of 0.86, (iii) denitrification rate of 0.0595 kg N kg$^{-1}$ VSS d$^{-1}$ and (iv) biomass concentration in the anoxic reactor of 2.5 kg TSS m$^{-3}$, with a VSS:TSS ratio of 0.86.

Case 2 implies the specific treatment of the anaerobic digester centrate by means of a SBR used for PN and subsequent recirculation of the NO$_2^-$-rich water to the head of the WWTP. Again, an aerobic and an anoxic reactor are required. The first one, a SBR, has a design load of 1.2 kg N m$^{-3}$ d$^{-1}$ (determined in the studies shown in the following section). A denitrification rate of 0.0595 kg N kg$^{-1}$ VSS d$^{-1}$ and a biomass concentration in the anoxic reactor of the main line of 2.5 kg TSS m$^{-3}$, with a VSS:TSS ratio of 0.86, have been considered.

Finally, Case 3 represents the specific treatment of the anaerobic digester centrate by means of a SBR used for PN and denitrification. Two carbon sources were assessed herein for denitrification: ethanol (Case 3a), and a subproduct of a chemical industry consisting in a mixture of polyethylene glycol and sorbitol (Case 3b). The only reactor considered, a SBR, has a design load of 0.47 kg N m$^{-3}$ d$^{-1}$ (determined in the studies shown in the following section).

Among the considered costs, the ‘Equipment’ concept represents the purchase cost of the aeration and mixing systems. Among the operating costs, the electrical cost is mostly due to the aeration and mixing systems. A kWh cost of 0.1 € kWh$^{-1}$ was estimated. The ‘sludge treatment’ concept was only considered in Case 2. However, due to the high alkalinity of the anaerobic digester centrate, only 40% of the stoichiometric amount would be required. No alkalinity will be required in Case 3a or 3b, due to the presence of OH$^-$ generated during denitrification. A HCO$_3^-$ concentration of 3,640 mg L$^{-1}$ was determined in the anaerobic digester centrate used in the present study. NaOH, with a cost of 0.15 € kg$^{-1}$, was selected as a source of alkalinity. Maintenance costs have been considered to represent 4% of the final amount.

**RESULTS AND DISCUSSION**

**Period I: Start-up phase**

During the start-up phase (Period I), PN was assessed in SBR A under the conditions shown in Table 1. The nitrite-oxidizing bacterial population was inhibited by means of a methodology based on the control of the NH$_4^+$-N concentration. Figure 1 shows the performance of the biological...
reactor during this stage. Initially, the system was fed a nitrogen load of 0.1 g N L\(^{-1}\) d\(^{-1}\). The N-ammonium concentration in the inlet waste water was around 40 mg NH\(_4\)-N L\(^{-1}\). After 1 week of operation, the nitrogen load was stepwisely increased up to 1.2 g N L\(^{-1}\) d\(^{-1}\). This value is inside the common load range fed to PN systems (Fux & Siegrist 2004; Ciudad et al. 2005; Gali et al. 2007; Ganigué et al. 2009). Progressively increasing the nitrogen load allowed for an adequate ammonium-oxidizing biomass adaptation, with no clear substrate inhibition.

Figure 1 shows that at the end of this first stage, around 88% of the nitrogen supplied as NH\(_4\)-N was recovered in the effluent as NO\(_2\)-N. Similar values have been found in different previous studies, as in Gali et al. (2007) and Ganigué et al. (2009). However, those studies were performed under operating temperatures around 10 °C higher. Regarding the duration of the start-up phase (around 50 d), this is also a regular value in biological nitrifying reactors. Similar start-up times were observed by Arnold et al. (2000), Villa-verde et al. (2000) and Xue et al. (2009), among others. This time is significantly higher than those commonly found in systems for COD removal, due to the much lower growth rate of nitrifying bacteria with respect to heterotrophic microorganisms. Wiesmann (1994) proposed values for these growth rates of 0.033 and 0.3 h\(^{-1}\), respectively. Overall, these results remark the high efficiency of the PN strategy followed.

**Period II: Partial nitrification followed by heterotrophic denitrification**

Afterwards, in Period II, PN was complemented with heterotrophic denitrification for the following 30 d. In order to achieve this, anoxic cycles were incorporated into SBR A operation and ethanol (1.0 L/d) was fed as organic carbon source. In order to shorten the adaptation period, SBR A was inoculated with 50 L of biomass extracted from a municipal Spanish WWTP. The same strategy for PN as in the previous section was used. The duration of each stage was as follows: 45 min for settling, 30 s for draining, 15 min for filling and successive cycles aerobic/anoxic of 60 min each. Initially, 12 of these cycles were performed before settling. However, as the load increased, the number of cycles was decreased down to seven. The average NH\(_4\)-N load supplied was 0.47 g N L\(^{-1}\) d\(^{-1}\). As Figure 1 shows, a progressive acclimation of the biomass to the new conditions was observed, leading to nitrogen removal efficiencies around 90% after 2 weeks under the abovementioned operating conditions.

**Period IIIa: Partial nitrification**

Finally, in Period III the denitrification stage was suppressed, with the purpose of assessing the resiliency of the system. The nitrite-rich water produced during this stage was used as affluent to a denitrification SBR (SBR B). The results found on this second SBR are presented in the following section. Figure 1 shows that less than 2 weeks after regular PN was resumed, SBR A had achieved again ammonium elimination capacities above 1.2 g N L\(^{-1}\) d\(^{-1}\), with a PN efficiency ([NO\(_2\)-N]:[NH\(_4\)-N] ratio) above 95%.

**Period IIIb: Denitrification using affluent from the biological treatment inlet of a WWTP as carbon source**

Simultaneously to the studies described in the previous section, a series of denitrification assays aimed at determining the reliability of using alternative and more sustainable organic carbon sources for denitrification were undertaken in SBR B. During Period IIIb, SBR B was fed with effluent from SBR A and affluent (biological treatment inlet) from a municipal Spanish WWTP. During the first 12 d of this stage, the ratio of these liquids was 1:11 (v:v), while from day 12 onwards this ratio was changed to 1:16 (v:v). SBR B was operated in 24 h-cycles, under anoxic conditions.

Figure 2 shows the inlet and outlet TNN, COD and TOC concentrations. It can be seen that the TOC and TNN were reduced around 16 and 49% on average, respectively. Considering the low biodegradability of this water, the amount of organic carbon needed would be 6.4 g TOC g\(^{-1}\) NO\(_2\)-N. This value is significantly higher than others cited in the literature, such as the value of 2.4 g COD g\(^{-1}\)
NO₂⁻N reported by Van Kempen et al. (2001), or a value of 2.2 g COD g⁻¹ NO₂⁻N found by Fux et al. (2005). However, in those studies, the carbon source employed was methanol or ethanol, much more easily biodegradable. Overall, a denitrification rate of 0.0595 g NO₂⁻N g VSS⁻¹ d⁻¹ was found during this stage.

**Period IV: Denitrification using a real liquid industrial waste as carbon source**

In this stage, NO₂⁻N denitrification was assessed in SBR B using a liquid waste from a nearby chemical plant (mixture 1:1:5 of polyethylene glycol:sorbitol:water, m:m) as carbon source. Effluent from SBR A was used as nitrite source.

Again, SBR B was operated in 24 h-cycles, under anoxic conditions. During the first 15 d of this stage, the daily volume fed to SBR B was around 11 L (1 L of effluent from SBR A + 10 L of an acetate solution + 20 mL of the polyethylene glycol and sorbitol solution). The acetate solution was added with the purpose of facilitating the adaptation of the microorganisms. From day 15 onwards, the acetate fed was stopped and the daily volume supplied of the polyethylene glycol and sorbitol mixture was decreased to 10 mL. The main results of this 45-d stage (Figure 3) show that average N-nitrite and TOC removal efficiencies of 87 and 25% were achieved, respectively. The amount of organic carbon needed for the denitrification was 4.5 g TOC g⁻¹ NO₂⁻N, around 50% lower than when effluent (biological treatment inlet) from the WWTP was used as carbon source. Overall, a denitrification rate of 0.26 g NO₂⁻N g VSS⁻¹ d⁻¹ was found during this stage. This value is concordant with others in which commercial carbon sources were employed (Timmermans & Van Haute 1985; Henze 1991; Lee et al. 1995). These results show that under adequate conditions a carbon-rich waste can be used as feed for denitrifying bioreactors with good results.

**Economic assessment of different alternatives for anaerobic digester centrate treatment**

An in-depth economic assessment of different alternatives for real-scale anaerobic digester centrate biological treatment was developed, taking into account both investment and operating costs. Three alternatives (Case 1: anaerobic digester centrate return to the head of the WWTP and consequent upsize of biological reactors; Case 2: PN in a SBR and NO₂⁻N-rich water return to the head of the WWTP and Case 3: PN + denitrification in a SBR) were considered. Within this last alternative, two different carbon sources (methanol, Case 3a, and a mixture of polyethylene glycol and sorbitol, Case 3b) were assessed.

Table 2 shows the main investment and operating costs calculated for each case. It can be seen that PN followed by denitrification in a SBR may represent yearly savings around 53% with respect to the most widely used alternative (anaerobic digester centrate return to the head of the WWTP and upsize of biological reactors). If methanol is used as carbon source instead, the difference with respect to Case I will represent a yearly saving around 35%. These results manifest that PN followed by heterotrophic denitrification can be a reliable and cost-effective alternative for the treatment of anaerobic digester centrate in a WWTP.

**CONCLUSIONS**

Results presented herein show that PN followed by heterotrophic denitrification represents a technically feasible and reliable alternative for the treatment of anaerobic digester centrate in WWTPs. In the present study, N-ammonium loads above 1.2 g N L⁻¹ d⁻¹ were nitrified in a SBR with overall NH₄⁺-N removal efficiencies above 90%. The time required for the start-up of the PN system was 50 d, similar to values found in previous PN systems. After coupling this PN process with heterotrophic denitrification, N removal efficiencies above 90% were observed. Only around 16% of the organic carbon present in the effluent (biological treatment inlet) of a municipal WWTP could be used as carbon source for denitrification, under the operating conditions of SBR B. The denitrification rate was 0.0595 g NO₂⁻N g VSS⁻¹ d⁻¹. Subsequently, a liquid residue from a chemical industry containing polyethylene glycol and sorbitol was fed to SBR B, under anaerobic conditions. The main results of this 45-d stage (Figure 3) show that average N-nitrite and TOC removal efficiencies of 87 and 25% were achieved, respectively. The amount of organic carbon needed for the denitrification was 4.5 g TOC g⁻¹ NO₂⁻N, around 50% lower than when effluent (biological treatment inlet) from the WWTP was used as carbon source. Overall, a denitrification rate of 0.26 g NO₂⁻N g VSS⁻¹ d⁻¹ was found during this stage. This value is concordant with others in which commercial carbon sources were employed (Timmermans & Van Haute 1985; Henze 1991; Lee et al. 1995). These results show that under adequate conditions a carbon-rich waste can be used as feed for denitrifying bioreactors with good results.

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sorbitol in equal quantities was employed as carbon source for denitrification. In this case, a TOC requirement of 4.5 g TOC$_{\text{removed}}$ g$^{-1}$ NO$_2$-N$_{\text{removed}}$ was obtained. The denitrification rate was significantly higher than in the assays with WWTP affluent (biological treatment inlet), with a value of 0.26 g NO$_2$-N g VSS$^{-1}$ d$^{-1}$. Average NO$_2$-N removal was around 87%. Overall, these results show that a carbon-rich waste can serve as a no-cost and environmentally friendly feed for denitrifying bioreactors.

Also, PN followed by heterotrophic denitrification proved to be a remarkably cost-effective option, as it can suppose yearly savings above 50% with respect to the traditional alternative of returning the anaerobic digester centrate to the head of the facility.

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