Anammox, biochar column and subsurface constructed wetland as an integrated system for treating municipal solid waste derived landfill leachate from an open dumpsite

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ARTICLE INFO

Keywords:
Anammox
Constructed wetland
Municipal solid waste
Green technology
Biochar

ABSTRACT

This study aims to treat nitrogen-rich landfill leachate from Karadiyana open dumpsite, Sri Lanka, through an integrated treatment train consists of an anammox process, Municipal Solid Waste derived biochar column followed by a biochar embedded subsurface constructed wetland. Characterization of leachate was done and the leachate pollution index (LPI) was estimated. Meanwhile, leachate was treated through a treatment system comprising an anammox reactor having 140 mm diameter and 250 mm height, a biochar reactor having the same dimensions with 1.3 kg of MSW biochar, and a laboratory-scale constructed wetland of $1 \times 0.3 \times 0.45$ m. The influent and effluent quality was assessed for the samples taken in 24 h intervals. The analysis indicated that the leachate was high in COD (4000–14,000 mg/L), ammonia (760–900 mg/L), nitrate (60–126 mg/L), and phosphorus (33–66 mg/L). Ammonia and nitrite were removed 94 and 99% by anammox unit, respectively. Nitrate, phosphate, COD and conductivity were significantly removed by the constructed wetland system in 78, 70, 65 and 61%, respectively, whereas biochar barricades extended support for removal of the contaminants and color. The combined treatment system demonstrated treatment efficiencies as 100% of ammonia, 98.7% of nitrite, 98.2% of nitrate, 80.9% of phosphate, 79.7% of COD, and 69.9% of conductivity. Thus, it can be concluded that the anammox, combined with biochar embedded treatment train is promising to treat landfill leachate, having a high pollutant index.

1. Introduction

Landfill leachate is a liquid produced via decomposition of organic waste when water passes through waste (Stefanakis et al., 2014). It promotes the process of decomposition by bacteria and fungi (Olarewaju et al., 2012). Leachate treatment is one of the significant problems, which must be confronted in the context of waste management strategies, as the available leachate treatment systems are highly expensive on account of the complexity of chemical constituents and much time-consuming performance (Vadillo et al., 1998; Schoeman et al., 2003; Foo and Hameed, 2009). Most developing countries, such as Sri Lanka, municipal solid waste is dumped to open dumpsites as the easy common practice, and these dumpsites are located adjacent to surface water bodies (Wijesekara et al., 2014). Leachate generated from these dumps discharges directly to the environment without any treatment due to the cost involved in treatment, lack of skilled labor and complexity of leachate.

Landfill leachate composition varies spatially and temporally due to the differences in moisture content, waste composition, weather conditions, site hydrology, amount of precipitation, the interaction of leachate with the environment, waste compaction etc. (Kulikowska and Klimiuk. 2008; Umar et al., 2010). Pollutants generated from Municipal Solid Waste (MSW) can be divided into four groups; as inorganic macro components, dissolved organic matter, Xenobiotic Organic Compounds (XOCs) and heavy metals originating from industries or households (Kjeldsen et al., 2010; Luo et al., 2019). Previous studies have characterized landfill leachate collected from several MSW dumpsites in Sri Lanka. Those results indicate that the leachate from dumpsites are rich in pollutants (NH_4^+, Fe^{3+}, Se^{5+}, Pb^{2+}, F^-, Cl^-, PO_4^{3-}, BOD_5 and COD) exceeding the maximum tolerance limits provided by the Sri Lanka

Standards Institute (Sewwandi et al., 2016; Wijesekara et al., 2014). The status and extent of the leachate pollution can be assessed by the Leachate Pollution Index (LPI), particularly in places where leachate is draining to the sensitive environment posing a high risk of contamination (Manimekalai and Vijayalakshmi. 2012). This LPI value indicates the strength of leachate quality and helps to identify whether the leachate is hazardous for discharging, which pressurizes to develop a sustainable leachate treatment method and to recognize a suitable landfill design (Sharma et al., 2008). Although, previous studies have assessed the quality of leachate in open dumpsites in Sri Lanka; no assessment of leachate pollutant potential of the open dumpsites has been reported (Nayanthika et al., 2018; Kumarathilaka et al., 2016).

The nitrification/denitrification system is potentially the cheapest and most efficient process to extract leachate nitrogen (Cema et al., 2007). However, the use of the biological treatment process for municipal landfill leachate usually results in low BOD and COD removal. This is mainly because of the high ammonia concentration and the presence of bio-refractory organics (such as humic substance or surfactants) (El-Gohary and Kamel, 2016). Generally, conventional leachate treatment methods consist of coagulation, flocculation, settling (Torretta et al., 2016; Kamaruddin et al., 2017), reverse osmosis (Koh et al., 2008) and air stripping (Ferraz et al., 2013) and because of their relatively high expenditure and operating costs, these systems do not seem economically feasible (Gao et al., 2014). Consequently, these techniques in an application of large-scale treatment are not economically acceptable for developing nations in the tropics (Wiszniowski et al., 2006). Therefore, a sustainable technical approach is required for treating leachate to mitigate the impact on the surrounding environment.

A novel microbial nitrogen extraction process called anaerobic ammonium oxidation (anammox) has been reported over the past decade, which is capable of oxidizing ammonium into nitrogen gas under anaerobic nitrite as the electron acceptor (Van de Graff et al., 1995; Phan et al., 2017; Wu et al., 2018; Lin et al., 2011). On the other hand, Municipal Solid Waste Biochar (MSW–BC) has received recent attention for its ability on volume reduction of waste and adsorbing nutrients (Ghezzehei et al., 2014; Jayawardhana et al., 2016; Gunarathne et al., 2018). A few studies have shown that the MSW–BC could be used effectively to mitigate contamination such as heavy metals and dyes (Agraioti et al., 2014; Parshetti et al., 2014; Li et al., 2015). Furthermore, MSW–BC, appears to have the ability to be used as an effective adsorbent in the removal of contaminants from landfill leachate, particularly, Volatile Organic Compounds (VOCs) (Jayawardhana et al. 2019a, Jayawardhana et al., 2019b). However, these data are solely from batch adsorption experiments, whereas column and laboratory-scale integrated treatment experiments are lacking.

Furthermore, no records exist in the literature on cost-effective hybrid treatment systems consisting of anammox and biochar for the treatment of landfill leachate with a high ammonia concentration. At the same time, limited work has been conducted on landfill leachate treatment using real leachate mostly research reports are on artificial leachate (Szymański et al., 2018; Wimalasuriya et al., 2011). Therefore, the objectives of the study are to assess the capacity of an anammox system as a combination of the MSW–BC-based column and constructed wetland to treat real landfill leachate. In the scope of this analysis, an anammox system, biochar reactor and designed wetland processes are sequentially used to assess the quality of the leachate treatment specific to its LPI. Pollutant extraction was quantified in each phase to determine treatment efficiencies. More information on the characterization of leachate, experiments on the pilot and bench-scale and key design considerations are addressed.

2. Materials and methods

2.1. General description of the study area

A land between Boralesgamuwa – Borupana in Colombo District, Sri Lanka has been used as an open dumping site for more than 25 years. This open dumpsite is located within a wetland and known as Karadiyana open dumpsite, surrounded by surface water sources. No treatment has been undertaken to eliminate toxins from leachate and therefore pollute the underlying soil and surface water directly. This contributes to numerous adverse environmental effects such as degradation of the ground and surface water, air, visual/aesthetic emissions (EIA report, 2015).

2.2. Determination of leachate discharge

The sampling location was chosen by careful analysis around the garbage dump considering topography as well as the leachate path where maximum flow may occur. It was almost impossible to determine the exact leachate flow direction during heavy rain. Therefore, a proper drainage route has been identified for the measurement of surface discharge of leachate which runs along the edge of the waste dump. A V-notch weir was established across the leachate path close to its discharge point to assess the surface discharge, as shown in Fig. 1. The precipitation data were obtained through an on site rain gauge station to track variations of the surface discharge. Flow measurements were taken from April to September 2018.

Leachate samples were obtained from the landfill site from the locations showed in Fig. 2. Dissolved Oxygen (DO), pH and temperature were measured in-situ (Simpson et al., 2017). The samples were then transported in an ice cooler to the laboratory and stored at 4 °C. The list of parameters analyzed are provided in Table 1. The specimens were allowed to return to room temperature before analysis and measurements were performed for parameters of leachate. Leachate characterization was done in 2018 and 2019. The solid content of the leachate (TSS, VSS, TS, and VS) was obtained by membrane filter paper techniques. A portable data logging colorimeter (Model DR/890, HACH, USA) were used in measuring COD while carbon type was obtained by the TOC analyzer (Shimadzu Japan). Cations and heavy metal content were determined using an Inductively Coupled Plasma Emission Spectrometer (ICP-OES 7000 Thermo, USA) as well as anions by ion chromatography (Shimadzu CDD 10 A). Quantitative analysis of VOCs was performed using static headspace equipped gas chromatography coupled mass-spectrometer (Shimadzu QP 2010).

2.3. Calculation of leachate pollution index

The Leachate Pollutant Index (LPI) was calculated using a weight arithmetic approach that helps identify the state of the leachate condition (Kumar and Alappat. 2005; Rafizul et al., 2012; Mishra et al., 2018). It was measured using the following formula based on the Delphi Rand Corporation methodology (Kumar and Alappat. 2005).

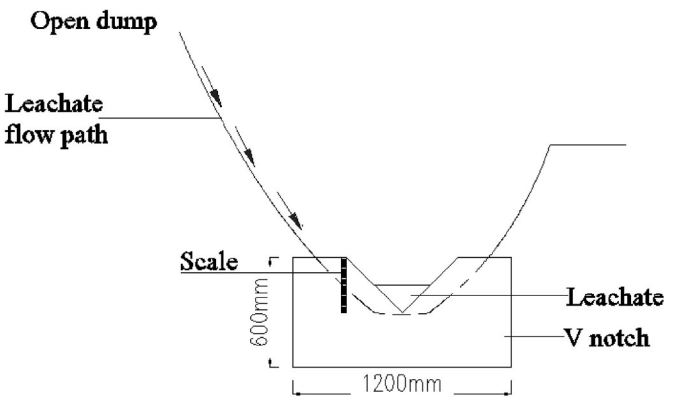

Fig. 1. V notch across the leachate path.

Fig. 2. Karadiyana open dumpsite and leachate flow (left) and map of the sampling locations (right).

Table 1
Average values of leachate quality parameters in Karadiyana dumpsite.

Parameter	L1		L2		L3		WDS
	Average	SD	Average	SD	Average	SD	
pH	7.95	0.20	8.18	0.09	8.24	0.39	6–8.5
EC (mS/cm)	14.29	11.30	12.72	6.77	9.87	2.997	–
NO_3–N (mg/L)	140.00	67.82	112.00	26.88	105.83	24.99	10
NO_2–N (mg/L)	2.28	1.83	0.73	0.10	3.20	5.20	–
PO_4 (mg/L)	71.75	59.10	92.58	119.16	48.25	28.85	5
SO_4(mg/L)	217.50	195.34	145.00	103.76	186.25	96.21	1000
NH_4–N (mg/L)	1867.50	739.07	5325.00	7662.19	1000.00	346.41	50
BOD (mg/L)	957.10	54.18	856.13	538.71	1329.03	994.25	30
COD (mg/L)	33,325.00	12,436.87	11,377.50	3756.07	8350.00	4625.65	250
TOC (mg/L)	508.75	321.29	865.41	503.69	238.19	118.61	–
TSS (mg/L)	2229.67	2071.19	202.50	44.80	54.67	7.49	50
Cd (μg/L)	4.20	0.45	3.61	0.85	2.75	0.46	100
Zn (μg/L)	1184.06	51.12	1270.53	376.47	968.40	166.42	5000
Ni (μg/L)	91.23	0.96	182.40	88.06	116.11	28.91	3000
Cu (μg/L)	266.14	17.01	210.43	22.27	309.45	41.15	3000
Pb (μg/L)	23.78	5.29	35.95	18.52	14.96	5.56	100
Cr (μg/L)	96.98	7.18	428.67	105.59	119.97	13.55	100
As (μg/L)	67.18	30.14	42.24	13.37	24.58	6.29	200
Fe (mg/L)	47.95	6.84	21.45	2.03	5.76	0.76	3
Al (mg/L)	6.69	0.49	4.27	1.16	0.95	0.54	–
Mn (mg/L)	1116.13	4.07	172.76	23.60	97.84	5.35	–
Si (mg/L)	11.26	0.98	10.855	0.93	6.40	1.34	–

(SD – Standard Deviation, WDS - Wastewater discharge Standards - National Environmental Regulations by Central Environment Authority, Sri Lanka).

$$LPI = \frac{\sum_{i=1}^{m} W_i P_i}{\sum_{i=1}^{m} W_i}$$

where, W_i is the weight factor for the ith pollutant variable, P_i is the sub-index score of the i^{th} pollutant variable, and m is the number of known concentrations of leachate contaminant variables., A total of 18 parameters of leachate, were selected for inclusion in LPI, and their weight factors are taken based on significance levels provided by the panelists on a scale of 1–5 and are summarized in. Supplementary Information Table S1. The sub-index values have been determined from the average sub-index curves shown and stated in the literature (Kumar and Alappat. 2005). The eighteen variables of leachate pollutants are grouped into three components in order to formulate three sub-LPIs such LPI organic, LPI inorganic and LPI heavy metals. Using the above formula, the three sub-LPI scores are computed separately. Ultimately, these LPI values have been aggregated using the following formula to determine the total LPI (Kumar and Alappat. 2005).

$$LPI = 0.232\ LPI_{or} + 0.257\ LPI_{in} + 0.511\ LPI_{hm}$$

where LPI is the overall LPI, LPI_{or} is the sub leachate pollution index organic component value; LPI_{in} is the sub-leachate pollution index inorganic component value and LPI_{hm} is the sub-leachate pollution index heavy metal component value.

2.4. *Biochar production*

Biochar was obtained in the absence of oxygen through the process of thermal decomposition called pyrolysis. In order to improve the production, biochar was produced by using the constructed pyrolizer, as shown in Fig. 3. The pyrolyzer was made of brick walls with enough to fill four 200 L barrels.

Four barrels were placed at once and fired about 3 h by using wood. The temperature was maintained inside the pyrolizer at ~450 °C Woods were arranged inside the pyrolizer in order to maintain the constant temperature and regularly checked by using a thermocouple thermometer.

2.5. *Enrichment of anammox bacteria*

Anammox bacteria were grown under the laboratory condition. The anaerobic digester was filled with 2 L of anammox culture and 5 L of wastewater for a cycle of reaction for four days. The discharge was 2 L/day. The reactor was made of cylindrical plastic material with working volume of 8 L and sealed with cover to maintain an anaerobic environment. Few outlets were present and those were used for gas, samples collections and sludge. The reactor was flushed with Ar/CO_2 (95/5%) gas mixture to maintain anaerobic conditions during the experiment and support CO_2 for anammox bacteria. Inside the reactors, the liquid temperature was regulated to be within the 35 °C range (Nitisoravut and Chamchoi, 2007). A synthetic medium containing $NaNO_2$ (30 mg N/L), $NaNO_3$ (0–14 mg N/L), NH_4Cl (30 mg N/L), $KHCO_3$ (0.4 g/L), $MgSO_4.7H_2O$ (0.25 g/L), KH_2PO_4 (0.040 g/L) and $CaCl_2$ (0.30 g/L) and micro-nutrients dissolved in demi-water was used to feed anammox bacteria (Hendrickx et al., 2014). Ammonia and nitrite influent and effluence have been determined to confirm their existence and incubation in two weeks.

2.6. *Pilot plant setup*

The pilot-scale treatment plant was established consisting of an anammox reactor, biochar column reactor and a constructed wetland. For this, an anaerobic digester consisted of two reactors having the same dimensions; 140 mm diameter and 250 mm height were used. One was used as an anammox reactor and the other one was used as a biochar reactor of the pilot plant. Before feeding leachate, it was diluted 1:10. The first reactor was filled with 3 L of anammox bacteria. The second reactor was filled as alternative layers starting from the bottom upward with MSW biochar and in a ratio of 2:1. The biochar was added in small layers while adding water. Each layer was compacted well with a rod to remove air voids. A 50% of laterite was filled in treating the area in the reactor and remaining was filled in the top of the reactor to prevent biochar floating.

Finally, a constructed wetland was established with dimensions of 1 × 0.3 × 0.45 m, so that the effluent of the biochar tank can flow through the constructed wetland. The tank was filled with from bottom upwards a 100 mm layer of gravel, 50 mm layer of laterite, 100 mm layer of 1:10 biochar/sand mix, 50 mm layer of sand and 25 mm layer of pebbles. *Canna indica* was planted based on aesthetically pleasing to remove contaminants from the biochar reactor effluent. The combined treatment system shows in Fig. 4.

The flow rate of the pilot treatment plant was 2 L/day. Leachate was diluted in 1:10 ratio in order to reduce ammonia concentration and to monitor anammox bacteria behavior. Due to the deficient concentration of nitrite in leachate relative to ammonia, sodium nitrite has been applied to the leachate as the concentration of ammonia and nitrite is 1: 1.3 ratio. Leachate was initially fed to an anammox reactor, so it passed through the column reactor of biochar and constructed wetland.

3. Results & discussion

3.1. *Leachate discharge and composition*

Leachate discharge has been calculated using a V-notch to determine basin parameters. Fig. 5 illustrates how precipitation influences the discharge of leachate. The leachate generated from the open dump is intensified by the rainfall over the catchment of the dumpsite. The total discharge of leachate was found to be 2.4 L/s. Upon modifying the topography of the dump pile, the total leachate discharge was reported as 1.1 L/s with the maximum 46 mm/day rainfall. The pollutant leachate concentration can change with the rainfall intensity.

Leachate displays relatively constant pH with small variations and may range from 7.5 to 9. Nonetheless, the results show a slightly simple

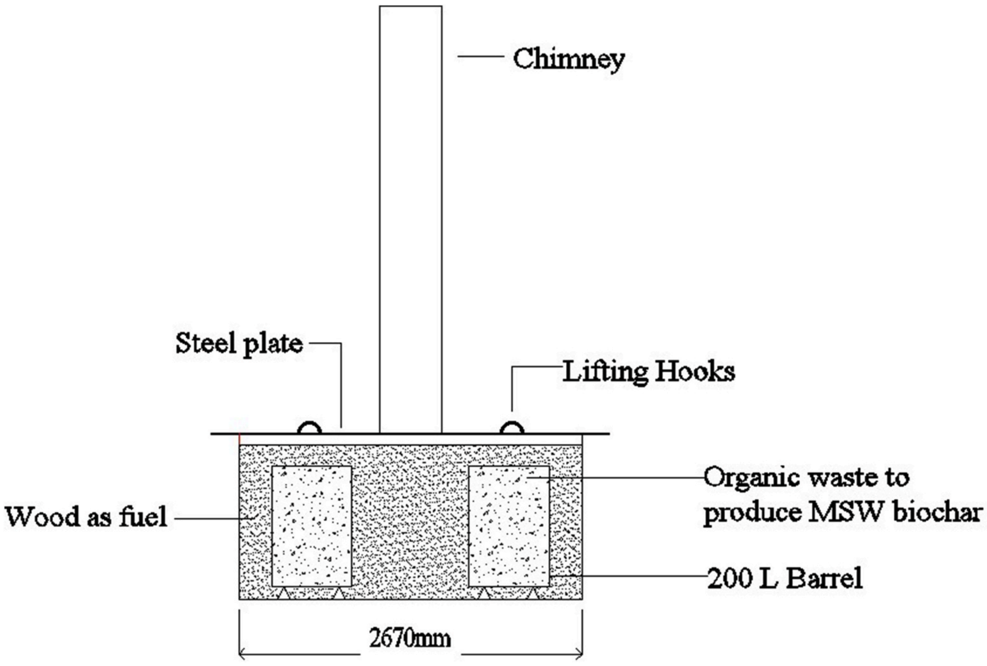

Fig. 3. Constructed pyrolizer.

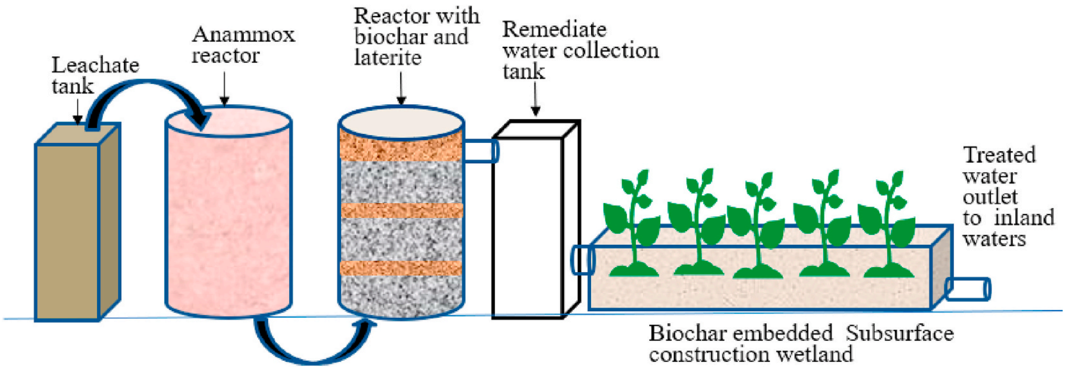

Fig. 4. Schematic diagram of pilot treatment systems.

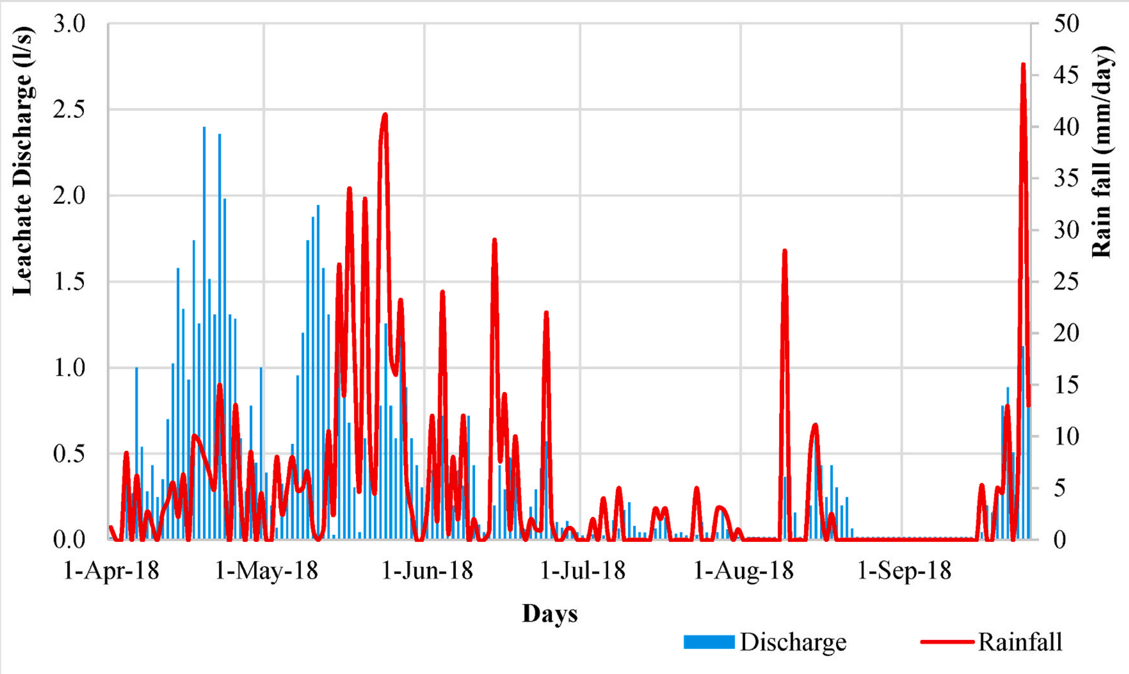

Fig. 5. Leachate discharge vs. rainfall from April to September 2018.

pH value for leachate (range: 7.4–8.8), which indicates an initial methanogenic stage (Kulikowska and Klimiuk, 2008). It has also been shown in previous research that significant landfill sites in Sri Lanka experience methanogenic conditions under the same climatic conditions (Wijesekara et al., 2014). Also, high levels of phosphate in leachate may be due to the organic refuse load containing phosphorus. During its biodegradation, this organic material (mainly phospholipids and phosphoproteins) releases phosphorus and ultimately raises the levels of phosphate. According to the leachate quality parameters, the phosphate levels varied considerably within wide ranges exceeding the country's standards values for wastewater discharge to the inland waters as 5 mg/L (Table 1) [National Environmental (Protection and Quality) Regulations 2008]. Comparatively, a lower value noted in location 3 as 48.25 mg/L while the higher value noted in location 1 as 71.75 mg/L. This concentration level may be due to the enhanced leaching of phosphate from the waste dumped at the site during the precipitation, followed by the dilution effect of rainfall. Also, this can be varied with the dump waste composition in each and every location.

The COD value in leachate was found in range 8350–33,325 mg/L (Table 1), indicating high contamination, which is considered to be an essential parameter for allowing landfill leachate to be discharged in most countries (Koshy et al., 2008). Also, the BOD_5 concentration in leachate varied within the range of 856–1329 mg/L, which was found to be higher than the norm for inland leachate disposal 30 mg/L [National Environmental (Protection and Quality) Regulations 2008]. These data suggest higher organic and inorganic pollutant loading in landfill leachate from Karadiyana dumpsite.

The levels of selected trace metals such as Cu, Ni, Cd, Zn, Pb, and Cr in raw leachate are varied within the range of 210–309, 91–182, 2.7–4.2, 968–1270, 15.0–36.0 and 97.0–428.7 μg/L (Table 1), respectively, in different locations at the Karadiyana open dumpsite. However, only the chromium concentration exceeding the permissible value adopted by National Environmental (Protection and Quality) Regulations 2008. This can be mainly attributed mainly to the disposal of chemical plants, oil and coal. Concentration of Zn may indicate the presence of fluorescent tubes, batteries, and a variety of food wastes. Besides, Fe level varied from 5.8 to 47.9 mg/L, indicating a high concentration in leachate even exceeding country's wastewater discharge permissible level: 3 mg/L. Also, aluminium in leachate varied within the range of 0.95–6.69 mg/L, indicates higher value than the country's wastewater discharge permissible level of 0.2 mg/L [National Environmental (Protection and Quality) Regulations 2008]. The sources of aluminium may be attributed to waste coming from engines, cables, etc. Accordingly, heavy metal results expressing a treat to the surrounding environment since the concentration are high. Similar statistics were found from a previous landfill leachate characterization on municipal solid landfill leachate, Gohagoda, in Sri Lanka (Wijesekara et al., 2014). According to the results, heavy metals can be transferred from soil to biotic environment and these polluted areas are not suitable as food due to the potential risk of bioaccumulation. Therefore, the potential risk of these contaminants cannot be ignored due to their adverse effects on groundwater and plants (Wijesekara et al., 2014).

3.2. Evaluation of leachate pollution index for karadiyana landfill leachate

The three sub-LPIs were determined using the above method and reported in Table 2. The concentration of certain parameters: Arsenic and Cyanide were presumed based on past test results from the Zoology Department, Sri Jayewardenepura, Sri Lanka (Test report, 2017). In the estimation of LPI_{hm} values, some errors may be anticipated as the concentration of some of the parameters in this subgroup was assumed. The error may not be high as the concentration/sub-index values of the parameters assumed are dependent on the concentration of contaminants. It has been reported that if the sub-index values of the missing contaminants are either not too high or too low, errors in measuring LPI values may not be significant (Kumar and Alappat. 2005).

It has been found that the LPI of Karadiyana landfill leachate is 28.88 in 2018, which is relatively high (Fig. 6). Three different locations were selected and determined the leachate pollutant index variation with time. All three locations show a similar variation in LPI value, as shown in the graph.

The LPI value of the leachate is highly variable depending on the dump composition (Palaniandy et al., 2009), site hydrology, sampling procedures, waste compaction, the interaction of leachate with the environment, amount of precipitation and landfill design and operation (Reinhart and Grosh, 1998). LPI values have no considerable difference among the sampling points and remains at a high level throughout the study period (Fig. 6).

High LPI value suggests a toxic, un-stabilized landfill site and poor environmental status (Esakku et al., 2007). High concentrations of BOD, COD, ammonia, nitrogen, and TDS value of the leachate samples are

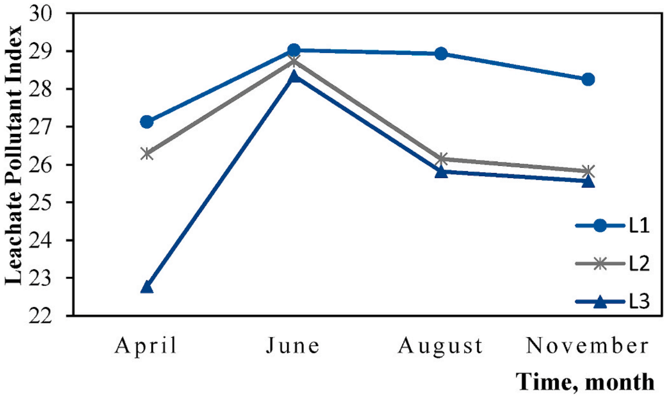

Fig. 6. Variation of leachate pollutant index.

responsible for high individual and cumulative pollution rating. It is, therefore, necessary to consider the LPI based on these criteria before deciding on the method of treatment.

LPI_{hm}'s low value indicates that the heavy metals in the leachate are low in concentration and do not pose a potential hazard to biological leachate treatment. The low value of LPI_{hm} also means that the waste collected in the landfill is screened for metals before landfilling, and the average value of LPI_{in} suggests that the inorganics in the leachate are not high in concentration (Kumar and Alappat, 2005). However, the LPI does not consider the accumulation factor with regard to the leachate generation and therefore, it is impossible to obtain an idea related to the environmental accumulation of toxic metals from landfill leachate. Nevertheless, when constructing a leachate treatment system for this landfill, account should be taken of the high concentration of ammonia nitrogen. Such three sub-LPI values also mean that the best treatment choice for this leachate would be biological (Kumar and Alappat. 2005).

3.3. Anammox process

The extraction performance of ammonia and nitrite was initially noted as 84.6 and 97.3% at the beginning and as 85 and 97.8% after two weeks, respectively. It indicates the presence of anammox bacteria in the reactor against the composition of leachate. The improvement in the

Table 2
Three sub LPIs and overall LPI of the leachate at Karadiyana open dumpsite, Sri Lanka.

Index	Parameter	Weight factor (W_i)	Pollutant concentration	Sub index value (P_i)	W_i. P_i
LPI organic	COD (mg/L)	0.267	5990	67.5	18.0225
	BOD_5 (mg/L)	0.263	1022	30	7.89
	Phenolic compounds	0.246	21.24	30	7.38
	Total coliform	0.224	2100	79	17.696
	Summation	1			
	LPI_{or}				50.9885
LPI inorganic	pH	0.214	8.24	2.5	0.535
	TKN (mg/L)	0.206	1718	60	12.36
	Ammonia nitrogen (mg/L)	0.198	972	97	19.206
	TDS (mg/L)	0.195	11	24	4.68
	Chloride (mg/L)	0.187	3954	30	5.61
	Summation	1			
	LPI_{in}				42.391
LPI heavy metals	Chromium (mg/L)	0.125	1.00	5.5	0.6875
	Lead (mg/L)	0.123	0.70	6.5	0.7995
	Mercury (mg/L)	0.121	0.00	5	0.605
	Arsenic (mg/L)	0.119	0.087	5	0.595
	Cyanide (mg/L)	0.114	0.00	5	0.57
	Zinc (mg/L)	0.110	10.37	5.5	0.605
	Nickel (mg/L)	0.102	0.497	5	0.51
	Copper (mg/L)	0.098	1.183	6.5	0.637
	Iron (mg/L)	0.088	31,422	80	7.04
	Summation	1			
	LPI_{hm}				10.874
Overall LPI	$0.232\ LPI_{or} + 0.257\ LPI_{in} + 0.511\ LPI_{hm}$				28.88

reactor removal output was seen for for two weeks. It shows anammox bacterial abundance for 14 days. The growth rate of anammox bacteria was estimated in previous reports, based on the yield of biomass and the rate of removal of nitrogen during their cultivation (Nitisoravut and Chamchoi, 2007; Zhang and Okabe, 2017; Ciesielski et al., 2017). The increasing rate of nitrogen load was lowin the stationary stage and therefore the time for doubling anammox bacteria was also observed as low. Hence, there are no data on the direct counting of anammox bacteria for the measurement of doubling time (Nitisoravut and Chamchoi, 2007). Nevertheless, the present results indicate the anammox bacteria's survival and enrichment under the given condition.

3.4. *Treatment performance*

Samples were taken separately after the treatment through an anammox reactor, biochar column and constructed wetland and investigated for the removal of conductivity, nitrate, nitrite, ammonia, COD and phosphate. To prevent the inhibitory effect of free ammonia, it is necessary to maintain a pH range between 7.0 and 8.0 in the anammox reactor. The pH range in the anammox reactor was within the range initially and showed a slight increase after 15 days Fig. 7(a). The pH of the biochar column effluent was remaining in the alkaline range due to the alkalinity of MSW-BC (Hossain et al., 2011). However, the pH was reduced by the constructed wetland, which may be due to the removal of the contaminant ions through the constructed wetland. Fig. 7(b) indicates the variation of conductivity through the treatment train. The total conductivity removal was noted as 69.9%. Although, this removal separately through the anammox reactor, biochar reactor, and constructed wetland was 4.5, 4.4 and 61.0%, respectively. According to the results, it can be noted that there is a more significant reduction through the constructed wetland, and this may be due to the change in the ion content and electrolytes. Influent COD range was noted in the range of 290–370 mg/L. There is a slight removal of COD through the anammox reactor and biochar reactor, indicating removal efficiency as 7.8 and 7.1%, respectively, as in Fig. 7(c). Although, the effluent concentration of constructed wetland is below the acceptable value for inland surface water value 250 mg/L given by NEA, the COD removal efficiency through the constructed wetland was 64.8% and altogether total removal was 79.7%. Constructed wetlands have a strong ability to purify the organic matter through the wetland matrix sedimentation and may be with the rhizospheric microbial support. Thus, it can be concluded that this treatment train is suitable for removing COD in the leachate.

Ammonia removal was greatly achieved by the anammox reactor and it was 93.7%. Biochar involved in removing ammonia at the beginning when the effluent of an anammox reactor reached its peak value, as shown in Fig. 7(d). This ammonia removal efficiency achieved by the biochar reactor by absorbing and it was noted as 6.3%. Although the effluent ammonia concentration to the constructed wetland has become zero. This suggests the success of ammonia removal through the Anammox reactor.

The average influent nitrite concentration was about 15 mg/L. The effluent nitrite concentrations of the reactor during the initial phase showed a higher value than the influent concentrations, as shown in Fig. 7(e). The levels of effluent nitrite have decreased over time after the conversion of nitrite to nitrogen gas under initial oxygen-limited conditions. Nitrite removal efficiency through an anammox reactor was noted as 89.3%. Moreover, the biochar reactor shows a promising result over adsorbing the increased nitrite concentration at the beginning. This pollutant removal efficiency was noted as 8.8%. Finally, 0.6% of total removal was achieved by the constructed wetland. Altogether, the total pollutant removal efficiency was noted as 98.7%, and therefore, it can be concluded that the whole treatment system is efficient for nitrite removal.

The average influent nitrate concentration was about 55 mg/L. There is an increment in nitrate concentration in effluent of anammox, as shown in Fig. 7(f). Ammonium serves as an electron donor and nitrite as an electron acceptor in the anammox cycle and eventually converted by anammox bacteria anaerobically into primarily nitrogen gas and some nitrate such that 10% of the N-feed to nitrate (Van de Graff et al., 1995). Hence, as a by-product, this process produced nitrate. That is why there is a spike in nitrate. Also, in some initial samples, the overshooting of effluent nitrate concentration was observed. It was believed that the phenomenon was associated with oxidation (Nitisoravut and Chamchoi, 2007). Because of this, nitrate concertation was increased by 5.3% in the effluent of the anammox reactor. Altogether, total nitrate concentration in biochar influent was increased. Biochar reactor reduced this by 25.8% and constructed wetland by 77.7%. Nitrate removal efficiency through the whole system was 98.2%. It can be noted that the efficiency of pollutant removal through the biochar reactor has been reduced over time. However, the constructed wetland shows a greater involvement in the process by reducing nitrate concentration. The nitrate in constructed wetland may be removed by nitrification and denitrification of microorganisms, absorption of plants and adsorption and filtering of the matrix. Phosphate removal performance through an anammox reactor, biochar reactor and wetland system are indicated in Fig. 7(g). Phosphate removal efficiency is higher through constructed wetlands; 70% when compared to the biochar reactor; 9.2%. There is no considerable removal of phosphate through the anammox reactor as it was 1.7%. Phosphate concentration in the effluent of an anammox reactor was almost the same as the influent. Anammox bacteria were not involved in reducing phosphate in the leachate. Further, phosphate removal efficiency through the biochar reactor was higher at the beginning as it absorbs pollutants effectively at the beginning but with the time passes the removal capacity was reduced. However, phosphate removal was substantially achieved by the constructed wetland as it absorbs pollutants to the plants and absorbs and precipitate to matrix and total removal efficiency was observed as 80.9%. In conclusion, biochar column showed a saturation in 15 days for conductivity, COD, nitrate and phosphate (Fig. 7) which suggests the need to use of few columns or barriers or switching the columns every 5 days if the system is used in the field.

The color variation of influent and effluent was measured from the National Water Supply & Drainage Board, Sri Lanka and results shown in Fig. 8. Since the Y-axis unit indicates the extinction per meter optical path length, considerable color variation can be described. This removal was noted as 82.35% in blue, 84.04% in red and 85.02% in yellow. A considerable color reduction can be seen through the whole system. Past research has shown regarding biochar embedded subsurface constructed wetland, which gives promising results (Athapattu et al., 2017) followed by the biochar barricades.

In Sri Lanka, there are many small-scale dump sites belonging to local authorities, which are practicing composting as a solution for solid waste management (Welikannage and Liyanage, 2009). But the leachate generation is ignored during the composting process. Therefore, the proposed treatment train for leachate can be a promising technique for small composting sites.

4. Conclusion

This study highlights the performance of anammox bacteria followed by biochar reactor and biochar embedded constructed wetland for the treatment of landfill leachate from an open dumpsite in Sri Lanka. The results showed that leachate is not suitable for an environmental discharge without treatment as some parameters of water quality such as TDS, ammonia, COD, nitrate and heavy metals were found above the acceptable limit. The high values of landfill leachate LPIs showed a substantial amount of contaminants present in the leachate. The results presented in this study show that a combination treatment system comprising an anammox reactor, biochar reactor and constructed wetland allowed reliable discharge effluent quality by improving the removal efficiency of many pollutants. Although biochar column showed a saturation in 15 days, a combination of columns with changing

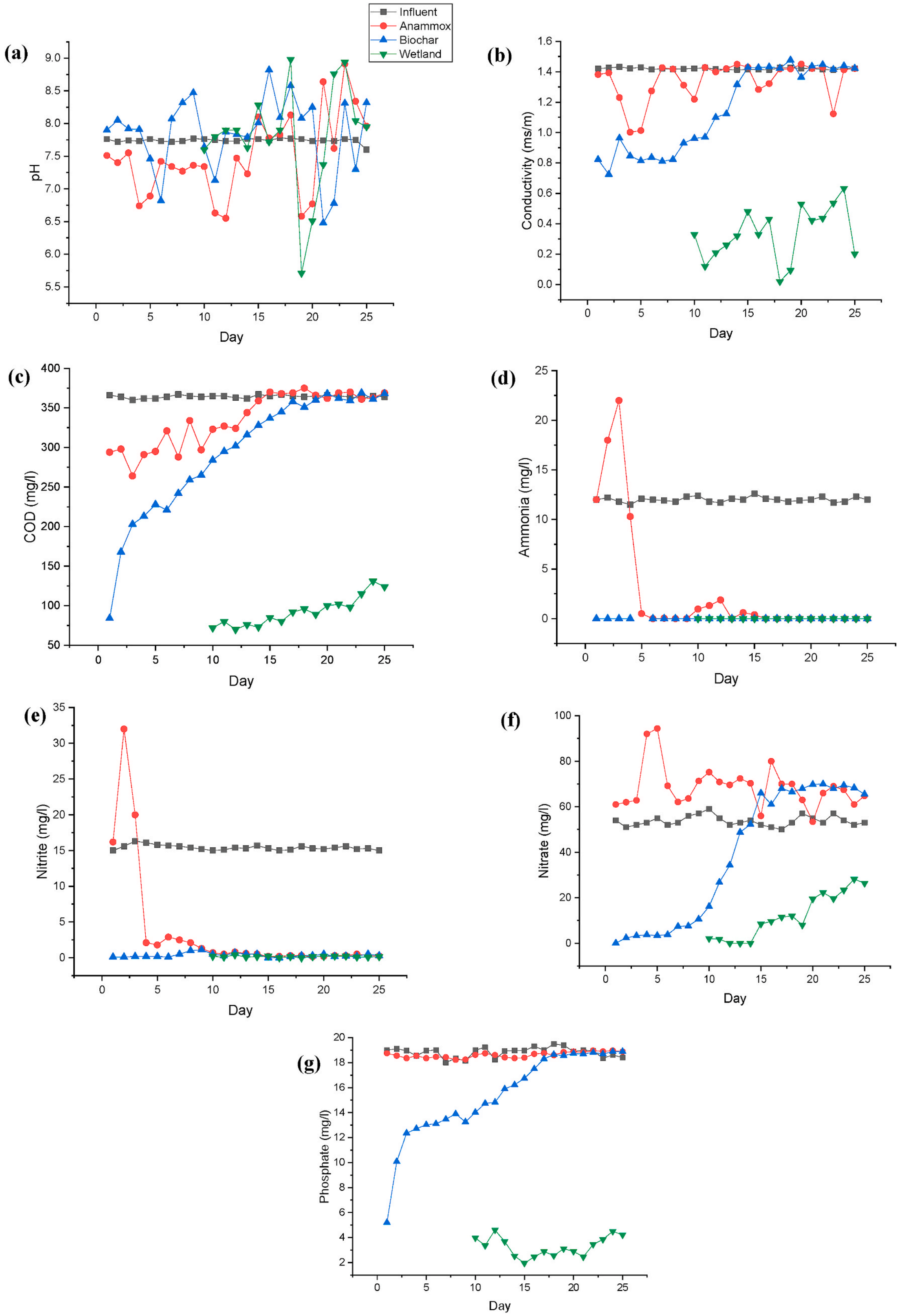

Fig. 7. Comparison of parameters between influent and effluent (a) variation of pH; (b) variation of conductivity; (c) variation of COD; (d) variation of ammonia; (e) variation of nitrite; (f) variation of nitrate; (g) variation of phosphate.

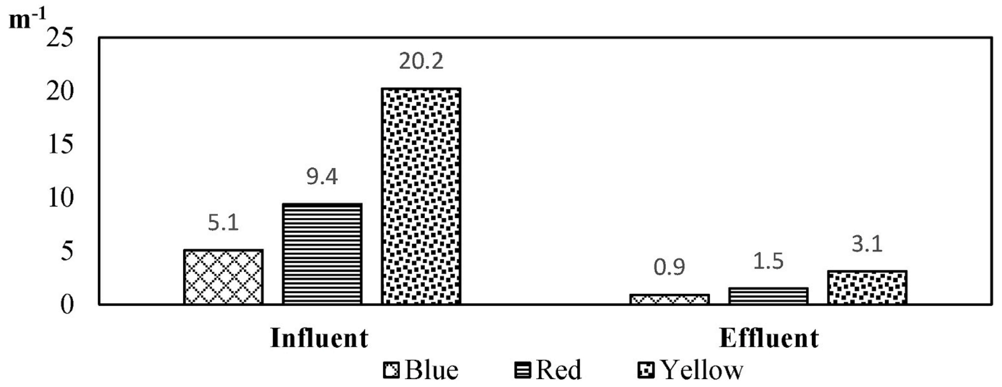

Fig. 8. Color variation of influent and effluent. (For interpretation of the references to color in this figure legend, the reader is referred to the Web version of this article.)

their position with respect to the influent, may improve the removal capacity. Therefore, it can be concluded that the present integrated process is an ideal technology for treating leachate especially for medium and small scale in open dumpsites.

Credit author statement

S.M.R. Joseph: Experimentation, data interpretation, data validation, writing the first draft, Prabuddhi Wijekoon: Experimentation, data interpretation, data validation, B. Dilsharan: Experimentation, data interpretation, data validation, N.D. Punchihewa: Experimentation, data interpretation, data validation, B.C.L. Athapattu: Conceptualization, writing the first draft, Supervision, Funding acquisition, reviewing and editing, Meththika Vithanage: Conceptualization, Supervision, Funding acquisition, writing-reviewing and editing.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgments

Authors acknowledgment NRC TO grant 18–021, National Research Council, Sri Lanka and the grant from UNESCO and the International Development Research Centre, Ottawa, Canada. The views expressed herein do not necessarily represent those of UNESCO, IDRC or its Board of Governors.

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