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POLLUTANT TRANSFORMATIONS IN LANDFILL LAYERS

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The behaviour of pollutants, in particular organic carbon and nitrogen, in solid waste landfill layers and factors affecting their decomposition have been studied using four large scale simulated lysimeters in four landfill thicknesses, and operated under semi-aerobic conditions. This paper presents about 800 days of data through November 1991. Data in this study indicate that the thicker solid waste layer has lower production rates of Total Organic Carbon (TOC) and Total Nitrogen (T-N) than does the thinner layer. If the layer, however, becomes too thick, no significant increase in the removal rate of T-N is observed, as nitrogen is leached out in the form of NO_3^--N without being denitrified at the bottom of the landfill layer. It is also assumed that temperature and pH in the landfill layer play important roles not only in microbial degradation, but also in dissolution of organic carbon and nitrogen.

Key Words—Solid wastes, semi-aerobic landfill, simulated landfill, landfill thickness, lysimeters, leachate, oxygen penetration.

1. Introduction

In recent years, with a rapid increase in population and an improvement of living standards, solid wastes arising from human activities have increased, and their proper treatment and disposal in some places is an urgent problem. As this trend is likely to continue, the sustainability of many large cities depends on how to solve solid waste problems, such as the difficulty of acquiring land for landfill.

The activities associated with the management of solid wastes can be generally classified into collection and transport, recovery and processing including shredding, incineration, etc., and final disposal (landfilling). Of these activities, the prior decomposition and stabilization of solid wastes placed in landfills has been considered an important subject from an environmental protection point of view, and many studies have been undertaken.

In previous studies it has been found that the microbial degradation of pollutants is influenced by allowing a landfill to become aerobic (Sagou *et al.* 1981, Hanashima 1985). Besides, recirculation of leachate (Leckie *et al.* 1976, Matsufuji *et al.* 1986, Shimaoka *et al.* 1989) and high density waste emplacement where each new waste layer is placed over an existing one (Collins *et al.* 1973, Ham *et al.* 1982, Stegmann 1983, Lee *et al.* 1991), both have been studied as landfilling techniques that can be applied to an actual landfill. Therefore, in order to establish an effective landfill technique with efficient degradation characteristics it is necessary to understand the mechanisms that occur within the landfill. In general, the pollutants released from solid wastes migrate downwards through the unsaturated landfill layer, and are finally discharged from the system. In this way, the pollutants undergo not only the dynamic physicochemical processes such as

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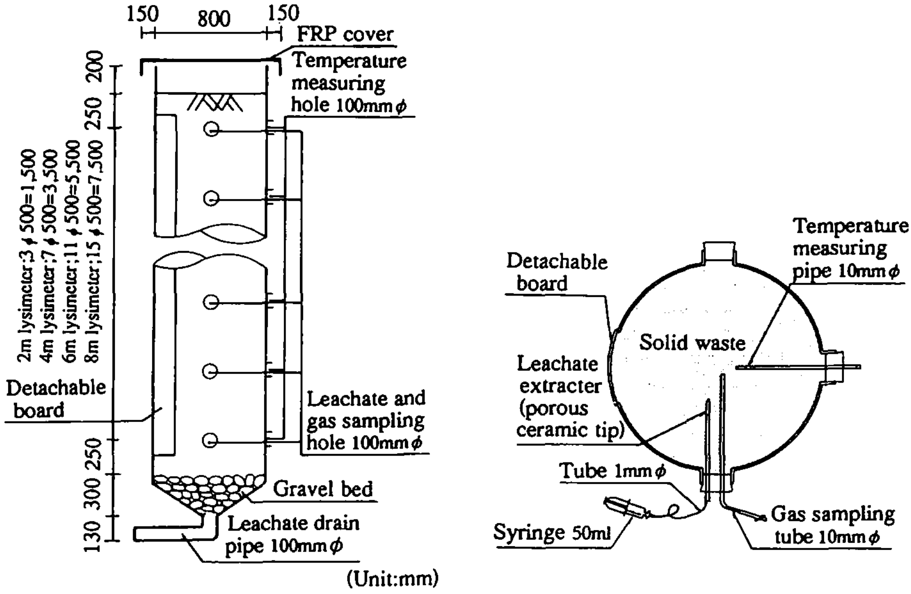

Fig. 1. Cross section through a landfill lysimeter.

dissolution, dilution, oxidation and reduction, and adsorption/desorption, but also biological processes such as microbial degradation. It has been reported that these processes can depend considerably on many factors, including landfill thickness (Qasim *et al.* 1970), moisture content, landfill temperature, waste density and waste composition. Most of the studies on the decomposition characteristics of pollutants within landfills conducted up to now employed a pilot/plant on a comparably small scale, so that the information obtained could not conveniently be transferred into practice. Furthermore, very little information about the interaction between degradation processes within a landfill has been accumulated, as most concern has been centred on the leachate quality from the bottom and gas flux from the surface.

The objectives of the study undertaken was to evaluate the effect of landfill thickness on the degradability of pollutants in a landfill layer by using four large scale simulated lysimeters with different waste thicknesses, and to investigate the factors affecting the decomposition of pollutants in each.

2. Materials and methods

2.1 Experimental conditions

Four cylindrical lysimeters were constructed: 3 m; 5 m; 7 m; and 9 m in height respectively, each of 80 cm diameter. Figure 1 depicts a cross section through a lysimeter. All of the lysimeters were operated under semi-aerobic landfill conditions by opening the bottoms to the atmosphere to let the air enter and to discharge the retentive seepage water. Free drainage conditions at the bottoms of the lysimeters were simulated by a 30 cm thick layer consisting of 10 cm diameter crushed stone. Solid waste was then placed and compacted thoroughly using a hand tamper in lysimeters of 2 m, 4 m, 6 m,

TABLE 1
Landfilling conditions

	Lysimeter A	Lysimeter B	Lysimeter C	Lysimeter D
Landfill lift thickness (cm)	200	400	600	800
Total weight of solid waste (t)	1.06	2.06	3.07	4.09
Moisture content (%)	23.6	23.0	22.1	21.9
Apparent density ($t\,m^{-3}$)	1.08	1.05	1.05	1.04
Net weight of solid waste (t)	0.81	1.59	2.39	3.19
Ignition loss (%)	16.7	16.6	16.0	15.9
C/N ratio	17.6	18.1	18.3	18.4

and 8 m, and were designated as lysimeters A, B, C, and D respectively. All the lysimeters were equipped with temperature measuring pipes, gas sampling tubes, and porous cups at 50 cm intervals to monitor the major factors such as temperature, gases/ and constituents of leachate.

In order to simulate natural average Japanese landfill conditions as close as possible with respect to carbon and nitrogen contents and to monitor the decomposition and removal phenomena of pollutants in the solid waste landfill layers, incinerator residue, crushed waste, and compost in the ratio 65:15:20 dry weight were thoroughly mixed. Incinerator residue used in this experiment was obtained from the residue hopper of a municipal solid wastes incineration plant in Fukuoka, Japan, operating in a Stokers' continuous-feed mass-fired incinerator (treatment capacity of 600 t day^{-1}, HCl gas removal process by spraying NaOH against waste gas generated). Crushed waste was collected from a crushing treatment centre (treatment capacity of 250 t day^{-1}) also in Fukuoka, and compost fermented from separated municipal solid wastes originated from a high rate composting plant (treatment capacity of 6 t day^{-1}) in Asakura town.

A total of 10 tonnes of solid wastes were placed in the lysimeters at an average density of 1.0 t m^{-3}. The average moisture contents ranged from 20 to 21% on wet weight basis. The other filling conditions are given in Table 1. All of the lysimeters were subject to environmental temperature fluctuations but natural precipitation was prevented by a FRP (Fiberglass Reinforced Plastic) cover, and was simulated by means of artificial irrigation. Tap water was irrigated in a series, twice a month for 8 hours at 82 mm of precipitation corresponding to a total 2000 mm year^{-1}, based on the recent 10-year annual average precipation of Fukuoka.

In order to produce outflows at the same time from the lysimeters of different landfill thickness, the initial moisture contents were brought up to 26% at field capacity in response to artificial rainfall for all the lysimeters, prior to the beginning of the experiment. Field capacity was assumed to be achieved when further addition of water resulted in an outflow of a significant volume of leachate.

2.2 Sampling and analysis

Before mixing the three kinds of solid wastes, each sample was collected and analysed for moisture content, carbon and nitrogen content. The physical and chemical characteristics of each solid waste used are summarized in Table 2. As the solid wastes were being

TABLE 2
Qualitative properties of each solid waste

	Incinerator residue	Crushed solid waste	Compost
Moisture content (%)	15.01	10.91	52.90
Cl^- ($mg\,l^{-1}$)*	209.2	542.4	3314.6
TOC ($mg\,l^{-1}$)*	179.9	218.2	2126.0
T-N ($mg\,l^{-1}$)*	8.0	19.7	34.9
Chloride ($mg\,kg^{-1}$)†	2331.5	4272.6	31382.6
C content (%)‡	3.055	11.433	35.153
N content (%)‡	0.088	0.496	2.708

* solid/liquid ratios of 1:10 by weight and continuous shaking for 6 hours.
† measured in accordance with the JIS K 0102.
‡ measured with a CN coder made by Yanagimoto Co. Ltd. model MT-600.

placed, solid waste samples at every 50 cm depth were collected and analysed for moisture content, C/N ratio and ignition loss.

To monitor the decomposition of pollutants in the landfill lysimeters, both leachate from the bottom and seepage water moving through the landfill layers were sampled. Seepage water at 50 cm depth was collected by inserting porous ceramic cups into the lysimeters for 2 or 3 days whilst exerting a negative pressure. The samples were filtered by a 0.45 μm membrane filter before chemical analysis.

Primary leachate composition parameters monitored included chlorides, biological oxygen demand (BOD), total organic carbon (TOC), and different forms of nitrogen. In addition, pH, electric conductivity (EC), oxidation and reduction potential (ORP), and alkalinity were measured or analysed to define leachate characteristics. Gases such as CO_2, CH_4 and O_2 were also collected from perforated gas tubes and analysed periodically.

3. Results and discussion

3.1 Qualitative and quantitative characteristics of leachate

Cumulative generation rates of leachate collected at the bottom with time are presented in Fig. 2. At the first irrigation, it was found that there were significant differences in the generation rate of leachate, varying between 5 and 35% among the lysimeters, governed by the initial moisture content of the landfill layer. The cumulative leaching rates, however, tended to increase to about 70 to 80% with additional irrigation. The result shows that the thicker the landfill layer, the lower the cumulative generation rate of leachate. The reason for this seems to be the increase in the landfill layer's water holding capacity and the evapotranspiration rate of the water held as the landfill layer becomes thicker.

The volume of leachate over time for all lysimeters after one irrigation is presented in Fig. 3. The generation volumes of leachate in lysimeters A and B (containing the lower landfill thicknesses) increased dramatically in a few hours after the start of the irrigation. They then decreased quickly after peaking sharply. In contrast to lysimeters A and B, lysimeters C and D (containing the thicker landfill layers) did not exhibit distinct peaks and showed a tendency towards a retarded leaching. It was noted that the thicker the

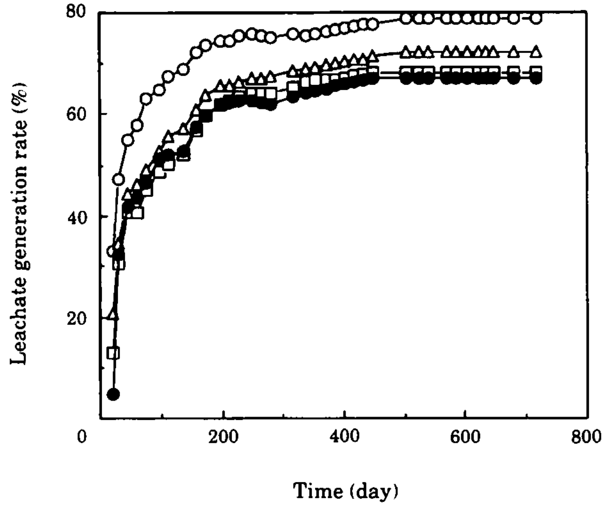

Fig. 2. Cumulative generation rates of leachate with time for lysimeters A(○); B(△); C(□); and D(●).

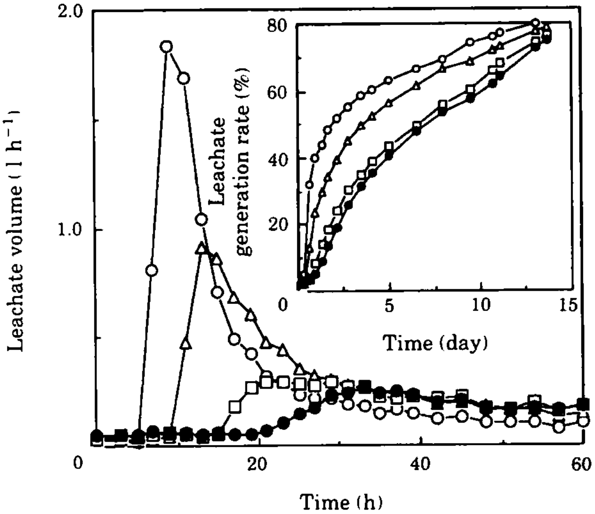

Fig. 3. Generation volume of leachate with time for 4 lysimeters for one irrigation. Key as Fig. 2.

landfill layer is, the slower the leachate begins to elute, and the maximum generation rate of leachate and the temporal variations in generation rate become small. By assessing the temporal changes of the cumulative generation rate (= cumulative generation volume/ cumulative irrigation volume × 100) of leachate in one irrigation, as shown in Fig. 3, it can be seen that the thicker the landfill, the greater the retention time of seepage water within the latter. For the cumulative generation rate to reach 50% it took 44, 85, 156, and 169 hours for lysimeter A, B, C, and D, respectively, depending on the landfill thickness.

Thus, because the maximum generation rate of leachate and the variation of the generation rate of leachate become smaller by rendering the landfill layer thicker, it is

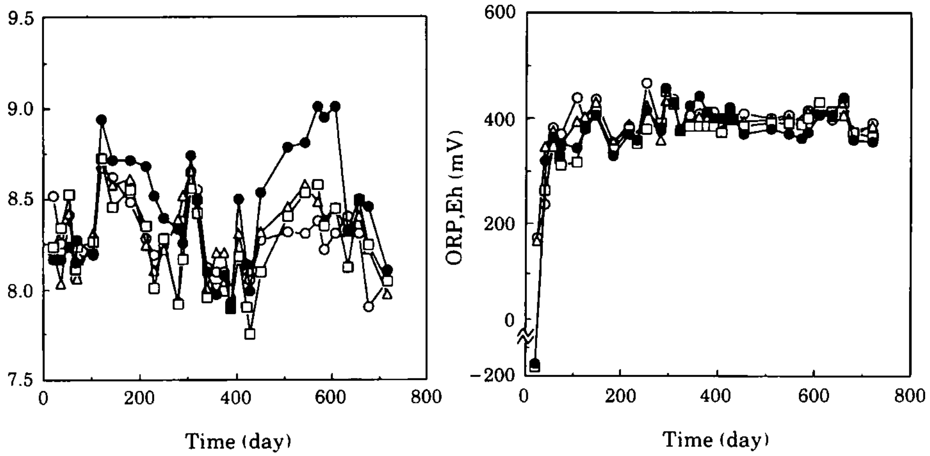

Fig. 4. Leachate pH and ORP(Eh) histories for 4 lysimeters. Key as Fig. 2.

possible to reduce the generation rates of leachate at the actual landfill sites, and thereby to reduce the design scales of the collecting and draining facilities of leachate and the leachate treatment facilities. It is also easier to maintain leachate treatment plants.

Figure 4 depicts the leachate pH and ORP(Eh) histories for all four lysimeters. The leachate pH levels for each lysimeter remained stable, at pH 8 to 9 for the duration of the experimental period. It can be assumed from these data that various inorganic salts, such as CaO and Na_2O, in the solid wastes were neutralized by the organic acids generated by hydrolysis reactions. The pH, however, decreased slightly coinciding with the summer periods—from the 1st day to the 50th day and from the 350th day to the 450th day—for all lysimeters. For this reason it seems that, as the temperature of the landfill layer goes up, the alkalinity is consumed by nitrifying bacteria (Anthony *et al.* 1980), and then the pH falls. In addition, the pH value has a tendency to increase slightly as the landfill layer thickens, but a significant difference with respect to the landfill thickness was not seen.

The temporal changes of ORP(Eh) of lysimeters A and B were higher than 200 mV for the duration of the experiment. It can be inferred from these data that the two lysimeters were under oxidizing conditions from the beginning of the experiment. Lysimeters C and D, however, showed reducing conditions [ORP(Eh) = − 160 mV] in the first stage of the experiment, as the dissolved oxygen of the seepage water within the landfill was depleted (Rovers *et al.* 1973). As time progressed, oxygen penetrated through the leachate drain pipe at the bottoms of the lysimeters and the insides of the landfill layers tended to become more aerobic. As a result, ORP(Eh) > 350 mV in all lysimeters from approximately 40 days after the start of the experiment, with no differences in ORP(Eh) among them.

Figure 5 presents the leachate TOC and T-N histories for the four lysimeters. At the beginning of the experiment TOC and T-N concentrations were high, about 4000–7000 mg l^{-1} and 1000–2500 mg l^{-1}, respectively, due to high initial loads from readily soluble salts. However, the two parameters decreased exponentially with time, towards 500 mg l^{-1} and 250 mg l^{-1}, respectively. The initial peak concentrations and the steady-state concentrations of TOC and T-N of the leachate tended to become higher as the landfill layer became thicker, but these values were not proportional to the landfill thickness.

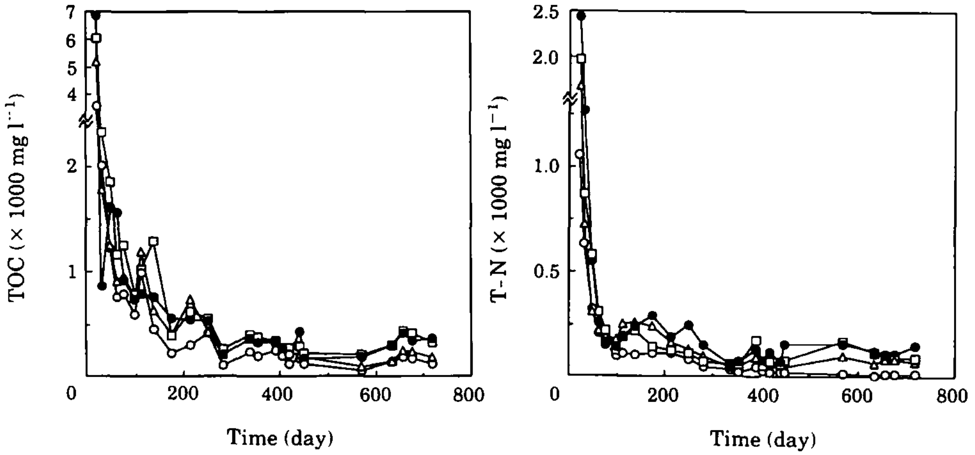

Fig. 5. Leachate TOC and T-N histories for 4 lysimeters. Key as Fig. 2.

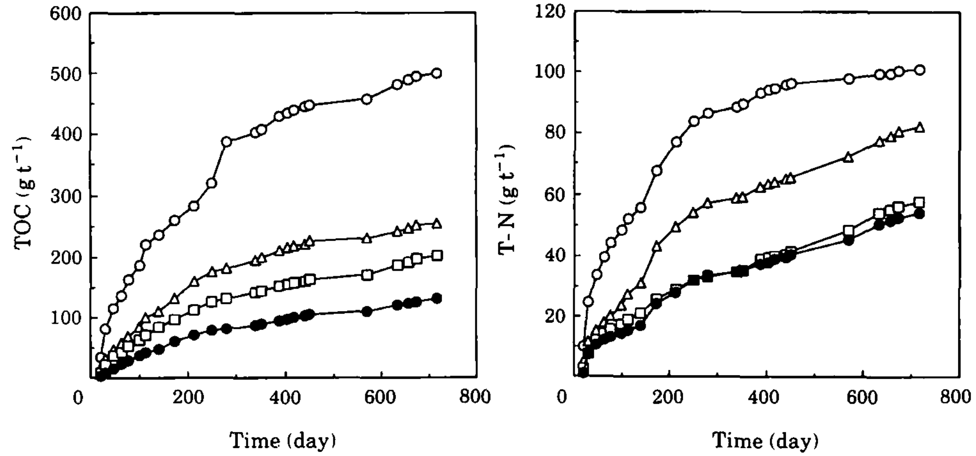

Fig. 6. Cumulative productions of TOC and T-N per tonne solid waste basis with time for 4 lysimeters. Key as Fig. 2.

3.2 Relationships between landfill thickness and pollutant removal

As stated previously, the generation rate of leachate tended to decrease as the landfill layer became thicker, and there were also small differences in leachate qualities, independent of the landfill thickness. In this section, the effects of landfill thickness on the production of pollutants and the pollutant removal characteristics in various landfill thickness are discussed, using an expression of the leaching amount of pollutant to leachate per dry solid waste basis.

Cumulative production of TOC and T-N to leachate per dry solid waste basis with time for all four lysimeters are presented in Fig. 6. The cumulative TOC produced for lysimeter A was about 500 g t^{-1} over 718 days, while lysimeter D leached about 100 g t^{-1} during the same period. This shows that the cumulative TOC production is in inverse proportion to the thickness of the landfill layer. It seems likely that as the layer thickens,

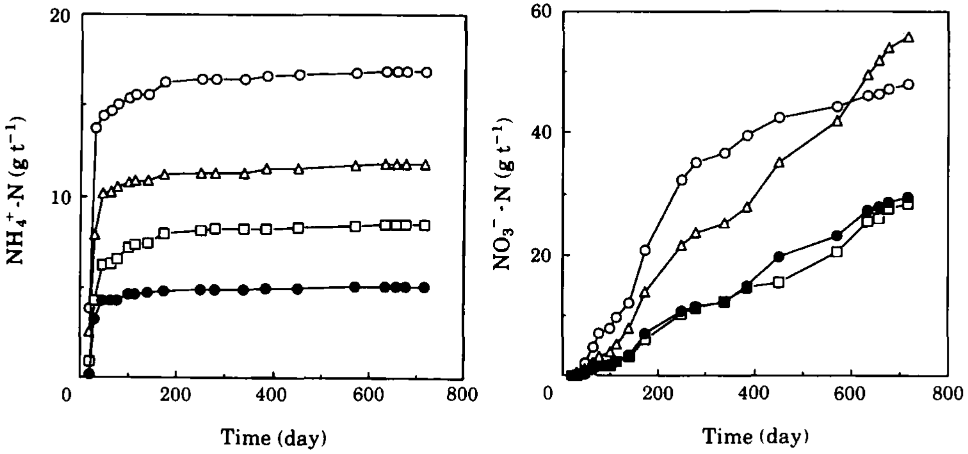

Fig. 7. Cumulative productions of NH_4^+-N and NO_3^--N per tonne solid waste basis with time for 4 lysimeters. Key as Fig. 2.

an increase in the amount of TOC removed in the landfill layer occurs, due to the length of retention time of the seepage water in the layer (Qasim *et al.* 1970).

The cumulative T-N produced on a dry solid waste basis also tends to decrease as the layer becomes thicker. This relationship is similar to that between the cumulative TOC production and the landfill thickness. However, when the latter was 8 m thick, in lysimeter D the cumulative T-N production did not differ greatly from that of lysimeter C, in which the waste layer was only 6 m thick. Therefore, we will discuss further the leaching characteristics of T-N according to the form of nitrogen. Figure 7 presents the cumulative production of NH_4^+-N and NO_3^--N per dry solid waste basis, except that of NO_2^--N which is comparatively small, with respect to time for the four lysimeters. The cumulative NH_4^+-N production was reduced as the layer thickened, and tended to be in inverse proportion to the landfill thickness. The relationship between the cumulative production and the landfill thickness for T-N was the same as that for TOC. However, when the thickness of the landfill layer increased to 8.0 m, the cumulative production of NO_3^--N became indistinct. From this, it appears that when a landfill layer becomes thick the nitrification of NH_4^+-N predominates over the denitrification of NO_3^--N, due to an increase in the aerobic area at the bottom of the landfill, as is presented in Fig. 11.

Figure 8 summarizes the relationships between the landfill thickness and total TOC and T-N produced. In general, the thicker the layer, the lower the amount of TOC eluted on a dry solid waste basis, presumably because the ability to remove it is proportional to the thickness of the layer. T-N also maintained a removal ability in proportion to the landfill thickness, and reached equilibrium at a thickness of 6 m.

3.3 Qualitative transformation characteristics of pollutants in a landfill layer

Figure 9 illustrates the TOC and T-N vertical concentration distributions through the landfill layer for lysimeters A, B, and D. Some concentration peaks can be identified in both TOC and T-N profiles corresponding to each lysimeter.

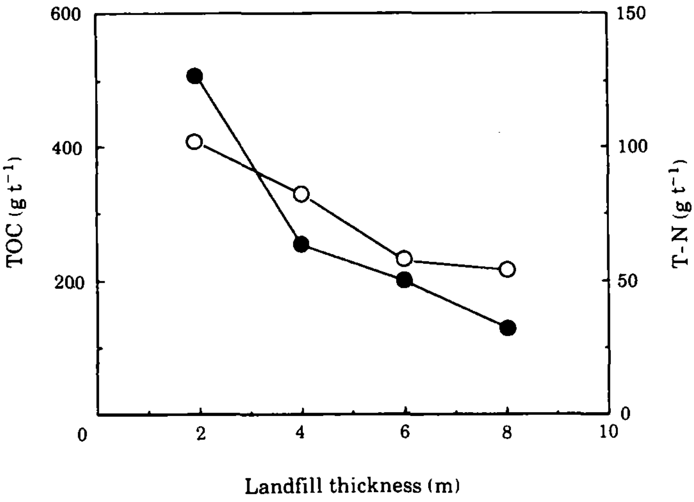

Fig. 8. Relationships between landfill thickness and total produced TOC (●), and T-N (○).

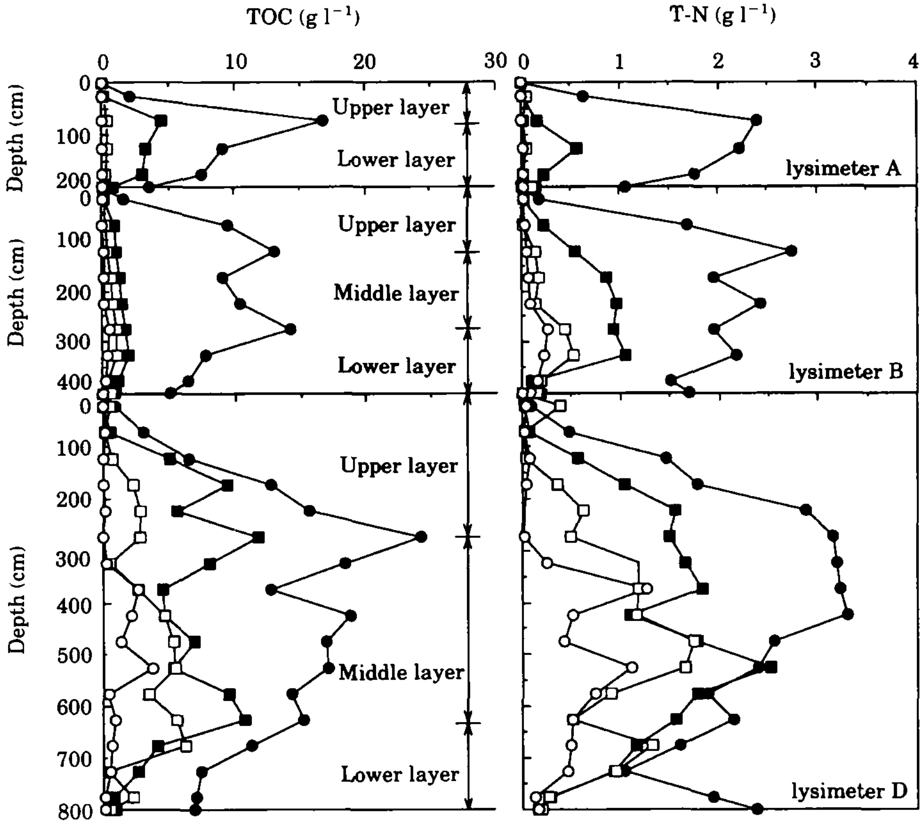

Fig. 9. TOC and T-N profiles for lysimeters A, B, and D. (●) 21 day; (■) 77 day; (□) 213 day; (○) 570 day.

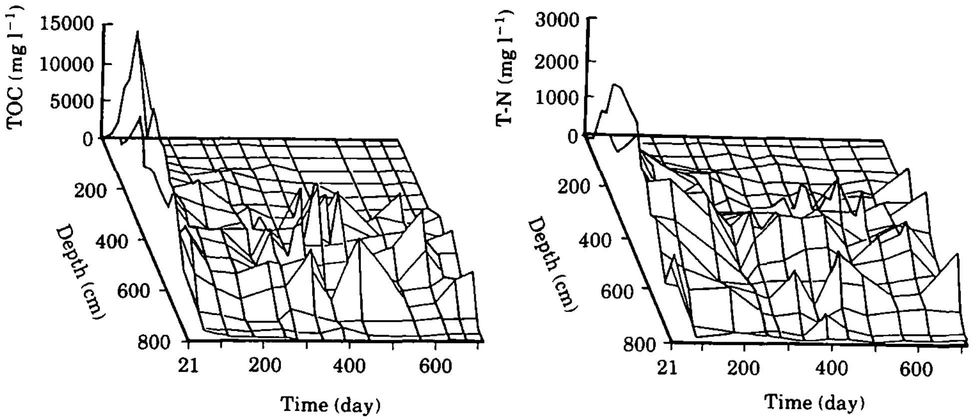

Fig. 10. TOC and T-N profiles with time for lysimeter D.

In lysimeter A, there appeared to be a peak for each component. The authors divided the original waste layer into an upper layer and a lower layer with the concentration peak as a border. It seems likely that TOC and T-N are dissolved from the solid wastes in the upper layer, and flow down with the movement of seepage water, increasing in concentration. Meanwhile, the degradation of TOC is enhanced by aerobic microorganisms in the lower layer, as the air penetrates from the leachate drain pipe in the bottom. At the same time, the rapid decrease in T-N concentration occurs in the lower layer, due to the denitrification activity at partially anaerobic microsites (Timothy 1987, Seech *et al.* 1988).

In lysimeters B and D, the concentration distributions of TOC and T-N varied from those in lysimeter A. The landfill can be divided into three layers: upper (from surface to the depth the first peak appears); middle (from the depth the peak appears to the depth the last peak appears), and lower (from the depth the last peak appears to the bottom). Depths at which the borders existed differed a little with regard to each component, but the upper and lower layers had the same properties as those in lysimeter A. The middle layer became apparent when the landfill thickness was larger than 2 m, and the concentrations of TOC and T-N increased and decreased vertically. This suggests that the concentrations of these components, which are increased in the upper layer, decrease rapidly due to the degradation by microorganisms in the middle layer, and then increase due to dissolution in the seepage water.

Furthermore, the removal of T-N as well as TOC predominated in the lower layer, especially at a depth of 150 cm from the bottom of all the lysimeters, even though there were differences in the layer thickness. This indicates that, in a semi-aerobic landfill, the degradation of organic matter and nitrogen around the leachate drain pipe or oxygen source point, governs the leachate quality.

The TOC and T-N profiles with respect to time for lysimeter D (Fig. 10) showed concentration distributions which repeated the increases and decreases over time. However, as time passed, the depth of upper layers enlarged, encroaching on the middle layer until it finally ceased to exist. In lysimeter D it seems likely that TOC and T-N components in the landfill layer were dissolved and microbially degraded repeatedly, so that complicated concentration distributions were established vertically throughout the depth.

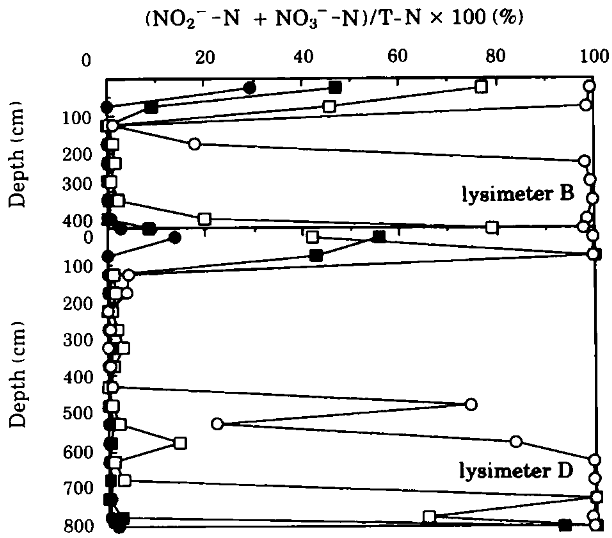

Fig. 11. NO_2 -N and NO_3 -N percentage profiles for lysimeters B and D. (●), 31 day; (■), 77 day; (□), 250 day; (○), 635 day.

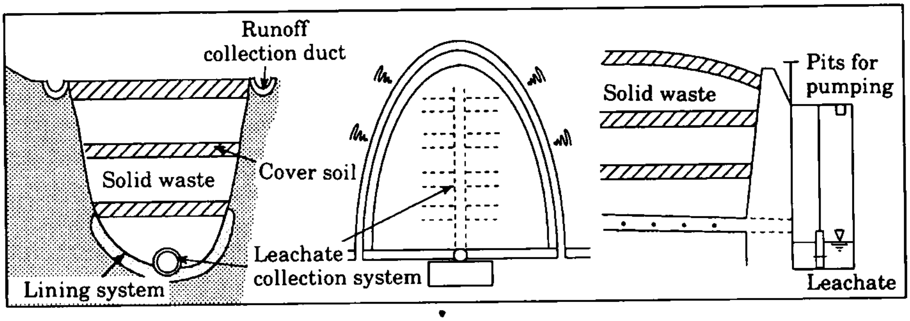

Fig. 12. Schematic diagram of semi-aerobic landfill structure.

Figure 11 shows the NO_2^--N and NO_3^--N concentration percentage distributions with time for lysimeters B and D. At the beginning of the experiment NH_4^+-N was dominant over the entire depth in both lysimeters. After 77 days nitrification began to occur at some depths in the upper and the lower layers, and NO_2^--N and NO_3^--N made up the major part of T-N. As time progressed, this area spread gradually from the surface of the upper layer and the bottom of the lower layer to the inner part of the layer. The percentage profiles of NO_2^--N and NO_3^--N on the 635th day show that nitrification occurred up to approximately 1 m below the surface in both lysimeters. Nitrification occurred up to about 2 m from the bottom of lysimeter B and about 4 m from the bottom of lysimeter D, while in both lysimeters nitrification occurred up to 1 m from the surfaces. Consequently, NO_3^--N was leached out of the system without denitrification, as described earlier.

3.4 Penetration characteristics of oxygen into a landfill layer

In Japan, approximately 53% of actual landfill sites have been semi-aerobic as shown in Fig. 12. Some of the advantages of semi-aerobic landfills are that solid waste is more

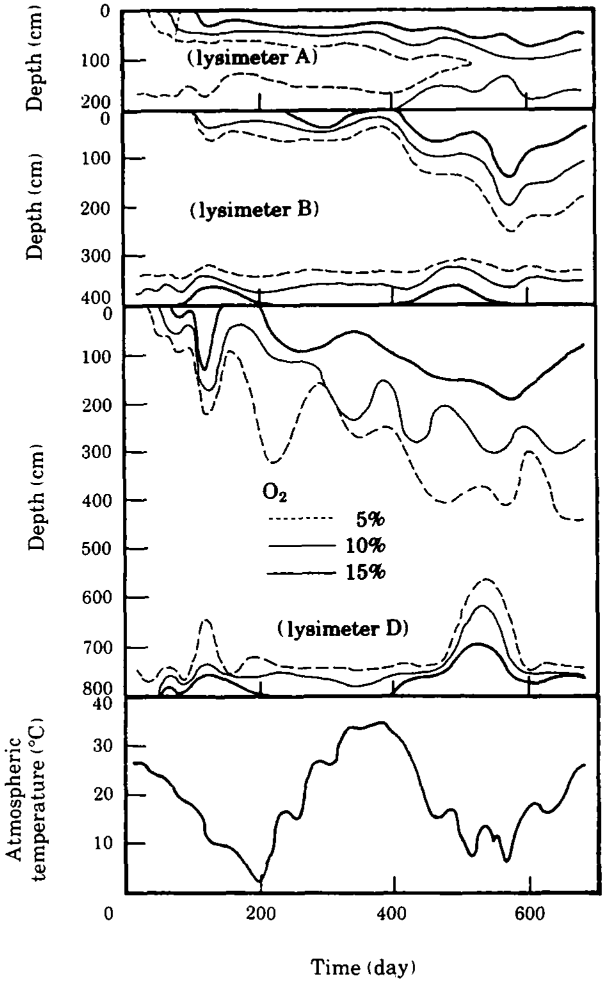

Fig. 13. Isoconcentration lines of O_2 with time for lysimeter A, B, and D.

quickly stabilized than in anaerobic landfills and can be maintained at a low cost. As a general characteristic of semi-aerobic landfills, the retentive seepage water can be drawn off rapidly from the landfill layer through the leachate drain pipe (a perforated pipe) located in the bottom. At the same time air (oxygen) can be supplied to the landfill layers through the same pipe by oxygen diffusion (Tanaka *et al.* 1986) or natural convective flows (Hanashima *et al.* 1981) so that aerobic degradation of organic matter in the landfill is enhanced. Therefore, it is necessary to clarify the mechanism of oxygen penetration; the stabilization of a semi-aerobic landfill is governed by the supply and consumption of oxygen.

Figure 13 shows isoconcentration lines of O_2 in the landfill layer with time for lysimeters A, B, and D. At the beginning of the experiment the oxygen concentration was low—under 1%—over the entire depth in all lysimeters. As time progressed the oxygen began to penetrate into the landfill layer from both the surface and the bottom. In particular, in lysimeter A, the oxygen concentration was more than 5% over the entire depth after approximately 500 days. It was confirmed that the lysimeter was fairly aerobic. For lysimeters B and D in the initial stages of the experiment, apparently due to

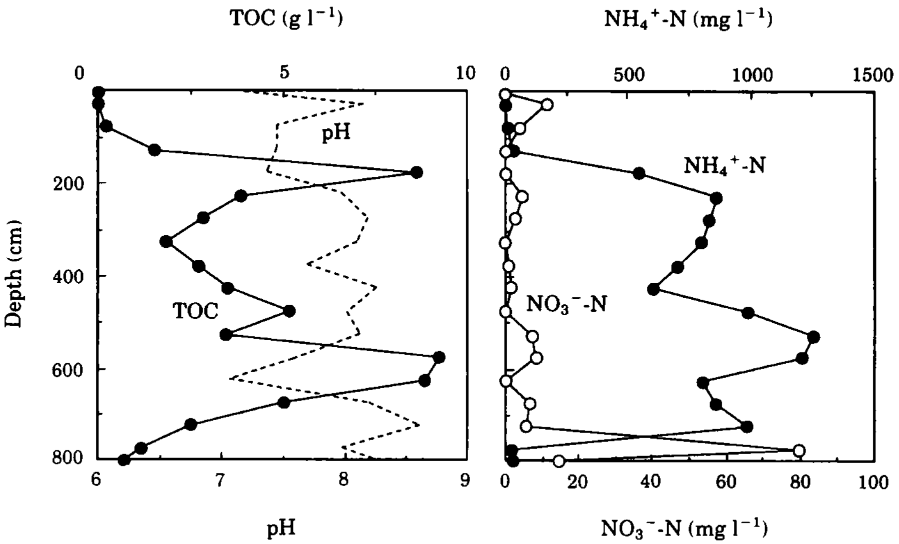

Fig. 14. TOC, pH, NH_4^+-N, and NO_3^--N profiles for lysimeter D.

the presence of pollutants in high concentrations throughout the entire landfill layer, the penetrating oxygen was consumed quite quickly. Thus, the oxygen penetration depth appeared to be quite small. The concentrations of pollutants, however, decreased with time due to microbial degradation and the consumption of penetrated oxygen was reduced. This resulted in an increase in the depth to which oxygen penetrated.

On the other hand, it was found that as the landfill layer became thicker, the depth at which the oxygen concentration was over 5% increased due to microbial activities—approximately 2 m in lysimeter B and 4 m in lysimeter D. However, from the bottom of the landfill layer the corresponding thickness of over 5% oxygen concentration was about 1 m, independent of landfill thickness. As mentioned in the previous section, oxygen consumption by microorganisms in the upper and middle layers decreased to a minimal level with time due to lack of substrate, and the depth to which oxygen penetrates increased. This was because the pollutants released from the landfill layers flow down with the seepage of water. In the lower layer where microbial activities are high, due to being continuously supplied with substrates from the upper and middle layers, the oxygen penetrating through the leachate drain pipe is consumed rapidly so that the oxygen penetrating depth is short.

With respect to yearly variation, the oxygen penetrating depth increases when the temperature lowers, and decreases when the temperature rises. This increase in oxygen penetrating depth may be caused by an increase in the natural convective flows of air due to the pressure gradient between the inside of the layer and the atmosphere, which is caused by the decrease in the consumption of oxygen and the production of gas. Conversely, when the temperature within the landfill layer goes up, the depth to which oxygen penetrates is reduced, due to the increase in the consumption of oxygen by the microbial activities and the dissolution of biodegradable matter from the solid wastes.

3.5 Factors affecting the vertical concentration distributions of pollutants

In this section, the factors affecting the dissolution and microbial activities are discussed. Figure 14 presents the NH_4^+-N, NO_3^--N, TOC, and pH profiles for lysimeter D. From

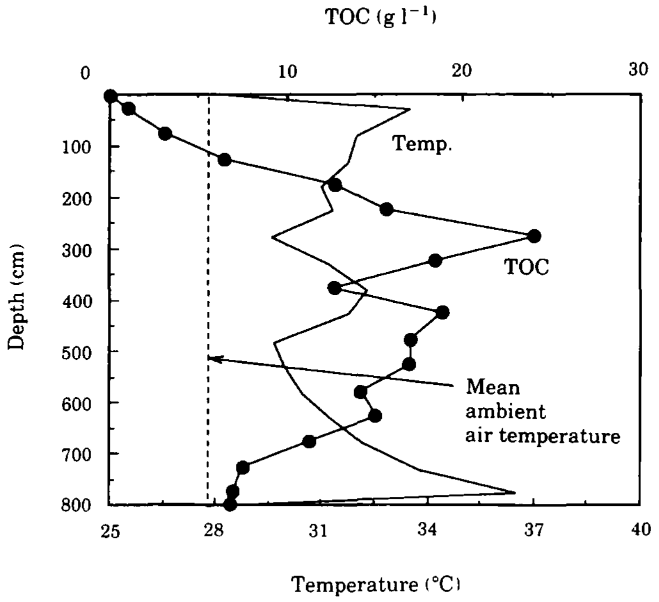

Fig. 15. Temperature and TOC profiles for lysimeter D.

the surface to a depth of about 175 cm, the pH rapidly increased to a level of about 8.0, resulting from alkali components dissolving into seepage water. The pH then decreased to a value of about 7.0 as the alkali components were consumed in nitrification of NH_4^+-N (Anthony *et al.* 1980), and were neutralized by the organic acids generated by hydrolysis reactions. Between the depths of 175 to 625 cm, microbial activities became quite high due to the decrease in pH. As a result, the degradation of TOC was accelerated, so that the pH again increased. It would seem that when the pH goes up to about 8, microbial activities fall. From the depth of 625 cm to the bottom, the pH tended to increase surprisingly by the microbial degradation of TOC, but decrease and increase slightly from the bottom to a depth of 100 cm by the nitrification activity and the denitrification activity, respectively.

Figure 15 shows the temperature and TOC profiles for lysimeter D. Temperature was generally high in the upper and lower layers where TOC removal was facilitated by aerobic microorganisms. In particular, the concentration of substrate was higher in the lower layer than in the upper, and the temperature was also higher in the lower layer than in the upper and middle layers. In the middle layer, the temperature went up at the depth where the concentration of TOC decreased, and down at the depth where the concentration of TOC increases. This supports the contention that the decrease of TOC in the landfill layer was caused by microbial activities.

As shown in Fig. 10 in section 3.3, the concentrations of TOC and T-N were high in the landfill layer during the summer periods (from the 1st to the 50th day and from the 350th to the 450th day) because microbial activities were high during these periods. Thus it was confirmed that the dissolution from the solid wastes of both components was considerably influenced by the temperature.

The data show that the pH and temperature distributions are vertical through the lysimeters due to microbial activities and the dissolution of pollutants in the landfill layer.

4. Summary and conclusions

By investigating the behaviour of four large scale simulated lysimeters of different landfill thickness it was possible to discover both the effects of landfill thickness with respect to the production rate of pollutant on a dry solid waste basis, and the spatial interactions and phenomena in the landfill previously little known. The results of the present experiment can be summarized as follows:

- The thicker the landfill layer of solid waste, the lower the cumulative generation rate of leachate, and the smaller the maximum generation rate of leachate and temporal variations.
- The initial peak concentrations and the steady-state concentrations of TOC and T-N of the leachate tended to increase as the landfill layer became thicker, but these values were not proportional to the landfill thickness.
- In a semi-aerobic landfill, the thicker the landfill layer, the greater the removed amount of TOC within the layer. T-N also maintained a removal ability in proportion to the landfill thickness up to a thickness of 6 m. However, above this thickness there appears to be a finite limit to removal ability with increasing layer thickess, as the nitrification of NH_4^+-N seemed to predominate over the denitrification of NO_3^--N, due to an increase in aerobic area at the bottom of the landfill.
- The depth to which oxygen penetrates into the landfill layer increased as the temperature decreased due to the natural convective air flow which arises from the pressure gradient between the inside of the layer and the atmosphere. Conversely, oxygen penetrating depth into the landfill layer decreased as the temperature increased due to the increase in consumption of oxygen by microbial activities and the dissolution of biodegradable matter from the solid wastes.
- Vertical pH and temperature distributions throughout the landfill layer were variable, apparently in response to microbial activities and dissolution of pollutants in turn governed by the pH and temperature distributions.

The above results demonstrate a possibility for the structural improvement of conventional landfills. They also support the development of a landfilling process with respect to an optimal landfill thickness and semi-aerobic state in order to utilize the degradation capacity of a landfill efficiently, and may contribute significantly towards the earlier stabilization of landfills. The results from this experiment support the possibility of developing more accurate models for predicting the behaviour of pollutants within landfill layers, and making the functional design of leachate treatment facilities more efficient.

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References

Anthony, F. G., Elizabeth, T. G. (1980) *Microbiology for Environmental Scientists and Engineers*, McGraw-Hill Series in Water Resources and Environmental Engineering, pp. 504–505.

Collins, H. J., Spillmann, P. (1973) Lysimeters for simulating sanitary landfill. *Journal of the Environmental Engineering Division*, ASCE, **108**, 852–863.

Ham, R. K., Bookter, T. J. (1982) Decomposition of solid waste in test lysimeters. *Journal of the Environmental Engineering Division*, ASCE, **108**, 1147–1170.

Hanashima, M. (1985) Aerobic landfill of solid wastes. Eng.D. Dissertation, University of Kyushu, Japan (in Japanese).

Hanashima, M. *et al.* (1981) Heat and gas flow analysis in semiaerobic landfill. *Journal of the Environmental Engineering Division*, ASCE, **107**, 1–9.

Leckie, J. O., Pacey, J. G. & Halvadakis, C. (1976) Landfill management with moisture control. *Journal of the Environmental Engineering Division*, ASCE, **105**, 337–355.

Lee, N. H. *et al.* (1991) Decomposition characteristics of the existing landfill layer in the building up of solid waste. *Journal of Japan Waste Management Association*, **43**, 287–295 (in Japanese).

Matsufuji, Y. *et al.* (1986) Practical applications of the recirculatory semi-aerobic landfill. *Proceedings of the 7th Annual Conference of the Japan Waste Management Association*, 204–207 (in Japanese).

Qasim, S. R., Burchinal, J. C. (1970) Leaching from simulated landfills. *Journal WPCF*, **42**, 371–379.

Sagou, M. *et al.* (1981) Effective landfill techniques of solid wastes (Ver. 1). *Journal of Japan Waste Management Association*, **34**, 294–313 (in Japanese).

Seech, A. G., Beauchamp, E. G. (1988) Denitrification in soil aggregates of different sizes. *Journal of Soil Science Society of America*, **52**, 1616–1621.

Shimaoka, T. *et al.* (1989) Nitrogen removal from leachate by self-purification in solid waste layer. *Proceedings 1 of Sardinia '89–2nd International Landfill Symposium, Sardinia, Italy*. pp. XLIV1–XLIV12.

Stegmann, R. (1983) New aspects on enhancing biological processes in sanitary landfill. *Waste Management & Research*, **1**, 201–211.

Tanaka, N. *et al.* (1986) Penetrating mass flux of oxygen and aerobic zone depth in sanitary landfill layer. *Proceedings of Environmental and Sanitary Engineering Research*, JSCE, **22**, 37–48 (in Japanese).

Timothy, B. P. (1987) Soil-microsites as a source of denitrification variability. *Journal of Soil Science Society of America*, **51**, 1194–1199.

Rovers, F. A., Farquhar, G. J. (1973) Infiltration and landfill behavior. *Journal of the Environmental Engineering Division*, ASCE, **99**, 671–690.