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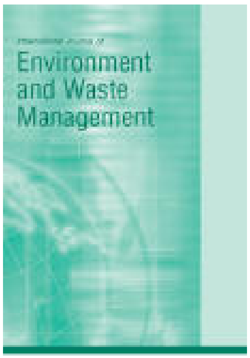

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Changes in physical, chemical and biological variables of the leachate generated from municipal solid waste subjected to saline solution application

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Abstract: The main characteristic of Brazilian municipal solid waste (MSW) is the high organic matter content that can accelerate the biodegradation process. This study aims to analyse the physical, chemical and biological variables of the leachate generate from a Brazilian MSW confined in five lysimeters over 1,022 days. In three of the lysimeters, the MSW was subjected to application of saline solution to evaluate its effect on the leachate characterisation. As a result, the anaerobic biodegradation process of the MSW subjected to saline solution remained in the anaerobic acid phase for almost the entire monitoring period, delaying the methanogenic phase in 350 days; however not completely inhibiting the biodegradation process. Moreover, the application of saline solution resulted in a decrease of the nitrogen ammonia concentrations in the leachate, nonetheless, it increased the values of electrical conductivity and total solids, and provided the slowest and gradual decrease in chemical oxygen demand.

Keywords: anaerobic biodegradation; lysimeter; municipal solid waste; MSW; saline solution; leachate; nitrogen ammonia.

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1 Introduction

Sanitary landfills are subjected to at least four phases of municipal solid waste (MSW) decomposition: an initial aerobic phase, an anaerobic acid phase, an initial methanogenic phase, and a stable methanogenic phase (Christensen and Kjeldsen, 1995; Kjeldsen et al., 2002; Barlaz et al., 2010). The MSW moisture content is the most important factor for its decomposition, since it accelerates the biodegradation process, providing conditions for microbial activity (Reinhart and Townsend, 1998; Al-Kaabi et al., 2009).

Bioreactor landfills are sanitary landfills designed and operated to enhance MSW decomposition, gas production, and MSW stabilisation (readily and moderately decomposable organic waste constituents) (Warith, 2002; Hossain et al., 2010, 2014). The fundamental process involved in the operation of bioreactor landfills is the recirculation of leachate and/or addition of water into the MSW mass (Reinhart et al., 2002; Hossain et al., 2014). Bioreactor landfills can be an important alternative to the conventional sanitary landfill in arid and semi-arid regions with seawater available, and coastal regions. However, operating a bioreactor landfill requires the availability of a sufficient amount of water close to the landfill facility, which is problematic in these regions where fresh water supplies are scarce (Al-Kaabi et al., 2009; Hossain et al., 2014).

A solution to the scarcity of fresh water could be the use of saline or brackish water from the ocean or underground. However, this kind of water may contain high salinity levels and other constituents that may be inhibitory to biodegradation processes (Khoury et al., 2000; Chen et al., 2008).

The inhibitory effect of sodium in anaerobic waste treatment has been investigated by Kugelman and McCarty (1965) and more recently by Anwar et al. (2016). This toxicity associated with the high salt concentrations can cause an increase in osmotic pressure (Khoury et al., 2000; Al-Kaabi et al., 2009), and, therefore, dehydration of bacterial cells (De Baere et al., 1984; Yerkes et al., 1997), increase in acclimation time, and increase in lag time (Al-Kaabi et al., 2009). In general, these aspects adversely affect the cellular level of microorganisms and delay or inhibit the anaerobic biodegradation process, reducing methane production and increasing stabilisation time (Khoury et al., 2000; Chen et al., 2008; Al-Kaabi et al., 2009; Hossain et al., 2014). However, some microorganisms are more susceptible to osmotic pressure than others (De Baere et al., 1984). According to McCarty (1964) the term ‘toxic’ is relative and the concentration at which a material becomes toxic or inhibitory may vary from a fraction of some mg L^{-1} to several thousand mg L^{-1}. At some very low concentration, the salt stimulates the biological reactions, but at some high concentration the biological activity, approaches zero (McCarty, 1964).

In addition, knowledge of the toxic effects of the salt on the MSW anaerobic digestion process will contribute indirectly to the research related to saline wastewater treatment (Rinzema et al., 1988; Soto et al., 1993; Omil et al., 1995; Feijoo et al., 1995).

In Brazil, the research related to the effects of saline water on the process of anaerobic digestion of MSW is uncommon. The main characteristic of this MSW is high organic matter content, whose biodegradation process is accelerated.

This study aims to analyse the anaerobic biodegradation process of a Brazilian MSW confined in five lysimeters, through physical, chemical and biological characterisation of leachate generated overtime (1,022 days, i.e., two years and ten months). In three of these lysimeters, the MSW were subjected to saline solution application and the other two were not. This monitoring served the purpose of understanding the effects of saline solution application in the variables present in the leachate, such as hydrogen potential (pH), total alkalinity, electrical conductivity (EC), biochemical oxygen demand (BOD), chemical oxygen demand (COD), oxidation-reduction potential (ORP), nitrogen ammonia, total solids and volatile fatty acids (VFAs).

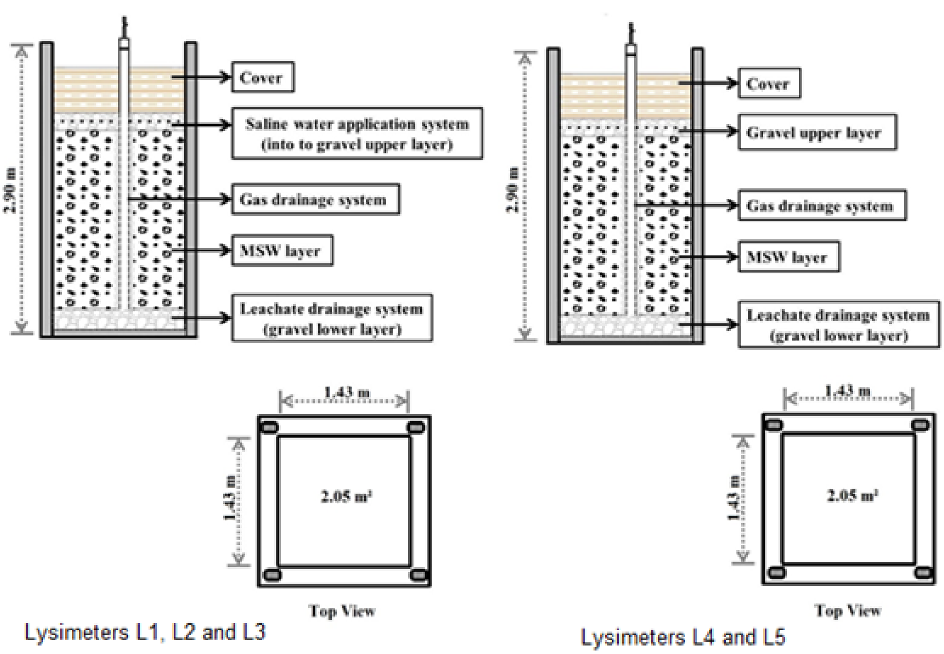
2 Materials and methods

The study was performed in the city of Campinas, located in the state of São Paulo, Southeastern Brazil. Campinas has an estimated population of 1,194,094 inhabitants and occupies an area of 794.6 km^2 (IBGE, 2019). The city is the third most populous municipality of the state and the 14th most populous in the country (IBGE, 2019). Its climate is tropical (type CWA, according to Koppen).

Five lysimeters were built in an area of the campus of the University of Campinas (Unicamp), in order to study the anaerobic biodegradation process of MSW. All lysimeters were subjected to rainfall. Nonetheless, three lysimeters (L1, L2, L3) were farther subjected to the application of saline solution, and whereas two lysimeters (L4 and L5) were not subjected to application of any liquid except rainfall and simulated conventional sanitary landfills (Manzatto and Miguel, 2019).

The lysimeters were built including leachate drainage system (gravel lower layer), compacted MSW layer, gas drainage system, gravel upper layer, saline solution application system (only in L1, L2 and L3) and cover (Figure 1). The capacity of each lysimeter was approximately 2.25 tons of MSW, and their dimensions are presented in Figure 1.

Figure 1 Lysimeters schematic design (see online version for colours)



The lysimeters were filled with fresh and full-sized MSW from the regular waste collection in the city of Campinas. The MSW was inserted into the lysimeters with layers of 12 cm, and were then compacted with a hammer until reaching a thickness of 6 cm (Favery et al., 2016). Gravimetric characterisation of the MSW is shown in Table 1 and was obtained by means of the methodology described by Miguel et al. (2016).

Table 1 MSW characterisation

Category	*%*	*Category*	*%*
Organic matter	41.9	Diapers and sanitary pads	3.6
Paper	7.7	Miscellaneous*	6.2
Cardboard	3.4	Wood	2.0
Hard plastic	3.7	Fabric	4.6
Soft plastic	8.8	Toilet paper	3.5
Metal	1.9	Construction waste	1.1
Glass	1.8	Pruning	7.9
Tetra Pak®	1.9	-	-

Notes: *Products such as rubber, foam, shoes, mixed material with more than one category, and hazardous waste such as paint cans, light bulbs, and batteries.

During the filling of the lysimeters L1, L2 and L3, the MSW layers were subjected to saline solution (45 litres per layer) after compaction. The saline solution was prepared with 32 g L^{-1} of salt (sodium chloride), which represents the salinity of sea water, in accordance with McCarty (1964). The saline solution was applied using a hose and evenly distributing the solution over the entire area of the MSW layer. After that, a new MSW layer was added, and so forth. After filling the lysimeters with the compacted MSW, a gravel upper layer was installed. A system consisting of 14 irrigation pipes was installed into this gravel upper layer for application of saline solution during the lysimeters operation. On this irrigation system was installed a compacted soil layer (cover) to unit weight close to 18 kN m^{-3}. The gas drainage system was installed at the time the lysimeters were filled. Further details are presented in Favery et al. (2016). Table 2 presents the final characteristics of the five lysimeters.

Table 2 Characteristics of the lysimeters

Variables	*Unit*	*Lysimeters*				
		L1	*L2*	*L3*	*L4*	*L5*
Cover thickness	cm	53	48	57	47	47
Soil unit weight (cover)	kN m^{-3}	17.93	18.60	16.54	18.46	18.76
Gravel upper layer	cm	22	18	13	15	18
MSW layer thickness	cm	192	176	167	160	185
MSW weight	kg	2,247	2,245	2,253	2,248	2,245
MSW unit weight	kN m^{-3}	5.75	6.47	6.60	8.69	5.97
Leachate drainage system thickness	cm	30	23	22	22	22

Saline solution application by irrigation system was initiated at 61 (L2), 63 (L1) and 68 (L3) days after the total filling of lysimeter, being conducted every fortnight. However, in the latter part of the study, a sequence of four consecutive weekly applications was adopted. Initially, the saline solution was prepared with 120 litres of water and 10 kg of salt (83 g L^{-1}), and subsequently with 24 kg salt (200 g L^{-1}) in order to evaluate inhibition of the biodegradation process. The amount of salt added in lysimeters L1, L2 and L3 was respectively 262.64 kg, 247.20 kg and 228.88 kg. The ratio of salt and MSW (kg salt/kg MSW) was 0.10 to 0.12. In total, L3 received 63 days of saline solution application, while L2 received 62 days and L1 received 61 days of this application.

Every day, since filling the lysimeters with MSW, measurements of ambient temperature, MSW temperature and rainfall were made. The monitoring occurred at 850 days for ambient temperature measures, 905 days for the mass temperature of MSW measures, and 930 days for rainfall measures.

The leachate samples were subjected to physical, chemical and biological analyses based on standard method (APHA, 2012): hydrogen potential (SM20 4500 H^+B*), EC (SM 2510 B*), BOD (SM20 5210 B*), COD (SM20 5520 D*), ORP (SM20 2580 B*), nitrogen ammonia (SM20 4500*) and total solids analysis (SM 2540D). Moreover, these leachate samples were subjected to total alkalinity and VFAs analysis, according to Ripley et al. (1986) and DiLallo and Albertson (1961), respectively. The analyses were performed only once for each variable in each sample and weekly, totalling 50 analyses. Sampling and analysis began from the third day after MSW filling for L2 and L4, after the fifth day for L1 and L3, and finally after 15 days for L5.

3 Results and discussion

3.1 Ambient and MSW temperatures and rainfall

The ambient temperature and cumulative monthly rainfall measurements over the monitoring period are presented in Figure 2. There was a direct relationship between ambient temperature and rainfall, that is, temperatures decreased during dry periods (0 to 180 days and 360 to 570 days) and increased during periods of high and intense rainfall (180 to 360 days and 570 to 720 days). Ambient temperature varied within the range of 12°C and 37°C and the cumulative monthly rainfall ranged from 0 mm to 321 mm. The months of greatest rainfall were December 2015 (321 mm) and January 2017 (297 mm).

The MSW temperatures and cumulative monthly rainfall measurements over the monitoring period are showed in Figure 3. There were no differences between the MSW temperature values measured in the lysimeters with and without the application of saline solution. The behaviour of MSW temperatures followed the same trend of ambient temperature behaviour over time, as evidenced in Mahler et al. (2003).

During the first days of MSW deposition in the lysimeters (up to 20 days), the temperatures presented high values, between 30°C and 41.2°C. After this period, temperatures fell and remained in the range of 20°C to 31°C.

According to Madigan et al. (2002), MSW temperatures between 20°C and 45°C occurred under action of mesophilic microorganisms. Al-Kaabi (2007) considered that the optimum temperature for enhancing MSW biodegradation is within the range of 30°C and 40°C.

Rainfall also influenced MSW temperatures directly. In dry periods, MSW temperatures were lower than those in the wet periods were. Mannapperuma and Basnayake (2004) reported that in tropical regions, ambient temperatures and higher average annual rainfall enhance both aerobic and anaerobic microorganisms, leading to rapid stabilisation of MSW landfill. Paixão Filho and Miguel (2017) stated that the MSW biodegradation process is accelerated under tropical conditions and, consequently, leachate composition presents peculiar characteristics.

Figure 2 Ambient temperature and cumulative monthly rainfall

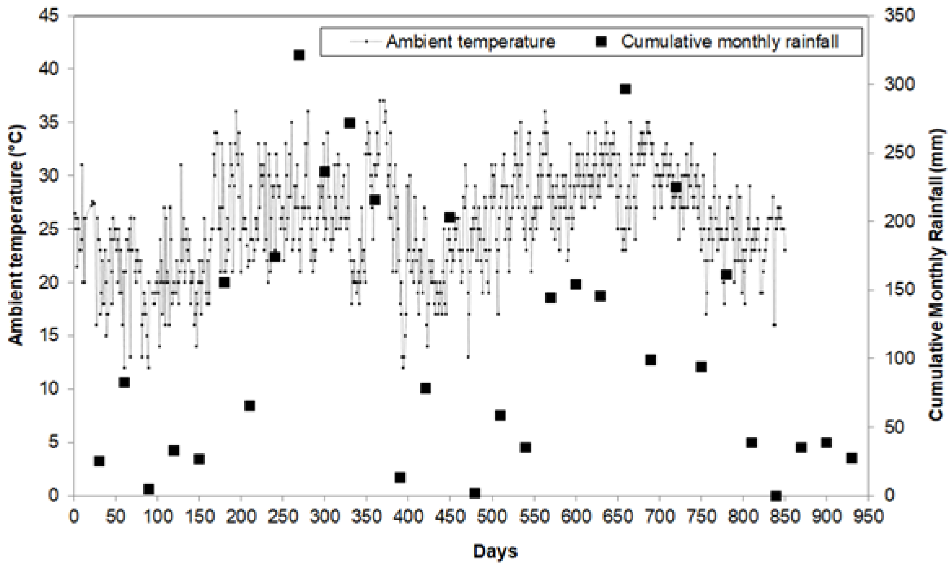
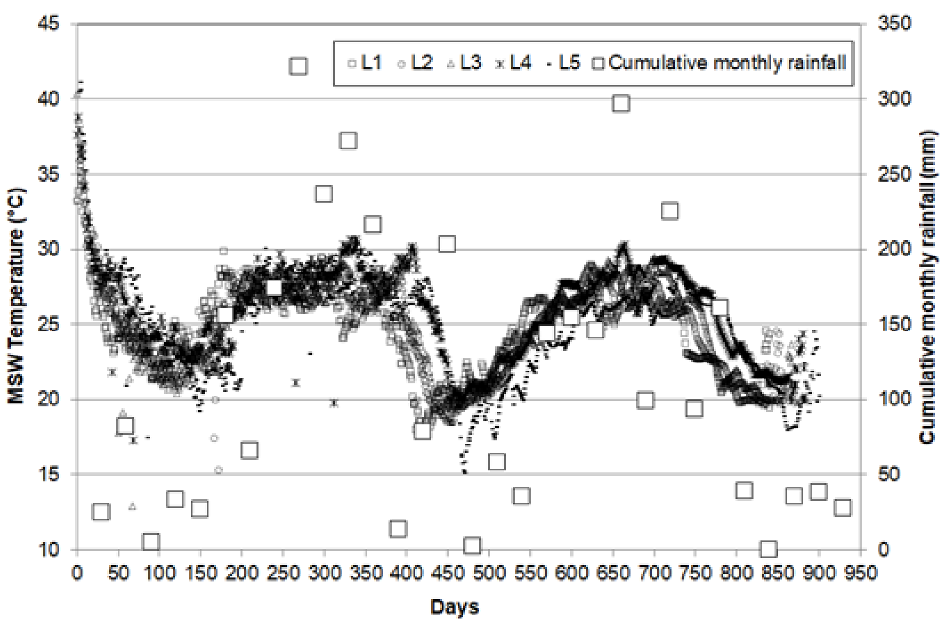


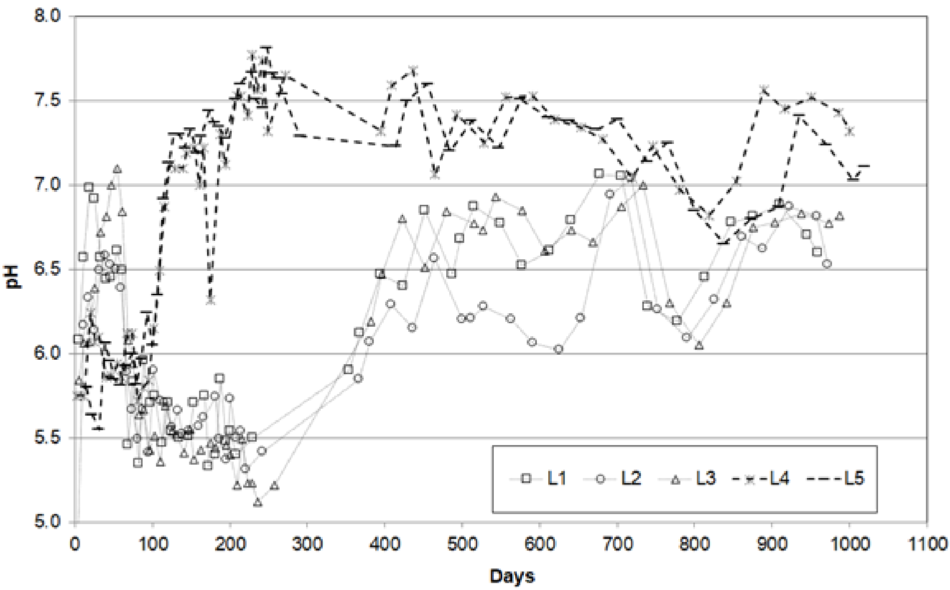
Figure 3 MSW temperature and cumulative monthly rainfall



3.2 Leachate characterisation and MSW biodegradation process

The results of leachate analysed for pH, VFA, alkalinity, EC, BOD, COD, ORP, nitrogen ammonia and total solids are presented in Figures 4, 5, 6, 7, 8, 9, 10, 11 and 12, respectively.

Figure 4 Leachate pH values



3.2.1 Behaviour of the leachate variables from L4 and L5

The behaviour of leachate variables from L4 and L5 during the monitoring period (Figures 4 to 12) showed a strong relation with the MSW biodegradation process. It was possible to identify three paths with specific behaviours of the MSW biodegradation process over time. Although high MSW temperatures between 30°C to 41.2°C (Figure 3) were obtained, the initial aerobic phase was not observed.

Path 1 (0 until 100 days) represented the anaerobic acid phase, based on the degradation of complex organic compounds into simpler organic acids keeping the potential hydrogen under acid conditions (Figure 4). VFA, BOD and COD concentrations (Figures 5, 8 and 9) suffered many variations, however maintaining high values, as well as alkalinity values (Figure 6). EC values were low (Figure 7). ORP values maintained negative values (Figure 10). Nitrogen ammonia values started to rise (Figure 11). Total solids kept high values that gradually decreased (Figure 12). According to Kjeldsen et al. (2002), the concentrations of the leachate variables are generally higher in the acid phase due to the formation of dissolved organic matter and the release of nitrogen ammonia, stemming from the biodegradation of protein and nitrogen compounds into the leachate.

Figure 5 Leachate VFA concentrations

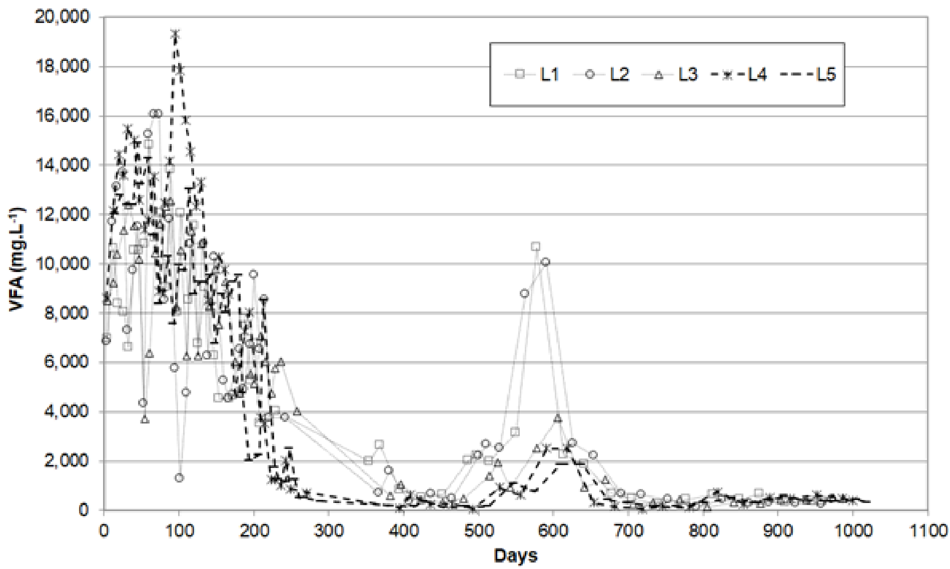


Figure 6 Leachate alkalinity concentrations

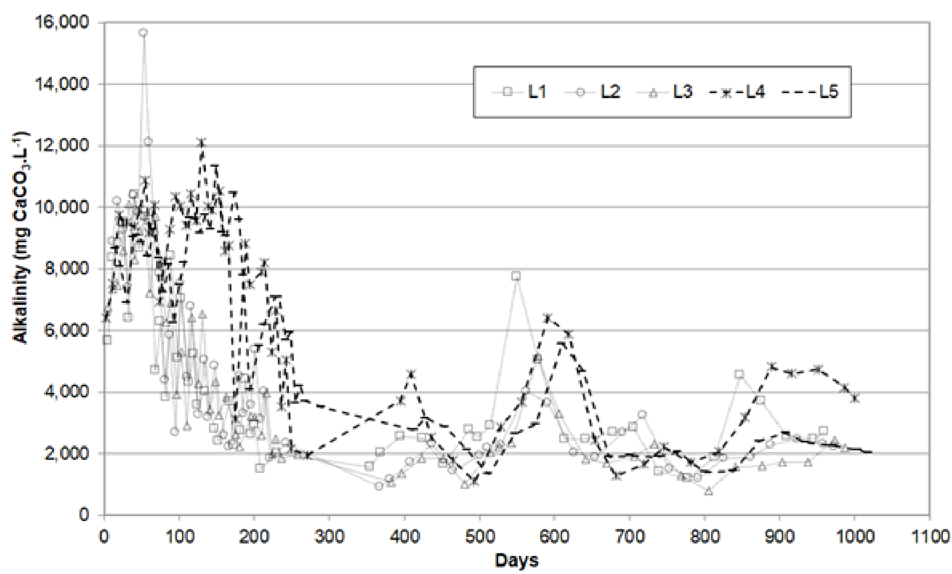


Figure 7 Leachate EC values

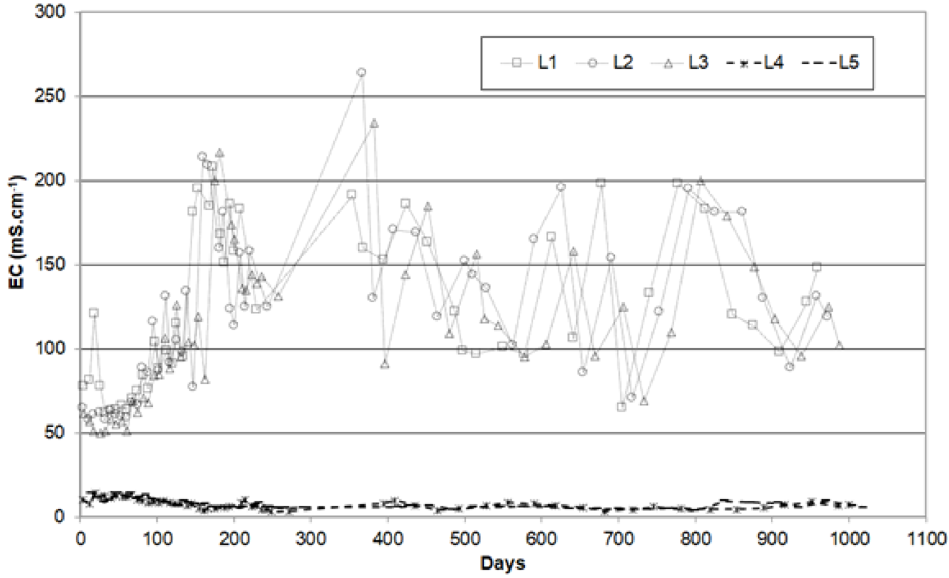


Figure 8 Leachate BOD concentrations

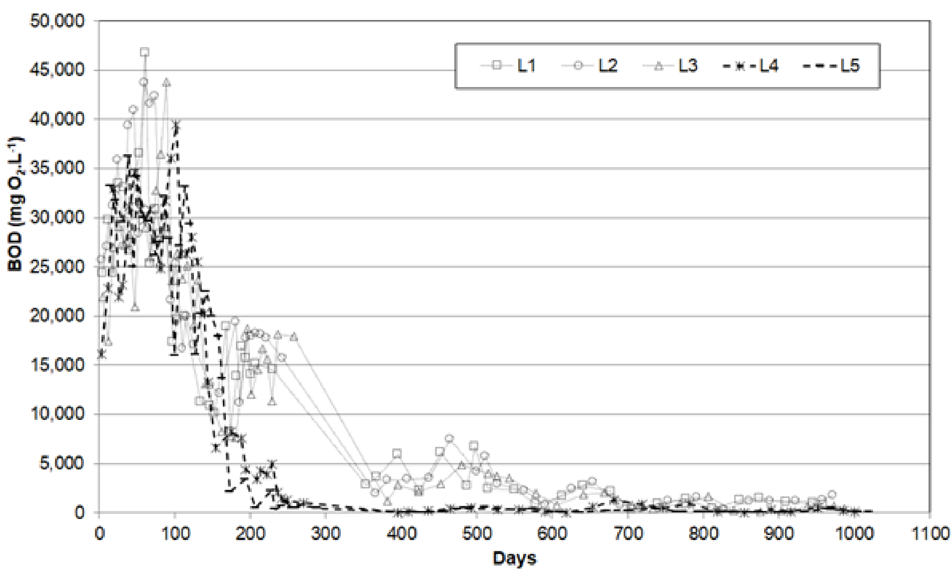


Figure 9 Leachate COD concentrations

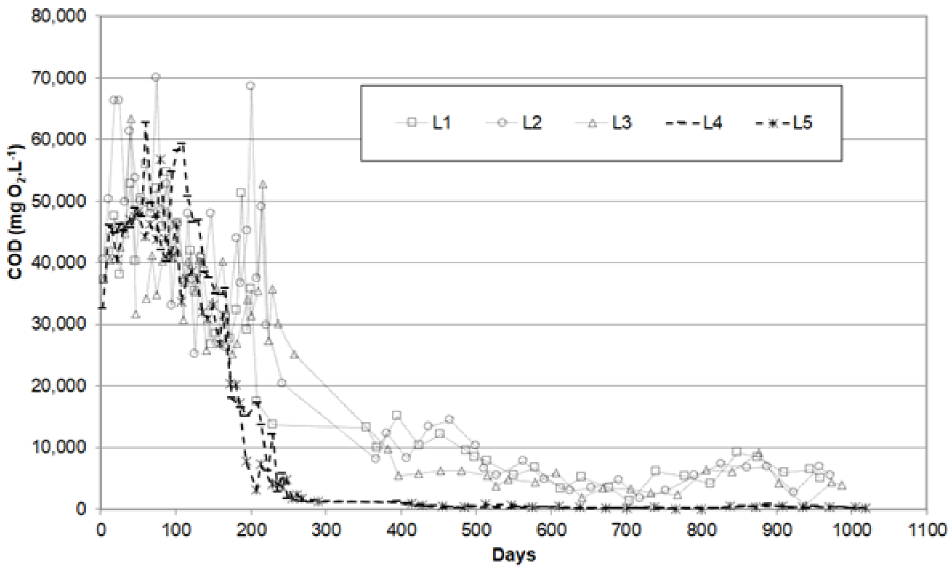
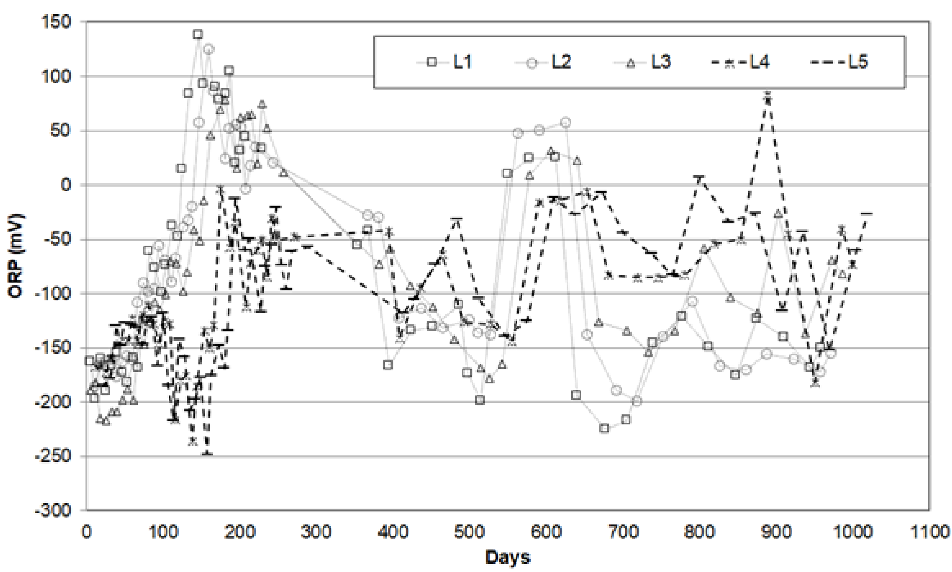
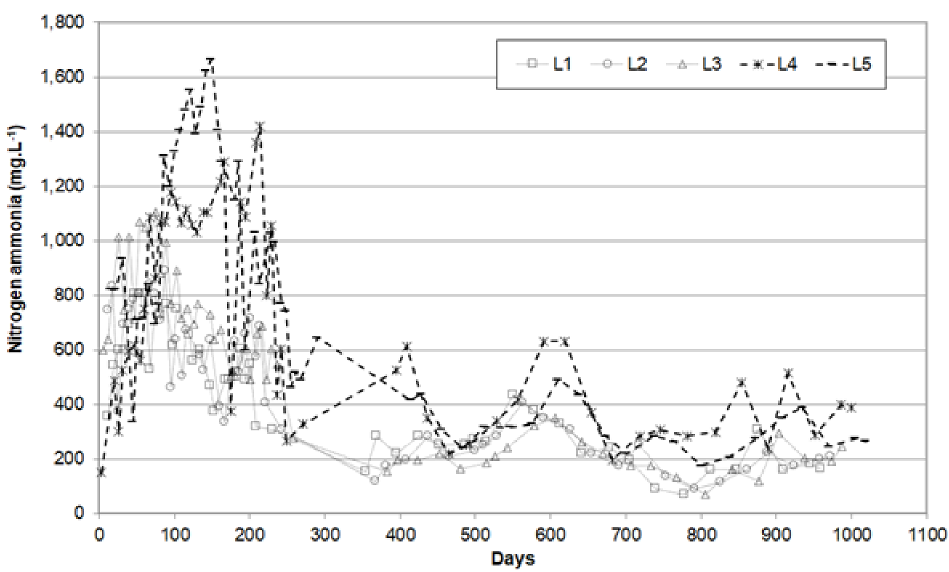


Figure 10 Leachate ORP values



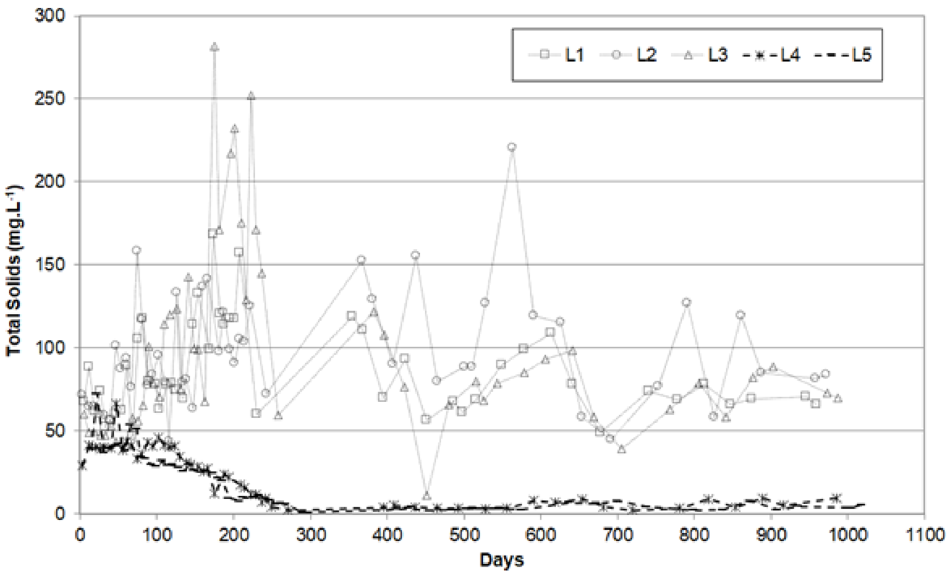
In path 2, between 100 until 260 days, there was strong growth in pH values, which reached the maximum pH value of 7.8 in 248 days (Figure 4). The optimum pH of methanogenic microorganisms is in the range from 6.8 to 7.4 (Barlaz et al., 1990), then from the 100 days the favourable conditions were reached, allowing to define this path as an initial methanogenic phase. In this phase, the content of dissolved organic matter significantly decreases and the composition of organic matter suffers changes. However, nitrogen ammonia does not present the same decreasing trend, because there is no mechanism for its degradation under methanogenic conditions (Robinson, 1995; Burton and Watson-Craik, 1998). VFA, BOD and COD values decreased rapidly (Figures 5, 8 and 9), characterising organic matter consumption. The reduction of VFA concentrations over time is because the acids formed are converted into carbon dioxide, water, hydrogen, acetate and organic compounds (McBean et al., 1995). In addition, there is a decrease in organic matter and consequent reduction of acids production. Starting from 100 days, alkalinity concentrations increased (Figure 6) due to increase in pH values during the same period (Figure 4), which means less alkalinity consumption. After 170 days, alkalinity values decreased up to 260 days (Figure 6). ORP values were negative and decreased from 100 up until 175 days, which then started to increase (Figure 10). According to Qasim and Chiang (1994), a decrease in ORP could indicate the transition from the anaerobic acid phase to the initial methanogenic phase. Nitrogen ammonia values increased and reached a maximum values in 150 days and then, presented a tendency to decrease (Figure 11). EC values also decreased because there was an increase in pH that results in a decrease of acidic ions dissolution (Figure 7). Total solids decreased for values close to 1.0 mg.L^{-1} (Figure 12).

Figure 11 Leachate nitrogen ammonia concentrations



The last path, defined as path 3, presented characteristics of the stable methanogenic phase. This phase started in 260 days and was maintained until the end of monitoring. The pH values oscillated between 6.6 and 7.7 (Figure 4). VFA and alkalinity concentrations (Figures 5 and 6, respectively) tended to stabilise with the exception of data close to 600 days, during which a concentration peak was observed. BOD and COD concentrations also stabilised (Figures 8 and 9, respectively). EC concentrations did vary, but also tended to stabilise with values below 10 mS cm^{-1} (Figure 6). ORP values increased with some peaks presenting positive values (Figure 10). Nitrogen ammonia concentrations decreased but maintained varied values (Figure 11). The concentrations of total solids remained lower, tending to stabilise (Figure 12).

Figure 12 Leachate total solids concentrations



Nitrogen ammonia originates from protein molecules of the organic matter and is indirectly related to the amount of organic matter in disposed waste. Nitrogen ammonia has low biodegradability, which makes its concentration remain high even with a long time in the MSW biodegradation process.

Many researchers reported that the nitrogen ammonia concentration shows a slow increase throughout the MSW biodegradation process, remaining high concentration even in leachate from older landfills low in organic content (Ehrig, 1983; Kruempelbeck and Ehrig, 1999; Kjeldsen et al., 2002; Kulikowska and Klimiuk, 2008; Castrillón et al., 2010; Baawain et al., 2017; Arunbabu et al., 2017). Gupta and Rajamani (2015) reported that high nitrogen ammonia concentration may be due to the process of deamination of amino acids during organic compound decomposition. Because of this, nitrogen ammonia concentration is one of the major long-term pollutants in landfill leachate (Robinson, 1995; Kruempelbeck and Ehrig, 1999; Christensen et al., 1994, 1998).

Nitrogen ammonia is one of those responsible for leachate toxicity to most organisms, along with organic load (Bernard et al., 1997). Nitrogen ammonia at higher concentrations could also damage the photosynthetic system, thus resulting in a reduction in photosynthetic efficiency in plants (Wang et al., 2008).

The three paths that express the behaviours of the MSW biodegradation process over time are presented in Table 3, as well as the values of the leachate variables obtained for the time intervals suggested.

3.2.2 Behaviour of leachate variables from L1, L2 and L3

The behaviour of the leachate variables from L1, L2 and L3 during the monitoring period (Figures 4 to 12) was different from the behaviour of L4 and L5 leachate variables. This fact indicated an influence of the saline solution application on the MSW biodegradation process. Three paths for the specific behaviours of the biodegradation process of MSW confined in L1, L2 and L3 are suggested.

Table 3 Behaviour of the MSW biodegradation process for L4 and L5

	L4 and L5		
	Path 1 – Anaerobic acid phase	*Path 2 – Initial methanogenic phase*	*Path 3 – Stable methanogenic phase*
Leachate variables	0–100 days	100–260 days	260 days–end
pH	5.6–6.2	6.0–7.8	6.8–7.3
VFA (mg.L^{-1})	7,560–19,300	500–17,800	60–2,500
Alkalinity (mg.L^{-1})	6,300–10,400	2,160–12,100	1,100–6,400
EC (mS cm^{-1})	7.8–14.7	3.1–9.6	3.5–9.8
BOD (mg.L^{-1})	16,000–36,000	532–34,900	18–1,270
COD (mg.L^{-1})	32,600–62,700	1,774–59,364	138–1,800
ORP (mV)	–180—120	–250—3	–182–+83
Nitrogen ammonia (mg.L^{-1})	150–1,300	267–1,660	197–630
Total solids (mg.L^{-1})	29–72	5.8–46.2	3.0–9.62

The initial aerobic phase was also not observed, despite the high MSW temperatures between 30°C to 41.2°C (Figure 3) obtained.

Path 1 (0 until 61 days) represented the anaerobic acid phase. In this path, the saline solution application had not been applied in the lysimeters. The behaviours of leachate variables were similar to those of L4 and L5, with the exception of pH, EC values and BOD, COD and total solids concentrations (Figures 4, 7, 8, 9 and 12, respectively), which were higher.

After 61 days, application of saline solution occurred and pH leachate values decreased, being kept within the range of 5.2 to 6.0 up to 360 days (Figure 4). Path 2 was defined by time interval 61 up to 360 days. This behaviour is contrary to the L5 and L4 leachates and is related to the application of saline solution that indirectly causes pH reduction of the leachate, as occurred in studies by Khoury et al. (2000) and Hossain et al. (2014). According to Khoury et al. (2000), low pH values in this case indicate instability between production and consumption of acid by the microorganisms responsible for the MSW anaerobic biodegradation process, resulting in accumulation of acids and delay of the progress to the methanogenic phase. High salt concentrations can be inhibitory to microorganisms of MSW biodegradation processes because it can cause an increase in osmotic pressure, dehydration of bacterial cells, increase in acclimation and lag times (De Baere et al., 1984; Yerkes et al., 1997; Khoury et al., 2000; Al-Kaabi et al., 2009). Alkalinity concentrations suffered strong decrease (Figure 6). Higher alkalinity consumption occurred so that the pH remained stable. In this path, VFA, BOD and COD concentrations (Figures 5, 8 and 9, respectively) followed the decrease seen in the L4 and L5 leachates up to 170 days. Henceforth until 360 days, their concentrations decreased but were higher than those found in leachate from L4 and L5, for the same period. Leachate ORP values had rapid growth, reaching positive values after 125 days (Figure 10). Nitrogen ammonia concentrations decrease gradually, maintaining lower values than those obtained in L4 and L5 leachates (Figure 11). Concentration of total solids grew, reaching high concentration peaks (Figure 12). In general, the behaviour of leachate variables in path 2 did not follow a pattern as it did with MSW confined in L4 and L5. The values of variables indicated that this path stayed on the anaerobic acid phase.

Path 3 (360 days to end monitoring) presented a tendency towards stabilisation of leachate variables. The value of pH increased and remained within the range from 5.8 to 7.1 until the end of the monitoring period. In the course of time, there was a natural reduction of organic matter and consequent decrease of its conversion into acids, resulting in an increase in pH and the possibility of progress to the initial methanogenic phase. The VFA and alkalinity behaviours decreased and tended to stabilise. Some concentration peaks were observed between 500 and 600 days (wet period). BOD and COD concentrations also decreased and tended to stabilise. At the beginning of the path 3, the COD and BOD values were still very high. From 360 days to approximately 500 days, COD values were around 10,000 mg.L^{-1}, indicating recalcitrance. EC concentrations also decreased, varying between 65 to 234 mS cm^{-1}. Leachate ORP values returned to negative. ORP concentrations in the leachate showed positive values only between 500 and 640 days. According to Farquhar and Rovers (1973), ORP increases in periods of excessive rainwater infiltration due to the presence of dissolved oxygen. In the case of lysimeters L1, L2 and L3, the application of saline solution and intense rainfall occurrence contributed to the increase of the amount of oxygen dissolved in the medium, consequently influencing ORP values. Nitrogen ammonia concentrations decreased after 360 days and oscillated between 70 mg.L^{-1} and 435 mg.L^{-1}. These values are lower than those obtained in L4 and L5 leachate. Saline solution application resulted in a decrease in nitrogen ammonia concentrations in the leachate, however, this fact does not eliminate the need for its treatment. Finally, total solids stayed high and variant values, probably due to precipitation from the saline solution, tending towards a slow stabilisation (Figure 12).

In Table 4, time intervals are suggested for the three paths for each specific behaviours of the MSW biodegradation process confined in L1, L2 and L3. Moreover, the values of the leachate variables obtained for these intervals are also presented.

3.3 Influence of saline solution on MSW biodegradation

The anaerobic acid phase was observed for all MSW confined in lysimeters (L1, L2, L3, L4 and L5). For L4 and L5, without saline solution application, this phase occurred between 0 and 100 days, however, for L1, L2 and L3, with saline solution application, this phase was observed for a longer period, between 0 and 360 days.

Saline solution application started in 61 days. Until 61 days, the behaviours of leachate variables were similar for all lysimeters. Nonetheless, starting from 61 days, when the saline solution started to be applied to L1, L2 and L3, the leachate variables presented behaviours different from L4 and L5 leachate variables. The pH and EC variables suffered greater influence from the saline solution. ORP values were higher between 61 and 360 days, whereas the values of alkalinity, BOD and nitrogen ammonia were lower. VFA and COD concentrations were higher especially between 200 and 360 days.

Table 4 Behaviour of the MSW biodegradation process for L1, L2 and L3

Leachate variables	*L1, L2 and L3*		
	Path 1 – Anaerobic acid phase	*Path 2 – Anaerobic acid phase*	*Path 3 – Initial methanogenic phase*
	0–61 days	*61–360 days*	*360 days–end*
pH	5.8–7.1	5.1–6.1	5.9–7.1
VFA (mg.L^{-1})	3,700–15,240	721–16,050	125–10,700
Alkalinity (mg.L^{-1})	5,600–15,700	940–9,700	780–7,700
EC (mS cm^{-1})	51–121	62–264	65–234
BOD (mg.L^{-1})	17,470–46,700	1,900–43,800	63–7,400
COD (mg.L^{-1})	31,700–70,000	8,100–68,600	680–15,050
ORP (mV)	–209—161	–150—30	+10—225
Nitrogen ammonia (mg.L^{-1})	356–1,070	120–1100	70–435
Total solids (mg.L^{-1})	42–101	59.7–281.5	12.5–221

For lysimeters without saline solution application (L4 and L5), the initial methanogenic phase was reached after 100 days, but for lysimeters with saline solution application, it was reached after 360 days. The stable methanogenic phase was reached after 260 days for L4 and L5. However, this phase was not observed for L1, L2 and L3.

It is worth noting that all lysimeters were subjected to typical rainfall of tropical regions, characterised by wet periods of high rainfall and dry periods of low air humidity. According to Rafizul and Alamgir (2012), the tropical seasonal variation influenced the degradation of MSW confined in lysimeters, indicating that the highest degradation occurred during the rainy season and the lowest during the dry season.

Moreover, L1, L2 and L3 were subjected to application of 120 litres of saline solution per 15 weeks, since 61 days up to the end of monitoring. In total, there were 7,560 litres of saline solution, in 63 days for each of the lysimeters. Therefore, for these lysimeters, the anaerobic acid phase comprised between 61 and 360 days was characterised by intensive leaching caused by high rainfly and application of saline solution. As a result, many solids presented in MSW were removed by leachate, not necessarily degraded. Concentrations of total solids showed higher values than those obtained for the L4 and L5 leachate (Figure 12), corroborating this fact.

Considering the optimum range of pH of methanogenic microorganisms being from 6.8 to 7.4 according to Barlaz et al. (1990), then up to 450 days there were no favourable conditions for methanogenic microorganisms' growth in the lysimeters L1, L2, and L3. This fact is probably related to high salt concentrations, which can be inhibitory to biodegradation microorganisms, delaying progress to the methanogenic phase.

The initial methanogenic phase for the MSW confined in L4 and L5 (without saline solution application) started after about 100 days of leachate monitoring. For the MSW confined in L1, L2 and L3 (with saline solution application), the initial methanogenic phase started after about 450 days of leachate monitoring. Therefore, a delay of 350 days (about 11 months and 20 days) was caused by saline solution application.

Ogata et al. (2016) showed that the nitrogen ammonia concentration in the leachate is a key inhibitor of anaerobic degradation waste in landfills with leachate recirculation, recommending a reduction in this concentration for the efficiency of this anaerobic waste degradation, and, consequently, the improvement of stabilisation of landfills with leachate recirculation. As demonstrated in this study, the application of saline solution with concentrations around 200 $g.L^{-1}$ resulted in a decrease of the nitrogen ammonia concentrations in the leachate generated by the MSW confined in the lysimeters (L1, L2 and L3). Therefore, application of saline solution could be analysed as an alternative to decrease nitrogen ammonia concentrations in landfills with leachate recirculation, considering this application after the leachate recirculation process. However, this application increased the values of EC and total solids, and provided the slowest and gradual decrease in COD. All these observations do not rule out the need for treatment of leachate generated by the wastes submitted or not to the application of saline solution.

4 Conclusions

Analyses of leachate generated by MSW confined in lysimeters revealed that the application of saline solution resulted in the accumulation of acids and delaying the progress to the methanogenic phase. This interference is related to an increase in osmotic pressure that induces dehydration of bacterial cells.

The application of saline solution (with concentrations of about 200 $g.L^{-1}$) influenced on the anaerobic biodegradation process of MSW confined in lysimeters, delaying the progress to the methanogenic phase, however not completely inhibiting the biodegradation process. As a result, the MSW anaerobic biodegradation process of lysimeters L1, L2 and L3 (subjected to application of saline solution) remained in the anaerobic acid phase in almost the entire study period. Initial methanogenic phase were reached in lysimeters L1, L2, L3, especially regarding pH and ORP, with a delay of 350 days.

During the monitoring period (1,022 days), a decrease in the concentrations of nitrogen ammonia in the leachate from MSW confined in lysimeters subjected to saline solution application was observed. The nitrogen ammonia concentration is one of the major long-term pollutants and is a major toxicant to organisms in the landfill, therefore, this decrease can be considered a positive result. However, EC and total solids values were high, and COD concentrations decreased more slowly and gradually. Therefore, the treatment of the leachate must also be considerate.

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