Co-treatment of old landfill leachate and municipal wastewater in sequencing batch reactor (SBR): effect of landfill leachate concentration

Kshitij Ranjan, Shubhrasekhar Chakraborty, Mohini Verma, Jawed Iqbal and R. Naresh Kumar

ABSTRACT

Sequencing batch reactor (SBR) was assessed for direct co-treatment of old landfill leachate and municipal wastewater for chemical oxygen demand (COD), nutrients and turbidity removal. Nitrogen removal was achieved by sequential nitrification and denitrification under post-anoxic conditions. Initially, SBR operating conditions were optimized by varying hydraulic retention time (HRT) at 20% (v/v) landfill leachate concentration, and results showed that 6 d HRT was suitable for co-treatment. SBR performance was assessed in terms of COD, ammonia, nitrate, phosphate, and turbidity removal efficiency, pH, mixed liquor suspended solids, mixed liquor volatile suspended solids (MLVSS), and sludge volume index were monitored to evaluate stability of SBR. MLVSS indicated that biomass was able to grow even at higher concentrations of old landfill leachate. Ammonia and nitrate removal efficiency was more than 93% and 83%, respectively, whereas COD reduction was in the range of 60–70%. Phosphate and turbidity removal efficiency was 80% and 83%, respectively. Microbial growth kinetic parameters indicated that there was no inhibition of biomass growth up to 20% landfill leachate. The results highlighted that SBR can be used as an initial step for direct co-treatment of landfill leachate and municipal wastewater.

Key words | co-treatment, landfill leachate, microbial growth kinetics, nitrification-denitrification, SBR

INTRODUCTION

Unscientific and inappropriate disposal of municipal solid wastes (MSW) is a major environmental problem particularly in poor and developing countries. Major problems associated with MSW landfills are leachate generation which contains a variety of solutes with high pollution potential. Inherent moisture present in wastes and rainwater percolation in landfills leads to leachate generation whereas decomposition of solid wastes by physical, chemical, and biological processes determines the leachate quality (Hassan et al. 2016). The quantity and quality of landfill leachate generated depends upon factors such as moisture content, level of compaction, waste composition, landfill age, co-disposal of liquid wastes, pre-treatment, particle size, density, precipitation, groundwater intrusion, leachate recirculation, waste settlement, vegetation, landfills cover, gas and heat generation and transport (Trabelsi et al. 2013).

Landfill leachate treatment options include spray irrigation on grasslands nearby, recirculation in landfills, co-treatment with municipal wastewater and evaporation ponds. Physico-chemical treatments such as aeration, coagulation-flocculation, filtration, adsorption, etc., and biological treatments such as anaerobic treatment, aerated lagoons, activated sludge plants, sequencing batch reactor (SBR) and rotating biological contactor (RBC), etc., have been widely studied (Schiopu & Gavrilescu 2010). Physico-chemical treatment technologies are expensive and generate by-products which require
further treatment before disposal. Biological treatment is routinely used for chemical oxygen demand (COD) and ammonia removal from young landfill leachate. Biological treatment may be ineffective for old landfill leachate with low BOD$_5$/COD ratio and a high concentration of toxic substances (Schiopu & Gavrilescu 2010). However, in a sequential scheme, biological treatment as a first step can offer several advantages, such as lesser sludge generation, COD and ammonia removal, and later, physico-chemical processes can be applied for non-biodegradable organic matter degradation.

Leachate treatment can be challenging for those landfills which are not located close to wastewater treatment plants. Often, an on-site landfill leachate treatment facility may not be economically and practically viable at all the locations. Moreover, depending on the climatic conditions, leachate production also ceases during dry months of the year, which works against an on-site treatment facility. Further, in developing countries like India, most of the MSW dumping sites are far from the city, which can pose challenges in leachate treatment. Thus, co-treatment of landfill leachate with municipal wastewater appears to be a promising option. Although there have been some earlier studies, a great deal of scope still exists for research to develop an efficient landfill leachate treatment. Co-treatment of landfill leachate with municipal wastewater could be a cost-effective alternative where the degradation of organic pollutants would also be favored due to dilution (Trabelsi et al. 2013). Co-treatment of landfill leachate with municipal wastewater will enhance BOD/COD ratio and make wastewater conducive for biological treatment. There have been studies where combined treatment of municipal wastewater and landfill leachate was successfully achieved, but most of the studies applied some pre-treatment or combined treatment before subjecting the wastewater mixture to a biological system (Aziz et al. 2011; El-Fadel et al. 2013; Trabelsi et al. 2013).

SBR has been reported to be a feasible technology for a variety of industrial wastewater treatments (Flapper et al. 2001). SBR cycles include: fill, react, settle, decant, and idle. SBR reaction phases can be adapted to achieve organic matter, suspended solids, and nutrients' removal in one reactor. In comparison to other biological treatment configurations SBR is advantageous, mainly due to factors such as better flexibility, limited space requirements, no requirement of settling tanks, and ease of automation (Bu et al. 2010; Aziz et al. 2011).

Biological treatment systems are influenced by environmental and engineering design factors such as pH, temperature, organic loading rate, solids retention time, etc. Experimentally optimizing all these parameters can be challenging, mainly due to the large number of experimental runs. To overcome such issues, experiments can be designed statistically by using central composite design or response surface methodology (RSM). RSM is often used to select the optimum process influencing parameters (Aziz et al. 2011). RSM collects mathematical and statistical tools which are useful for modeling and analysis of research problems where responses of interest are influenced by some variables (Bas & Boyaci 2007). The present scientific study was designed mainly to get preliminary data on SBR for leachate co-treatment. Based on the results of this work, further studies can be designed applying statistical tools such as Box–Behnken experimental design using RSM to assess the effects of key variables.

Widely used SBR variants at industrial level are C-Tech (cyclic activated sludge treatment) for biochemical oxygen demand (BOD) and ammonia removal, intermittent cycle extended aeration system, applied for removal of BOD, N, P, etc., and aquaSBR also applied for BOD, N, and P removal. Laboratory-scale studies have proved that SBR can be effective for the removal of COD, ammonia, and nitrate in landfill leachate. Most of the studies on landfill leachate treatment or co-treatment using SBR have focused mainly on ammonia and COD removal, whereas in the present study we also assessed nitrate, phosphate, and turbidity removal (Uygur & Kargi 2004; Spagni & Marsili-Libelli 2009; Capodici et al. 2014). Moreover, studies on the effect of landfill leachate concentration on SBR performance are scarce. The major objective of this study was to carry out direct landfill leachate and municipal wastewater co-treatment using SBR for COD, nitrogen (ammonia and nitrate), phosphate, and turbidity removal at different concentrations of landfill leachate.

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**MATERIALS AND METHODS**

**Landfill leachate, municipal wastewater, and activated sludge**

Landfill leachate was collected from MSW dumpsite located at Jhiri in Ranchi, Jharkhand, India. On a daily basis, 700
tons of mixed solid wastes generated in the city are collected and disposed of at the unlined open dumpsite without any treatment or processing. Municipal wastewater was collected from the wastewater treatment plant at the Birla Institute of Technology, Mesra, Ranchi, where wastewater comes from offices, hostels, faculty and staff quarters. Return activated sludge (RAS) collected from the aerobic wastewater treatment plant was used for starting SBR.

**SBR design and operation**

Experiments were carried out in a laboratory-scale SBR, fabricated using acrylic sheets with dimensions of 14.8 cm (L) × 9.4 cm (W) × 30 cm (H). Total volume of SBR was 4.2 L and the working volume was 3 L. The outlet of the SBR for effluent removal was provided at 4.5 cm from the bottom to prevent unwanted loss of active biomass. Agitation in the SBR was provided by using a magnetic stirrer throughout the reaction period. Air was supplied to the SBR using an aquarium pump with two stone diffusers located at the bottom of the reactor. Every day’s cycle was divided into 5 min of filling phase, 23 h of reaction phase (aeration and/or agitation), 50 min of settling phase, and 5 min of decant phase. The total reaction phase was divided into a 17 h aerobic (nitrification) phase and a 6 h anoxic (denitrification) phase. Dissolved oxygen was maintained at 3–4 mg/L throughout the oxic phase of the cycle. During the anoxic phase, aeration was turned off in the SBR to achieve conditions favorable for denitrification. Hydraulic retention time (HRT), sludge retention time (SRT), and other conditions were optimized one by one to achieve better treatment efficiency. Whenever any condition was changed, SBR was allowed to establish by adding external nutrients such as glucose and sodium acetate, ammonium chloride, and potassium di-hydrogen phosphate at COD/N/P ratio of 100:6:2. Once stable biomass concentration was achieved in the SBR, HRT optimization experiments were started.

Optimization studies were initiated at 6 d HRT by adding 2% (v/v) of landfill leachate to municipal wastewater which was continued until a leachate concentration of 20% (v/v) was reached. Thereafter, the landfill leachate concentration was kept at 20% (v/v) and HRT was lowered gradually from 6 to 2.5 d. The SBR was operated at four different HRT (6, 5, 3, and 2.5 d) to evaluate organic matter, nutrients, and turbidity removal efficiency. Glucose was added at the start of the aerobic phase to establish a BOD/COD ratio >0.5 since even after mixing of municipal wastewater with landfill leachate, the BOD/COD ratio remained lower than that required for biological treatment. Sodium acetate was added as substrate at the start of the anoxic phase to enhance denitrification efficiency. Following HRT optimization, the SBR was re-started where 6 d HRT was maintained. To study the influence of landfill leachate concentration, co-treatment in the SBR was started by adding 2% (v/v) of landfill leachate to municipal wastewater. Landfill leachate concentration in the influent fed to the SBR as daily volumetric exchange ratio was slowly increased to 5, 10, 15, 20, 25, 30, 35, and 40% (v/v).

**Analysis**

Samples were collected from the SBR at the beginning and end of each treatment cycle to assess SBR stability and organic matter, nutrients, and turbidity removal efficiency. pH was measured using a multiparameter meter (Horiba, Japan) on a regular basis. Total suspended solids (TSS), mixed liquor suspended solids (MLSS), mixed liquor volatile suspended solids (MLVSS), and sludge volume index (SVI30) were determined as per *Standard Methods* (APHA 1992). COD, ammonia, nitrate, phosphate, and turbidity analyses were carried out as per *Standard Methods* (APHA 1992).

**Microbial growth kinetics**

Microbial growth kinetics helps in performance assessment of biological treatment systems (Kulikowska et al. 2007;
El-Fadel et al. 2013). Data were analyzed using mathematical equations to determine growth kinetic parameters, namely, biomass yield coefficient (\( Y_{\text{obs}} \)) and biomass decay rate (\( k_d \)).

### Biomass yield coefficient

Biomass yield coefficient (\( Y_{\text{obs}} \)) quantifies biomass generated per unit mass of COD consumed. \( Y_{\text{obs}} \) was calculated using the following equation (Kulikowska et al. 2007; El-Fadel et al. 2013):

\[
Y_{\text{obs}}(\tau) = \frac{X_{\text{org}} \times (V_w/\tau_e) + X_e \times (V_{\text{eff}}/\tau_e)}{(C_s - C_e) \times (V_{\text{in}}/\tau_e)}
\]

where,

- \( X_{\text{org}} \) = volatile suspended solids in SBR (mg VSS/L);
- \( V_w \) = volume of suspended solids disposed in SBR operating cycle (L);
- \( \tau_e \) = time of SBR operating cycle (d);
- \( X_e \) = effluent volatile suspended solids concentration (mg VSS/L);
- \( V_{\text{eff}} \) = volume of effluent from SBR operating cycle (L);
- \( V_{\text{in}} \) = volume of influent to SBR operating cycle (L), \( V_{\text{in}} = V_{\text{eff}} + V_w \);
- \( C_s \) = influent COD (mg COD/L);
- \( C_e \) = effluent COD (mg COD/L).

### Biomass decay rate

Endogenous metabolism of bacteria results in decaying of biomass, which is routinely expressed by biomass decay rate (\( k_d \)). \( k_d \) establishes correlation between sludge age, biomass yield coefficient (\( Y_{\text{obs}} \)), and biomass loss. Biomass decay rate was calculated using the following equation (Kulikowska et al. 2007; El-Fadel et al. 2013):

\[
\frac{1}{\theta} = Y \times \frac{(C_s - C_e) \times (V_{\text{eff}}/\tau_e)}{V \times X_{\text{org}}} - k_d
\]

where,

- \( \theta \) = solids retention time (SRT) (d);
- \( Y \) = biomass yield coefficient (mg VSS/mg COD);
- \( C_s \) = influent COD (mg COD/L);
- \( C_e \) = effluent COD (mg COD/L);
- \( V_{\text{eff}} \) = volume of influent in SBR operating cycle (L);
- \( \tau_e \) = time of SBR operating cycle (d);
- \( V \) = SBR working volume (L);
- \( X_{\text{org}} \) = volatile suspended solids in SBR (mg VSS/L);
- \( k_d \) = biomass decay rate (d).

### Data analysis

Data were processed using MS Excel and SigmaPlot (Version 15) for preparing figures and tables. Tests for differences in mean concentrations of COD, ammonia, nitrate, and phosphate at different landfill leachate concentrations were performed using univariate analysis of variance (ANOVA). If significance was found, mean concentrations of sub-categories were compared using ANOVA followed by Tukey post hoc analysis. ANOVA was also performed to determine the variance of microbial growth kinetic parameters at different leachate concentrations. Data were also subjected to correlation analysis to understand the relationship between different landfill leachate concentrations and HRT with treated wastewater characteristics. All the statistical tests were carried out using Microsoft Excel with the Analysis Toolpak.

### RESULTS AND DISCUSSION

#### Characterization of landfill leachate and wastewater

Landfill leachate was in a stabilized state as indicated by low BOD/COD ratio and slightly high pH (Table 1). High ammonia concentration and comparatively low nitrate concentration in landfill leachate also proved that the MSW dumpsite is in a methanogenic phase (Schiopu & Gavrilescu 2010). Municipal wastewater contained moderate BOD concentration and due to this reason, even after wastewater addition to leachate, BOD/COD ratio remained <0.5. In order to overcome this, glucose was added to the wastewater mixture to maintain BOD/COD ratio >0.5 required for biological treatment. Such issues of low BOD/COD ratio may not be evident at field-level application where municipal wastewater might contain high BOD concentration;
Table 1 | Mean (±SD) physico-chemical characteristics of municipal wastewater and landfill leachate

<table>
<thead>
<tr>
<th>Parameters</th>
<th>Municipal wastewater</th>
<th>Landfill leachate</th>
</tr>
</thead>
<tbody>
<tr>
<td>pH</td>
<td>7.0 ± 0.3</td>
<td>7.8 ± 0.4</td>
</tr>
<tr>
<td>EC (mS/cm)</td>
<td>0.74 ± 0.2</td>
<td>8.9 ± 1.9</td>
</tr>
<tr>
<td>ORP (mV)</td>
<td>60 ± 10</td>
<td>167 ± 20</td>
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<tr>
<td>TDS (mg/L)</td>
<td>450 ± 40</td>
<td>6,700 ± 2,160</td>
</tr>
<tr>
<td>TSS (mg/L)</td>
<td>460 ± 20</td>
<td>6,400 ± 2,500</td>
</tr>
<tr>
<td>BOD (mg/L)</td>
<td>240 ± 100</td>
<td>70 ± 45</td>
</tr>
<tr>
<td>COD (mg/L)</td>
<td>440 ± 140</td>
<td>4,000 ± 1,950</td>
</tr>
<tr>
<td>Ammonia (mg/L)</td>
<td>40 ± 2.2</td>
<td>290 ± 110</td>
</tr>
<tr>
<td>Nitrate (mg/L)</td>
<td>0</td>
<td>22 ± 9</td>
</tr>
<tr>
<td>Phosphate (mg/L)</td>
<td>10 ± 4</td>
<td>51 ± 45</td>
</tr>
</tbody>
</table>

*n = 20 for wastewater; n = 30 for landfill leachate.

furthermore, the volume of landfill leachate addition will be lower when compared to the volume of wastewater in sewage treatment plants.

**HRT optimization**

HRT was gradually varied from 6 to 2.5 d to assess the SBR performance for co-treatment at 20% (v/v) leachate concentration for COD, ammonia, and nitrate removal (Figure 1(a)–1(e)). SVI<sub>50</sub> was always >40 mL/g during experimental runs, and lower SVI could be attributed to microbial growth inhibitory effects of mature landfill leachate. However, MLVSS in the SBR was always >2,000 mg/L which is in agreement with the literature. MLVSS concentration was slightly higher at 6 d HRT when compared with 3 and 2.5 d HRT, which supports the fact that higher landfill leachate concentration at these HRT may have affected microbial growth. COD removal efficiency was >60% at all HRT studied. Ammonia removal efficiency dropped with the decrease in HRT. Ammonia removal was mainly due to nitrification and not by other mechanisms, such as volatilization which requires high pH, and this fact is substantiated by the measured nitrate concentration (Figure 1(e)). Correlation coefficient at different HRT and effluent characteristics indicated a strong negative relation between HRT and ammonia concentration in the effluent (−0.84) (Table 2). Correlation analysis showed that increasing HRT resulted in significant ammonia removal. There was significant negative correlation between ammonia and nitrate (−0.69) which supports nitrification as a major ammonia removal mechanism. HRT results subjected to one-way ANOVA exhibited a statistically significant difference between treatments (F<sub>16,38</sub> = 74.7, P < 0.001). Thus, for assessing the effects of different leachate concentrations on SBR performance, 6 d HRT was selected whereas SRT was 30 d.

**Effect of landfill leachate concentration on SBR performance**

During co-treatment, pH in the SBR was in the range of 7.2 to 8. There was always a slight increase in pH at the end of every day’s cycle under anoxic conditions. It is known that CO<sub>2</sub> is released in anoxic phase reactions from a SBR which leads to an increase in pH (Zhang et al. 2006). A decrease in pH at the end of the aerobic phase in the SBR is due to the release of H<sup>+</sup> ions during nitrification, whereas an increase in pH after the anoxic phase is due to denitrification (Melidis 2014).

Variations in sludge characteristics, organic matter, nutrients, and turbidity removal efficiency during co-treatment of landfill leachate and municipal wastewater in the SBR is presented in Figure 2(a)–2(g). SVI is a major parameter used to determine sludge settling properties in activated sludge processes. SVI <100 mL/g is of prime importance in any activated sludge system to achieve better effluent clarification. SVI and sludge volume (SV) were determined for each cycle to assess sludge settling characteristics (Figure 2(a)). SVI was 80 mL/g when only municipal wastewater was fed into the SBR, whereas during co-treatment when landfill leachate was added, SVI dropped considerably. Lower SVI in the SBR compared to the usual ~100 mL/g could be due to the inhibitory effects of old landfill leachate. Old landfill leachate is known to contain a variety of dissolved organic matter such as humic substances, hydrocarbons, proteins, lipids, carbohydrates, carboxylic acids, amino acids, and hydrophilic acids (Leenheer & Croué 2005; Bu et al. 2010). These substances can have a serious impact on microbial activity and on pollutant removal efficiency. These substances can also become converted to toxic byproducts such as poly-aromatic hydrocarbons,
adsorbable organic halogens, polychlorinated biphenyls during treatment (Bu et al. 2010; Fudala-Ksiazek et al. 2014). Biological treatment can be affected by these toxic substances and/or by the presence of some other non-biodegradable organic compounds like surfactants, detergents, wetting agents, emulsifiers, foaming agents, and dispersants (Wiszniowski et al. 2006).

Table 2 | Correlation coefficient between HRT and effluent characteristics

<table>
<thead>
<tr>
<th>HRT</th>
<th>COD</th>
<th>Ammonia</th>
<th>Nitrate</th>
</tr>
</thead>
<tbody>
<tr>
<td>HRT</td>
<td>1</td>
<td></td>
<td></td>
</tr>
<tr>
<td>COD</td>
<td>-0.15</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>Ammonia</td>
<td>-0.84*</td>
<td>-0.26</td>
<td>1</td>
</tr>
<tr>
<td>Nitrate</td>
<td>0.57*</td>
<td>-0.29</td>
<td>-0.69*</td>
</tr>
</tbody>
</table>

*Denotes significant correlation.
Janczukowicz et al. (2001) have also reported SVI in the range of 30–60 mL/g while assessing activated sludge settling properties in SBR. Low SVI in the SBR during co-treatment led to conditions which favored effective sludge settling. Generally, it has been reported that SVI <70 mL/g leads to conditions which result in turbid effluent. However, in the present study, considerable turbidity removal was achieved (see Figure 2(c)). Initially, SV was 369 cm³ with municipal wastewater alone, but it decreased with the increase in landfill leachate concentration. Decrease in SV could be due to high leachate load which can increase the concentration of inhibitory substances in SBR. MLSS and MLVSS were monitored regularly to assess the viability of biomass during SBR runs. Stable and steady increase in MLSS was recorded at the end of the daily cycle. MLVSS was always >2,000 mg/L at different landfill leachate concentrations. MLSS and MLVSS were not much affected with landfill leachate up to 30% (v/v), whereas both declined at 35 and 40% leachate concentrations.

COD removal was in the range of 50–85% during the co-treatment process (Figure 2(d)). Maximum COD removal was 73% at 10% landfill leachate concentration. COD removal efficiency declined slightly with the increase in landfill leachate concentration. Comparable COD removal efficiencies have been reported in other studies on SBR (Uygur & Kargi 2004; Neczaj et al. 2007). However, Capodici et al. (2014) reported 88% COD removal at 50% landfill leachate concentration when SBR was used as post-treatment. Slightly lower COD removal efficiency achieved in the present study, when compared to that reported in the literature, could be due to higher concentration of refractory matter in old landfill leachate (Capodici et al. 2014). Moreover, higher SRT maintained in the SBR for ammonia, nitrate, and phosphate removal could be responsible for moderate COD removal (El-Fadel et al. 2013). For instance, effluent from biological treatment will contain non-biodegradable fractions of COD initially present and additional non-biodegradable fractions (both termed as soluble microbial products) produced during biological treatment (Kulikowska et al. 2007). Sludge age has been reported to play an important role in generation of soluble microbial products in activated sludge systems (Pribyl et al. 1997). In order to achieve higher COD removal, SBR should be followed by alternative physico-chemical treatments such as adsorption, filtration, electro-coagulation, ozonation, etc.

Nitrification efficiency in SBR was 97% when fed with municipal wastewater alone, whereas the efficiency ranged between 56 and 90% during landfill leachate co-treatment (Figure 2(e)). High ammonia removal (90%) could be achieved at 25% leachate concentration and there was no considerable decline in nitrification efficiency with the increased leachate concentration. Optimum pH for ammonia-oxidizing bacteria is 7–8.5 and for nitrite oxidizing bacteria is 6.0–7.5 (Groeneweg et al. 1994). During the present study, pH in the SBR was always between 7 and 8, which supports nitrification as the dominant mechanism of ammonia removal. Ammonia removal achieved in the present study is in agreement with that reported in the literature for landfill leachate treatment (Neczaj et al. 2007; Spagni & Marsili-Libelli 2009).

Post-anoxic mode denitrification was applied for nitrate removal with sodium acetate as substrate to increase the process efficiency. Maximum nitrate reduction (83%) occurred at 25% landfill leachate concentration (Figure 2(f)). Denitrification efficiency dropped considerably at 40% landfill leachate concentration. Increased leachate concentration may produce adverse effects on the growth of denitrifying bacteria. Similar denitrification efficiency has been reported by several researchers (Diamadopoulos et al. 1997; Debsarkar et al. 2006; Melidis 2014).

Most of the studies on landfill leachate treatment or co-treatment using SBR have mainly focused on ammonia and COD removal, whereas in the present study, phosphate removal was also assessed. Maximum phosphate removal (90%) occurred at 10% and 15% landfill leachate concentrations and the P removal efficiency ranged from 70 to 80% (Figure 2(g)). The majority of phosphate removal occurred in the aerobic phase and the anoxic phase P concentration did not show any significant change, which is in good agreement with that reported in the literature (Melidis 2014).

Correlation coefficient between different landfill concentrations and effluent characteristics (Table 3) showed positive correlation between leachate concentration and effluent COD concentration (0.87). This indicates that COD reduction efficiency decreased with increased leachate concentration. Effluent characteristics were also subjected
Figure 2 | Mean (n = 3, ±SD) variations in (a) SV and SVI, (b) MLSS and MLVSS, (c) turbidity, (d) COD, (e) ammonia, (f) nitrate, and (g) phosphate at different landfill leachate concentration during co-treatment with municipal wastewater in SBR.
to univariate ANOVA test, where significant difference was found between different landfill leachate concentrations ($F_{0.99} = 57.07$, $P < 0.001$). Post hoc t-test revealed that COD removal was not significantly different up to 15% leachate concentration, whereas above this concentration, data showed significant differences. Post hoc t-test for ammonia removal showed that up to 25% leachate concentration there was no statistically significant difference whereas above this concentration there was significant difference.

**Microbial growth kinetics of co-treatment process in SBR**

In activated sludge processes (SBR, ASP, etc.), COD and nutrients are removed by different types of bacteria, such as aerobic ammonia oxidizers, nitrite oxidizers, anaerobic ammonia oxidizers, phosphate-oxidizing bacteria, and other heterotrophic bacteria (Jungles et al. 2014; Melidis 2014). Using mixed microbial culture as inoculum for the reactor is helpful for a establishing diverse microbial community (Wiszniewski et al. 2006). Mixed culture enhances biomass retention where compact and dense microbial granules can be produced, and has the ability to resist high organic loading rate and toxic pollutant loadings (Yang et al. 2004).

Biological kinetics is important for determining the overall treatment design and operation. Several operating parameters, such as MLVSS ($X_{mg}$), sludge wasting/day ($V_w$), cycle time ($t_c$), effluent MLVSS ($X_e$), volume of influent ($V_{in}$), volume of effluent ($V_{eff}$), COD of influent ($C_i$), and COD of effluent ($C_e$) were first compiled to determine the kinetic parameters.

First, biomass yield coefficient ($Y_{obs}$) was estimated to quantify biomass produced per unit of substrate utilized. During co-treatment studies, significant variation was found in biomass yield coefficient (Table 4). Maximum $Y_{obs}$ of 0.64 mg VSS/mg COD was found at 20% landfill leachate concentration, whereas increased leachate concentration resulted in lowering of $Y_{obs}$ (0.15 mg VSS/mg COD) at 40% landfill leachate. Kulikowska et al. (2007) reported $Y_{obs}$ 0.55–0.6 mg VSS/mg COD for SBR operated at different HRT (2 to 12 d). These authors also highlighted that sludge age in an SBR was an important parameter to maintain VSS concentration. The literature indicates that biomass yield coefficient in reactors with anoxic phase would be 35–52% higher than reactors with aerobic phase alone (Lin et al. 2000).

Biomass decay rate ($k_d$) was in the range of 0.33 to 0.55 d$^{-1}$. Nitrifying bacteria performance in activated sludge systems with varying oxidation reduction potential (ORP) highlighted average autotrophic decay rate of aerobic, anoxic, and anoxic/aerobic reactors at 0.15, 0.1, and 0.06 d$^{-1}$, respectively (Lee & Oleszkiewicz 2003). More than a five-fold increase in $k_d$ values have been reported under aerobic conditions compared to anoxic conditions (Kulikowska et al. 2007). Kinetic parameters of SBR

<table>
<thead>
<tr>
<th>Leachate concentration</th>
<th>COD</th>
<th>Ammonia</th>
<th>Nitrate</th>
<th>Phosphate</th>
</tr>
</thead>
<tbody>
<tr>
<td>Leachate concentration</td>
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<td>0.87*</td>
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<tr>
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<td>1</td>
</tr>
<tr>
<td>Nitrate</td>
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<td>-0.21</td>
<td>1</td>
<td>1</td>
</tr>
<tr>
<td>Phosphate</td>
<td>-0.21</td>
<td>1</td>
<td>1</td>
<td>1</td>
</tr>
</tbody>
</table>

*Denotes significant correlation.

Table 3 | Correlation coefficient between landfill leachate concentration and effluent characteristics

<table>
<thead>
<tr>
<th>Correlation coefficient</th>
<th>COD</th>
<th>Ammonia</th>
<th>Nitrate</th>
<th>Phosphate</th>
</tr>
</thead>
<tbody>
<tr>
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<td>1</td>
</tr>
<tr>
<td>Ammonia</td>
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<td>-0.21</td>
<td>1</td>
<td>1</td>
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<tr>
<td>Nitrate</td>
<td>-0.21</td>
<td>1</td>
<td>1</td>
<td>1</td>
</tr>
</tbody>
</table>

Table 4 | Mean ($n = 3$, ±SD) variations in biomass yield coefficient and decay rate during co-treatment in SBR at different leachate concentrations

<table>
<thead>
<tr>
<th>Landfill leachate (%)</th>
<th>$Y_{obs}$ (mg VSS/mg COD)</th>
<th>$k_d$ (d$^{-1}$)</th>
</tr>
</thead>
<tbody>
<tr>
<td>0</td>
<td>0.31 ± 0.05</td>
<td>0.36 ± 0.01</td>
</tr>
<tr>
<td>5</td>
<td>0.34 ± 0.01</td>
<td>0.35 ± 0</td>
</tr>
<tr>
<td>10</td>
<td>0.50 ± 0.03</td>
<td>0.33 ± 0</td>
</tr>
<tr>
<td>15</td>
<td>0.46 ± 0.02</td>
<td>0.52 ± 0.02</td>
</tr>
<tr>
<td>20</td>
<td>0.64 ± 0.03</td>
<td>0.44 ± 0.01</td>
</tr>
<tr>
<td>25</td>
<td>0.63 ± 0.04</td>
<td>0.52 ± 0.03</td>
</tr>
<tr>
<td>30</td>
<td>0.50 ± 0.02</td>
<td>0.55 ± 0.03</td>
</tr>
<tr>
<td>35</td>
<td>0.14 ± 0.01</td>
<td>0.43 ± 0.01</td>
</tr>
<tr>
<td>40</td>
<td>0.13 ± 0.01</td>
<td>0.45 ± 0.01</td>
</tr>
</tbody>
</table>
indicate that in spite of increased leachate concentration, MLVSS was within the effective range for aerobic and anoxic phases necessary to maintain reactor efficiency. ANOVA (post hoc t-test) indicated that there was no statistically significant difference in \( Y_{obs} \), up to 30% landfill leachate concentration, whereas above 30% leachate concentration \( Y_{obs} \) was significantly affected. On the other hand, for \( k_d \), up to 15% landfill leachate showed no significant difference, whereas at \( \geq 20\% \) leachate concentration there was significant difference.

Overall, co-treatment of landfill leachate and municipal wastewater in SBR was able to remove ammonia and phosphate which met the standards for discharge in inland surface water (Table 5) (source: CPCB, India). Although SBR was largely effective for co-treatment, effluent COD and nitrate concentration was not up to discharge standards. Nevertheless, considerable COD and nitrate reduction could be achieved and further sequential treatment with a physico-chemical process might prove efficient in meeting the discharge standards.

CONCLUSIONS

Results of the study proved that SBR can be an efficient technology for direct co-treatment of landfill leachate and municipal wastewater. Activated sludge from a wastewater treatment plant was found to be an effective inoculum for SBR startup as indicated by stable biomass concentration. HRT optimization revealed that the SBR produced optimum results at 6 d HRT and 30 d SRT at 20% (v/v) landfill leachate concentration. Co-treatment showed significant COD, ammonia, nitrate, phosphate, and turbidity removal up to 20–25% landfill leachate concentration. SBR treatment efficiency dropped at 35 and 40% landfill leachate concentration. Microbial growth kinetic parameters revealed that growth and MLVSS concentration was within the range for effective SBR performance. Biomass yield coefficient and biomass decay rate was 0.64 mg VSS/mg COD and 0.44 d\(^{-1}\), respectively, at 20% leachate concentration, which indicated the efficacy of biomass for organic matter removal up to this concentration. SBR can be used as an efficient process to treat complex landfill leachate along with municipal wastewater in a sustainable way without any pre-treatment. However, co-treatment was not effective as a stand-alone process and further physico-chemical treatment will be necessary to meet the discharge standards.

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REFERENCES


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