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Comparing young landfill leachate treatment efficiency and process stability using aerobic granular sludge and suspended growth activated sludge

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ABSTRACT

The treatment of young landfill leachate was investigated using two 3 L sequencing batch reactors (SBR) with different biomass: aerobic granular sludge (GSBR) and the suspended growth activated sludge (ASBR). According to the nature of aerobic granular sludge, high settling velocities were expected in the GSBR and it was confirmed with 60 ml/g VSS of 5-min sludge volume index (SVI_5). However, the activated sludge required 30 min for the ASBR to achieve a SVI of 42 ml/g VSS. Compared to the ASBR, the GSBR was also more efficient in nitrogen and carbon removal. During the steady period of the experiment, 99% of total ammonium nitrogen (TAN) was removed through nitritation and nitrification in the GSBR with an average influent TAN concentration of 498 mg/L. With high influent TAN concentration, the GSBR could achieve a full nitrification efficiency of 56 ± 12% without accumulating nitrite. On the contrary, complete nitrification was not achieved in the ASBR as it was exposed to high concentrations of free ammonia (FA) and free nitrous acid (FNA), 16 and 0.2 mg/L respectively. Partial nitrification (nitritation) with the efficiency of 77 ± 10% was observed in the ASBR. The GSBR also presented higher efficiency in denitrification compared to the ASBR. 23% denitrification was observed in the GSBR during the anaerobic phase. Moreover, higher chemical oxygen demand (COD) removal efficiency was observed in GSBR than ASBR. Phosphorus removal efficiency was almost identical in both reactors. Overall, compared with the activated sludge, the aerobic granular sludge showed the best nutrients removal performance and higher tolerance to toxic compounds in young landfill leachate.

1. Introduction

In the last decade, waste management strategies have been directed to waste minimization (reduction, recover, reuse and recycling), encouraging final disposal alternatives to landfilling. The European Commission has proposed to Member States that a maximum of 10% of municipal solid wastes should be landfilled by 2030 [1]. Nonetheless, sanitary landfills still represent the most often used final disposal alternative for municipal solid waste in the world. Although landfilling comprises a well-established solution for waste management, it causes environmental impacts such as greenhouse gas and leachate production [2]. Leachate is a wastewater with a diverse composition, including inorganic salts, heavy metals, high levels of total ammonium nitrogen (TAN), both biodegradable and refractory organic matter, and xenobiotic organic compounds. Based on its composition, leachate may be classified as young or old. Young leachate contains more volatile fatty acids (VFA) and has higher level of BOD (Biological oxygen demand)/COD ratio (> 0.3) [2,3]. Old leachate usually contains high TAN concentrations and low BOD/COD ratio (< 0.3) as a result of organic matter stabilization under anaerobic conditions [2].

Leachate treatment strategies include on-site treatment plants [4], transport to wastewater treatment plant and co-treatment with domestic wastewater [5] and reinjection or recycle to the landfill cell [3]. Physico-chemical, membrane and biological processes are among the successful methods reported for leachate treatment.

Air striping has been commonly used to reduce the TAN concentration in leachate. Ferraz et al. [6] achieved 88% of TAN removal treating 12 L of leachate for 72 h using an aerated packed tower. Yuan et al. [3] found that adjusting the pH to 11 enhanced air stripping efficiency, whereas it was neutralized to 7.5 as leachate was further treated by a biological process. Accordingly, proper pH adjustments prior to biological process are necessary to prevent nitrification inhibition. Tang and Chen [7] reported 95% of nitrification inhibition when the aeration tank of an activated sludge reactor treating domestic wastewater presented pH 6.5, whereas at pH 7.2 nitrification inhibition was less than 40%. Among other physico-chemical methods for leachate treat-

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Table 1
Composition of the synthetic leachate.

Inorganic Compounds		Organic Compounds		Trace metal solution	
Components	Per litre	Components	Per litre	Components	Per litre
NaCl	2000 mg	Acetic acid	0.075 ml	$CoSO_4 \cdot 7H_2O$	150 mg
$CaCl_2$	700 mg	Propionic acid	0.075 ml	H_3BO_4	50 mg
$NaHCO_3$	2000 mg	Butyric acid	0.075 ml	$ZnSO_4 \cdot 7H_2O$	50 mg
NaOH	297 mg	Acetone	0.0525 ml	$CuSO_4 \cdot 5H_2O$	40 mg
K_2HPO_4	32.5 mg	Ethanol	0.0525 ml	$MnSO_4 \cdot 7H_2O$	500 mg
NH_4Cl	120–500 mg	Propanol	0.0525 ml	$(NH_4)_6Mo_7O_{24} \cdot 4H_2O$	50 mg
Trace metal	0.02 ml	Phenol	0.02 ml	$Al_2(SO_4)_3 \cdot 16H_2O$	30 mg
				$NiSO_4 \cdot 6H_2O$	500 mg
				96% H_2SO_4	1 ml

ment, electrochemical process could remove 82% of TAN and 87% of COD from raw leachate at a high current level (200 mA/cm^2) [8]. Membranes are typically used at final stage of leachate treatment, improving the quality of the pre-treated leachate in order to attend local discharge limits. A forward osmosis (FO) membrane system applied to a pre-treated leachate removed 98.6% of COD, 96.6% of total phosphorus (TP) and 76.9% of TAN [9].

Regarding biological treatment, the co-treatment of leachate with domestic wastewater has been extensively reported by the literature, presenting COD removal efficiencies up to 90% and satisfactory nutrients removal [3,5,35,36]. Despite the good results related to leachate co-treatment with domestic wastewater, there are important concerns about this alternative. As most existing wastewater treatment plants were not originally designed to treat leachate, is it unclear how they will perform at long-term receiving leachate. Another concern is the possibility of leachate to be simply diluted instead of effectively being co-treated. In fact, it was reported that raw leachate was most probably diluted with domestic wastewater rather than biodegraded in a SBR when co-treated at a volumetric ratio of 5% [5].

Alternatively, leachate can be treated on-site where facilities designed to attend its specific characteristics are used. Significant results have been reported from leachate treatment by biological systems that could be installed on-site, including: fungi, activated sludge and aerobic granular sludge [10,11,12]. White rot fungi could remove 78% of color and 52% of COD after 4 days of immobilization [13]. High removal efficiencies for COD (85%) and total nitrogen (90–95%) were obtained by full-scale sequencing batch reactor (SBR) plant treating leachate [37]. Promising results were also obtained when leachate was treated by aerobic granular sludge (AGS): COD removal was up to 83% and TAN removal was up to 44% [11].

Reactors based on the aerobic granular sludge present advantages over the conventional activated sludge process, such as: high settling velocity, compact structure, simultaneous nutrient removal, and ability to sustain high biomass concentration [14]. In spite of those advantages over activated sludge, the literature is lacking the application of aerobic granular sludge for leachate treatment, which has motivated the current research. This study aimed to compare the performances of activated sludge SBR (ASBR) and aerobic granular sludge SBR (GSBR) in treatment of a young leachate, focusing on organic matter and nutrients removal.

2. Materials and methods

2.1. Experimental set up

The returned activated sludge (RAS) was collected from the Southend Water Pollution Control Centre in Winnipeg, MB, Canada. The aerobic granular sludge was gradually cultivated from the RAS in a SBR. The GSBR consisted of a plastic column with a 45 cm height and a 12 cm internal diameter. The ASBR consisted of a glass jar with a 30 cm height cm and a 15 cm diameter. Total and working volumes of both reactors were the same: 5 L and 3 L, respectively. Both reactors were provided with an up-flow feeding. The air flow was kept between 2 and 3 L/min during the aerobic phase. The mixed liquor pH was kept higher than 6.5.

2.2. Reactor operation process

The ASBR and the GSBR were operated at 3 cycles per day. For the GSBR, each cycle consisted of 0.5-h feeding period, 1.5-h anoxic/anaerobic period, 5.5-h of aerobic period, 5-min settling period and 12-min decanting period. For the ASBR, each cycle consisted of 1.5-h anoxic/anaerobic feeding period, 5.5-h of aerobic period, 40-min settling period and 12-min decanting period. During mixing, there were 2 min for wasting the sludge in ASBR. For both reactors, at each cycle, 1.5 L of supernatant (treated effluent) was withdrawn and 1.5 L of fresh feed was pumped into reactors, keeping an exchange ratio of 50%.

2.3. Synthetic leachate composition

The synthetic young leachate recipe was prepared according to Vangulck et al. [15]. As shown in Table 1, the components contained three kinds of volatile fatty acids (VFA), hardly biodegradable organic compounds, different salts and trace metals. Every four days the feed was prepared in 20 L jars and stored at 4 °C. In order to increase the influent TAN, the concentrations of NH_4Cl used for feed preparation varied from 120 to 500 mg/L (Table 1). For the GSBR, the influent TAN at the beginning of the experiments was 100–150 mg/L, being gradually increased along with time to the ranges of 300–380 mg/L and 400–500 mg/L. The same procedure was applied to the ASBR and the tested influent TAN ranges were 130 mg/L, 300–330 mg/L and 400–450 mg/L.

2.4. Analytical methods

The experiment was conducted at room temperature (20 ± 2 °C). TAN, nitrate (NO_3^-—N), nitrite (NO_2^- —N) and phosphate (PO_4^-—P) concentrations were detected by Lachat Instrument QuikChem 8500. Hach COD kits were used to determine the soluble COD from feed and effluent samples. Mixed liquor suspended solids (MLSS), mixed liquor volatile suspended solids (MLVSS) and sludge volumetric index (SVI) were performed according to the Standard Methods [16]. SVI was measured in 5 min for the GSBR (SVI_5) and in 30 min for the ASBR (SVI_{30}). Extracellular polymeric substances (EPS) were extracted according to Adav et al. [14]. Polyprotein (PN) and polysaccharides (PS) concentration in EPS were measured by the modified Lowry assay kits and phenol-sulfuric acid colorimetric method [17]. Sludge particle size was determined by Malvern Mastersize 2000 analyzer. Free-ammonia (FA) and free-nitrous acid (FNA) were calculated according to Eqs. (1) and (2) [18], while nitrite accumulation rate (NAR) was determined by Eq. (3) [19]. Denitrification and simultaneous nitrification-denitrifica-