

Impacts of aeration and active sludge addition on leachate recirculation bioreactor

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Abstract

Stabilization of municipal solid waste (MSW) is affected by moisture, nutrients, oxygen, pH and accumulation of inhibitory fermentation products, etc. Optimization of these parameters could create a favorable environment that promotes the rapid development of the desired microbial population and acceleration of decomposition of MSW. The objectives of this work was to determine the feasibility of enhancing phase separation through intermittent aeration strategy throughout the treatment process; to demonstrate the potential of active sludge for in situ nitrogen removal; to examine the efficiency and evaluate the possibility of in situ removal of contaminants from leachate. The results indicate that the removal ratio of COD, BOD₅, NH₄⁺ and total nitrogen are over 80, 81, 75, and 74%, respectively, in the leachate recirculation reactors with aeration; the removal efficiency of NH₄⁺ and total nitrogen of the reactor which were added active sludge were 88 and 84%, respectively. Therefore, aeration strategy has positive impacts on the solid waste stabilization; addition of active sludge in reactor is favorable for the remediation of the nitrogen; using landfill itself for in situ attenuating the contaminants from leachate is feasible.

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1. Introduction

The cities especially metropolitan cities have significant problems associated with management of municipal solid waste (MSW) in China. The amount of solid waste generation is increased by 9% in recent years. The MSW generated is less than 1 kg per capita per day, lower than in many developed countries. China has large population and the total amount of MSW is about 230 million tones annually [1]. Most of landfills are co-disposed with industrial wastes, incinerator ashes, mine wastes and household hazardous substances such as batteries, paints, dyes, inks, etc. Therefore, leachate is rich in various contaminants such as organics, inorganics, heavy metals, etc. The theoretical amount of leachate produced from the landfill is 2 m³ ha⁻¹ per day for the first 5 years after encapsulation, and 5 m³ ha⁻¹ per day for the succeeding years [2]. This means that large amount of leachate is generated and if not managed well can pose serious threat to the environment, especially to the groundwater. Therefore, landfills

should be operated in a manner which minimizes environmental pollution.

Concentrations of leachate contaminants such as COD, BOD₅, heavy metals, ammonium, etc. may accumulate to much higher levels which would create a leachate discharge problem [3–5]. Removal of pollutants from leachate is currently practiced ex situ. However, ex situ treatment can be difficult and costly. Therefore, the development of an in situ removal technique would be an attractive alternative, potentially yielding both economic and environmental advantages. Studies have shown that leachate recirculation produces stabilized leachates containing relatively low concentrations of degradable carbon compounds but high concentrations of ammonium due to the hydrolysis and fermentation of the nitrogenous fractions of biodegradable substrates. The ammonium nitrogen concentration can reach as high as 5000 mg L⁻¹ and such a high level of ammonium can create numerous environment problems [3,6]. A landfill with leachate recirculation may be regarded as MSW bioreactor treatment system. The landfill environment is a complex heterogeneous system in which different types of microorganisms coexist. These microbial populations are capable of a variety of reactions depending upon the prevailing environmental conditions and the

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organisms-substrate specificity [7]. In the reactors, leachate is treated by nitrification, denitrification, anaerobic fermentation and methanogenesis [8]. Recent study has shown that 99.5% ammonia in leachate was removed, coupled with elevated concentrations of nitrate and nitrite [32]; the removal specific rates was 0.196 and 0.117 mg N/day-g dry waste and half-saturation constants of 59.6 and 147 mg N/L for acclimated and unacclimated wastes, respectively [33]. It has Leachate recirculation enhances and accelerates conversion and stabilization of the MSW by increasing the uniformity of moisture, substrate and nutrient distribution and by creating an environment that promote the rapid development of the desired microbial population [3]. Thus, the landfill gas (LFG) generation rates are enhanced while the environmental impacts are reduced by containing the leachate and controlling the LFG emissions. Recent study has shown that leachate recirculating could reduce the organic concentration considerably, with a maximum reduction rate of COD over 95%; and, using a semi-aerobic process [34]. The volume of leachate is reduced by maximizing evaporative losses during recirculation which can reduce leachate treatment capital and operation cost, and thus the overall contaminating life span of the landfill is reduced [9,26]. Because of these advantages, more people paying attention to this technology and therefore it is bound to progress to advanced leachate recirculation concepts based on chemical and biological mechanisms of landfill ecology. For example, methanogenic microflora established anaerobic separate attached-film reactor can be used to treat leachate with high concentrations of volatile organic acids; the sequential linking of different aged leach bed cells takes advantage of adapted microflora and high alkalinity of leachate in old cells to treat leachate and inoculate new cells.

Although a landfill may have most advanced leachate management practices, ensiling process result in an imbalance between acidogenesis and methanogenesis in the landfill leachate [10]. Some reports have shown that methane-production remains inhibited for over 1 year, even in a landfill moistened by leachate recycle [11]. Recirculating leachate within the same landfill cell will not correct imbalance between acidogenesis and methanogenesis. Anaerobic treatment of landfill leachate may involve two-phase decomposition of organic matter, namely acid formation phase and methane fermentation phase. Experiments based on this concept typically result in rapid inoculation of the acid formation phase and its subsequent conversion to a methane-producing combined-phase system, which can eliminate the desired-phase separation and associated benefits [11]. Two-phase treatment is reportedly more rapid and more stable than combined-phase treatment. In theory, enhancement of phase separation could be attempted by pH control, sequencing of batch reactors, and aeration of the effluent, etc. With the maturity of aeration technology and relatively high diffusivity of gases, aeration of the landfill cell seems to be the most practical option for achieving phase separation in leachate recycling reactors and landfill cells [11,27]. Microorganisms in the acid formation phase and methane fermentation phase have different growth rates and optima for the environmental and nutrient; aeration of the landfill was proposed as a practical option to optimize these conditions in order to

improve performance and enhance solid waste stabilization rate in the first few years. Moreover, the possibility of achieving separation by aeration was based on the fact that methanogenic bacteria are generally believed to be more sensitive to oxygen and other inhibitors than other bacteria involved in the methane fermentation.

In addition, the leachate contains very high concentration of the nitrogen; therefore some consideration has also been given to in situ removal of nitrogen. The objectives of this research are to:

- (1) Determine the feasibility of enhancing phase separation through intermittent aeration strategy throughout the treatment process.
- (2) Evaluate the efficiency of adding active sludge for solid waste stabilization enhancement.
- (3) Examine the efficiency and demonstrate the possibility of in situ removal of contaminants from leachate.

2. Materials and methods

2.1. Sources of wastes and leachate

Samples of MSW and leachate were obtained from Paijia Landfill of Changchun, China. Their physical and chemical properties are shown in detail in Tables 1 and 2. All MSW samples were shredded to less than 1 cm prior to physicochemical analysis and the column study.

2.2. Experimental setup

Three landfill reactors simulators were constructed in the laboratory using three Plexiglas columns of length 60 cm and a diameter of 17 cm. The test system is shown in Fig. 1. The composition and operation of reactors are given in Table 3. The reactors were kept at a constant room temperature (24 °C) to enhance the growth of microorganisms. Simulated landfill reactors were filled with 7.5 kg of shredded and compacted synthetic MSW, in order to enhance the removal of the nitrogen, active sludge added in reactor C.

2.3. Experiment operation

Leachate recirculation frequency was three times per week for each reactor; reactors B and C were aerated intermittently, and aeration flow rate was approximately of 0.5 L/min. The

Table 1
Physical and chemical properties of the waste

Components	Value	Components	Value
Moisture content (%)	29.1	Lead (mg kg ⁻¹)	44.31
Volatile solid (%)	32.5	Cadmium (mg kg ⁻¹)	0.42
Fixed solid (%)	67.5	Cuprum (mg kg ⁻¹)	36.24
Total nitrogen (mg g ⁻¹)	3.58	Zinc (mg kg ⁻¹)	24.50
Total phosphorus (mg g ⁻¹)	8.78	Nickel (mg kg ⁻¹)	20.85
Chromium (mg kg ⁻¹)	36.24		

Table 2
Physical and chemical properties of the leachate

Components	Concentration (mg L ⁻¹)	Components	Concentration (mg L ⁻¹)
pH	6.72	Cd	0.011
COD	2258.6	Cr	0.0725
BOD ₅	767.9	Co	0.175
Ammonium	246.75	Ni	0.258
Nitrate	61.35	Ba	0.812
Total nitrogen	572.67	Pb	0.103
Hardness	1865.3	Cu	15.34
Conductivity (ms m ⁻¹)	105.8	Mn	8.03
Chloride	1559.3	Sr	4.51
Sulphate	1658.3	Zn	9.34
Fe	2.65	Al	2.03

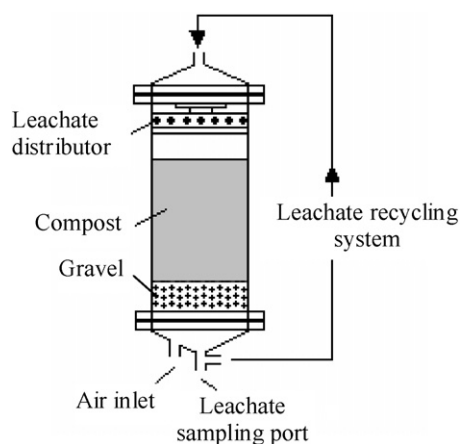


Fig. 1. Configuration and operational features of the reactor.

0.75 kg active sludge was added in reactor C. The maximum volume of the recirculated leachate did not exceed 1.8 L in order to avoid flooding of the reactor. Throughout the experiment study, water was added to the reactors at a constant rate of 600 mL per week in order to simulate the 20 cm annual infiltration.

2.4. Analytical methods

Leachate samples were collected at the bottom of the reactors and analyzed for chemical oxygen demand (COD), biological oxygen demand (BOD₅), pH, conductivity, chloride and the selected heavy metals (Fe, Cd, Ni, Zn, Cu, etc.). All these analyses were performed in accordance with standard methods for the examination of water and wastewaters [12]. Conductivity measured by conductometer (DDS-11A); pH measured by microprocessor pH meter, 211; Heavy metals were analyzed using Perkin-Elmer A Analyst 300 atomic absorption spec-

Table 3
Composition and operation of the reactors

Reactors	Weight (kg)	Height (cm)	Density (g cm ⁻³)	Active sludge	Operational ways
Reactor A	7.5	30	1.1	None	Leachate recycling
Reactor B	7.5	30	1.1	None	Leachate recycling and intermittent aeration
Reactor C	7.5	30	1.1	Yes	Leachate recycling and intermittent aeration

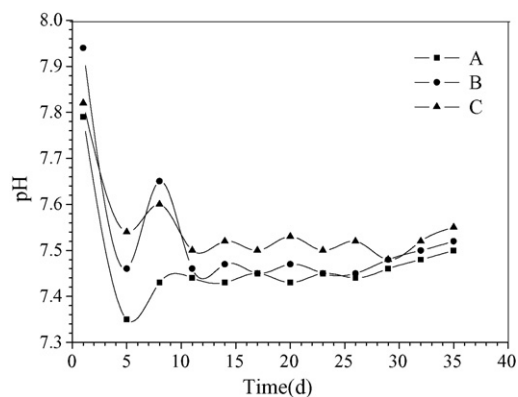


Fig. 2. Changes of pH over time.

trophotometer. Prior to the analysis, each sample was digested with concentrated HNO₃ and HCl (1:1) according to the ASTM (3010) Standard Method.

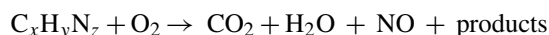
3. Results and discussion

3.1. pH

Fig. 2 shows the variation of pH over time. The first 5 days of the test represents the initial adjustment phase during which the pH levels of the reactors A, B and C dropped approximately from 7.79, 7.94 and 7.82 to 7.35, 7.46 and 7.54, respectively. Depression of pH during startup indicated that recirculation was insufficient to remove acids as fast as their production during the initial phase. During the second stage (from 5 to 11 days), catabolism of fermentation products might have occurred and resulted in fluctuation of pH value. In the subsequent days, pH remained constant. In addition, aeration can drive out CO₂ from the reactor pore spaces thus preventing pH decrease; active sludge addition can accelerate the degradation of MSW, which resulted in pH increase. This explains why pH values of reactors B and C were higher than that of reactor A.

3.2. Organics

The basic unbalanced equation for aerobic decomposition for complex organic compounds of the form C_xH_yN_z is:



The whole process of biodegradation can be divided into three basic phases. The first phase is called the phase of intensive aerobic biodegradation with exponential growth of biomass. It is characterized by the fast decay of organics in the solid phase (COD_s) for initial growth, and then decay of organics in liquid

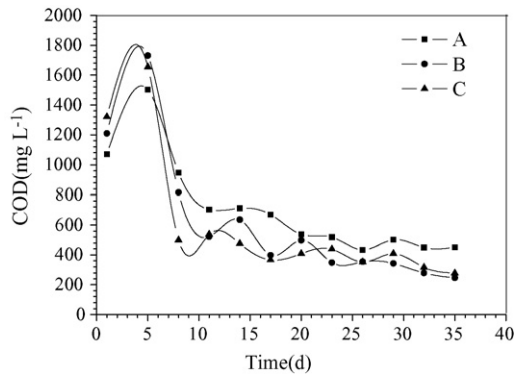


Fig. 3. Changes of COD concentrations over time.

phase (COD_L) and exponential growth of biomass. In the second phase, the limited aerobic biodegradation can be observed. The level of COD_S remains either constant or a slight decrease, the decay of COD_L continues, biomass approaches the stationary phase or slightly increases. Finally, the termination phase begins when low COD_L decrease is observed and biomass concentration remains constant or begins to die. COD concentration in the solid and liquid phase and biomass concentration are balanced, which can be presented in the form of ordinary differential equations [13]:

$$\frac{d([COD_S])}{dt} = -r_H \quad (3.1)$$

$$\frac{d([COD_L])}{dt} = r_H - r_D \quad (3.2)$$

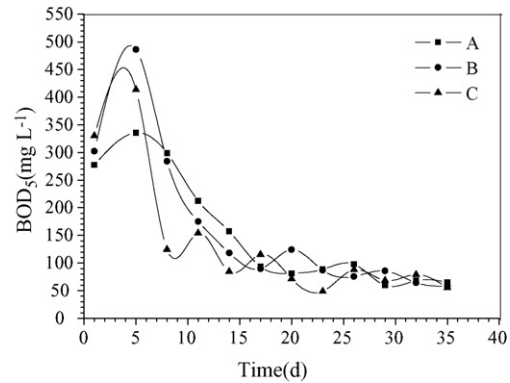
$$\frac{dX}{dt} = r_X \quad (3.3)$$

where r_X is the rate of biomass growth ($g L^{-1} h^{-1}$), r_H the rate of hydrolysis reaction ($g L^{-1} h^{-1}$) and r_D is the rate of biodegradation ($g L^{-1} h^{-1}$).

The results of the COD concentrations over time are shown in Fig. 3. The figure exhibits accumulation of COD during the first 5 days, which indicate that recirculation was also insufficient in removing COD in pace with its production during the initial stage. The COD values of the reactors A, B and C increased from 1070, 1209 and 1321 to 1502, 1731 and 1654 $mg L^{-1}$, respectively, in the initial adjustment phase. However, subsequent reduction of COD suggests that biological catabolism occurred, which in turn resulted in the increase in pH values for the three reactors until the 11th day, followed by a decrease to 703, 522 and 538 $mg L^{-1}$, respectively. The COD concentration stabilized in the 11th day on the whole. By the 35th day the COD concentrations had reached low values of 450, 246 and 278 $mg L^{-1}$ for reactors A, B and C, respectively. Table 4 shows that removal efficiency of the reactor B is 13.78% higher than for A, while reactor C is 5.11% higher than for B averagely. After 14 days, the COD values of all the reactors stabilized and their concentrations remained constant. Therefore, intermittent aeration can optimize microorganisms growth environment by adjusting pH, and have positive effect on the balance growth of acid-production phase and methane-

Table 4
Removal efficiencies of the reactors

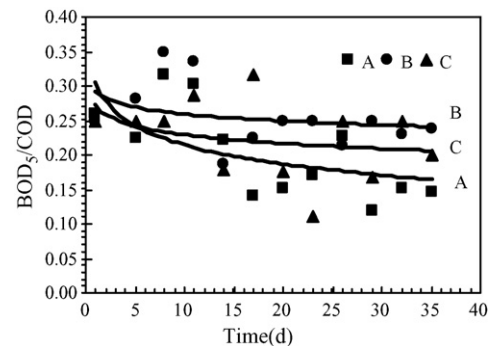
Reactors	Maximum removal efficiency (%)	Minimum removal efficiency (%)	Average removal efficiency (%)
Reactor A	71.23	36.89	60.59
Reactor B	85.78	52.84	74.37
Reactor C	88.11	67.49	79.48

Fig. 4. Changes of BOD₅ concentrations over time.

production phase, and favorable to the realization of the two-phase separation.

The BOD₅ results are displayed in Fig. 4. The BOD₅ results show similar trend with that of COD. The BOD₅ concentration for reactors A, B and C increase from 277, 302 and 330 $mg L^{-1}$ to 335, 486 and 413 $mg L^{-1}$, respectively, during the first few days, and then decrease down to 157, 118 and 84 $mg L^{-1}$ after about 14 days of degradation. All BOD₅ concentrations dropped to less than 60 $mg L^{-1}$ in the end.

The BOD₅/COD ratio indicates the amount of biodegradable compounds in the leachate. As the organic biodegradation occurs, the BOD₅/COD ratio decreases. Fig. 5 shows that BOD₅/COD ratios of the reactor B and C are higher than that of reactor A, which means that aeration can enhance the release of dissolved organics into the liquid. The BOD₅/COD ratio of reactor C is lower than for B; this may be as a result of biodegradation by microorganisms introduced by the addition of active sludge. The average BOD₅/COD ratio for reactors A, B, and C are 0.24, 0.31 and 0.27, respectively.

Fig. 5. Changes of BOD₅/COD over time.

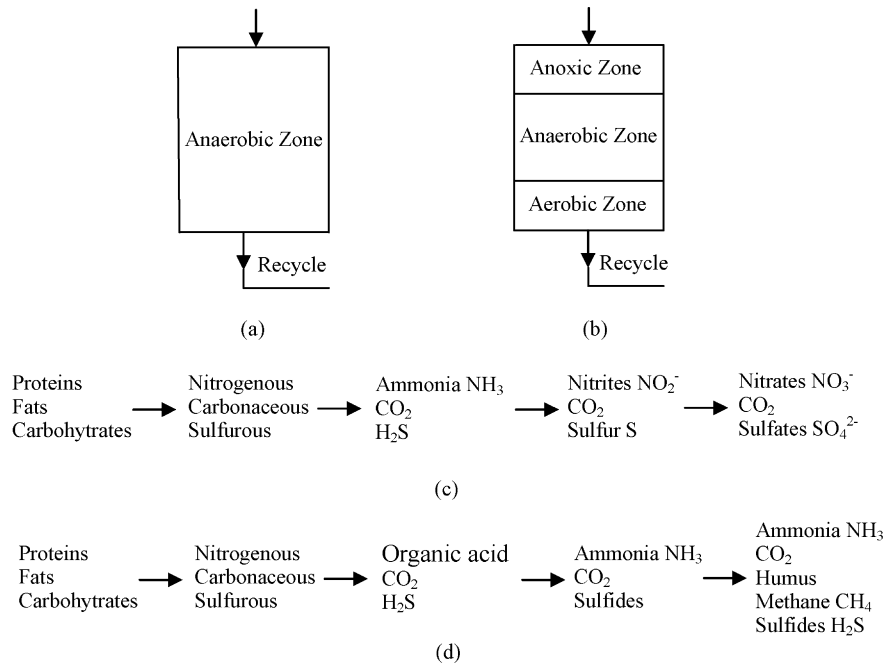


Fig. 6. Conceptual landfill configuration for nitrification and denitrification in reactors: (a) reactor A; (b) reactor B and C; (c) biochemical reactions occurring in aerobic zone and (d) biochemical reactions occurring in anaerobic zone.

3.3. Nitrogen

Biological nitrification/denitrification is commonly used for nitrogen removal in wastewater treatment technology. Organic nitrogen and ammonium are oxidized through a series of compounds ending in nitrate under aerobic condition. The reactors were developed to include anoxic, anaerobic and aerobic zones as illustrated in Fig. 6(a and b). Biochemical reactions occurring in aerobic and anaerobic zone are shown in Fig. 6(c and d); in anoxic zone, ammonia in the leachate is converted into nitrate by oxygen carried by recycled leachate, then followed by a denitrification process. These modifications to the usual design and operation of the landfill were intentionally incorporated for in situ attenuation of high nitrogen concentrations, especially ammonium, in leachate of the final maturation phase of landfill stabilization. Leachate recirculation was used to transport stabilization products from one landfill layer to the next and thereby enhance conversion on a continuum. The sources of carbon and nitrate necessary for denitrification could be supplied by utilizing leachate recycling to carry the residual C and N from the anaerobic zone into the aerobic zone, and subsequently to the anoxic zone at the top of the system. Accordingly, leachate nitrogen could be removed concomitantly with other attenuation, e.g., sulfur compounds [14,25]. Some studies have shown that the efficiency of nitrogen conversion is dependent on the operational stages. Both separate and combined-phase operation with internal leachate recycle around each reactor provided 95% nitrogen conversion; In contrast, combined-phase operation without internal recycle provided a conversion efficiency per cycle ranging between 30–52% for nitrification and 16–25% for denitrification [3,25].

3.3.1. Ammonium

The results for ammonium are displayed in Fig. 7. Ammonium concentration of reactor A did not decrease significantly, but fluctuated between 200 and 414 mg L⁻¹ throughout experiment. This illustrates that leachate recycling alone would have little impact on the ammonium removal in the landfill. The ammonium concentration in reactors B and C started decreasing from 318 and 324 to 80 and 40 mg L⁻¹, respectively, after the 35 days. The removal ratio of the ammonium reached 75 and 88% for reactors B and C, respectively. These results suggested that intermittent aeration is an effective way to remove ammonium; however, addition of active sludge showed a slight positive influence on ammonium removal.

In the initial stage, landfill leachate might contain too much refractory organics or heavy metals that might inhibit nitrification, which explain the slow change of ammonium concentration

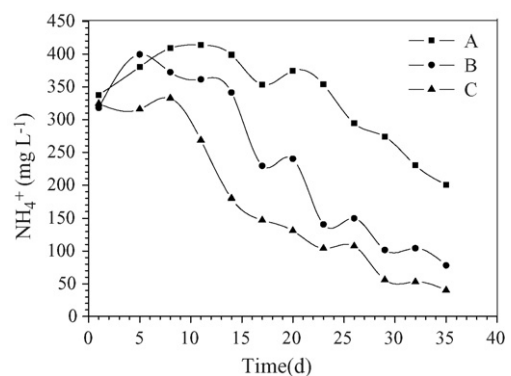


Fig. 7. Changes of ammonium concentrations over time.

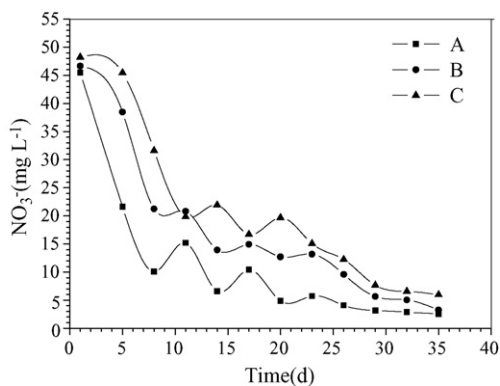


Fig. 8. Changes of nitrate concentrations over time.

at the initial stages in Fig. 7. After some time, the degradable COD in leachate was stabilized or removed, and nitrification occurred under the aeration condition where ammonium-N was oxidized to nitrate-N; and thus ammonium concentration started to decrease. Fig. 3 shows that COD of reactors B and C stabilized after 11 and 8 days, and ammonium concentration decreased after 14 and 11 days, respectively. In addition, Fig. 7 shows that reactor C, which had active sludge, was more effective in removal of ammonium compared to reactor B without sludge. Part of ammonium-N is utilized by the heterotrophs for biomass synthesis and the remainder is oxidized to nitrate-N by nitrifying bacteria [15].

Another nitrogen removal method is by Anoxic Ammonium Oxidation (Anammox). Anammox is an anoxic microbiological process in which ammonium, together with nitrite, is converted to nitrogen gas [16]. It has been shown that under oxygen-limited conditions, a considerable part of the nitrogen load was eliminated at the leachate treatment plant [17]. Seventy percent of nitrogen loss was also observed in a nitrifying rotating biological contactor treating ammonium-rich leachate from a hazardous waste landfill [18]. This process might have also occurred in reactors A, B and especially C.

3.3.2. Nitrate

Fig. 8 shows the changes in nitrate concentration for reactors A–C. The results suggest that they follow similar trend. The concentration of nitrate sharply declined in the first 8 days, and then stabilized over the subsequent period. The nitrate concentration of reactor A–C decreased from the initial values of 45.5, 46.7

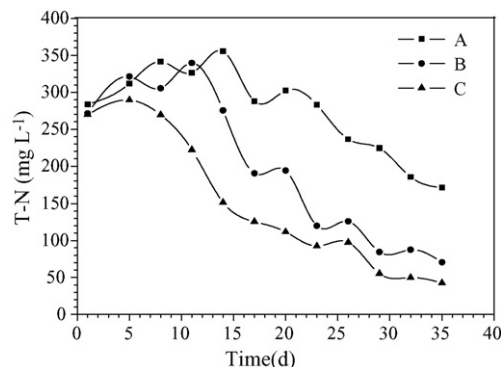


Fig. 9. Changes of total nitrogen concentrations over time.

Table 5

The concentrations of total nitrogen and removal efficiency

Reactors	A	B	C
Initial concentration (mg L^{-1})	283.6	271.3	269.8
Peak concentration (mg L^{-1})	355.2	339.6	289.6
Final concentration (mg L^{-1})	171.0	70.6	42.7
Removal efficiency (%)	52	79	85

and 48.3 mg L^{-1} down to below 2.54, 3.28, and 5.98 mg L^{-1} , respectively. In general, reactor A was more effective than both B and C in nitrate removal, which may be because it was mainly under the anaerobic conditions and denitrification might have occurred. On the contrary, reactor B and C nitrification may be the prevailing reaction due to aeration.

3.3.3. Total nitrogen

Fig. 9 shows the results of the leachate total nitrogen (TN) concentration changes. The change in total nitrogen for the three reactors followed similar trend. The concentration of total nitrogen peaked up after a certain period then decreased. It took 14, 11 and 5 days for reactor A, B and C to reach peak concentrations of 355.22, 339.64, and 289.56 mg L^{-1} with final removal ratio was 52, 79 and 85%, respectively. Reactor C took shortest time to reach peak concentration compared to both A and B and was the most effective in total nitrogen removal followed by B. Table 5 shows the different stages of total nitrogen concentration and removal ratio in detail. All these results indicated that an aerated landfill with leachate recirculation would remove total

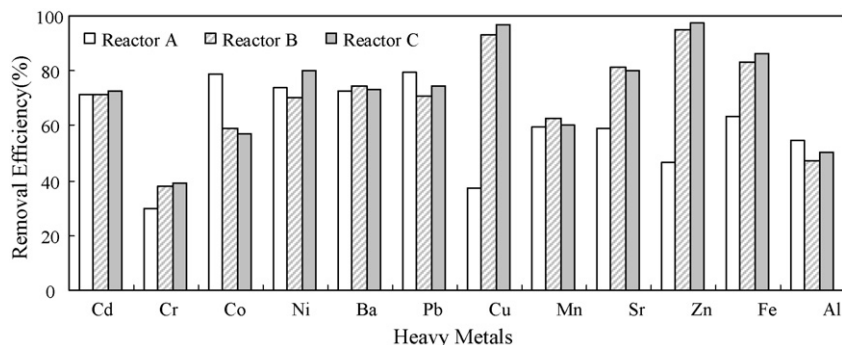


Fig. 10. Changes of removal efficiency of heavy metals.

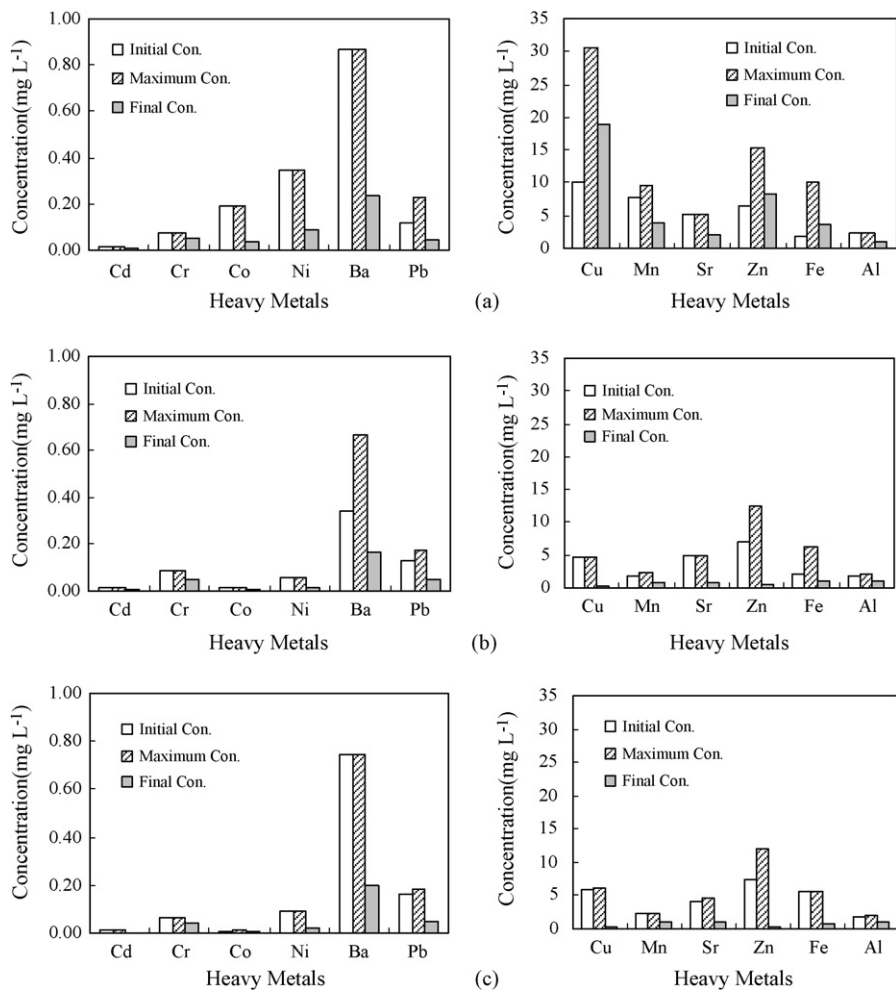


Fig. 11. Changes of samples heavy metals concentrations of (a) reactor A; (b) reactor B and (c) reactor C.

nitrogen effectively and addition of active sludge has positive impact on contaminants removal.

3.4. Heavy metals

The major sources of heavy metals in landfills are the co-disposed industrial wastes, incinerator ashes, mine wastes and household hazardous substances such as batteries, paints, dyes and inks, etc. They are rich in iron, cadmium, copper, zinc, lead, and nickel, etc. From Fig. 10 it can be noted that the three reactors have high removal efficiencies for most heavy metals especially for Cd, Ni, Ba, Cu, Zn and Fe, except Cr. Heavy metal removal ratio of reactor A, B and C was 29.6–79.2%, 37.9–94.9% and 39.2–96.7%, respectively. Most heavy metal removal ratios of reactor B and C were more than 70%. Therefore, Reactor B and C were more effective in heavy metal removal than that of reactor A. This observation might be attributed to effect of pH among other factors as it can be seen from Fig. 2 that pH was above 7.3, which may cause heavy metals to be retained by processes such as sorption, carbonate precipitation, and hydroxide precipitation, etc.

From Fig. 11(a–c), it also can be noted that reactor B and C are more effective in removal of heavy metals compared to

reactor A, which means that aeration does not always cause increase in heavy metal concentration; reactor B and C have a similar trend, which suggest that addition of active sludge has a slight impact on the heavy metals removal in the solid waste disposal. However, several studies have shown that aerobic conditions in landfill could result in increase of the heavy metal solubility in leachate because of the conversion of the heavy metals species [9,22]. Therefore, heavy metal concentrations in leachate not always increase under aerobic condition but other factors and processes may come into play. In this paper it was found that leachate recirculation and aeration have positive impacts on attenuation of most heavy metals. pH increase can enhance hydroxide and carbonate metal precipitation. Previous studies indicated that in leachate recirculation landfills, the primary metal removal mechanisms appear to be sulphide and hydroxide precipitation [20,21]. By establishment of highly reducing environment, about 90% of all heavy metals were attenuated within the first 10 days due to the sulphate reduction and subsequent heavy metal sulfide precipitation [23]. Leachate recirculation stimulated reducing conditions for the reduction of sulphate to sulphide, which moderated leachate metals to very low concentrations [24]. Hydroxide precipitates could form at neutral or above neutral. Therefore, the primary

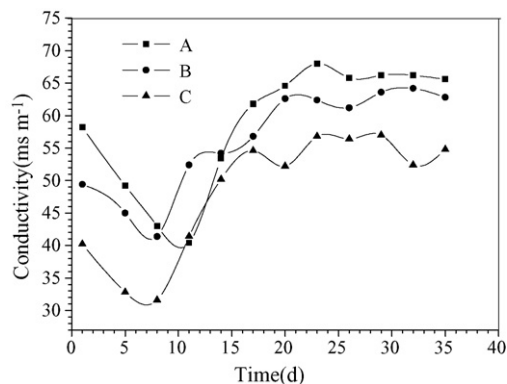


Fig. 12. Changes of conductivity over time.

metal removal mechanisms appear to be precipitation in this work.

Complexation, metal ions combine with non-metallic ligands by covalent bonds, is another process affecting the heavy metals concentration. A significant fraction of the heavy metals is found in colloidal matter and as organic complexes. The colloidal matter to a large extent consists of organic material, although also sulphides, carbonates, and other inorganic constituents have been determined in the colloidal fraction. The organic association is predominant for Cd, Cu, and Pb, but even significant for Ni [19,28–31]. High molecular weight humic-like substances are formed from waste organic matter tends to form strong complexes with heavy metals. In some instances, a remobilization of precipitated metals resulted from such complexation once the organic content has stabilized [9,19,22]. Inorganic complexes with carbonate, chloride, and hydroxide (Cr only) are also significant [28,30].

Besides precipitation and complexation process, sorption, redox condition, etc. also can affect the heavy metal concentrations in leachate.

3.5. Conductivity

The conductivity of leachate reflects its total concentration of ionic solutes and is a measure of the solution's ability to convey an electric current. Fig. 12 shows that the change in conductivity of leachate in the reactors over time followed a similar trend. The conductivity of reactor A decreased from the initial value of 58.2 to 40.4 ms m^{-1} after 11 days while for reactors B and C it decreased from 46.66 and 48.26 to 41.40 and 31.6 ms m^{-1} after 8 days, respectively. The decrease in conductivity was due to the washout of some easily mobilized ions such as metals, chloride and sulfate, combined with other factors such as the conversion of sulfate to sulfide under increasingly reducing conditions, and the subsequent precipitation of sulfide as heavy metal-sulfides which would tend to withdraw significant ionic strength from solution. When solid waste stabilized, easily mobilizable ions were washed out continuously and surplus oxygen gradually created oxidized conditions which resulted in partial dissolution of the precipitates in leachate and thus resulting in the increase in conductivity after the decrease in the initial phase. From Fig. 12 it can be also be seen that reactor C is the most effective, followed by reactor B.

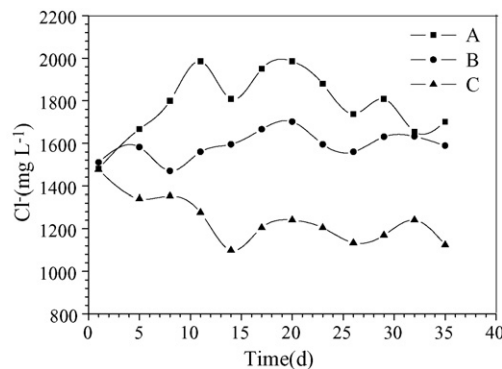


Fig. 13. Changes of chloride concentrations.

3.6. Chloride

Fig. 13 shows the changes of chloride concentration in the three reactors. The chloride concentration for reactor C was lower compared to A and B. The average chloride concentrations for reactors A, B and C were 1787.82, 1590.91 and 1238.85 mg L^{-1} , respectively. The results indicate that the chloride concentration was approximately constant, the phase of the solid waste degradation have slight influence on the chloride concentration change. However, operation measures have impact on the chloride concentration in the leachate. The chloride is washed out from the landfill via leachate recycling. Leachate recirculation only, would lead to chloride removal less than that dissolved in the leachate, and thus the leachate chloride concentration increase. This is the case in reactor A. Aeration might maintain a chloride concentration balance between the dissolution and removal, which as illustrated by reactor B, the chloride concentration remains basically constant all the time.

3.7. Temperature

Changes in temperature reflect the degree of solid waste degradation. Both aerobic and anaerobic degradation release energy, which would lead to an increase in solid waste temperature. The variation of temperature ranged from 25 to 39 °C. Fig. 14 shows the monitored changes of temperature for reactors A and B. The results suggest that the temperature changes in reactors A and B closely follow a similar trend. The rate at which aerobic bacteria degrade solid waste under aeration condition is higher than that of anaerobic bacteria under anaerobic condition

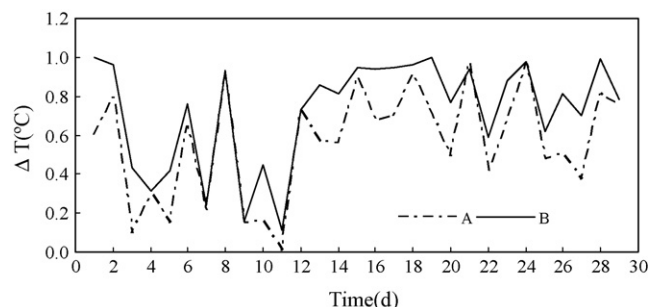


Fig. 14. Changes of temperature over time.

and so is the release more heat. Therefore, temperature change of the reactor B is a little higher than in reactor A, which indicate that aeration has positive impact on the municipal solid waste degradation.

4. Summary and conclusions

- (1) Intermittent aeration strategy is favorable for separation of acid formation phase and the methane fermentation phase, reduction of the acid-production time, optimization the microorganism growth environment and the acceleration of the stabilization rate of solid waste.
- (2) Intermittent aeration and addition of active sludge can accelerate the COD degradation effectively in leachate recirculation reactor; the removal ratio of the COD is 13.78% higher with aeration than that of only leachate recirculation. In leachate recirculation with aeration reactor, the removal ratio of the COD is 5.11% higher when active sludge was added than without. Aeration has positive impact on the release of the organics to leachate.
- (3) Intermittent aeration and addition of active sludge can enhance the transformation and removal of nitrogen. Total nitrogen removal efficiency of the leachate recirculation reactor is 40%, for the leachate recirculation reactor with aeration is 74%, while for leachate recirculation reactor with aeration and addition of active sludge is 84%.
- (4) Leachate recirculation strategies with aeration offer opportunities for more rapid waste stabilization and attenuation of heavy metals. Addition of the active sludge influence heavy metals removal slightly.

Therefore, leachate recirculation with aeration could enhance and accelerate the conversion and stabilization of solid waste by creating an environment that promotes the rapid development of the desired microbial populations of denitrifiers, nitrifiers, and methanogens. In a word, in situ removal of contaminants from leachate is effective and feasible.

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