Ammonium removal in landfill leachate using SBR technology: dispersed versus attached biomass
A. Sivic, N. Atanasova, S. Puig and T. Griessler Bulc

ABSTRACT
Large concentrations and oscillations of ammonium nitrogen (NH₄⁺-N) in municipal landfill leachate pose considerable constraints to its further treatment in central wastewater treatment plants. The aim of this study was to evaluate and optimize two technologies for the pre-treatment of 600 L/day of landfill leachate: in particular, to optimize their operational conditions for NH₄⁺-N removal up to a level appropriate for discharge to sewers, i.e. <200 mg/L. Both technologies were based on a sequencing batch reactor (SBR), with two different biomass processes: (A) SBR with dispersed/flocculated biomass and (B) SBR with biomass attached to carriers. The results revealed that both technologies successfully reduced the NH₄⁺-N from 666 mg/L (on average) at the inflow to below 10 mg/L at the outflow with alkalinity adjustment in a 12-hour cycle. Both technologies achieved 96% removal efficiencies for NH₄⁺-N. However, SBR with dispersed biomass showed higher flexibility under varying conditions due to the shorter adaptation time of the biomass.

Key words | ammonium nitrogen, biological treatment, biomass carriers, landfill leachate, SBR

INTRODUCTION
Municipal solid waste landfill sites, whether operating or closed, represent a constant threat of contamination of groundwater and surface water. They also reduce the performance of municipal wastewater treatment plants (WWTPs) due to leachate production. The need for efficient collection and treatment of leachate from landfill sites to avoid surface water and groundwater contamination is defined by EU legislation (2000/60/EC). The search for best available technology (BAT) remains a challenge for many experts, especially due to increasing changes in the technologies for processing solid waste and the specifics of each landfill site (Sil & Kumar 2017). Despite different views on leachate treatment, many experts agree that on-site treatment facilities are needed. They require less maintenance and power, and represent lower environmental and health risks in comparison with end-of-pipe treatment solutions (Yadav & Dikshit 2016).

Landfill leachate is difficult to characterize, because its composition and concentrations depend on a variety of factors, such as waste composition and treatment (e.g. mechanical biological treatment), landfill technology, waste age, landfill closure (semi- or totally water-tight), site geology, the waste’s moisture content and other seasonal and hydrological factors (e.g. rainfall, surface runoff coefficient) (Griessler Bulc & Zupancic Justin 2007; Mojiri et al. 2012). Leachate is characterized by a very complex composition, including high concentrations of several parameters, such as chemical oxygen demand (COD), biological oxygen demand (BOD₅), total organic carbon (TOC), ammonium nitrogen (NH₄⁺-N), chlorides, iron, manganese, phenols, and adsorbed organic halogens (AOX) (Mojiri et al. 2012).

In the early stages, acetogenic liquors have high BOD₅ and COD values, and high concentrations of NH₄⁺-N, which pose a serious pollution treat to the environment. In the later methanogenic phases of decomposition, these organic compounds are effectively converted to landfill gas, but high NH₄⁺-N concentrations remain in liquor containing relatively low levels of biodegradable organic material. At both stages of decomposition (acid and methanogenic), the reduction of high concentrations of NH₄⁺-N is often the bottleneck of the treatment (Bulc et al. 2004; Wang et al. 2015).

Nowadays, a wide range of leachate treatment processes have been applied, based on physico-chemical methods including coagulation with flocculation (Yadav & Dikshit 2016), membrane filtration and electrocoagulation (Suresh...


et al. 2016), electrocoagulation, electroflotation, and sedimentation (Hassani et al. 2016), advanced oxidation processes (Sharu et al. 2016), and biological processes including anaerobic treatment (such as an up-flow anaerobic sludge blanket; UASB) (Rosli et al. 2016), aerobic treatment procedures (such as treatment wetlands) (Lavrova et al. 2016), and a sequencing batch reactor (SBR) (Mojiri et al. 2012; Aziz et al. 2013; Fouad et al. 2016), etc.

Despite many treatment options, biological treatment has proven to be the most economical and sustainable method for the removal of biodegradable organic compounds (BOC), among which nitrification and denitrification are efficient for the reduction of high concentrations of organic nitrogen (N), especially in leachate with a high COD/N ratio. This treatment is efficient for leachate treatment of acidic landfill; however, high concentrations of COD and NH$_4^+$-N, high COD/BOD$_5$ ratio, and various metals can disturb treatment processes (Aziz et al. 2011). The need for effective nitrification/denitrification processes for leachate treatment supports SBR technology as a possible solution (Aziz et al. 2013; Yusoff et al. 2016). Aziz et al. (2013) proved that SBR is flexible with easily modified operational conditions, showing high removal efficiency, simplicity in operation and beneficial economical aspects.

The biological treatment processes under aerobic conditions in SBR can operate with dispersed or attached growth-activated sludge technology. In dispersed activated sludge technology, the biomass in the reactor is mixed and flocks are formed. Attached growth technology (i.e. moving bed biofilm reactor; MBBR) uses free-floating biomass carrier elements to which activated sludge is fixed and on which it grows, which results in longer sludge ages, larger biomass contained per volume of bioreactor, prevention of biomass washout and consequently more stable operation of the reactor (Ødegaard 2006). However, it requires more constant conditions and longer adaptation time (Qaderi et al. 2011). In high-nitrogen streams such as landfill leachate, the available influent alkalinity must be considered in order to ensure suitable autotrophic nitrification, which is achieved when the alkalinity and ammonium ratios are higher than 1.7 mg HCO$_3^-$-C mg$^{-1}$ NH$_4^+$-N (Metcalfe & Eddy 2003).

However, both processes (attached or dispersed biomass) have advantages and challenges regarding NH$_4^+$-N removal. Therefore, the optimization and adjustment of SBR operational strategies for reaction phases is needed to increase the efficiency of appropriate treatment processes.

This study assessed two SBR-based technologies: (A) with dispersed and (B) with attached biomass processes for an on-site pre-treatment of a maximum of 600 L/day of methanogenic municipal landfill leachate prior to treatment in a central WWTP for municipal wastewater. The operational strategies were adjusted in both technologies to reduce the cycle length for NH$_4^+$-N removal from leachate to meet Slovene legal requirements (OG RS No. 62/08, OG RS No. 98/15, OG RS No 36/16) for NH$_4^+$-N discharge into a sewer, i.e. <200 mg/L. European legislation for disposal and landfills is based on Council Directives 1999/31/EC and 2011/97/EU, which is summarized in the abovementioned Slovenian regulations.

MATERIAL AND METHODS

Landfill leachate characteristics

Landfill leachate was generated on the 35-year-old municipal solid waste landfill, Tenetiše (Kranj, Slovenia). The amount of leachate directly corresponded to the amount of rainfall because the waste was not covered daily with inert soil or other material to reduce leachate production. In the dry season, the leachate amounts ranged from 60 to 80 m$^3$ per day and in the rainy season from 300 to 600 m$^3$ per day on average. Maximum values were recorded in autumn (September–October) with 1,000 to 1,100 m$^3$ per day. Concentrations of NH$_4^+$-N varied according to the dilution by rainfall and significantly increased during dry-weather periods. The average NH$_4^+$-N concentration in collected leachate (in a retention tank) was 659 mg/L, with a peak of 900 mg/L in the dry season, while during heavy rain events it fell below 200 mg/L.

Table 1 presents the characteristics of the landfill leachate during this experimental study. Leachate samples were taken from the retention tank (SP-RT) and analysed according to Standard Methods (APHA/AWWA/WEF 2005) for the parameters listed in Table 1. The analysis of the chemical parameters showed that the leachate was poorly biodegradable (BOD$_5$/COD ratio below 0.3) and that it contained high levels of NH$_4^+$-N, which exceeded the limit concentrations for discharge into a sewer, i.e. <200 mg/L, (OG RS No. 98/15, OG RS No. 36/16) and caused additional negative impacts on the further treatment in the central WWTP.

Figure 1 presents a schematic overview of the experimental setup. The landfill leachate was firstly collected in a retention tank (RT) with a volume of 60 m$^3$. Then, 600 L of leachate was pumped to each reactor (A and B), of the pilot SBR system, using submerged pumps (A1, B2). The
rest of the leachate was pumped into the sewage system using the submerged pumps (P3, P4) for further processing by the central WWTP. The pilot SBR system consisted of two reactors, i.e. an SBR reactor with dispersed biomass (A) and an SBR reactor with attached biomass on PVA-gel beads from the Japanese company Kuraray (B). The
PVA-gel beads made from polyvinyl alcohol were slightly heavier than water, with a specific gravity of 1.025. The gel beads were 4 mm in diameter, hydrophilic in nature, had a porous structure with only 10% solids and a continuum of passages from 10 to 20 microns in diameter tunnelling through each bead (Kuraray 2005).

The pre-treatment systems in our experiment were designed to treat leachate collected during dry weather periods with the highest concentrations of NH₄⁺-N. During major rainfall events, the first flush of leachate combined with storm water drainage was collected in the retention tank to obtain the most polluted fraction of landfill leachate. The remaining water was bypassed directly into the sewage system connected to the central WWTP.

**The pilot SBR systems**

The SBR systems were located at the landfill site for on-site treatment, consisting of a polyester tank with a total capacity of 6 m³, divided into four equal parts with operational capacity of 900 L: Reactors A, B and two outflow tanks from Reactors A and B (Figure 1). In the pilot SBR system, the effectiveness levels of leachate treatment (NH₄⁺-N removal) with both SBR Reactors A and B were monitored simultaneously.

Each SBR treated 500 to 600 L/day of landfill leachate. Dissolved oxygen (DO) was controlled at 4–6 mgO₂/L with an on/off blower. The pilot SBR system was equipped with a control panel and process measurement equipment (A6, B6; Figure 1), specified in Table 2. This equipment was a key factor for the smooth progress of the research. The external additions were composed of two dosing pumps (A5, B5). In the middle of the aerobic phase, the dosing pumps maintained alkalinity by dosing bicarbonate in a liquid solution with a set point of pH 6.5–7.5 in each reactor, to achieve complete nitrification.

<table>
<thead>
<tr>
<th>Name</th>
<th>Measured item</th>
<th>Units</th>
<th>Manufacturer</th>
</tr>
</thead>
<tbody>
<tr>
<td>LDO process on-line probe</td>
<td>DO (dissolved oxygen) temperature</td>
<td>mg/L °C</td>
<td>Hach Lange</td>
</tr>
<tr>
<td>pHD-S differential probe</td>
<td>pH temperature</td>
<td>°C</td>
<td>Hach Lange</td>
</tr>
<tr>
<td>Electrical conductivity sensor</td>
<td>EC</td>
<td>μS/cm</td>
<td>Hach Lange</td>
</tr>
<tr>
<td>NH₄D sc Ammonium sensor</td>
<td>NH₄⁺-N</td>
<td>mg/L</td>
<td>Hach Lange</td>
</tr>
<tr>
<td>AMMON eco sc</td>
<td>Cleaning head for NH₄⁺-N sensor</td>
<td>–</td>
<td>Hach Lange</td>
</tr>
<tr>
<td>New HOAB</td>
<td>Cleaning system for NH₄⁺-N</td>
<td>–</td>
<td>Hach Lange</td>
</tr>
<tr>
<td>Controller SC 1000</td>
<td>Device controller</td>
<td>–</td>
<td>Hach Lange</td>
</tr>
</tbody>
</table>

**Operating conditions**

The experimental period of both SBRs (A and B) lasted for 145 days. The methodology for optimizing the SBR cycle for leachate treatment of Reactors A and B was divided into five test periods (Figure 2). Operational conditions for each of the test periods are presented in Table 3. The first (I) and the second (II) test periods were run on a 24-hour cycle without alkalinity adjustment; the first (I) was dedicated to biomass adaptation, while the second (II) was dedicated to achieving stable conditions, given that biomass had already adapted.

When a stable state of nitrification was achieved in Period II, bicarbonate was dosed to increase the nitrification efficiency (Period III). The SBRs ran during the third (III) period in a 24-hour cycle mode with pH adjustment (set point of pH 6.5–7.5) to fix the molar ammonium/bicarbonate ratio. Observations in this period showed that high NH₄⁺-N removal efficiency was achieved in the beginning of the second half of the cycle, which led to further cycle modifications in Period IV, in which the cycle was shortened to a 12-hour cycle with pH adjustment. Period IV exhibited an even shorter aeration time to reach stable nitrification rates. Therefore, the aeration time was shortened by 2 hours, thus increasing anoxic time (Table 3).

During the start-up period, the SBR reactors were filled with leachate (600 L), followed by 300 L of activated sludge (4 mg/L MLSS), from a central WWTP; 100 L of biomass carriers (PVA-gel beads) were inserted into the tank B. They were used at a filling fraction of approximately 10%, with a recommended maximum of 15%, to ensure effective mixing in the entire reactor volume; however, the actual surface area of the PVA-gel beads is not known because bacteria are cultivated inside the core of the beads and consequently it is difficult to give the rate in terms of biofilm area (Kuraray 2005).
Monitoring

Parameters such as NH$_4^+$-N, DO, pH, temperature and EC values were measured on-line every 5 min over 145 days in Reactors A (SP-A, Figure 1) and B (SP-B, Figure 1) with the process measurement equipment (A6, B6, Figure 1). NH$_4^+$-N values were measured alternately in Reactors A and B throughout the duration of experiment as follows: in Reactor A, days: 1–13, 19–24, 32–38, 48–60, 74–81, 96–110, and 124–145; in Reactor B, days: 14–18, 25–31, 39–47, 61–73, 82–95, and 11–123 (Figure 3(a)–3(c)). Additionally, there were five analytical measurements of COD, BOD, NH$_4^+$-N, NO$_2^-$-N and NO$_3^-$-N performed at the inflow and at the outflow of both reactors for 1 day in the test Period III and 4 days in the test Period IV.

The following treatment procedure was applied in the SBRs: after each treating cycle, one-third of the volume was decanted as effluent, while two-thirds of the treated
water remained in the reactor. Thus, in the following cycle, raw inflow was mixed with the two-thirds volume from the previous cycle. Accordingly, NH₄⁺-N concentrations were measured at three sampling points: NH₄⁺-N in the raw leachate (SP-RT), NH₄⁺-N in the SBR at the beginning of the cycle (SP-A, SP-B) and NH₄⁺-N at the end of the cycle (i.e. in the effluent, SP-A, SP-B). On average, the concentrations of NH₄⁺-N in the raw leachate were 659 mg/L, while the concentrations in the reactor at the beginning of cycles were on average 212 mg/L.

**RESULTS AND DISCUSSION**

The pilot system of both SBR-based technologies: with dispersed (A) and with attached (B) biomass on PVA-gel carriers, located at the Tenetišė municipal solid waste landfill (Kranj, Slovenia) ran for 145 days under different operational conditions, corresponding to five test periods (I–V). Figure 3 shows the performance values of NH₄⁺-N at the beginning and at the end of each treatment cycle and for each test period (I–V) for (a) Reactor A and (b) Reactor B, and (c) comparison of NH₄⁺-N treatment efficiencies between Reactors A and B for each test period during the experiment.

**Periods I and II: adaptation and achieving stable conditions**

The objective of Period I was to adapt the biomass to the operational conditions. In this period, which lasted for 7 days, the average NH₄⁺-N values in the reactors (SP-A and

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Figure 3 | Experimental results: (a) NH₄⁺-N values and efficiencies in Reactor A; (b) NH₄⁺-N values and efficiencies in Reactor B; vertical black lines indicate different test periods (I–V); (c) comparison of NH₄⁺-N treatment efficiencies between Reactors A and B for each test period during the experiment.
SP-B) at the beginning of the cycle were 332 mg/L and 141 mg/L on average at the end of the cycle, i.e. in the effluent, reaching 54% treatment efficiency on average.

At the end of Period II, the operation processes were stabilized and Reactors A and B achieved 76% and 73% NH$_4^+$-N removal efficiencies, respectively (Figure 4(a) and 4(b)). In Reactor A, the NH$_4^+$-N values decreased to 46 mg/L in 6 hours on average while in Reactor B, the value 77 mg/L was reached after 14 hours (Figure 4(a)). In Period II, the nitrification efficiency was 74% on average, so a surplus of alkalinity was required to increase the nitrification efficiency, to prevent free nitrous acid inhibition (FNA) and as a carbon source for ammonium oxidizing bacteria/nitrite oxidizing bacteria (AOBs/NOBs) (Puig et al. 2006; Ganigué et al. 2007). Free ammonia (FA, NH$_3$), not ammonium (NH$_4^+$), is the substrate of the AOB, and can be inhibiting at very high concentrations. pH control is necessary when dealing with wastewater containing high nitrogen concentrations, to prevent inhibition of AOB by FA (Ganigué et al. 2007, 2012; Vadivelu et al. 2007). Regulating pH is also crucial for achieving high NH$_4^+$-N removal efficiencies.

**Periods III, IV and V: enhancing nitrification efficiency and cycles adjustment**

A level of 74% of NH$_4^+$-N removal efficiency was the value achievable using all the influent inorganic carbon (IC) available in the leachate, after the alkalinity was raised the nitrification efficiency increased to 94%. In Period III from Day 30, bicarbonate was dosed during the aerobic phases and maintained the pH values between 6.5 and 7.5 in both reactors to limit high FA concentrations. The nitrification process was reinforced achieving 94% of ammonium oxidation. This efficiency was achieved after 8 hours in Reactor A and after 10 hours in Reactor B (Figure 4(a)), reaching effluent NH$_4^+$-N concentrations below 10 mg/L on average. This proved that the cycle length could be shortened to 12 hours instead of 24 hours in Period IV. Despite this change, the SBRs were capable of maintaining the NH$_4^+$-N removal efficiency of 95% in an extremely dry weather period (influent: 850 mg/L). This efficiency was reached in 8–9 hours on average in both reactors (Figure 4(b)).

Because high NH$_4^+$-N removal efficiency in Period IV was reached in only 8–9 hours, the aeration phase in Period V was shortened by 2 hours, increasing the anoxic phase from 30 to 150 min. In Reactor A, constant NH$_4^+$-N removal efficiencies of 96% were reached, while in Reactor B fluctuating NH$_4^+$-N removal efficiency was observed at the beginning of the Period V until Day 119 when the nitrification stabilized and reached 96% (Figure 5(c)). Moreover, after Day 119, the concentration of NH$_4^+$-N dropped from 200 to less than 20 mg/L in both reactors in the first 4 hours of the aeration phase, on average (Figure 4(b)). This was reached due to the introduction of a longer anoxic phase with mixing at the beginning of the cycle in Period V. Effective mixing enabled adequate contact between the activated sludge and raw leachate (pollutant particles) (Letterman et al. 1999). Consequently, the nitrification efficiency reached 96% in both reactors with a longer anoxic phase with mixing pattern in Period V.

Almost complete nitrification was achieved in the last three periods. Table 4 presents analytical results from Day 36 (Period III) and Days 69, 74, 78 and 99 (Period IV), when the nitrification process achieved almost total ammonium oxidation; in Days 36 and 69 all NH$_4^+$-N was converted to NO$_3^-$-N; in Days 74, 78 and 99, some denitrification can be noted despite only 30 min anoxic reaction time.

**DO and biomass adaptation to modifications**

Figure 5 shows DO concentrations and pH values in Reactors A and B in each test period. DO concentrations in
Reactor A in test Periods II, III and IV reached the maximum value after 4 hours; in Reactor B, DO concentrations increased more quickly and reached the maximum value of 4–6 mg/L after 2 hours of operation. This means that in the system with dispersed biomass, a bigger sludge volume index occurred which required a longer aeration period or higher intensity of the air supply.

In test Period V in Reactor A, DO concentration began to rise within the aeration phase and reached the maximum value after 3 hours on average. In Reactor B, DO concentrations began to rise after 4 hours and then, in only 50 min, DO concentration reached the maximum value of 4–6 mg/L. It means that the SBR with the attached biomass on PVA gel carriers (Reactor B) required more constant operating conditions and needed more adaptation time, which show lower operational flexibility. The results showed that it is difficult to compare the oxygen consumption in both reactors because they are used at different activated sludge volumes, and because the activated sludge volume in the PVA-gel beads is not known.

Table 4 | Analytical measurements during the experiment

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Unit</th>
<th>Day 36</th>
<th>Day 69</th>
<th>Day 74</th>
<th>Day 78</th>
<th>Day 99</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>In</td>
<td>Out A</td>
<td>Out B</td>
<td>In</td>
<td>Out A</td>
</tr>
<tr>
<td>COD</td>
<td>mgO2/L</td>
<td>870</td>
<td>454</td>
<td>585</td>
<td>916</td>
<td>523</td>
</tr>
<tr>
<td>BOD5</td>
<td>mgO2/L</td>
<td>215</td>
<td>50</td>
<td>190</td>
<td>220</td>
<td>150</td>
</tr>
<tr>
<td>NH4-N</td>
<td>mgN/L</td>
<td>450</td>
<td>5</td>
<td>5</td>
<td>470</td>
<td>2</td>
</tr>
<tr>
<td>NO2-N</td>
<td>mgN/L</td>
<td>0.1</td>
<td>0.1</td>
<td>0.1</td>
<td>0.1</td>
<td>0.1</td>
</tr>
<tr>
<td>NO3-N</td>
<td>mgN/L</td>
<td>4</td>
<td>430</td>
<td>415</td>
<td>4</td>
<td>465</td>
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</tbody>
</table>

Figure 5 | DO concentration and pH values in test Period (a) II, (b) III, (c) IV and (d) V for Reactor A versus Reactor B.
SBR systems for treating leachate – comparison

Table 5 presents the comparison of both technologies, dispersed (A) and attached (B) biomass; in Period II Reactor A demonstrated advantages over Reactor B due to the longer adaptation time needed for attached growth system. In Periods III and IV both reactors, A and B showed comparable results. In Period V, Reactor A achieved constant NH$_4^+$-N removal efficiencies of around 96%; while the average NH$_4^+$-N removal efficiency was 91% in Reactor B. However, once stabilized (on Day 119), the nitrification efficiency also reached 96% and the same ammonium removal capacity was achieved in both reactors (0.39 kgN/d).

The results showed that the attached biomass needed a longer adaptation time to overgrow the gel beads, due to their porous matrix. Biomass carriers can also improve the treatment process by increasing the biomass concentration in the reactor (Yusoff et al. 2016). In several studies, it has been reported that the attached-growth processes have demonstrated greater efficiency and stability than dispersed-growth processes, especially at low temperatures (4–5 °C) in the presence of inhibitory substances and under high or variable loading conditions (Levstek & Plazl 2009; Yusoff et al. 2016). The results showed that the attached biomass reactor is less adaptable to cycle changes (reaction phases) in comparison with the dispersed system, as is evident at the beginning of Period V, in which the fluctuating NH$_4^+$-N removal efficiency can be seen.

According to test Periods IV and V, despite some differences between the observed systems A and B highlighted in this study, both systems can be used for the pre-treatment of leachate from solid municipal waste landfill in order to be acceptable for further processing at the municipal WWTP. Both kept the effluent NH$_4^+$-N levels (8 and 21 mg NH$_4^+$-N/L for Reactors A and B on average) much lower than the legal Slovene requirements (OG RS No. 98/15, OG RS No. 36/16) (200 mg of NH$_4^+$-N/L for discharging into a sewer).

While optimal operation for NH$_4^+$-N removal in the presented case was obtained using a 12-hour cycle with reaction time of 150 min of anoxic phase with mixing patterns and 510 min of aeration phase with external bicarbonate supplied, the relevant literature reports substantial diversity of operational strategies in a SBR (Table 6), but for treatment targets different from the one presented here, i.e. NH$_4^+$-N conversion to nitrate: Monclús et al. (2009) and Kulikowska (2012) used conventional nitrification and denitrification processes, while Ganigué et al. (2007), Li et al. (2014) and Wei et al. (2017) used partial nitrification (PN) to convert ammonium to nitrite. Li et al. (2014) used PN-SBR with end-point pH control. This variety of applications results in a variety of operational strategies and removal efficiencies as presented in Table 6. The majority of the experiments in Table 6 used dispersed biomass, except for Wei et al. (2017), using attached (sequencing Batch Biofilm Reactor (SBBR)) with synthetic inflow.

Similar NH$_4^+$-N removal efficiencies under the same cycle length were achieved by Monclús et al. (2009); however, an additional carbon source and longer anoxic phase was required for denitrification, while Kulikowska (2012) used a longer cycle (24 hours) to achieve a slightly higher removal efficiency.

As is evident from Table 6, different DO concentrations were used in the studies presented. In our experiment, the DO was maintained at a relatively high level (4–6 mg/L) to avoid any impact of the DO concentration on the

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Unit</th>
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<th>IV</th>
<th>V</th>
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<tbody>
<tr>
<td>NH$_4^+$-N</td>
<td>mgN/L</td>
<td>A</td>
<td>B</td>
<td>A</td>
<td>B</td>
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<tr>
<td>NH$_4^+$-N removal efficiency</td>
<td>%</td>
<td>76</td>
<td>73</td>
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<tr>
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<td>hour</td>
<td>24</td>
<td>24</td>
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<td>24</td>
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<tr>
<td>Anoxic reaction</td>
<td>hour</td>
<td>6.5</td>
<td>6.5</td>
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<tr>
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<td>16.0</td>
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<tr>
<td>Needed aeration</td>
<td>hour</td>
<td>6</td>
<td>14</td>
<td>8</td>
<td>10</td>
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<tr>
<td>Adaptation time</td>
<td>day</td>
<td>22</td>
<td>22</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Ammonium removal capacity</td>
<td>kgN/d</td>
<td>0.13</td>
<td>0.12</td>
<td>0.18</td>
<td>0.18</td>
</tr>
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</table>

*After stable conditions (Period II) – treatment efficiency >70%, Period (III, IV, V) – treatment efficiency >90%.
nitrification process. Thus, the results presented do not include optimized DO conditions. Further studies are necessary to estimate the optimal set point of DO concentration for each of the technologies.

**CONCLUSIONS**

This study provides an insight into two SBR-based technologies: (A) with dispersed biomass and (B) with attached biomass on PVA-gel carriers to reduce the NH$_4^+$-N concentration in the leachate from a solid municipal waste landfill. Despite some differences between A and B, both technologies can be used for pre-treatment of leachate from a solid municipal waste landfill to be acceptable for further processing by the municipal WWTP. The most effective technology was the one with dispersed biomass using the 12-hour cycle and pH adjustment with an anoxic phase at the beginning of the cycle. The results showed that 150 min of the anoxic phase followed by 510 min of aeration with pH adjustment were sufficient for effective nitrification, since the NH$_4^+$-N values dropped from 666 mg/L to below 10 mg/L in Reactors A and B. Both technologies achieved 96% removal efficiencies. However, technology A was more flexible due to shorter adaptation times in contrast to technology B, which required longer adaptation times to overgrow the gel beads due to their porous matrix. The results indicate that the attached biomass technology needs more constant conditions, as it is less adaptable to cycle changes in comparison with the dispersed system. The leachate treatment by the pilot SBR systems was very effective in comparison with other SBR systems found in the literature, achieving low effluent concentrations (below 10 mg/L) of NH$_4^+$-N/L in both reactors, while treating 600 L of leachate per day. This work presents results from a real case of landfill leachate. Due to the specific character of different leachate sources, the employed technologies are transferable to similar types of landfill leachate. Thus, the study has provided the relevant basis for the design of a full-scale plant to treat leachate on-site.

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