

ORIGINAL ARTICLE

Characteristics of CO₂, CH₄ and N₂O emissions from a multi-soil-layering system during wastewater treatment

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Abstract

In the present study, characterization of greenhouse gas emissions, such as CO₂, CH₄ and N₂O, from a multi-soil-layering (MSL) system during wastewater treatment was conducted, using a laboratory scale system (width 50 × depth 10 × height 120 cm). Net fluxes of CO₂, CH₄ and N₂O from the system were in the range of 21.2–48.0 g m⁻² day⁻¹, –66.4–0.02 mg m⁻² day⁻¹ and 41–246 mg m⁻² day⁻¹, respectively. Carbon dioxide emission generally increased as the air temperature increased. The net flux of CO₂ was not appreciably different among the treatments, even though the hydraulic loading rates (HLR) differed widely. These findings indicated that the decomposition of organic matter loaded through wastewater was more efficient in the low HLR treatments. Mean CH₄ emission was lower than the amount of CH₄ brought by the influent. Methane was consumed in the MSL system during most of the study period, and the consumption tended to be more efficient in the treatments at high HLRs. Wastewater quality was not markedly related to the characteristics of the CO₂ and CH₄ emissions. Nitrous oxide was mainly released into the treated water as dissolved gas. Nitrous oxide emission per 1,000 L of wastewater decreased as the HLRs increased in the wastewater containing a high level of contaminants (HWW), while in the case of wastewater containing a low level of contaminants (LWW), such a decrease in the emission was not observed. Mean N₂O conversion rates of the LWW treatments ranged from 1.10 to 1.79% and were higher than those of the HWW treatments, which ranged from 0.19 to 1.95%. Nitrous oxide emission could be reduced in the HWW treatments.

Key words: carbon dioxide, methane, nitrous oxide, multi-soil-layering system, wastewater treatment.

INTRODUCTION

The characteristics and results of various types of wastewater treatments using a multi-soil-layering (MSL) system have been reported in several papers (Luanmanee *et al.* 2002; Masunaga *et al.* 2002a, 2003; Wakatsuki *et al.* 1993). These authors reported extensively the characteristics of the MSL system and its applicability to various types of wastewater or polluted water treatments. However, the mechanisms of wastewater

treatment inside the MSL system have not yet been fully elucidated, although attempts had been made in a number of recent papers (Masunaga *et al.* 2002b; Sato *et al.* 2002a; Sato 2005a,b). The fate of the contaminants removed in the MSL system has also not been fully clarified. While phosphorus was absorbed in the MSL system, organic matter and nitrogen were reported to be decomposed and released from the system (Luanmanee *et al.* 2001; Sato *et al.* 2002b). However, these authors did not determine the forms of the decomposed organic matter and nitrogen. Transformation of organic matter and nitrogen during wastewater treatment could lead to the release of greenhouse gases such as CH₄ and N₂O (Czepiel *et al.* 1995; Intergovernmental Panel on Climate Change 1995). Therefore, because domestic wastewater and its treatment process play an important role in

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various sources of greenhouse gases, there has been a growing concern about greenhouse gas emissions from wastewater treatment systems. As large quantities of carbon and nitrogen flow into domestic wastewater treatment systems, significant amounts of CH₄ and N₂O emissions are likely to occur. Emissions of these greenhouse gases vary with the treatment conditions in sewage treatment plants based on the active sludge method (Itokawa *et al.* 1996; Kimochi *et al.* 1998). However, there have been very few studies on greenhouse gas emissions from the soil trench system, which is a soil-based wastewater treatment system (Kong *et al.* 2002), although there are several reports on greenhouse gas emissions from reclaimed effluent applied to the soil (i.e. secondary treated sewage) (Master *et al.* 2003). As the amount of wastewater discharged into a MSL system is usually 10–100-fold higher than that in the soil trench system or the reclaimed effluent applied to land, it can be assumed that the characteristics of greenhouse gas emissions from a MSL system will be different.

The purpose of the present study was to characterize and evaluate greenhouse gas emissions, such as CO₂, CH₄ and N₂O, from a MSL system during wastewater treatment. Characterization of the removal efficiency of contaminants, such as organic matter, nitrogen and phosphorus, has been reported in a previous paper using the same wastewater treatment experiments (Masunaga *et al.* 2007).

MATERIALS AND METHODS

Description of the experimental conditions

Figure 1 shows the structure of the laboratory-scale MSL system used in the present study. Acrylic boxes, height 139 cm × width 50 cm × depth 10 cm in size, were prepared. The MSL system consisted of soil mixture layers and zeolite layers with a diameter of 1–3 mm. The soil mixture layers were composed of volcanic ash soil rich in organic matter classified into Andisol (United States Department of Agriculture (USDA) 1999), sawdust and granular metal iron at a volume ratio of 75%, 12.5% and 12.5%, respectively. These layers were 8 cm thick and were covered with a 2 cm thick layer of powder charcoal and were arranged in a brick-layer-like pattern surrounded by zeolite layers. The top of the MSL system was fully covered by an acrylic chamber, height 20 cm × width 50 cm × depth 10 cm, 10,000 cm³ in size for the collection of the greenhouse gases emitted from the system to the atmosphere. The chamber was closed only during the period of gas sampling. Although two aeration pipes were installed in the system, aeration was not applied during this experiment.

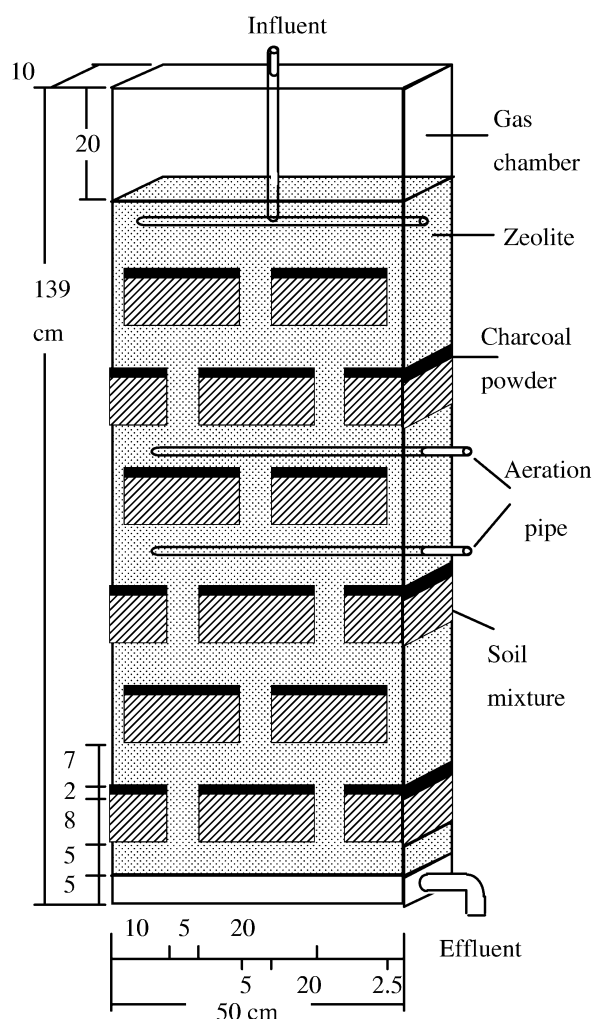


Figure 1 Structure of the multi-soil-layering system.

Domestic wastewater from a housing development was diluted (8 or 4-fold) with well water. These two types of wastewater contained relatively low or high levels of contaminants and were hereafter designated as LWW and HWW, respectively. The mean properties of LWW and HWW were as follows: 32.1 and 69.5 mg L⁻¹ for Biochemical Oxygen Demand (BOD), 6.3 and 9.6 mg L⁻¹ for the total nitrogen content (T-N) and 0.76 and 1.47 mg L⁻¹ for the total phosphorus content (T-P). Detailed data were presented in table 1 of our previous paper (Masunaga *et al.* 2007). Each type of wastewater was discharged into the MSL system continuously for 24 h at hydraulic loading rates (HLRs) of 500, 1,000, 1,250, 1,500 and 2,000 L m⁻² day⁻¹ using tube pumps (IWAKI PST-1120, Asahi Techno Glass Co Ltd, Tokyo, Japan). Therefore, the number of total treatments was 10 with a combination of five HLRs and two

grades of wastewater. Hereafter, the treatments are designated as L-500 to L-2000 and H-500 to H-2000. During the 14-month period of the wastewater treatment experiment, clogging occurred during some of the treatments. At that time, wastewater discharge into the clogged system was interrupted for 1–2 or 8 weeks for recovery. Details on the clogging were described in our previous paper (Masunaga *et al.* 2007).

Sampling and analyses

The emissions of the greenhouse gases CO₂, CH₄ and N₂O released into the atmosphere (EA) were determined using a static chamber technique from the top of the system. A 200-mL gas sample was placed into a 1-L Tedlar bag at 0 and 30 min for CO₂ and CH₄ and at 0, 1, 2 and 3 h for N₂O after the chamber lid was closed. The gas emission was calculated based on the changes in the gas concentration in the chamber in relation to the closure time.

The dissolved gas (CO₂, CH₄ and N₂O) concentrations in both the wastewater and the treated water were determined using a headspace equilibration technique (Liikanen *et al.* 2002; Senga *et al.* 2001). The gas emission from treated water (ET) was calculated as the product of the dissolved gas concentration and the volume of the treated water released from the system. For the determination of dissolved CO₂ and CH₄ concentrations, a 25-mL water sample was introduced into a 60-mL plastic syringe equipped with a silicon stopper without any gas-phase, and 1 mL of a 0.1 mol L⁻¹ HgCl₂ solution was immediately added to the sample to stop biological activity. Then 1 mL of a 0.5 mol L⁻¹ H₂SO₄ solution was added to lower the pH and to convert all the aqueous carbonate species to H₂CO₃. Twenty-five milliliters of 99.9% grade N₂ gas was added and the syringe was shaken for one minute and was kept at 20°C for equilibration. Carbon dioxide and CH₄ concentrations in the gas-phase were then determined. For the determination of the dissolved N₂O concentration, a water sample was placed into a 70-mL glass vial, and the vial was then sealed with a butyl-rubber stopper and an aluminum cap without any gas phase, followed by the injection of 1.9 mL of formaldehyde (1% final concentration) to stop biological activity. The headspace contained 30 mL of 99.9% grade N₂ gas. After the glass vial was shaken for one minute and kept at 25°C for equilibration, the N₂O concentration in the gas-phase was determined. The dissolved gas concentrations were calculated according to Henry's law using the solubility constants of CO₂ and CH₄ (Lide 2001) and the solubility formula of N₂O (Weiss and Price 1980). Sampling of the gases and water, which usually started at 10:00 hours, was conducted over a period of 10 months at intervals of approximately 2 weeks. For water sampling, wastewater was collected

from the storage tank and the treated water was collected immediately after it was drained from the system for dissolved gas analyses for the oxidation–reduction potential (ORP) measurement (TOA HM-14P, Tokyo, Japan).

The concentrations of the gases were determined using gas chromatography (GC). The CO₂ and CH₄ concentrations were measured using a gas chromatograph (GC, Shimadzu GC-14A, Kyoto, Japan) equipped with a flame ionization detector (FID), a thermal conductivity detector (TCD), and a 2-m stainless column packed with divinylbenzene–ethylvinylbenzene–ethylene glycol dimethacrylate (50/80 mesh). The temperatures of injection, column, FID and TCD were 80, 60, 100, 80°C, respectively. A carrier gas, ultrapure N₂, flowed at a rate of 60 mL min⁻¹. The nitrous oxide concentration was measured using a GC (Shimadzu GC-8A) equipped with an electron capture detector (ECD) and a 2-m stainless column packed with Unideads C (80/100 mesh). Temperatures of injection and ECD were 200 and 300°C, respectively. A carrier gas, ultrapure grade N₂, flowed at a rate of 50 mL min⁻¹.

It was assumed that approximately 90% of the CO₂ dissolved in the water samples occurred as HCO₃⁻ because the pH values of the water samples were almost neutral (Masunaga *et al.* 2007). However, we expressed the amount of all the aqueous carbonate species as CO₂ (dissolved CO₂) in the present paper because we considered that this form would be suitable for evaluating the characteristics of the emission of total CO₂ produced in the wastewater treatment processes in the MSL system.

Net gas (CO₂, CH₄ and N₂O) flux was calculated as: net gas flux = EA + ET – [amount of dissolved gas in the wastewater], and the N₂O conversion rate was defined and calculated as: N₂O conversion rate = [net N₂O flux] / [total amount of nitrogen loaded].

RESULTS AND DISCUSSION

Variations in CO₂, CH₄ and N₂O concentrations in water and gas emissions

Figure 2 shows the changes in the air temperature and ORP of treated water, and the concentrations of the CO₂, CH₄ and N₂O gases dissolved in LWW and HWW and in the L-500 and L-2000 and H-500 and H-2000 waters. Figure 3 shows the variation in the gas fluxes from the top surface of the system into the air (EA). Although the data of the other treatments are not shown in Figs. 2,3, the pattern of the fluctuations in the graphs was generally identical. The L-2000 and H-2000 wastewater treatments were interrupted around February to May because of clogging (Fig. 3c). Although the influent stopped in the system during that period, the EA values of the gases were measured and are shown in Fig. 3.

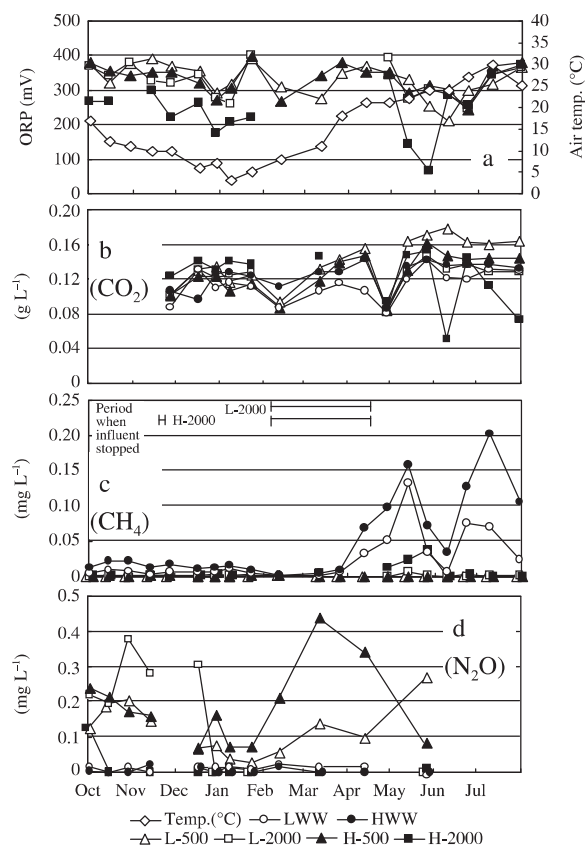


Figure 2 (a) Variations in air temperature and oxidation-reduction potential (ORP) of treated water and concentrations of (b) CO₂, (c) CH₄ and (d) N₂O dissolved in the L-500, L-2000, H-500 and H-2000 treatment waters, and in wastewater containing low (LWW) and high (HWW) levels of contaminants. The period during which the influent stopped because of clogging in the L-2000 and H-2000 treatments is shown in (c).

The air temperature fluctuated from 3°C in February to 30°C in July over the study period. The ORP values ranged from -41 to 377 mV for the wastewater and from 66 to 398 mV for the treated water. The ORP values in the treated water were generally lower in the HWW treatments and in the treatments at higher HLRs (Fig. 2a). The relationship between the air temperature and ORP was not clear. Within the range of observed air temperatures, the atmospheric-equilibrium concentrations (AEC) of CO₂, CH₄ and N₂O in pure water were estimated to be approximately 1.26–0.49 mg L⁻¹ for CO₂, 0.07–0.04 g L⁻¹ for CH₄ and 0.81–0.30 g L⁻¹ for N₂O (at 0–30°C), based on the global mean concentrations of these gases in 2002 (Japan Meteorological Agency 2004).

Dissolved CO₂ concentrations in the water samples ranged from 0.08 to 0.14 mg L⁻¹ for wastewater and from 0.05 to 0.18 mg L⁻¹ for treated water. These concentrations were approximately 100-fold as high as

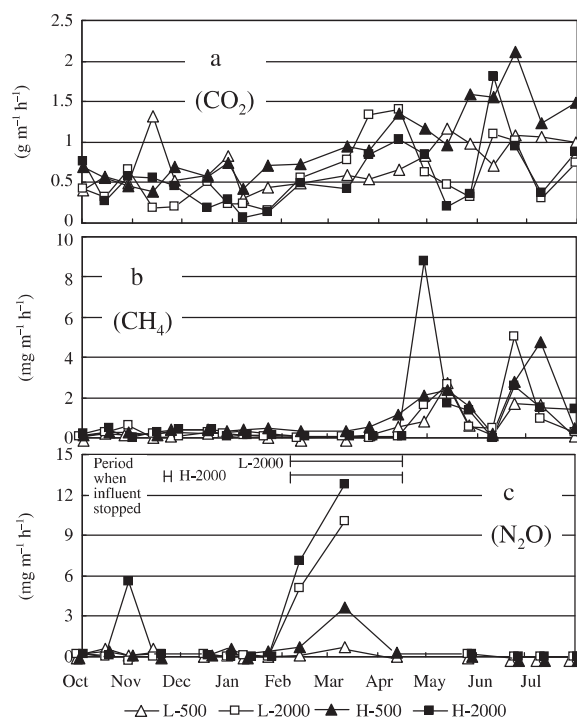


Figure 3 (a) Variations in the emission rates of CO₂, (b) CH₄ and (c) N₂O into the atmosphere during the L-500, L-2000, H-500 and H-2000 treatments. The period during which the influent stopped because of clogging in the L-2000 and H-2000 treatments is shown in (c).

the AEC of CO₂. Unless there was a supersaturation of CO₂ in the wastewater, the concentration appeared to remain constant or even increased during the treatment process in the MSL system. The EA values of CO₂, which ranged from 0.06 to 2.11 mg m⁻² h⁻¹, generally increased when the air temperature increased (Fig. 3a). Although detailed results are not shown in the present paper, EA and air temperature showed a positive correlation in all treatments. The dissolved CO₂ concentration in the L-500 treatment was distinctively higher than that in the LWW treatment from April to July (Fig. 2b), which indicated that CO₂ production in the system increased during the warm period. Although such a trend was not clear for the other treatments, the net CO₂ flux tended to increase with the increase in the air temperature because of the increase of EA. This increase of CO₂ emission into the treated water and the atmosphere corresponded to a natural phenomenon, that is, decomposition of organic matter is higher in warmer periods. Although the influent stopped because of clogging from February to April (Fig. 3a), approximately 0.4–1.4 g m⁻² h⁻¹ of CO₂ was released from the L-2000 and H-2000 treatments through the decomposition of accumulated organic matter in the system.

Dissolved CH₄ concentrations in the treated water were in the range of not detected (ND)–0.04 g L⁻¹ and remained very low compared with those in the wastewater, which ranged from 0.002 to 0.20 g L⁻¹ throughout the experimental period (Fig. 2c). The CH₄ concentration in the wastewater was 5000-fold higher than the AEC (maximum) in July. Because of this high degree of supersaturation, it is possible that dissolved CH₄ was released into the atmosphere as soon as the wastewater was discharged into the MSL system. The EA values of CH₄ (ND–8.81 mg m⁻² h⁻¹) apparently increased from April to July, showing a very similar pattern to that of the CH₄ concentration in LWW and HWW (Fig. 2c,3b). When the wastewater was not discharged because of clogging during the L-2000 and H-2000 treatments, CH₄ was not released from the system (Fig. 3b). Sparrow and Cochran (1995) reported the effect of soil temperature on CH₄ consumption. The temperature of the MSL system probably changed with changes in the air temperature. However, the CH₄ emission from the MSL system did not appear to be influenced by changes in the temperature in the system.

Dissolved N₂O concentrations in the treated water were in the range of ND–0.44 µg L⁻¹ and tended to decrease as the air temperature decreased from October to February (Fig. 2a,d) and then increased when the air temperature increased from February. The dissolved N₂O concentration in the treated water was approximately 1000-fold higher than the AEC (maximum) in March. In contrast, the dissolved N₂O concentrations in the wastewater were in the range of ND–0.02 g L⁻¹ and remained very low throughout the experimental period. The EA values of N₂O ranged from ND to 12.8 mg m⁻² h⁻¹ and were relatively high from February to March, especially when clogging occurred during the L-2000 and H-2000 treatments. Accumulation of organic matter and nitrogen in the system in the cold season and the increase in the air temperature from February to March may have stimulated the denitrification process and N₂O production in the MSL system. Although the ORP value of treated water is often related to the conditions of nitrogen treatment in a MSL system (Masunaga *et al.* 2007), the relationship between the ORP values and N₂O concentrations in the treated water was not clear in the present study.

Characteristics of CO₂, CH₄ and N₂O emissions in relation to HLR

Figure 4 shows the ET and EA values of CO₂, CH₄ and N₂O per 1,000 L of wastewater, while Table 1 shows the net fluxes of the gases in the steady state of the wastewater treatments. Most of the CO₂ released from the system originated from the wastewater (Fig. 4a). Carbon dioxide was released from the system mainly in a dissolved form in the treated water (ET). Although the

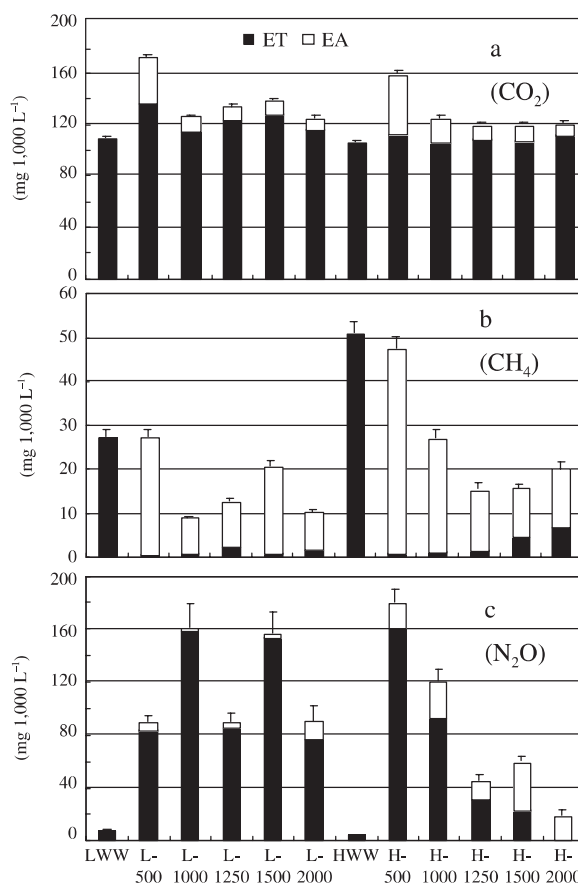


Figure 4 Mean amounts of (a) CO₂, (b) CH₄ and (c) N₂O emitted into the atmosphere (EA) and treated water (ET) per 1,000 L of wastewater. The values for wastewater containing low (LWW) and high (HWW) levels of contaminants correspond to the amounts of gases dissolved in 1,000 L of wastewater. Vertical bars indicate the standard errors of the total values (EA + ET) during the experiment.

ET values of CO₂ per 1,000 L of wastewater did not change appreciably with the HLRs, the EA values were significantly higher in the L-500 and H-500 treatments. Therefore, the total CO₂ emission per 1,000 L of wastewater was also high in the L-500 and H-500 treatments. Although HWW exhibited a higher BOD, no significant differences between the LWL and HWW treatments were observed (Fig. 4a). If the percentages of organic matter decomposition were similar in all the HLR treatments, net CO₂ emission should have increased with the increase in HLR. However, such a trend was not observed (Table 1). The net CO₂ flux was the highest in the L-1500 treatment. Although the amount of wastewater discharged in the L-1500 treatment was threefold as high as that in the L-500 treatment, the flux of L-1500 was only 1.5-fold higher than that of L-500. The CO₂ emission in the H-500, H-1000, H-1250 and H-1500

Table 1 Net fluxes (mean \pm standard error) of CO₂, CH₄ and N₂O from the systems in the steady state of the wastewater treatments[†]

| Loading rate | CO ₂ (g m ⁻² day ⁻¹) | CH ₄ [‡] (mg m ⁻² day ⁻¹) | N ₂ O (mg m ⁻² d ⁻¹) |
|--------------|---|---|---|
| L-500 | 31.2 \pm 0.6 | 0.02 \pm 0.2 | 41 \pm 3 |
| L-1000 | 21.2 \pm 0.9 | -11.1 \pm 1.9 | 168 \pm 27 |
| L-1250 | 32.5 \pm 0.9 | -18.4 \pm 1.8 | 105 \pm 9 |
| L-1500 | 48.0 \pm 0.6 | -8.3 \pm 1.7 | 246 \pm 34 |
| L-2000 | 31.0 \pm 1.1 | -36.3 \pm 6.4 | 191 \pm 33 |
| H-500 | 26.7 \pm 0.7 | -1.7 \pm 0.7 | 90 \pm 6 |
| H-1000 | 26.3 \pm 1.3 | -28.6 \pm 3.4 | 136 \pm 19 |
| H-1250 | 27.6 \pm 1.3 | -24.1 \pm 5.2 | 51 \pm 6 |
| H-1500 | 28.8 \pm 1.2 | -33.5 \pm 6.3 | 46 \pm 7 |
| H-2000 | 36.6 \pm 0.7 | -66.4 \pm 22.4 | 68 \pm 27 |

[†]Steady state corresponds to the period of flow excluding the period of clogging. [‡]Negative values indicate the gas consumption in the system.

treatments was not appreciably different and was relatively low compared with that in the LWW treatments. These results indicated that the efficiency of organic matter decomposition was higher at lower HLRs and in the LWW treatments. The results also suggested that some part of the organic matter loaded into the system remained undecomposed and accumulated in the system, especially during the cold period when the CO₂ emission was relatively low (Fig. 2b,3a). This was probably related to the occurrence of clogging in the MSL system. In the HWW treatments, clogging occurred earlier than in the LWW treatments. In the H-2000 treatment, clogging occurred after 4 months (November), while in the L-2000, H-1000, H-1250 and H-1500 treatments clogging occurred after 7 months (March), and in the L-1250 treatment clogging occurred after 9 months (May). In the other treatments, clogging did not occur throughout the experimental period (Masunaga *et al.* 2007).

Methane emission from the system was lower than the amount originally contained in LWW and HWW, except for the L-500 and H-500 treatments, and was mainly released as EA (Fig. 4b), indicating that CH₄ was consumed in the MSL system, based on the net emission (Table 1). Although it has been reported that N fertilization frequently reduced CH₄ consumption in soil (King and Schnell 1994; Mosier *et al.* 1991), N supply into the MSL system through LWW and HWW discharge did not appear to affect CH₄ consumption in the system. Kong *et al.* (2002) reported that the CH₄ emission from a soil trench system increased with a decrease in ORP in the system when the duration of the treatment period increased. In the MSL system, the ORP value of treated water generally remained above 250 mV (Fig. 2a). Although ORP inside the MSL system was not directly measured, it was assumed that the system remained under aerobic

and oxidative conditions. The high ORP value appeared to contribute to efficient CH₄ oxidation. However, in the L-500 and H-500 treatments, the emission was relatively high (Fig. 3b), although the net fluxes were almost zero (Table 1), presumably because the gas exchanges between the system and the atmosphere were more frequent in the lower HLR treatment. In the MSL system, the water content increased when the HLR increased (Fig. 2; Masunaga *et al.* 2007), and a greater amount of organic matter remained undecomposed and accumulated at higher HLRs. The accumulation of water or organic matter in the system possibly closed the pore spaces in the water permeable layer, especially in the top layer where most of the suspended solid (SS) was removed. This may have prevented gas exchanges between the system and the atmosphere at HLRs above 1,000 L m⁻² day⁻¹. Methane gas released from the wastewater into the gas phase inside the MSL system might have remained for a longer period of time because of the less frequent gas exchanges, and it was efficiently oxidized in the gas phase inside the MSL system. Although the ET values tended to increase at higher HLRs, especially in the HWW treatments, probably because of the shorter water retention time, the CH₄ flux was clearly lower at HLRs above 1,000 L m⁻² day⁻¹ compared with the values in the L-500 and H-500 treatments (Fig. 4b).

Nitrous oxide was mainly released from the system as ET in all the treatments, except for the H-1500 and H-2000 treatments. The EA values in the HWW treatments were higher than those in the LWW treatments. The emission per 1,000 L of wastewater tended to decrease as HLR increased in the HWW treatments, while it was not affected by the HLR changes in the LWW treatments. Net N₂O flux was relatively high at high HLRs in the LWW treatments compared with the HWW treatments at high HLRs. Nitrous oxide is produced by both nitrification and denitrification processes. In the wastewater treatments of the MSL system, the HWW treatments, especially at higher HLRs, led to a lower N₂O emission (Fig. 4c), which might result from the lower ORP values in the system (Table 1; Masunaga *et al.* 2007). The conditions in the HWW treatments at higher HLRs probably led to a decrease of N₂O production by the enhancement of the reduction of N₂O to N₂ in the denitrification process. The ET values of N₂O were significantly reduced in the H-1250 H-1500 and H-2000 treatments (Fig. 4c).

Nitrogen removal characteristics and N₂O emission

Table 2 shows the nitrogen removal efficiency and the N₂O conversion rate in each treatment. Nitrogen removal rates tended to be high at high HLRs and in the HWW treatments. The N₂O emission was also higher at higher

Table 2 Characteristics of nitrogen treatment and N₂O conversion rates in the multi-soil-layering system

| | N loading rate (g N m ⁻² day ⁻¹) | N removal (%) | Net N ₂ O flux (g N m ⁻² day ⁻¹) | N ₂ O conversion rate (%) |
|--------|--|------------------|---|---|
| L-500 | 3.4 | 31 | 0.04(0.01–0.09) | 1.10(0.19–2.63) |
| L-1000 | 6.9 | 52 | 0.13(0.001–0.74) | 1.79(0.02–9.50) |
| L-1250 | 8.6 | 49 | 0.09(0.004–0.20) | 1.16(0.05–2.61) |
| L-1500 | 10.3 | 41 | 0.18(0.002–0.93) | 1.78(0.02–8.03) |
| L-2000 | 13.7 | 65 | 0.13(0.001–0.39) | 1.17(0.01–3.56) |
| H-500 | 5.0 | 41 | 0.07(0.02–0.20) | 1.95(0.40–9.25) |
| H-1000 | 10.0 | 67 | 0.08(ND-0.25) | 1.08(0–4.25) |
| H-1250 | 13.3 | 76 | 0.04(ND-0.15) | 0.28(0–1.08) |
| H-1500 | 16.0 | 67 | 0.03(ND-0.11) | 0.19(0–0.66) |

HLRs for the LWW treatments, while in the HWW treatments, the opposite was recorded and the release of N₂O was lower than that in the LWW treatments. Mean N₂O conversion rates were relatively high in the LWW treatments and ranged from 1.10 to 1.79%, while in the HWW treatments they ranged from 0.19 to 1.95%. These values were relatively low compared with those of the N₂O conversion rates, 2.4–4.0%, in the soil trench system (Kong *et al.* 2002). Kimochi *et al.* (1998) reported that N₂O emission increased at a low carbon substrate/nitrogen ratio in the influent in the sewage treatment process. The ratio of carbon substrate and nitrogen in wastewater also affected the nitrogen removal efficiency in the MSL system. The mean Chemical Oxygen Demand (COD)/T-N ratio of HWW, which was approximately 12.6, was higher than that of LWW (7.0) (Masunaga *et al.* 2007). The HWW treatments led to a higher N removal efficiency. This phenomenon is in agreement with previous results showing that treatment of wastewater with a higher COD/T-N ratio led to a higher T-N removal efficiency in a MSL system where domestic wastewater with COD/T-N ratios of 1 to 6 was treated (Luanmanee *et al.* 2001). In addition, the HWW treatments resulted in a lower N₂O emission compared with the LWW treatments (Table 2). It is likely that the higher availability of carbon substrates and lower ORP conditions, namely anaerobic conditions, in the HWW treatments led to the enhancement of the denitrification processes and the reduction of N₂O to N₂. In terms of nitrogen removal, the H-1250, H-1500 and H-2000 treatments removed more than 63% of T-N from the wastewater with average N₂O conversion rates of 0.19–0.34%. These treatments enhanced the nitrogen removal efficiency and decreased N₂O emissions.

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