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Methane oxidation in water-spreading and compost biofilters

This study evaluated two biofilter designs to mitigate methane emissions from landfill vents. Water-spreading biofilters were designed to use the capillarity of coarse sand overlain by a finer sand to increase the active depth for methane oxidation. Compost biofilters consisted of 238-L barrels containing a 1 : 1 mixture (by volume) of compost to expanded polystyrene pellets. Two replicates of each type of biofilter were tested at an outdoor facility. Gas inflow consisted of an approximately 1 : 1 mixture (by volume) of CH_4 and CO_2. Methane output rates (J_{out}; g m^{-2} day^{-1}) were measured using the static chamber technique and the Pedersen *et al.* (2001) diffusion model. Methane oxidation rate (J_{ox}; g m^{-2} day^{-1}) and fraction of methane oxidized (f_{ox}) were determined by mass balance. For methane inflow rates (J_{in}) between 250 and 500 g m^{-2} day^{-1}, the compost biofilter J_{ox}, 242 g m^{-2} day^{-1}, was not significantly different ($P = 0.0647$) than the water-spreading biofilter J_{ox}, 203 g m^{-2} day^{-1}; and the compost f_{ox}, 69%, was not significantly different ($P = 0.7354$) than water-spreading f_{ox}, 63%. The water-spreading biofilter was shown to generally perform as well as the compost biofilter, and it may be easier to implement at a landfill and require less maintenance.

David K. Powelson
Jeffery Chanton
Department of Oceanography, Florida State University, Tallahassee, Florida, USA

Tarek Abichou
Jose Morales
Department of Civil and Environmental Engineering, The Florida A&M University and the Florida State University College of Engineering, Tallahassee, Florida, USA

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Corresponding author: David K. Powelson, Department of Oceanography, 0102 OSB, West Call Street, Florida State University, Tallahassee, Florida 32306, USA.
Tel: 850 645 6925; fax: 850 644 2581
e-mail: powelson@eng.fsu.edu

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Introduction

Methane has 62 times the global warming potential of carbon dioxide on a mass basis over the next 20 years (Ramaswamy *et al.* 2001). Biofilters can be used to oxidize methane emissions from landfill vents and thereby reduce flux of this greenhouse gas to the atmosphere. High emissions may be used for power generation or flared, but these options are not economically feasible for low flows from smaller or older landfills. In these cases bacteria can be used to oxidize methane and other organic gases as well (Humer & Lechner 1999, Scheutz & Kjeldsen 2003).

There are hundreds of methanotrophic bacterial species, and they are found in nearly all samples taken from soils, oceans, and waste materials (Hanson & Hanson 1996). Except for bacterial consortia that reduce sulphate, all methane oxidizers are aerobic (Hanson & Hanson 1996). Methanotrophic bacteria are most common at the interface between aerobic and anaerobic, methanogenic environments (Ehrlich 1996). Researchers have studied a variety of biofilter materials in columns with passive diffusion of oxygen from the surface, and found maximum methane oxidation rates (J_{ox}) ranging from 54 to 500 g m^{-2} day^{-1} and fraction of methane oxidized (f_{ox}) ranging from 47 to 100% (Table 1). Methanotrophic bacteria need oxygen and water, but uniform porous media are likely to become too dry at the top and too

Table 1: Passively-aerated methane biofilter and column studies.

Biofilter material	Methane oxidation		Reference
	Maximum rate ($g\ m^{-2}\ day^{-1}$)	Fraction (%)	
Coarse sand	166	61	Kightley *et al.* 1995
Compost	168	100	Humer and Lechner 1999
Agricultural soil	171	82	De Visscher *et al.* 1999
Landfill cover soil, sandy loam	290	79	De Visscher *et al.* 1999
Landfill cover soil, sandy loam	125	47	Hilger *et al.* 2000
Loamy sand	435	83	Park *et al.* 2002
Loamy sand	210	81	Scheutz & Kjeldsen 2003
Leaf compost	500	95	Wilshusen *et al.* 2004
Compost and sand	54	98	Berger *et al.* 2005
Coarse sand	267	72	This study
Compost and polystyrene pellets	423	58	This study

wet at the bottom, thereby restricting the zone for effective methane oxidation. Volumetric water content (θ_v) within the range 0.025 to 0.20 was not found to influence methane oxidation rate (Gebert *et al.* 2003). However, the methane 'oxidation horizon' in uniform media has been observed to be just 15 cm thick (Humer & Lechner 1999, Wilshusen *et al.* 2004). This narrow zone may be bypassed or become clogged with exopolymeric substances (EPS), thereby reducing biofilter efficiency (Wilshusen *et al.* 2004, Huber-Humer 2005).

The water-spreading biofilter was designed to improve the distribution of air and water for methane oxidation. The principle of the water-spreading biofilter is to use water tension and adhesion (capillarity) in a fining-upward gradient of soil pores to resist the pull of gravity on soil water. This arrangement increases θ_v near the top and gas porosity near the bottom, in comparison with a uniform media. A sharp transition between large pores and small pores should be avoided, however, to prevent formation of a nearly saturated layer at the bottom of the finer sand that blocks oxygen diffusion (also known as a 'capillary barrier'; Berger *et al.* 2005). A water-spreading biofilter installed at a landfill would look like a rounded ridge that extends for whatever length is required to treat the amount of methane coming from a pipe. This design has other potential advantages over a contained, compost-based biofilter: (1) the ridge design would allow oxygen penetration from the sides as well as the top; (2) construction may be less expensive because it does not require a containment structure; (3) it can be made to look like a natural feature that blends with the final landfill landscaping; (4) the sand media will not decompose and settle like compost; and (5) it can be used in areas where compost is limited or regulations limit its use. It is also possible that changes in speed or direction of the wind would change the location of EPS production in the ridge design, thereby reducing pore clogging. The water-spreading design might also be incorporated in alternative landfill cover designs known as capillary barrier covers (Berger *et al.* 2005).

The purpose of this research was to develop a simple and reliable biofilter that would expand the habitat for methanotrophic bacteria to oxidize methane emissions from vents in municipal solid waste landfills. The objective was to compare the water-spreading biofilter design with a compost biofilter design. Methane oxidation rates and fraction oxidized were determined by mass balance from replicate biofilters exposed to the weather and fed with artificial landfill gas.

Materials and methods

Biofilter construction

Two replicates of each biofilter design were set up at an outdoor facility in Tallahassee, Florida, USA. This location permitted accurate measurements of inflow and outflow of methane, while exposing the biofilters to the weather. The gas supply and metering equipment were housed in an adjacent instrument shed.

Compost biofilters ('N' and 'S') were constructed from 238-L plastic barrels (90 cm high, 58 cm in diameter; Figure 1). The gas-inflow tube was connected to a slotted pipe that was covered with 16 cm of gravel to form the gas dispersion layer. The biofilter media consisted of expanded polystyrene pellets (also known as packing peanuts, 7 mL each) and compost. The compost was made at the local landfill from chipped yard waste (mostly leaves and branches), where it was windrowed on a closed section of the landfill for 4 years. The pH of compost in distilled water was 7.5. The pellets formed the structure of the media and compost filled the space between the pellets, thereby maximizing the amount of compost and minimizing settling as the compost decomposed. The porosity of

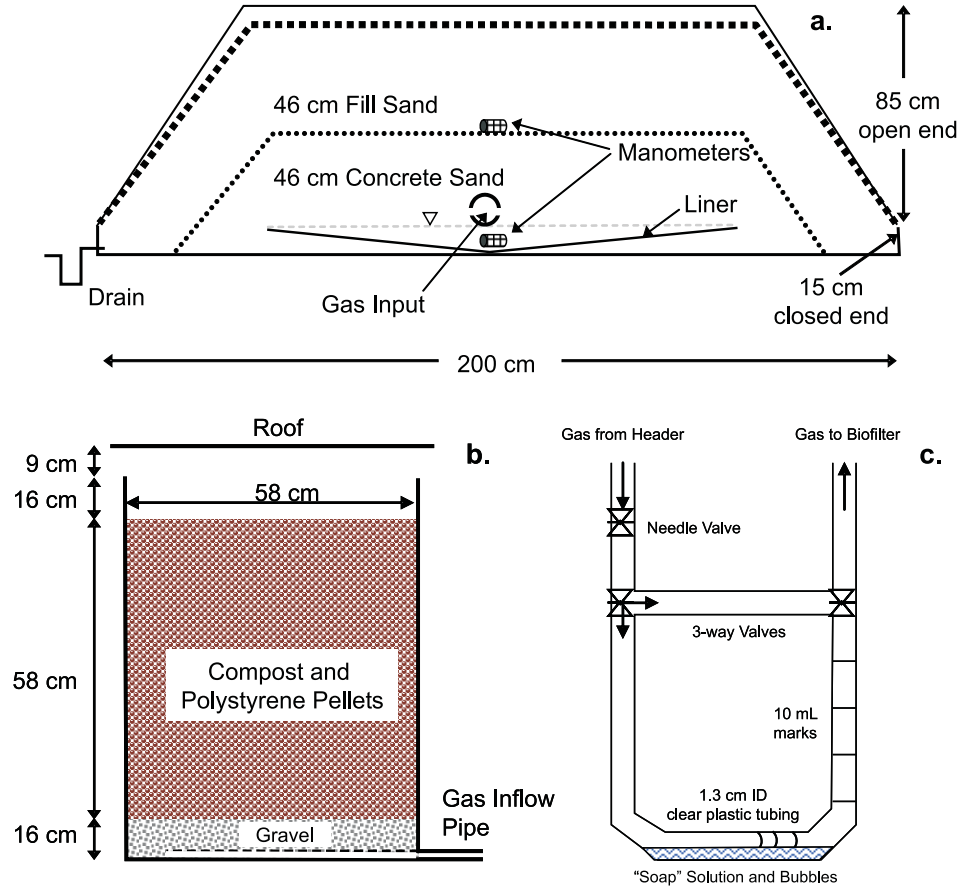

Fig. 1: Schematic cross-section diagrams of a water-spreading biofilter (a) and a compost biofilter (b). The water-spreading biofilter gas input and manometer tubing and frame struts are not shown for clarity. Inflow CO_2 and CH_4 were mixed, humidified, divided at a header, and sent to flow control and measurement devices (c), one for each biofilter. Flow measurements were taken by directing flow through a soap-film flow meter with three-way valves.

loosely packed pellets was 50%; consequently the compost biofilter medium consisted of equal volumes of expanded polystyrene pellets and compost. The compost biofilters had roofs supported by legs that shed rain but allowed free air circulation to the top of the biofilters. The roof was required in this climate to prevent the compost from becoming too wet for good oxygen penetration. During flux measurements the roof was replaced by the barrel lid that sealed the top except for a 4.3 mm inside diameter sampling tube in the centre.

The water-spreading biofilters consisted of two 25-cm-thick sections of the ridge design ('W' and 'E'). The sections were encased in open-topped frames of plywood [pressure-treated with ACQ (alkaline copper and quaternary ammonium compounds)]. The overall internal dimensions were 2 m wide by 1 m tall by 0.25 m deep (Figure 1). A water-filled tube was used to drain the bottom without gas leakage. During flux measurements plywood sheets were clamped to the top and sides of the trapezoidal frame, sealing the biofilter except for a 4.3-mm inside diameter sampling tube in the centre of the top sheet.

Two coarse sands, obtained from local suppliers, were used in the water-spreading biofilters: concrete sand for the bottom layer and fill sand for the top layer. The concrete sand was coarser, having 80% of its mass larger than 0.425 mm, in comparison with 60% for the fill sand (Table 2). The concrete sand had a pH = 7.6 and the fill sand had a pH = 8.0 in distilled water; the high pH was probably due to bits of shell

Table 2: Water-spreading biofilter sand characteristics.

	Concrete sand	Fill sand
Particle size (mm)	——mass %——	
> 2.000	3.14	0.26
2.000–1.180	8.68	0.38
1.180–0.425	68.02	59.81
0.425–0.250	12.32	32.84
0.250–0.106	7.65	6.20
0.106–0.075	0.14	0.24
< 0.075	0.05	0.27
	——pH——	
pH	7.6	8.0
	–dry bulk density (g cm^{-3})–	
Soil moisture curves	1.53	1.63
Biofilters	1.40 ± 0.03	1.41 ± 0.02

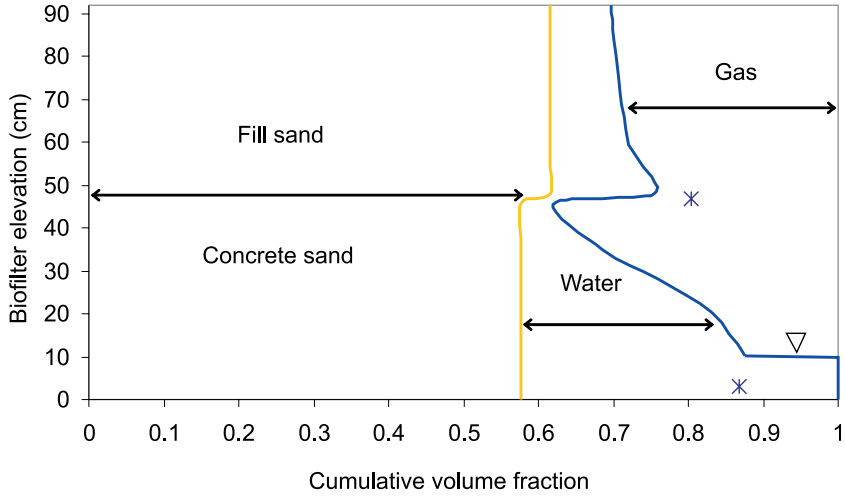

Fig. 2: Predicted volume fractions of solid, water, and gas for vertical elevations in the water-spreading biofilters. The gas distribution pipe was centered at 16 cm, and the boundary between the sand types was at 46 cm. The observed water contents (plus the solid fraction), calculated from average manometer readings over the course of the experiment, are indicated by the * symbol.

and limestone in the source material. The relationship between θ_v and pressure head (h_p, cm H_2O) for each of the sands was quantified by soil moisture release curves. Curves for h_p from 0 to –110 cm for both sands packed to a bulk density similar to that expected in the biofilter were determined using the hanging water column method (Dane & Hopmans 2002). In static conditions $-h_p$ for a volume of sand is equivalent to its height above a water table. Although static conditions would rarely occur in the field, where precipitation, evaporation, and water generated during methane oxidation affect the relationship between elevation and h_p, the use of static conditions for design planning is reasonable.

The predicted volume fractions of solid, water, and gas in the water-spreading biofilters are depicted in Figure 2. The solid volumes were calculated from the bulk densities and by assuming a particle density of 2.65 g cm^{-3}. Water volumes were calculated assuming a water table at 10 cm and no flow (static) conditions. The appropriate section of the soil moisture curves (h_p = 0 to –47 for concrete sand and h_p = –47 to –92 for fill sand) were used to form the curve between water and gases. Note that the concrete sand had some large pores that drained at a slight negative pressure so that just above the water table θ_v dropped from 0.42 to 0.30. In the concrete sand θ_v decreased sharply with elevation so that at 46 cm height θ_v = 0.04, but the finer fill sand above brought θ_v up to 0.14 at 48 cm. At the top (elevation 92 cm) θ_v = 0.08, whereas if the entire ridge had been concrete sand θ_v would have been 0.02.

Concrete sand was placed in the bottom of the biofilter frames to make a bed for the liner. The liner consisted of a double thickness of 0.2 mm polyethylene sheeting formed into a shallow pan, 150 cm wide, with overflow 10 cm above the bottom. The overflow elevation of the liner set a maximum height of the water table. The liner was not a reservoir, however, because horizontal pressure gradients 'wick' water over the edge of the liner, causing the water table to fall unless the drainage is replaced by rainfall or water from methane oxidation. Two centimetres of concrete sand was placed over the liner and the lower manometer was installed. Manometers consisted of water-filled 1-bar porous ceramic cups (Soilmoisture Corp., Santa Barbara, CA, USA) and clear plastic tubing (Young & Sisson 2002). More concrete sand was added to the 10 cm level. The gas input pipe was installed, which consisted of an 11-cm-diameter, corrugated and perforated plastic pipe wrapped in aluminium window screen. The gas input tube was inserted into the pipe. Concrete sand was added to the 46 cm level. The upper manometer was installed and fill sand was added to the 92 cm level. All tubing exited the biofilter from sealed ports.

Each water-spreading biofilter was inoculated with 7 L of water that was likely to contain methanotrophic bacteria. The water came from shallow puddles that formed after rain on the local landfill cover, and where gas was observed to bubble from the cover. A nutrient solution was added to the water-spreading biofilters to compensate for the nutrients that landfill gas and dust would provide in a landfill environment. The solution was made from 1.85 g L^{-1} of Peters Orchid Food (Spectrum Brands, Atlanta, GA, USA), and consisted of (mg/L): N 536, P 81, K 154, Mg 9, Fe 2, Cu 0.9, Mn 0.9, Zn 0.9, B 0.4, and Mo 0.009. The surface of each water-spreading biofilter received 1.9 L $week^{-1}$ of the nutrient solution.

Gas mixing and input

The synthetic landfill gas was a mixture of CH_4 and CO_2. Flows from compressed gas tanks were first approximately equalized using regulators, needle valves, and rotameters. The

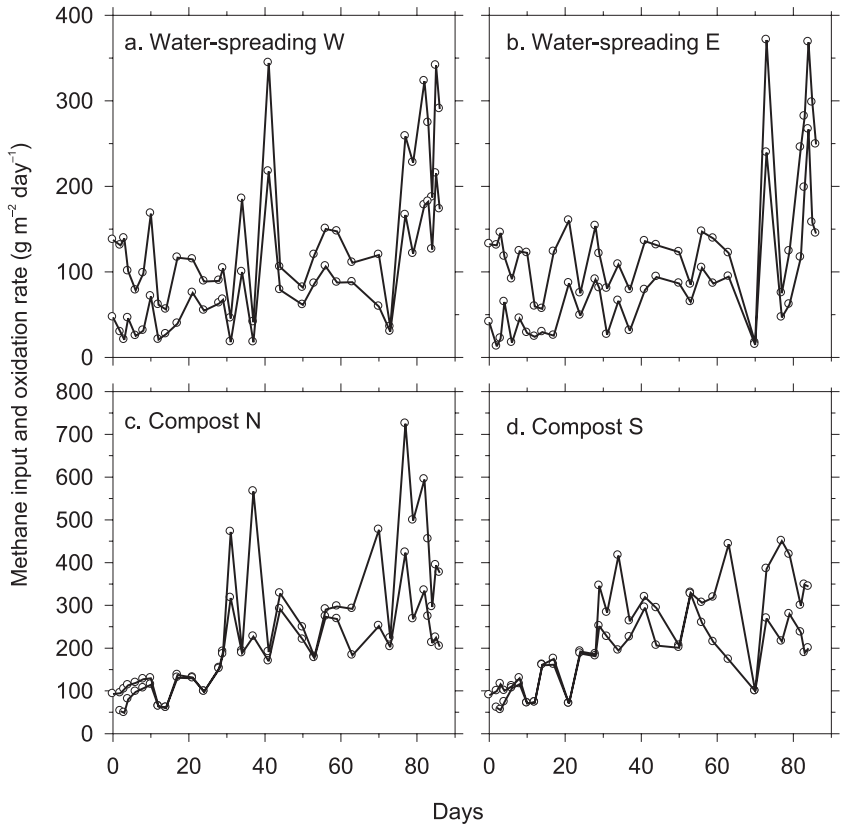

Fig. 3: Methane input (J_{in}, upper curves) and oxidation rate (J_{ox}, lower curves) for the four experimental biofilters.

gases were then combined and bubbled through a humidification flask where samples were taken to determine the actual inflow volumetric fraction of methane, f_v (average $f_v = 53 \pm 2\%$; the mean ± standard error of the mean will be used throughout this paper). The gas then flowed through a header that divided it between four sets of valves and soap-film flow meters (Figure 1c). The soap-film flow meters were a loop of 1.3 cm inside diameter clear plastic tubing that had been calibrated in 10 mL increments and contained a small amount of detergent solution. The amount of solution was not enough to restrict gas flow, but when the bottom of the loop was shaken detergent films were generated that spanned the tubing and were timed as they flowed from one mark to the next. The four gas streams then flowed to the biofilters.

The aluminium shed that housed the metering equipment was hot during the day, affecting the regulators and valves, and this resulted in considerable flow variability from day to day (Figure 3). It was necessary to ensure that inflow rate was nearly constant for several hours before flux measurements were taken. We were confident that flow overnight was adequately constant, although flow measurements were not taken, because the flow controls were not affected by solar heating. Consequently all measurements and sampling were done in the morning before sunrise. Although we had initially planned inflow to be constant, the variability obtained is more representative of actual landfill conditions. Inflow was intentionally increased to the compost biofilters after day 29 because the methane flux out (J_{out}, defined below) was very low from day 12 to 29 and a higher flow rate was needed to challenge these biofilters. Inflow to the water-spreading biofilters was increased on day 77 to more closely match that of the compost biofilters. The experiment lasted 86 days (9 August to 3 November 2005).

The inflow flux, J_{in} (g m^{-2} day^{-1}) was calculated by:

$$J_{in} = Q f_v M U_i P(A R K)^{-1} \quad (1)$$

where Q is the synthetic landfill gas flow rate (mL min^{-1}), f_v is the volumetric fraction of methane, M is the molar mass of methane (16 g mol^{-1}), U_i is the units conversion factor (1.44 L min(mL day)$^{-1}$), P is pressure at the flow meter (atm), A is the area covered by the biofilter (0.264 m^2 for the barrels, 0.500 m^2 for the water-spreading sections), R is the gas constant (0.08205 L atm(K mol)$^{-1}$), and K is air temperature (Kelvin).

Methane outflow and oxidation analysis

Flux measurements were made using 60-mL syringes fitted with stopcocks to take a series of gas samples from the head-

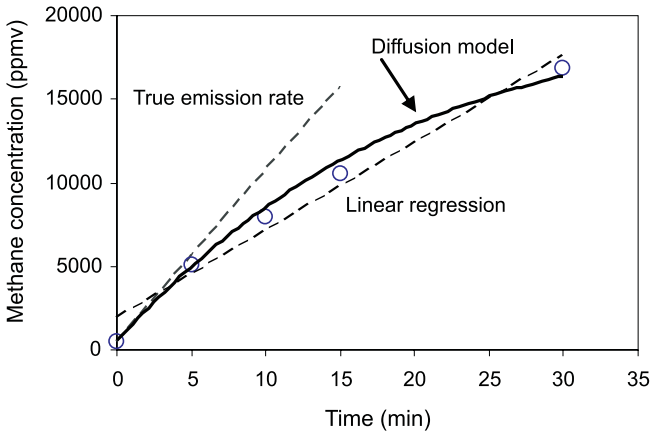

Fig. 4: An example of high out-flux data fit by the linear regression (d*C*/d*t* = 524 ppm/min) and the diffusion model (initial d*C*/d*t* = 1021 ppm/min), and the projected true emission (d*C*/d*t* = 1021 ppm/min).

space above the biofilters over a 25 to 30 min period. A flux measurement was started by covering the top of a biofilter, thereby sealing it except for the sampling tube. The sampling tube remained open to the atmosphere except when a sample was taken. This procedure is equivalent to the 'non-steady state' or 'static' chamber method used in the field (Hutchinson & Livingston 2002). Methane concentrations were quantified within 2 h using a Shimadzu 14A gas chromatograph with a flame ionization detector and Carbosphere column. Scott Specialty gases were used as standards.

Methane flux out of the biofilters, J_{out} (g m^{-2} day^{-1}), was determined from the change in concentration, C (ppmv), with time, t (min) calculated by:

$$J_{out} = (dC/dt)\, M\, V\, U_o\, P(A\, R\, K)^{-1} \quad (2)$$

where U_o is the units conversion factor (0.00144 L min(μL day)$^{-1}$) and V is the headspace volume (L). The rate d*C*/d*t* usually declined with time as methane built up in the headspace due to a declining concentration gradient. Consequently, it was necessary to estimate d*C*/d*t* at time zero. Pedersen *et al.* (2001) developed a stochastic diffusion model to find this initial rate. Their diffusion model does much better than linear regression in projecting the initial emission rate (Figure 4). When J_{out} was relatively low, d*C*/d*t* was nearly constant and the diffusion model estimation failed. In these cases linear regression was used to find d*C*/d*t*.

The methane oxidation rate, J_{ox} (g m^{-2} day^{-1}), is:

$$J_{ox} = J_{in} - J_{out} \quad (3)$$

The fraction of methane oxidized, f_{ox}, is:

$$f_{ox} = J_{ox} J_{in}^{-1} \quad (4)$$

The oxidation rate was not used to compare biofilter performance over the whole experiment because oxidation rate tended to parallel J_{in} and there was considerable variability in J_{in} (Figure 3). However, for a common range in J_{in} a two-way analysis of variance was used to compare J_{ox} and f_{ox}. The fraction oxidized was the primary variable used to compare biofilters using a repeated measures analysis of variance. Biofilter performance was better compared by using f_{ox} because the covariance of J_{ox} and J_{in} was normalized by using the ratio of the two [equation (4)]. There was a slight relationship between f_{ox} and J_{in} for the compost biofilters that will be discussed below. SAS statistical software was used for these analyses (SAS Institute 2005).

Results and discussion

Methane oxidation

The compost biofilters initially performed better than the water-spreading biofilters. From day 1 to day 29, f_{ox} in the compost biofilters ranged from 47 to 99%, and averaged 87 ± 4%. In the same period f_{ox} in the water-spreading biofilters ranged from 10 to 71% and averaged 41 ± 4% (Figure 5). After day 29 the compost biofilter f_{ox} declined, and the water-spreading biofilter f_{ox} continued to increase, but the compost biofilters still performed better until day 63. The compost biofilters initially performed well probably because the compost had a well-developed methanotropic community due to exposure to landfill gas while the compost aged. The reason for the decline in compost biofilter performance is less clear – EPS may have reduced gas penetration into active regions of the compost (Huber-Humer 2005). However, there was no change in flow-meter pressure over the course of the study, indicating that there was no important change in gas porosity. The increase in f_{ox} for the water-spreading biofilters was probably due to the development of a methanotrophic community in the sand media. It might be possible to hasten this development, and eliminate the need for inoculation and addition of fertilizer, by covering water-spreading biofilters with a thin layer of compost. For the overall experiment (29 sampling dates) the repeated measures analysis of variance showed that compost biofilters had a significantly greater f_{ox} ($P = 0.0022$), however the difference changed with time ($P < 0.001$). From day 0 to day 59 (21 sampling dates) the compost biofilters had greater f_{ox} ($P = 0.0022$), but from day 63 to day 86 (eight sampling dates) there was no significant difference in f_{ox} ($P = 0.5852$). For days 63 to 86 the average f_{ox} was 64 ± 3% for the water-spreading biofilters and 63 ± 4% for the compost biofilters.

The maximum J_{ox} and f_{ox} are listed in Table 1, but these are single observations. A better measure of performance may be the comparison of J_{ox} and f_{ox} for the same range in J_{in}. In

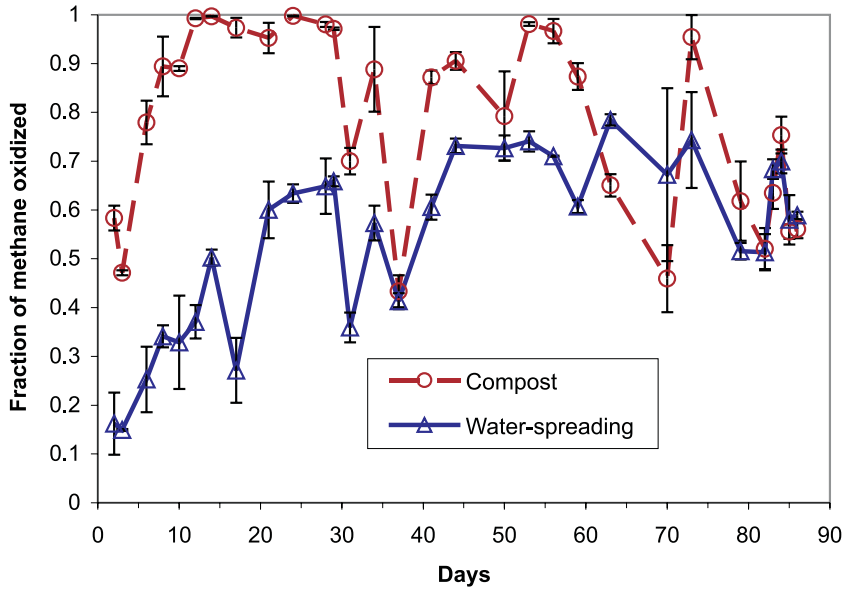

Fig. 5: Average fraction of methane oxidized (f_{ox}, ± standard error) in the compost and water-spreading biofilters.

the range 250 to 500 g m^{-2} day^{-1} there was no significant difference in J_{in} (P = 0.254; compost average J_{in} = 361 g m^{-2} day^{-1} and water-spreading average J_{in} = 321 g m^{-2} day^{-1}). Consequently, data from this range were used in two-way analysis of variance, in which biofilter identity and type were class variables, and J_{ox} and f_{ox} were response variables. For this analysis each measurement was assumed to be an independent observation and not correlated with time. The F-value for J_{ox} was 2.66 (df = 3,32 , P = 0.0647), indicating that the compost average, 242 g m^{-2} d^{-1}, was not significantly different than the water-spreading average, 203 g m^{-2} d^{-1}. The F-value for f_{ox} was 0.43 (df = 3,32, P = 0.7354), indicating that the compost average, 69%, was not significantly different from the water-spreading average, 63%.

Comparison of the biofilters described in this study with others listed in Table 1 is difficult because of variable methods and presentation of results. However, if steady-state J_{ox} and f_{ox} of other studies are compared with this study for J_{in} between 250 and 500 g m^{-2} day^{-1}, it appears that the biofilters had similar performance. For example, the leaf compost biofilter of Wilshusen *et al.* (2004) had average J_{ox} = 360 g m^{-2} day^{-1} and f_{ox} = 69%, which is similar to our compost biofilters (J_{ox} = 242 g m^{-2} day^{-1}, f_{ox} = 69%); and the coarse sand of Kightley *et al.* (1995) had average J_{ox} = 166 g m^{-2} day^{-1} and f_{ox} = 61%, which is similar to our water-spreading biofilters (J_{ox} = 203 g m^{-2} day^{-1}, f_{ox} = 63%).

The oxidation rate, J_{ox}, increased with J_{in} for both types of biofilters (Figure 6). The water-spreading biofilters showed a linear trend (J_{ox} = 0.623 J_{in}, r^2 = 0.944), whereas the compost biofilters showed a slightly declining rate of increase (J_{ox} = 4.24 $J_{in}^{0.687}$, r^2 = 0.855). By substituting these results for J_{ox} in equation (4), f_{ox} can be inferred to a constant (f_{ox} = 0.623) for the water-spreading biofilters with J_{in} between 17 and 371 g m^{-2} day^{-1}. The constant f_{ox} was surprising because we expected that low J_{in} would result in nearly complete oxidation of methane. For the compost biofilters f_{ox} was a function of J_{in} (f_{ox} = 4.24 $J_{in}^{-0.313}$), and 100% oxidation can be expected for J_{in} up to 101 g m^{-2} day^{-1}. The maximum oxidation capacity was not approached for either biofilter, however, since J_{ox} was still steeply increasing at the maximum J_{in} tested (371 g m^{-2} day^{-1} for the water-spreading biofilters and 725 g m^{-2} day^{-1} for the compost biofilters).

Water distribution in the water-spreading biofilters

The manometers responded to precipitation, but quickly returned to their usual values: lower $h_p \approx -2$ cm, and upper $h_p \approx -26$ cm. We had expected that the liner would remain full of water and the lower manometer h_p = +7 cm. Instead, the lower h_p = –3.0 ± 0.7 cm (W) and –0.6 ± 0.7 cm (E). The lower manometer pressures indicate that although the sand remained very wet, it was usually unsaturated (calculated average θ_v = 0.29 and gas porosity = 0.13). This may have been due to unusually hot and dry weather during the experimental period – the average temperature was 1°C higher than the normal of 24.8°C, and there was 10 cm less precipitation than the normal 34 cm. The upper manometers had higher pressure than static conditions (static upper h_p = –49 cm when lower h_p = –2 cm). Over the course of the experiment the upper h_p = –27.4 ± 0.4 cm (W) and h_p = –24.6 ± 0.6 cm (E), indicating that water always had a downward gradient. Although rainfall and the nutrient solution contributed water to the surface, the gradient was maintained for weeks with no rain. This indicates that water produced by methane oxidation, and the restriction in hydraulic conductivity across the

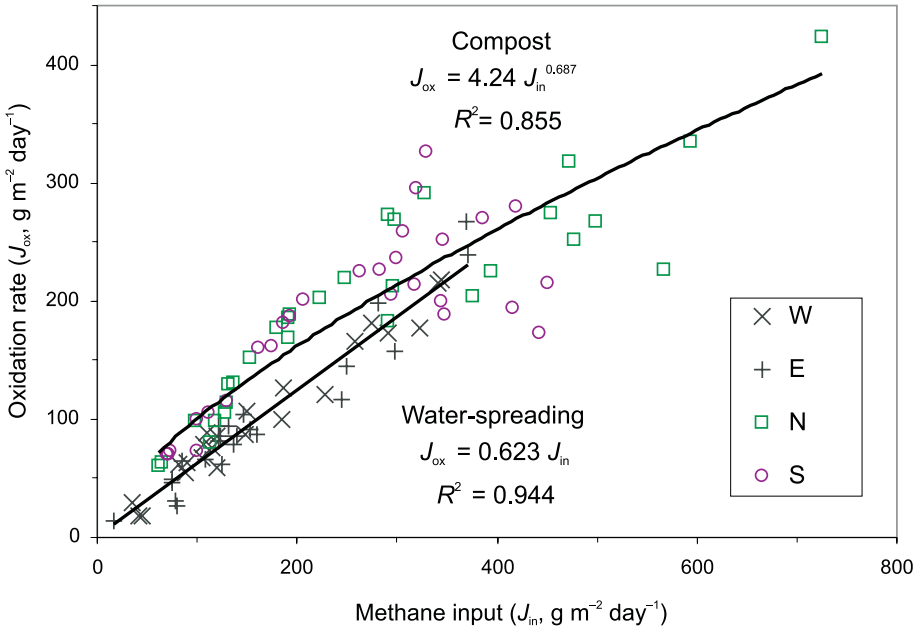

Fig. 6: Oxidation rates (J_{ox}), input flux (J_{in}), and regressions for the water-spreading (W and E) and compost (N and S) biofilters. Startup values are excluded (before day 21 for water-spreading and before day 4 for compost).

contact of finer sand over coarser sand, kept the sand at the level of the upper manometers wetter than static conditions (calculated average θ_v = 0.19, rather than 0.14). The water contents corresponding to the average manometer readings for the water-spreading biofilters are plotted in Figure 2.

Landfill gas would typically be warmer than the air-temperature gas used here (average air temperature 25.8°C). The condensation of water from the warmer gas would increase the water content in landfill biofilters. This should not affect the performance of the water-spreading biofilter because the sand was well-drained and water contents rapidly returned to usual values after a rain. Warmer biofilters might have lower oxidation rates, however. The optimum temperature for methane oxidation in peat soils was 25°C, although oxidation occurred at 35°C (from six studies referenced in Hanson & Hanson 1996).

Plans for additional study

We intend to construct a full-scale water-spreading biofilter at a landfill to evaluate construction techniques, performance, and reliability. This type of biofilter showed promise in the current 86-day study, but its long-term reliability needs to be tested under field conditions. Under passive flow conditions at a landfill where there is a wide range of inflow rates, the water-spreading biofilter is expected to perform better than a biofilter with uniform media because it provides a deeper zone that has a good balance of water and gas.

Conclusions

The water-spreading biofilter was designed to improve distribution of gas and water for methane oxidation in comparison with a barrel biofilter with compost media. The water-spreading biofilter had fining upward layers of sand to improve distribution of gas and water, and the sides were not enclosed to increase the surface area for diffusion. With the exception of the early start-up period, the water-spreading biofilters performed as well as the compost biofilters. From day 63 to the end of the experiment on day 86, the water-spreading biofilters oxidized 64 ± 3% of input methane and the compost biofilters oxidized 63 ± 4%.

Acknowledgement

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