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Effect of a geotextile filter on leachate quality in a recirculated landfill bioreactor

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ABSTRACT

Landfills cause various problems for local authorities, such as contamination of soil and water with toxins, the formation of leachate and the release of landfill gases. More economical and applicable innovative solutions to overcome these problems will be an advantage for local authorities. One of the problems that local authorities face during the operation of landfills is the cost of leachate treatment due to high energy consumption. The objective of this study is, with the use of two laboratory-scale anaerobic bioreactors, to improve leachate quality by using a polymeric geotextile (GT) material placed horizontally in the drainage layer of a lab-scale landfill bioreactor (LBR). The simulated LBR equipped with the geotextile filter (LBR-GT) achieved faster leachate quality improvement than the control reactor (LBR-C). Scanning electron microscope images showed that the GT filters allowed the biofilm to grow not only on the surface but also in the interior pores, which increased the interactions between the biomass and the organics. In this way, the leachate quality improved in a short time as a result of the high biomass growth in the GT filter. The chemical oxygen demand (COD), the 5-d biochemical oxygen demand (BOD_5), the pH, the oxidation reduction potential (ORP) and other operating parameters in the leachate were regularly monitored. The LBR-GT reached a –300 mV ORP value on the 54th day, while the LBR-C reached the same ORP value on the 145th day. After 208 d of anaerobic incubation, the removal rates for the COD and BOD_5 in LBR-C were 93% and 96%, respectively, whereas in the LBR-GT, the removal rates were 96% and 99%, respectively. The main result of this study was that the LBR-GT took only 90 d to reach 90% COD removal rate, whereas the LBR-C took 166 d to reach the same removal rate.

Keywords: Solid waste; Leachate quality; Geotextile; Landfill bioreactor

1. Introduction

In developing countries, landfilling is the major municipal solid waste (MSW) management method because it is easier to install and operate than composting, incinerating and recycling [1,2]. Solid wastes pose risks to the environment during their transfer, transport and final disposal. MSWs are one of the main sources of the contamination of air, soil and water. Engineers have designed many disposal methods, and experts have developed management strategies to minimise the detrimental effects of MSWs on both the environment and human health. These management strategies can be achieved by following integrated solid waste management (ISWM) plans. Unlike the conventional perspective, which aims to only avoid the adverse effects of MSWs on public health, the ISWM aims to manage MSWs more efficiently. ISWM is the selection and application of appropriate engineering technologies and management strategies to overcome all the potential dangers of solid wastes, and while doing this, it also aims to provide economic and social benefits. Four basic management strategies have been identified for ISWM: (1) waste prevention/source reduction, (2) recycling and composting, (3) combustion/incineration (waste-to-energy facilities) and (4) landfilling.

Closed landfills are capped with soil and impermeable layers to minimise rainfall infiltration. The leachates produced from the waste body in a landfill site have to be

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collected first and treated before being released into the environment. Conventional landfills are designed to restrict the exposure of air and water to the waste body in the landfill, thus reducing the potential health and environmental risks. However, this operation style considerably decelerates the biodegradation rates; therefore, the required time for the stabilisation of solid wastes becomes long. Stabilisation of a landfill is considered completed when landfill gas is in negligible amounts, at which point the leachate is no longer an environmental risk, and maximum settlement of the waste mass has taken place [3].

A closed landfill should be monitored by recirculating the leachate back to the waste mass until all the toxic compounds that are harmful to the environment are completely removed. This approach is possible only if the landfill is operated as a landfill bioreactor (LBR). The LBRs increase the moisture content of the waste; thus, the waste is decomposed and stabilised faster. The moisture is usually supplied from the leachate produced in the landfill. Through the recirculation of the leachate, the water content, nutrients, enzymes and the bacteria are evenly distributed in the landfill.

A conventional landfill, which is designed to store solid wastes, can be considered as a bioreactor by optimising the stabilisation process and creating the desired environment for microorganisms. LBRs are mainly classified as anaerobic, aerobic, facultative and hybrid bioreactors. Many studies and real case trials have been conducted on anaerobic and aerobic bioreactors, while facultative and hybrid types are in progress. However, the anaerobic bioreactor is the most common and economically preferable option. Anaerobic LBR technology mainly aims to increase landfill gas production, improve leachate quality and decrease its amount, acquire new air spaces and provide sustainability [4]. The factors that affect the efficiency of an anaerobic LBR, such as leachate recirculation period, recirculated volume, waste shredding and compaction, the pH/alkalinity adjustment, seeding with the anaerobic sludge, supplemental nutrient addition and codisposal with sewage sludge, have been investigated in many studies.

The infiltration of the rainwater and trapped water in the waste body generates the leachate in the landfill. Moreover, the formation of the leachate can be observed when the waste body reaches field capacity (v/v: 45%–65%), which is simply defined as the maximum moisture content level that a porous medium can retain [5]. If the moisture content of the waste body is above the field capacity, then the leachate production starts in the landfill. In a LBR, the main operational goal is to maintain the moisture content near the field capacity to accelerate the waste stabilisation [6,7]. MSWs consist of many different types of materials, such as food, plastic, textiles and paper. Therefore, the characterisation of the leachate is complex. The amount of leachate can be understood to be directly related to the amount of water that enters the landfill.

Operating landfills as a bioreactor is a new and promising strategy. LBRs also have a great potential for in situ leachate remediation, but this method currently receives relatively less interest at this time. One of the promising in situ leachate treatment materials that can be used in the drainage layers of landfills is geotextile (GT) fabrics. Many types of GTs are manufactured from polypropylene (PP) or polyester (woven, nonwoven, needle punched and heat bonded), which are used in many infrastructure projects for different purposes [8]. Highly permeable GT fabrics are used as an infrastructure material in landfill areas across the world mainly for protecting the impermeable geomembrane layer from gravel and waste particles. In addition, GTs are used for filtration, separation, drainage and for other purposes in civil, geotechnical and environmental engineering. GTs are thin, but strong, durable and permeable materials.

GT filters can clog through permeation with the leachate in the landfill sites as a result of high biomass growth [9]. From this point, using GTs as biofilters in the treatment of different wastewaters have been investigated [10]. The biomass formation in GTs was first encountered in the 1990s. The first study focused on how leachate filtration clogged the GT materials in landfill sites as a consequence of biomass formation [9]. A column packed with alternating layers of gravel, GT and sand was used in that study. The GTs clogged through permeation with the leachate using different types of GTs, leachates and test conditions. Findings indicated that the clogging of the GT was due to the formation of biofilm and was affected by the porosity and thickness of the GT material. The needle-punched nonwovens had the highest residual permittivity or permeability, the heat-bonded types were easily clogged by leachate permeation and the characteristics of the leachate sample affected the clogging [9,11].

In one study, alternating layers of nonwoven GT fabrics were used to filter wastewater [12]. In other studies, GTs were used to treat wastewater and storm water in different conditions; combined sewer overflows were treated with the GT baffle contact system (GBCS) [13], wastewater was treated with GBCS [14] and septic tank effluent was treated with GT biofilters [15]. In one study, the use of layered GTs was investigated in the treatment of septic tank effluent prior to ground infiltration, and the exhumed GTs were scanned with a scanning electron microscope (SEM) to show the biomass growth inside the GT [10,15]. Furthermore, one study investigated the effects of wastewater filtration on GT permeability [16].

Leachate has a complex and heterogeneous composition. Therefore, it may cause clogging as a result of physical, chemical and biological accumulations in both the gravel and GT layers in the landfills. In one study, three types of nonwoven GTs with different mass per unit areas were used to investigate GT permeability reduction [17]. The microorganisms in the porous media were identified and quantified, and after 90 d, the biofilm formation was detected by SEM [17].

In a different study, a cylindrical structure was wrapped by GT as the attached growth medium and used for wastewater treatment. Results showed high removal efficiencies (above 90%) of carbon, nitrogen and phosphorus after 250 d of operation [18].

Studies found that the biofilm is formed in the interior structure of the GT, and the dissolved organic compounds are biodegraded when exposed to the wastewater for a certain amount of time [16]. However, no study focuses on the use of highly porous GT filters as an attached growth or suspended growth media in landfills to remediate landfill leachate at the site where it is produced. Our hypothesis emerged because of the GT's high filtration and biomass accumulation capacity. Innovations of this study include its investigation of the applicability of GT materials in landfills for this purpose and the introduction of a new alternative LBR model.

2. Materials and methods

2.1. Experimental setup

2.1.1. Structure of reactors

To simulate LBRs, two lab-scale reactors were constructed with a height of 1 m and a diameter of 30 cm by using opaque polyvinyl chloride pipes. The reactors were equipped with several ports for the collection and distribution of the leachate. The reactors consisted of two main compartments: the upper compartment is designed to hold the waste mass, and the lower compartment consists of the drainage layer and interior leachate tank. A schematic of the system is shown in Fig. 1.

2.1.2. Design of drainage layers

The leachate produced from the waste body in the reactors was collected after passing through a specifically designed drainage layer. The total depth of the drainage layer was 10 cm, and the two types of gravel of different sizes were used at equal heights. The gravel with a 15 mm diameter was placed at the bottom of the drainage layer, and the gravel with the 10 mm diameter was placed at the top. In the LBR-GT, a specific GT fabric was inserted into the drainage layer. The technical specifications of the GT are given in Table 1.

2.1.3. Leachate collection, recirculation and distribution system

The bottom of the reactors was punctured to allow the leachate to transfer into the drainage layer. The leachate produced in the reactors was infiltrated by the drainage layer and collected in the 9 L interior leachate tank. The interior leachate tank was connected to another exterior leachate tank by a pipe. This exterior tank is made of transparent PP and has an opening for probe analysis. It also has a volumetric scale to measure the produced leachate volumes. Peristaltic pumps with a 2 L/h flow rate were used to recirculate the leachates. The recirculated leachates were distributed from the top of the reactors by using an equally perforated polyurethane pipe.

2.2. Operation

Table 2 summarises how the reactors were configured and operated.

2.2.1. Experimental startup

The reactors were placed in an isolated room and operated at 33°C–37°C. The ambient temperature was observed by a digital room thermometer, which showed the maximum and minimum measured temperatures. This thermometer was periodically reset, and the current values were monitored to ensure the ambient temperature was between the desired values.

2.2.1.1. Loading of municipal solid wastes The feed waste material was taken from a compost plant where mainly

Table 1
Technical specifications of the geotextile used in this study

Brand and model	TenCate TS40
Material	Polypropylene
Production method	Needle punch
Density	184 g/m^2
Thickness	1.95 mm
Permeability	115 L/s m^2
Opening size	116.75 µm

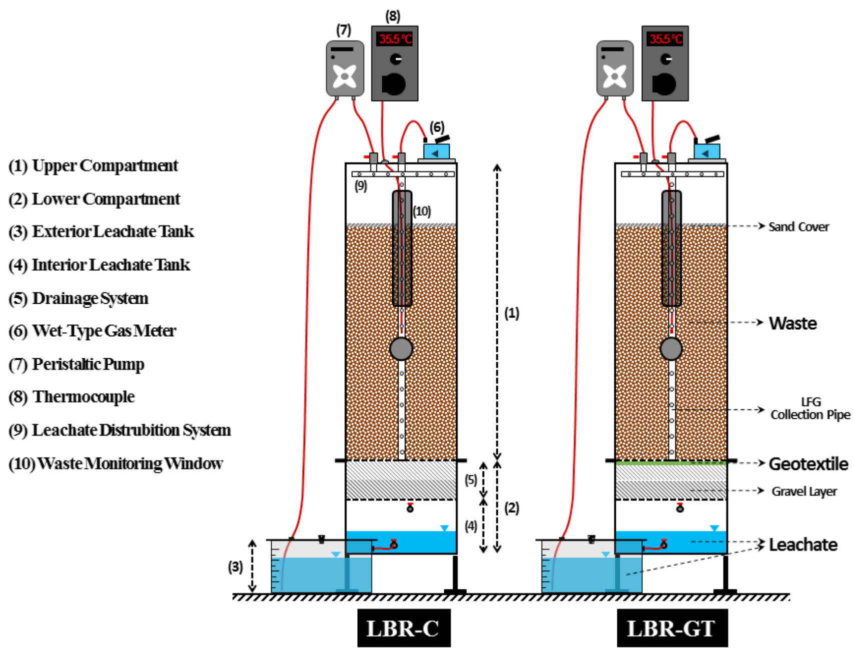

Fig. 1. Schematic diagram of lab-scale LBRs.

organic MSWs are used. The MSWs were taken from the outlet of the Ø 80 mm rotary screen at the entry of the compost plant.

Both reactors were filled with 30.8 kg of MSWs, and the waste was compacted to a density of 700 kg/m^3. The total height of the waste was maintained at 62.50 cm, and the total volume was calculated as 44 L. At the top of the waste body, 2–3 cm of Ø 4 mm coarse sand was placed to allow for the uniform distribution of the recirculated leachate.

After the reactors were filled with the MSW, 1 L of anaerobic seed sludge was introduced to the waste body to accelerate the anaerobic conditions. The operation of the LBRs was started by closing all the ports and lids to make sure that the reactors were both airtight and watertight.

2.2.1.2. Operational procedure In the first month of the operation, 1.1 L of distilled water was added every week to the reactors by using the peristaltic pumps to simulate the rainfall. A total of 5.5 L of distilled water was injected into the reactors in the first month of the operation. After 1 month, when the waste body reached the field capacity precisely, no more supplemental water was added. In both reactors, all the produced leachates were recirculated to the waste body three times a week. When the amount of leachate produced exceeded the required amount for recirculation, the extra leachate volume was removed.

2.3. Analytical procedure

2.3.1. Solid waste analysis

The composition of the solid waste was determined by separating each type of waste component in the feedstock. After separating the food, paper, textile, glass, metal, plastic and stone from the mixed waste, each component was weighed separately. Then, the percentages of each waste component were calculated. The analyses of total carbon, total organic carbon, total Kjeldahl nitrogen (TKN), ammonium nitrogen (NH_4–N), total phosphorus (TP), and oil and grease were performed by following the Turkish Standards Institution methods. The EPA 6010 C method was used to analyse the metals (Ba, Cd, Cr, Cu, Mo, Ni, Pb and Zn), and other solid waste parameters were analysed according to the Standard Methods [19].

Table 2
Configuration and features of the bioreactors

Parameter	LBR-C	LBR-GT
MSW wet quantity, kg	30.8	30.8
MSW volume, L	44	44
MSW density, kg/m^3	700	700
Seed sludge volume, L	1.0	1.0
Seed sludge ratio (V:V), %	2.26	2.26
Rainwater addition, L/week	1.1	1.1
Number of rainwater additions	5	5
Leachate recirculation	Yes	Yes
Recirculation frequency, times/week	3	3
GT filter in drainage layer	No	Yes

The solid waste characteristics significantly affect system performance, and the characterisation must be known to evaluate the results appropriately. The composition of the feed MSWs is listed in Table 3.

The water, total solids (TS), volatile solids (VS), fixed solids (FS), carbon, nitrogen and phosphorus contents and their subforms are important parameters in the LBR operation. The physical and chemical properties of the MSW are listed in Table 4.

The analysis of the results of the heavy metal contents of the MSW used in this study is presented in Table 5. The concentrations of the chromium and barium are comparatively higher than those obtained from the other metals, which

Table 3
Composition of the solid waste

Waste component	Value (%)
Food	62
Paper	16
Textiles	3
Glass	2
Metal	1
Plastic	8
Stones	8

Table 4
Physical and chemical properties of the solid waste

Parameter	Value
Water content, %	62
Total solids (TS), %	38
Volatile solids (VS), %	71
Fixed solids (FS), %	29
pH	6.36
Conductivity, mS/cm	4.43
Total carbon (TC), %	31.6
Total organic carbon (TOC), %	31.3
Total Kjeldahl nitrogen (TKN), mg/kg	15,173.2
Ammonia nitrogen (NH_4–N), mg/kg	619.7
Total phosphorus (TP), mg/kg	3,102.7
Oil and grease, mg/kg	7.7

Table 5
Metal contents of the solid waste

Parameter	Value
Barium (Ba), mg/kg	263.4
Cadmium (Cd), mg/kg	1.2
Chromium (Cr), mg/kg	81.5
Copper (Cu), mg/kg	122.9
Molybdenum (Mo), mg/kg	31.9
Nickel (Ni), mg/kg	29.0
Lead (Pb), mg/kg	30.4
Zinc (Zn), mg/kg	431.5

may be attributed to the dumping of Cr- and Ba-containing wastes with MSW.

2.3.2. Seed sludge analysis

Repetitive samples were taken from the anaerobic seed sludge and analysed for pH, conductivity, total dissolved solids (TDS), oxidation reduction potential (ORP), temperature, water content, TS, VS, FS, total suspended solids (TSS), volatile suspended solids (VSS), chemical oxygen demand (COD), total nitrogen (TN), nitrate (NO_3^-) and TP. All the measurements were performed by following the Standard Methods [19].

The physical and chemical properties of the anaerobic seed sludge are given in Table 6.

2.3.3. Leachate analysis

The leachate samples were taken from the produced leachates and stored at 4°C prior to analysis. The conductivity (85 assays), TDS (85 assays), pH (85 assays), ORP (75 assays), COD (78 assays), alkalinity (24 assays), TKN (24 assays), ammonia nitrogen (NH_3–N; 24 assays) and the chloride ion (Cl^-; 8 assays) analyses were conducted according to the Standard Methods [19]. The 5-d biochemical oxygen demand (BOD_5; 21 assays) was conducted by using the OxiTop (WTW, Weilheim, Germany) method. Sulphate (SO_4^{2-}; 8 assays) and fluoride (F^-) ions (8 assays) were measured by using ion chromatography (Shimadzu IC-SA2) with an anion column equipped with a CDD-10A conductivity detector. The mobile phase was prepared by dissolving 0.1908 g Na_2CO_3 and 0.1428 g $NaHCO_3$ in ultrapure water and diluted to 1 L. Then, it was filtered and subjected to ultrasound into an ultrasonic bath for 15 min. The flow rate of the mobile phase was 1 mL/min in isocratic conditions. The injection volume was 20 µL, and the testing time was 15 min. Metals (8 assays) were determined by using inductively coupled plasma optical emission spectroscopy (ICP-OES Optima 7000DV).

2.3.4. Biomass detection in geotextile

A precision balance was used for the residual biomass measurements. The GT filter was specifically cut for the reactor and weighed before the operation. After the completion of the operation, the drainage layer of the LBR-GT was removed from the upper compartment to exhume the GT filter. The GT was removed from the reactor and air dried overnight. Then, it was weighed again to calculate the residual biomass quantity, and duplicate samples with ~1 cm² area were taken from the GT to take the SEM pictures. The GT samples were coated in gold before the SEM analysis, which was conducted with a Philips XL30S-FEG device. In addition to the SEM pictures, the GT filter was photographed before the operation, after the operation and after air drying to observe the visual effect of the biomass formation on the GT material.

Table 6
Physical and chemical properties of the seed sludge

Parameter	Value
Total solids (TS), mg/L	8,110
Volatile solids (VS), mg/L	4,503
Fixed solids (FS), mg/L	3,607
Total suspended solids (TSS), mg/L	6,390
Volatile suspended solids (VSS), mg/L	3,120
pH	6.6
Conductivity, µs/cm	1,267.9
Total dissolved solids (TDS), mg/L	515.8
Oxidation reduction potential (ORP), mV	–301.5
Chemical oxygen demand (COD), mg/L	4,802
Total nitrogen (TN), mg/L	355
Nitrate (NO_3^-), mg/L	0.3
Total phosphorus (TP), mg/L	71

3. Results and discussion

The simulated lab-scale LBRs were monitored throughout the study to investigate the effect of the GT layer on the leachate quality. For this purpose, all the results from the leachate quantity and quality analysis and biomass detection analysis are given in this section, including a detailed discussion of the results [20].

3.1. Leachate quantity

3.1.1. Produced leachate amounts

Although 5.5 L of distilled water and 1 L of anaerobic seed sludge were added to each reactor in the first month of the operation period, different amounts of leachates were produced. Although the reactors were filled with the same MSW, which already had a high content, extra water was retained by the waste body in the early weeks of the operation. However, after 3–4 weeks, the waste body reached the field capacity, and almost no extra water was captured thereafter (Fig. 2).

3.1.2. Recirculated leachate amounts

All the produced leachates were periodically recirculated into the reactors. However, the amount of produced leachate for both reactors was equalised after 80 d of operation (Fig. 3). When the reactors reached the methanogenic phase, the waste body released some additional water probably as a result of the hydrogenotrophic methanogenesis activity. While these methanogenic bacteria use hydrogen to

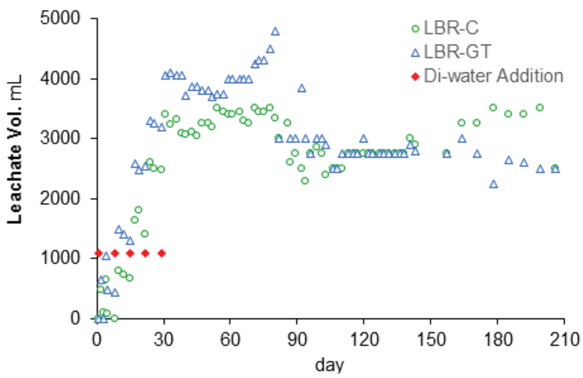

Fig. 2. Injected distilled water and leachate volumes over time.

produce methane, water is produced at the same time. When the leachate produced in the reactors exceeded the required amount for recirculation, the extra leachate was removed.

3.2. Leachate quality

The periodic leachate samples were taken to determine the change in the leachate quality, and numerous analyses were conducted. In this section, all the leachate quality parameters are explained.

3.2.1. pH

Generally, a stabilised leachate has a higher pH than that of a fresh leachate. The pH of fresh leachate is less than 6.5, while the pH of aged landfill leachate is higher than 7.5. The initial low pH is due to the high concentration of volatile fatty acids produced during the acid phase. The pH of a stabilised leachate is generally constant with small variations and may range between 7.5 and 9. The increase in pH suggests that steady-state conditions were reached between the acid and methanogenic phases in the landfill.

In this study, the leachate pH values were monitored three times a week by using the reference electrode. The leachate pH is directly related to the fatty acids and alkalinity of the system. The produced CO_2 content during the degradation process also has an effect on the pH. Cations, such as NH_4^+ and Na^+ tend to increase the pH and alkalinity, while the accumulation of volatile fatty acids (VFAs) and the produced CO_2 gas during anaerobic degradation have a decreasing effect on the pH [21].

The optimal pH range is generally accepted as 6.5–8.2 for the methanogens. However, in the early stages of stabilisation, the pH values are reported below this optimal range because of the high production of acids.

As can be seen in Fig. 4, the leachate pH values started at near-neutral conditions (pH ≅ 6.5) due to the effect of the injected distilled water during the first month of operation. Then, pH values decreased suddenly to around 5.7 for both reactors and were maintained until methanogenic conditions occurred. This decline was mainly due to VFA accumulation in the reactors.

While the pH started to rise and reached a neutral value after the 45th day in LBR-GT, the pH reached a neutral value after the 130th day in LBR-C. This increase in pH was mainly related to the consumption of volatile organic acids by the microbial community. The earlier pH increase in the LBR-GT suggests that the usage of GT as a biofilter accelerated the stabilisation process in the reactor. An appropriate environment was also provided for the methanogens when the pH values of the leachate increased to a neutral value. The maximum detected pH values were 7.91 and 8.04 for LBR-C and LBR-GT, respectively.

3.2.2. Oxidation reduction potential

The inoculation of the reactors with anaerobic seed sludge should have affected the initial low ORP values. Thus, in Fig. 5, a stable trend was not observed at the beginning of operation; however, both reactors reached their maximum ORP values almost at the same time (the 40th day). Then, the ORP values started to decrease as a result of the consumption of the available oxygen in the reactors. The decreasing trend for the LBR-GT was sharp after the 45th day, whereas the LBR-C did not show a similar decreasing trend in the ORP values. They both reached very low ORP values at different times, which is a simple indication of methanogenic conditions. The ORP value of the LBR-GT decreased below –300 mV on the 54th day, while in the LBR-C, it did not occur until the 145th day. The earlier ORP decrease in the LGR-GT suggests that the usage of GT filter accelerated the anaerobic conditions.

Fig. 5 shows the variations of ORP throughout the study.

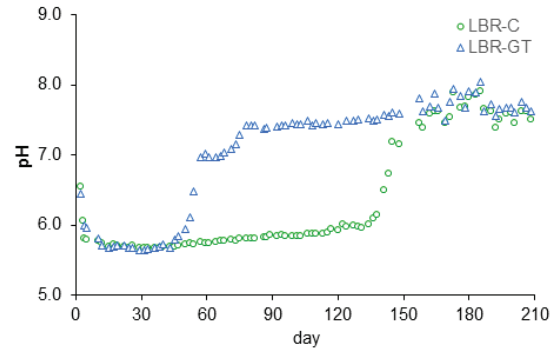

Fig. 4. Variations of pH values over time.

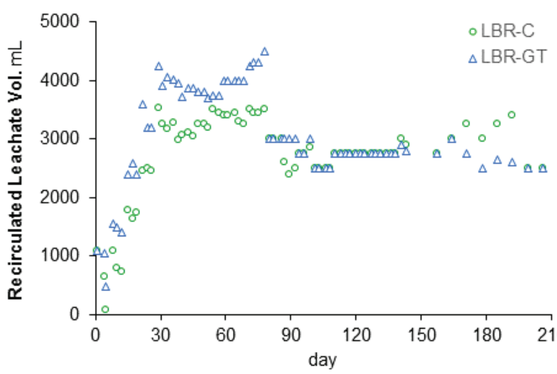

Fig. 3. Leachate recirculation amounts.

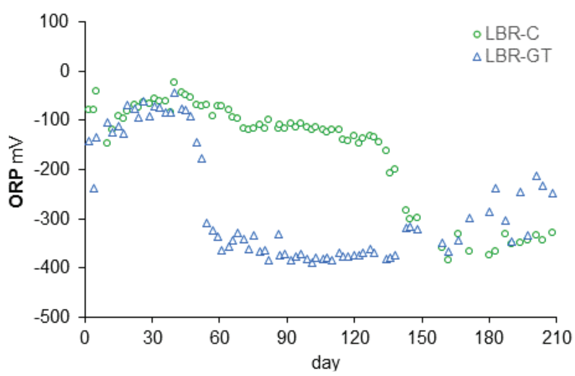

Fig. 5. Variations of ORP values over time.

3.2.3. Conductivity and total dissolved solids

Conductivity reflects the ability of electrical conduction and the total concentration of ionic solutes in a solution. The ions in the waste body (e.g., Cl^-, NO_3^-, SO_4^{2-}, Mg^+ and Ca^+) were washed out due to the recirculation of the leachate.

Fig. 6 presents the change in leachate conductivities for both reactors over time. The conductivity values reached their maximum values (LBR-C: 27.70 mS/cm and LBR-GT: 25.40 mS/cm) for both reactors in the earlier months of the operation. Thereafter, the conductivity of the leachates started to decrease at different rates. For the LBR-GT, the decreasing trend started on the 47th day, while the same trend started approximately 3 months later in the LBR-C. After 159 d of operation, the leachate conductivities were very close to each other with only a minimal difference. The final conductivity values of the leachate samples were 20.60 and 19.68 mS/cm for the LBR-C and LBR-GT, respectively.

TDS is an indicator of mineralisation during stabilisation and is directly related to electrical conductivity. TDS mainly includes all ions and carbonate species, which reflect the total concentration of the dissolved constituents in a water sample. The maximum and final TDS concentrations for the LBR-C and LBR-GT were 13.55 and 10.34 g/L and 12.83 and 9.75 g/L, respectively. The trend in the TDS was the same as that of the conductivity for both reactors (Fig. 7).

3.2.4. Organic constituents

Organic constituents are released from the waste body by the infiltration of water during the stabilisation period. The quantity and quality of leachate both significantly influence the organic mass effluent from landfills [22]. COD and BOD_5 are the main parameters that are used to determine the organic content of water samples in environmental sciences [19]. The variation of the dissolved organic constituents in the leachate must be monitored during the stabilisation of the MSWs in a LBR. The results are given below.

3.2.4.1. Chemical oxygen demand COD is a critical important parameter during the operation of LBRs. COD can be simply identified as the amount of oxygen required to stabilise organic matter by using a specified strong oxidant (dichromate; $Cr_2O_7^{2-}$) under controlled conditions.

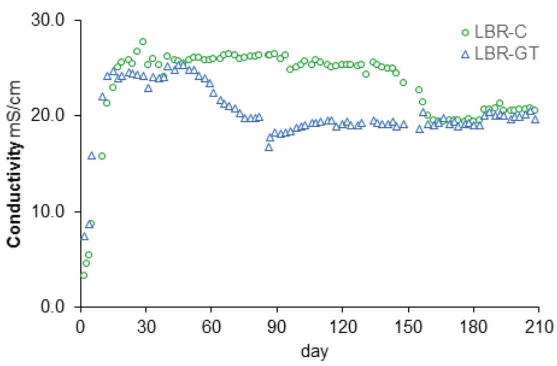

Fig. 6. Variations of conductivity over time.

COD is the most often used parameter for determining the organic strength of the leachate. While the decreasing trend of ORP, electrical conductivity and TDS started at almost the same time in the LBR-GT, a similar trend was observed for the COD after 50 d of operation. However, after the 19th day, the COD values in the LBR-C fluctuated between 50,000 and 70,000 mg/L until the 145th day. The maximum COD values of the LBR-C and LBR-GT were detected as 69,647 and 52,440 mg/L, respectively. As can be seen in Fig. 8, a significant difference exists in the final COD concentrations of the reactors. In the final leachate samples, 5,116 and 2,192 mg/L of COD concentrations were detected for LBR-C and LBR-GT, respectively.

Many studies have indicated that aerobic LBRs have greater organic removal efficiencies than anaerobic types, and only aerobic systems achieved above 90% COD removal [1,23,24]. However, the COD removal rate of 90% was reached in 3 months in the LBR-GT, which is an extraordinary performance for an anaerobic LBR. Similar COD removals for LBR-C were only reached after 5 months. This high organic removal rate in a shorter time period for LBR-GT obviously demonstrates that the GT filter enhanced the degradation of the organics in leachate.

3.2.4.2. Biochemical oxygen demand BOD is a similar parameter to COD, but the BOD test represents only the biodegradable portion of the organic matter in a water sample. However, BOD values generally show a similar trend to COD concentrations. Therefore, it provides additional information

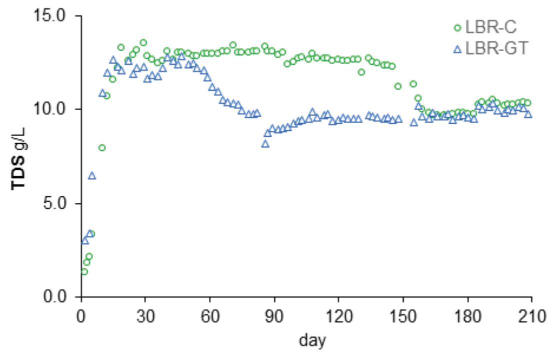

Fig. 7. Variations of total dissolved solids over time.

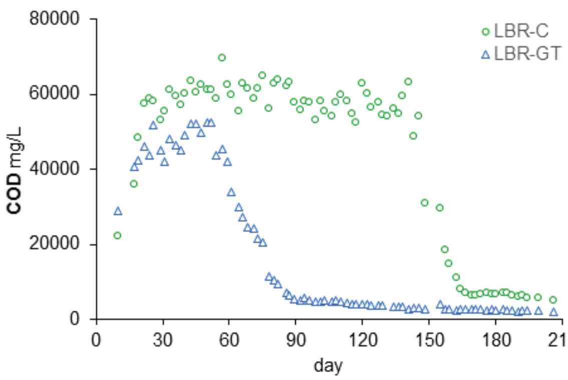

Fig. 8. Variations of chemical oxygen demand over time.

on the biodegradable fraction of the COD. Thus, the BOD_5 tests were conducted periodically during the operation.

Fig. 9 represents the change in BOD_5 over time. The maximum detected BOD_5 concentrations were 23,500 and 27,500 mg/L for LBR-C and LBR-GT, respectively. The BOD_5 concentrations started to decrease just after the first month of the operation in LBR-GT. The BOD_5 trend was similar to that of the COD for the LBR-C. The final BOD_5 concentrations were 900 and 250 mg/L for LBR-C and LBR-GT, respectively. More than 99% of BOD_5 removal was achieved in the LBR-GT.

Significant BOD_5 removal was achieved in LBR-GT in less than 3 months, while a similar BOD_5 removal was only reached after 6 months in the LBR-C. This high BOD_5 removal rate in a shorter time period for the LBR-GT demonstrates that the GT filter that contains the biomass enhanced the removal efficiency.

The BOD_5/COD ratio describes the degree of biodegradation and provides information about the age of a landfill. The low BOD_5/COD ratio indicates the high concentration of non-biodegradable organics and thus the difficulty to be biologically degraded.

The BOD_5/COD ratio is generally used for the proportion of the biodegradable organic content in the leachate of the landfills where the ratio is high at the earlier period of the stabilisation process. The biodegradable organic particles in the leachate, which are expressed as BOD_5, are consumed by the microorganisms more easily. The range of the BOD_5/COD ratio was determined between 0.18–0.42 and 0.11–0.65 for LBR-C and LBR-GT, respectively (Fig. 10).

3.2.5. Ions

Ions, such as sulphate, chloride and fluoride, were detected infrequently. In this section, the ion concentrations of the leachate samples will be given. The other ions measured in this study were either detected at very low concentrations or not detected. Therefore, only these three ions were included in the study.

3.2.5.1. Sulphate (SO_4^{2-}) The sulphate concentration of the leachate generally depends on the decomposition of the organic matter present in the solid wastes. The sulphur compounds are present as SO_4^{2-} and S^{2-} ions in the leachate samples. The SO_4^{2-} concentration is expected to decrease with the landfill age. This decrease is due to the reduction of SO_4^{2-} to S^{2-} when the anaerobic conditions in the landfill occurred. Therefore, the SO_4^{2-} concentration in the leachate can also be used as an indicator of waste stabilisation within the landfill.

Fig. 11 shows another indicator of the rapid generation of the methanogenic conditions in LBR-GT. In the first month of operation, the SO_4^{2-} concentration reached its maximum value in the LBR-GT (3,531.16 mg/L) on the 31st day of the operation, but during the methanogenic phase, the concentration sharply decreased and remained below 1,000 mg SO_4^{2-}/L until the end of the operation. In LBR-C, the concentrations increased and reached 2,794.37 mg SO_4^{2-}/L on the 87th day. Then, it started to decrease when the methanogenic conditions were maintained in the reactor.

The final sulphate concentrations were 248.57 and 66.98 mg SO_4^{2-}/L for LBR-C and LBR-GT, respectively. This finding means that 98% of the sulphate was removed in both reactors at the end of the study.

3.2.5.2. Chloride (Cl^-) Chloride ion is a non-biodegradable and a persistent constituent that is generally used to estimate the dilution effects on the leachate. After the start-up period, no supplemental water was injected into the system, and the leachates produced from the reactors were recirculated periodically. Therefore, a significant change was not expected in the Cl^- concentrations. Fig. 12 shows the change in the chloride ion concentrations over time for both reactors. The maximum concentrations were 3,499 and 2,499 mg Cl^-/L,

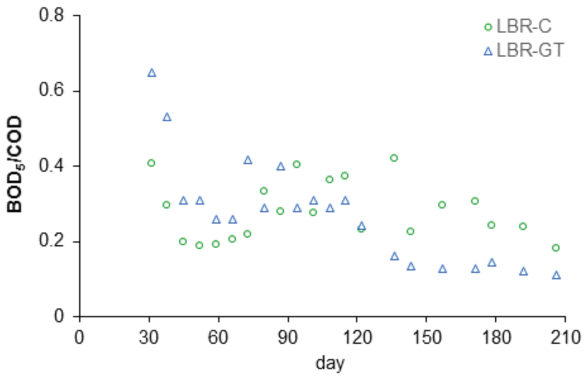

Fig. 10. Variations of BOD_5/COD over time.

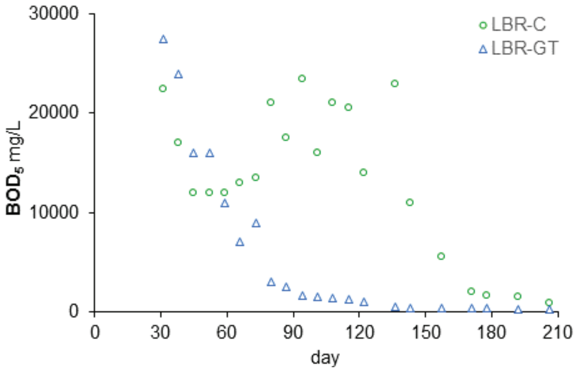

Fig. 9. BOD_5 variations over time.

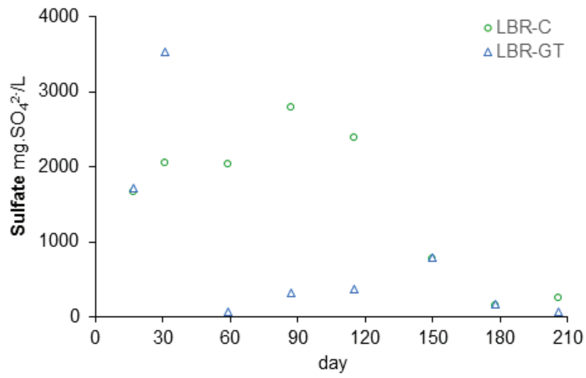

Fig. 11. Sulphate ion concentrations over time.

the minimum concentrations were 1,560 and 1,639 mg Cl^-/L for the LBR-C and LBR-GT, respectively.

3.2.5.3. Fluoride (F^-) Fluoride ion, which is a toxic constituent for methanogens [25], was measured periodically. Fig. 13 shows the change in fluoride ion concentrations over time and indicates that a healthy anaerobic microbial population removed the F^- content in the leachate. The final concentrations were very low at 24.75 and 0.74 mg F^-/L, whereas the maximum values were 1,085.03 and 895.46 mg F^-/L for the LBR-C and LBR-GT, respectively. Almost all the fluoride content was removed from the leachate by the end of the study in both the reactors.

3.2.6. Alkalinity

Total alkalinity is a measure of the buffering capacity in the system. The recirculation of the leachate in the landfills may provide high alkalinity in the leachate because it is known as a beneficial condition for buffering the pH that helps the landfill stabilisation process [22,26]. Alkalinity has a key function in anaerobic degradation. Methanogens can be inhibited if the acid concentrations exceed the total alkalinity in the system, and as a result, the LBR system may fail [27].

The alkalinity change over time is given in Fig. 14. The maximum alkalinities were recorded as 12,750 and 11,625 mg/L for LBR-C and LBR-GT, respectively. After the methanogenic conditions were maintained in the system, the final concentrations were 8,000 and 6,875 mg $CaCO_3$/L for LBR-C and LBR-GT, respectively. During the operation, the total alkalinity concentrations were at sufficient levels for a healthy anaerobic environment.

3.2.7. Ammonia nitrogen (NH_3–N) and total Kjeldahl nitrogen

Nitrogen has a long-term pollution potential in landfills. If no degradation pathway exists for the ammonia nitrogen in landfills, then ammonia nitrogen may accumulate in the system. Particularly in recirculated LBRs, supplemental water addition and/or recirculation of the leachates cause higher ammonification rates, which result in higher ammonia concentration compared with conventional landfills [28]. The majority of nitrogen content in the leachate is in the form of ammonia, which is considered as one of the most significant long-term pollutants in landfill leachate [5]. Ammonia nitrogen is in the form of ammonium (NH_4) at lower pH values where no significant adverse effect occurs on the anaerobic process, while at higher pH values (>8–9), a high ammonia nitrogen (NH_3) content may show an inhibitory effect on the anaerobic degradation processes [29].

Fig. 15 shows the change in ammonia concentrations in the reactors. No stable trend in ammonia concentrations was detected during the study. The initial ammonia concentrations were 814.8 and 943.6 mg/L for LBR-C and LBR-GT,

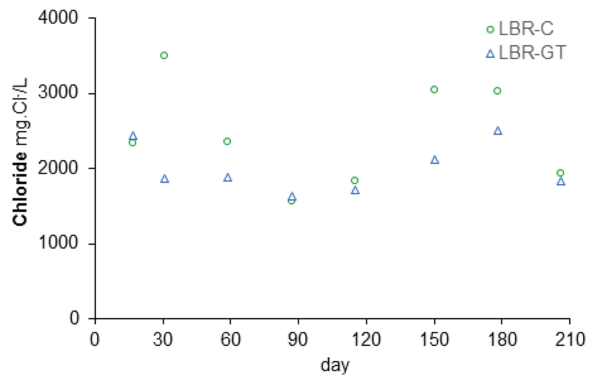

Fig. 12. Chloride ion concentrations over time.

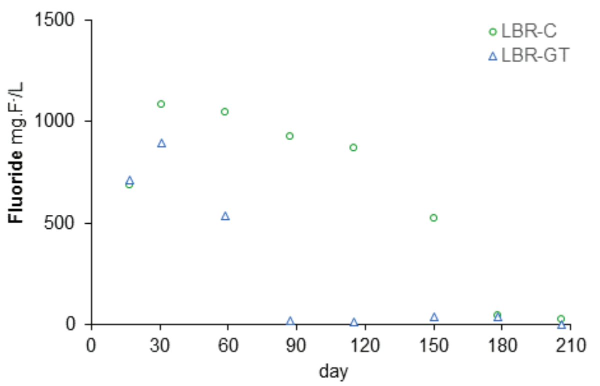

Fig. 13. Fluoride ion concentrations over time.

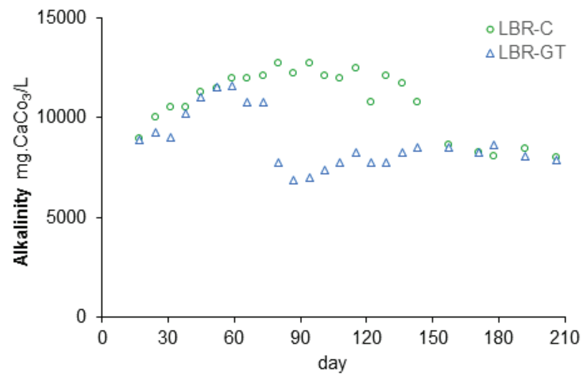

Fig. 14. Variations of total alkalinity over time.

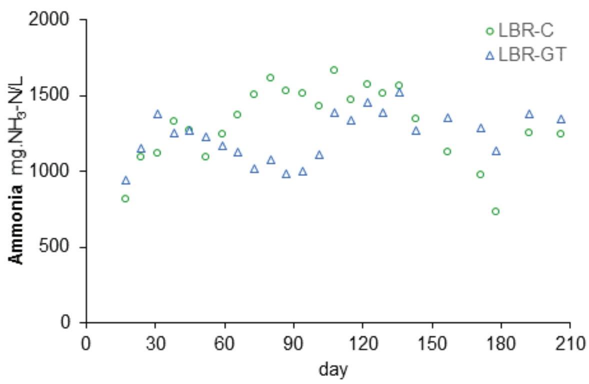

Fig. 15. Variation of ammonia nitrogen over time.

respectively, while the final ammonia concentrations were detected as 1,246.0 and 1,346.8 mg/L for LBR-C and LBR-GT, respectively.

The initial Kjeldahl nitrogen concentrations were very close to each other (LBR-C: 1,366.4 mg/L, LBR-GT: 1,388.8 mg/L). However, the change in the TKN over time was different, and no proper trend was documented, as seen in the ammonia concentrations during the operation (Fig. 16). The final TKN concentrations were very close to each other and detected as 1,820.0 and 1,758.4 mg/L for LBR-C and LBR-GT, respectively.

3.2.8. Metal contents

Generally, the concentration of the heavy metals in the leachate was reasonably low. The heavy metal concentrations in a landfill are usually higher in the earlier stages because of the higher metal solubility as a result of the production of organic acids. As a result of the increased pH, the metal solubility decreases, thereby resulting in a decrease in concentration in most heavy metals except for lead, because it is known to produce a heavy complex with humic acids.

The change in the concentration of the metals, including cobalt, chromium, copper, ferrous, nickel, lead and zinc, is given in Figs. 17 and 18 for the LBR-C and LBR-GT, respectively. The silver and cadmium concentrations were not added to the graph because all the Ag and Cd data were under the detection limits.

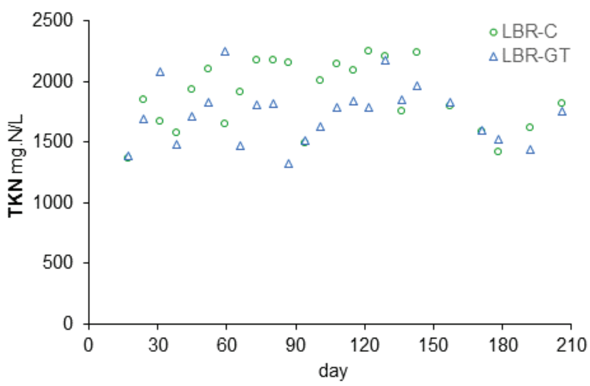

Fig. 16. Variation of total Kjeldahl nitrogen over time.

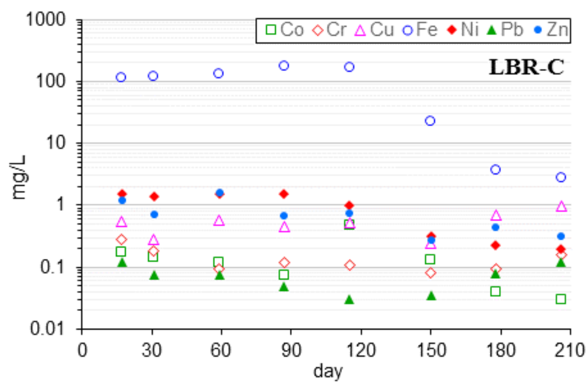

Fig. 17. Variations of metal content in LBR-C over time.

3.3. Biomass formation in geotextile

3.3.1. Residual biomass quantity on geotextile

The GT filter was weighed with a precision balance before the start-up and after 208 d of operation. Table 7 shows the weight of the clean GT and the GT with biomass residual after being air dried overnight. Subtracting these two values obtained the total residual biomass in the GT filter. Dividing the biomass weight by the surface area of the filter (~700 cm^2) obtained the biomass density.

3.3.2. Visual effect of biomass on geotextile

The prepared and specifically chosen GT filter used in this study was photographed before the start-up (Fig. 19(a)), after the operation (Fig. 19(b)) and after being exhumed and air dried overnight (Fig. 19(c)). The figure clearly shows a visible biomass residue in the GT filter. After long-term operation, almost every part of the GT filter surface was evidently darkened by the biomass residues, which indicates a good distribution of leachate over the waste body. After the GT filter was exhumed from the reactor (LBR-GT), it was air dried overnight, and then it turned a brownish colour. After air drying, viewing the biomass residue on the GT filter became easier.

3.3.3. SEM images of geotextile

Figs. 20(a) and (b) clearly show the structure of the nonwoven GT sample, which was taken from an unused GT material to observe the porous structure of the material at varying magnifications. Before the material was used as a filter medium, no particles were found on the complex fibre structure.

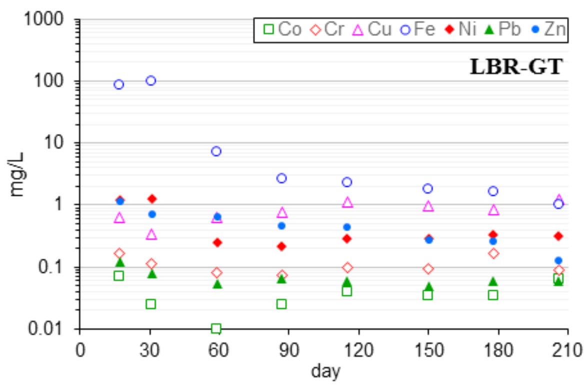

Fig. 18. Variations of metal content in LBR-GT over time.

Table 7
Biomass residual and density on the geotextile filter

Parameter	Value
Clean geotextile, g	31.25
Geotextile with biomass, g	31.76
Residual biomass, g	0.51
Biomass density, g/cm^2	0.0007

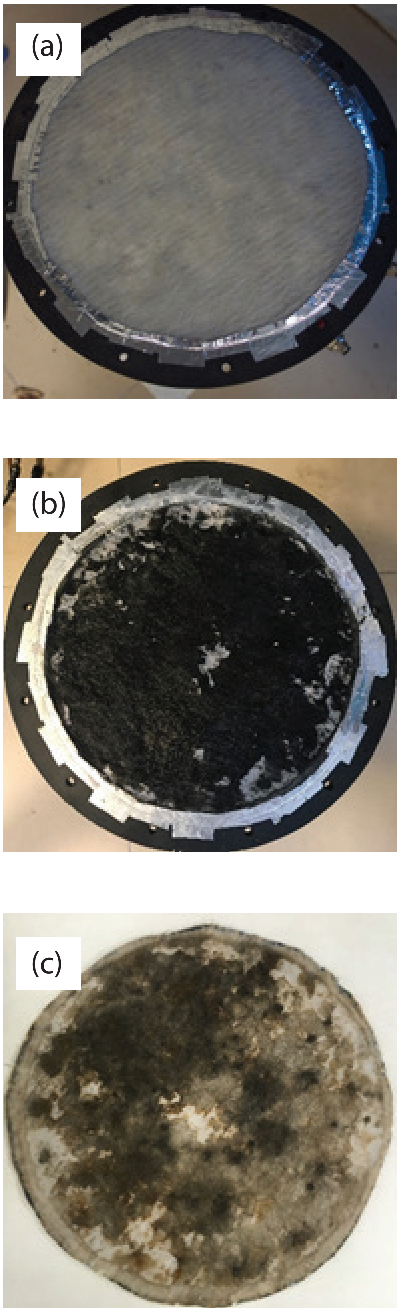

Fig. 19. Geotextile filter used in this study: (a) clean filter, (b) filter after 208 d of operation, and (c) exhumed filter after air drying.

After long-term operation of the bioreactor, some biomass formation was expected to have occurred in the GT filter. To detect this expected biomass, SEM pictures were taken from the exhumed and air dried GT sample. Previous studies demonstrated that the biomass can accumulate either inside the porous structure of the nonwoven GTs as trapped suspended flocs between the fibres or as an attached biofilm on the surface area of the fibres [9,10,30].

The complex biomass structure between the widely spaced fibres can be seen in Fig. 21. Fig. 21(b) clearly shows some attached formations on the fibres, which can indicate biofilm formation. In addition, some visible trapped particles

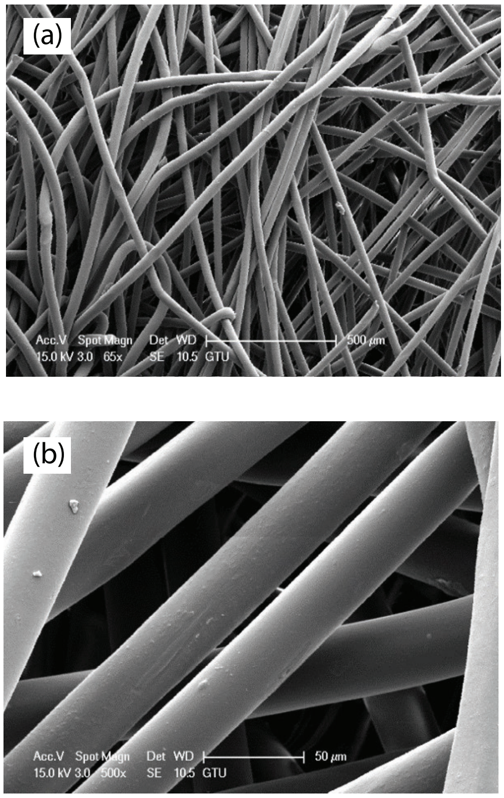

Fig. 20. SEM images of a clean geotextile sample: (a) 65× magnification and (b) 500× magnification.

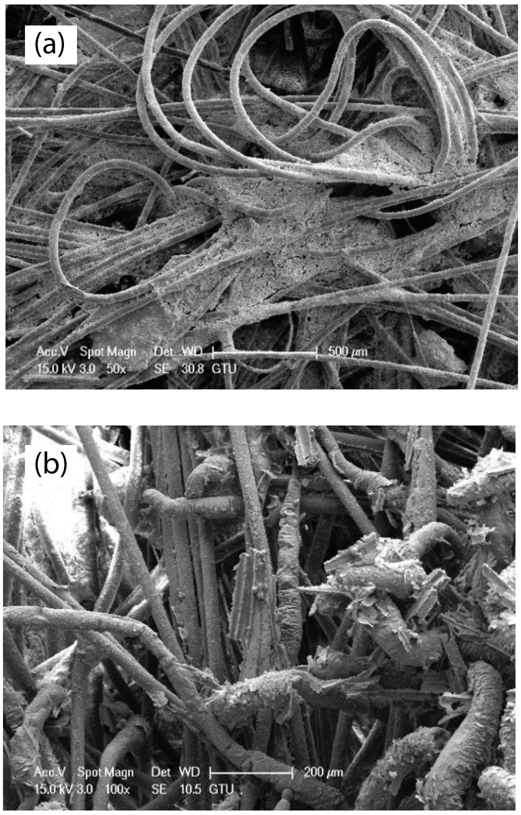

Fig. 21. SEM images of geotextile sample with biomass accumulation: (a) 50× magnification and (b) 100× magnification.

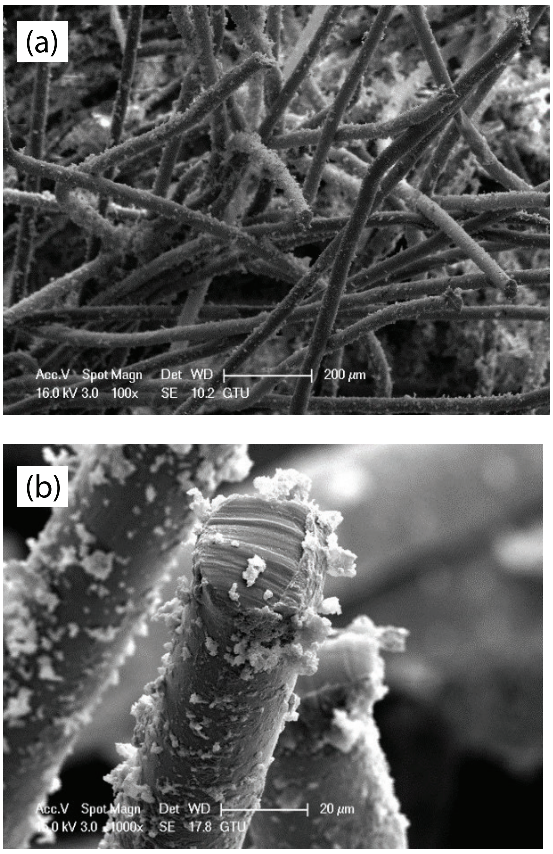

Fig. 22. Cross-section SEM images of biomass structure in the geotextile sample: (a) 100× magnification and (b) 1,000× magnification.

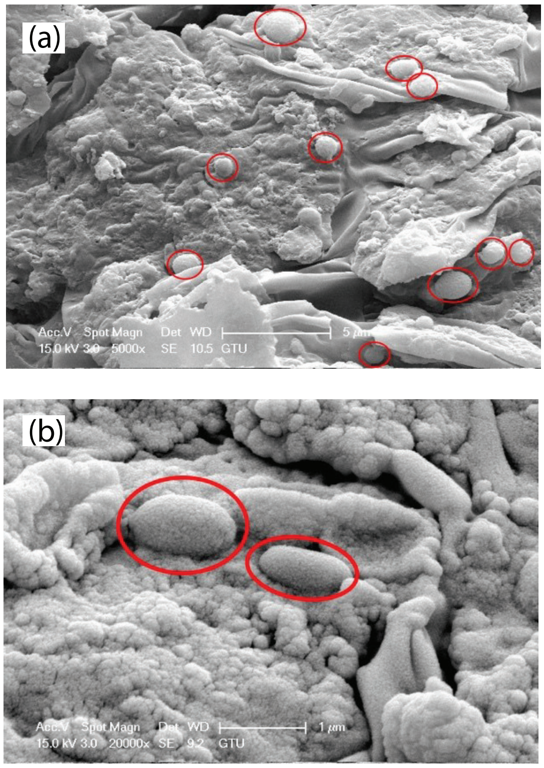

Fig. 23. SEM images of the microbial community on geotextile's fibres: (a) 5,000× magnification and (b) 20,000× magnification.

are found between the pores of the GT (Fig. 21(a)), which resemble a suspended growth as in the activated sludge process.

In Fig. 22, the cross-section views of the exhumed GT sample are given at varying magnifications. These images show that biofilm formation occurred not only on the surface of the GT material but also in the interior structure of the GT.

To see the microbial community clearly and closely, additional pictures were taken with much higher magnifications in the SEM system (Fig. 23). In Figs. 23(a) and (b), some bacillus and coccus bacteria can be clearly seen. Unlike in previous studies that investigated the morphology of methanogens [31–33], these bacillus and coccus bacteria are most probably the methanogenic bacteria groups.

A limit of organic accumulation is found on the GT filter before clogging or hydraulic failure occurs. However, this limit depends on how the biofilm forms in the GT. An idealised two-dimensional projection of a continuous biofilm that adheres to the GT fibres was developed [10]. In that model, increasing the biomass thickens the film and encroaches on the leachate transport channels in the GT filter.

PP GT fibres are hydrophobic and have complex pore structures. Thus, the leachate carrying attached microorganisms could be entrapped, where particle convection is limited to a point where it cannot move through a restrictive channel when percolating through a matrix with varying pore sizes of GT. The particles in the leachate have attached microorganisms, which would use those passing solutes as a substrate to construct not a continuous biofilm but an individual floc similar to suspended growth in the activated sludge systems. Biomass floc 'rattles' in a GT pore until it connects to another one in an adjacent pore. The result is the circulation of water conveying fresh substrate in the laminar flow around the biomass, which would have a higher specific surface for the substrate transfer than the continuous biofilm model [15].

4. Conclusions

In this study, two simulated lab-scale LBRs were used to investigate the effect of a GT filter on the leachate quality. The conclusions of this study can be summarised as follows:

- The sharp decreasing trend of ORP in the LBR-GT suggests that the usage of GT filter in the drainage layer accelerated the anaerobic conditions.
- The decreasing trend in the LBR-GT for COD and BOD_5 started 3 months earlier than in the LBR-C. Thus, the GT filter in the drainage layer of the bioreactor can be concluded to have accelerated the degradation of the organics in leachate.
- The SEM images of the GT filter with different angles and magnifications clearly showed that a healthy microbial biofilm was maintained in the porous structure of the GT filter.
- A high potential exists for in situ leachate remediation by using a GT filter, which can be placed around the leachate collection pipes in the drainage layer of the landfills.

In this way, the leachate first filtrates through the GT filter before entering the leachate collection pipe, thus capturing and degrading the organics in the leachate by the biomass already formed in the GT fabric.

- This study showed that GT filters allowed biofilm growth on the surface, as well as in the interior pores, thereby increasing the potential for biomass and organics interactions. In this way, the leachate quality was improved. Moreover, a high potential of energy saving in landfill sites exists due to the reduced leachate treatment cost.
- Although one type of nonwoven GT was tested in this study, the use of a broader range of GT samples would provide a better understanding of how manufactured properties such as apparent opening size and fibre texture would affect leachate treatment efficiency. In doing so, the GTs can be manufactured specifically for leachate treatment purposes.
- During the operation of the LBR-GT, no clogging was observed in the GT filter, which was thought to be the potential problem at the beginning of this study. The main reason the GT filter did not clog was probably the intermittent recirculation of the leachate to the reactor, which allowed the reopening of the pores filled with the biomass in the GT when no leachate recirculation existed.

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