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Moisture effects on greenhouse gases generation in nitrifying gas-phase compost biofilters

Guilherme D.N. Maia,*, George B. Day V, Richard S. Gates, Joseph L. Taraba, Mark S. Coyne

Abstract

Gas-phase compost biofilters are extensively used in concentrated animal feeding operations to remove odors and, in some cases, ammonia from air sources. The expected biochemical pathway for these predominantly aerobic systems is nitrification. However, non-uniform media with low oxygen levels can shift biofilter microbial pathways to denitrification, a source of greenhouse gases. Several factors contribute to the formation of anoxic/anaerobic zones: media aging, media and particle structure, air velocity distribution, compaction, biofilm thickness, and moisture content (MC) distribution. The present work studies the effects of media moisture conditions on ammonia (NH3) removal and greenhouse gas generation (nitrous oxide, N2O and methane, CH4) for gas-phase compost biofilters subject to a 100-day controlled drying process. Continuous recordings were made for the three gases and water vapor (2.21-h sampling cycle, each cycle consisted of three gas species, and water vapor, for a total of 10,050 data points). Media moisture conditions were classified into three corresponding media drying rate (DR) stages: Constant DR (wetter media), falling DR, and stable-dry system. The first-half of the constant DR period (0 to 750 h; MC = 65 to 52%, w.b.) facilitated high NH3 removal rates, but higher N2O generation and no CH4 generation. At the drier stages of the constant DR (750 to 950 h; MC = 52 to 48%, w.b.) NH3 removal remained high but N2O net generation decreased to near zero. In the falling DR stage (1200 to 1480 h; MC = 44 to 13%) N2O generation decreased, CH4 increased, and NH3 was no longer removed. No ammonia removal or greenhouse gas generation was observed in the stable-dry system (1500 to 2500 h; MC = 13%). These results indicate that media should remain toward the drier region of the constant DR (in close proximity to the falling DR stage; MC = 50%, approx.), to maintain high levels of NH3 removal, reduced levels of N2O generation, and nullify levels of CH4 generation.

* Corresponding author. Tel.: +1 859 608 7570.
E-mail addresses: gdnmaia@gmail.com (G.D.N. Maia), gday@bae.uky.edu (G.B. Day V), rsgates@illinois.edu (R.S. Gates), jtaraba@bae.uky.edu (J.L. Taraba), mscoyne00@email.uky.edu (M.S. Coyne).
1 Tel.: +1 859 257 3000x117; fax: +1 859 257 5671.
2 Tel.: +1 217 244 2791; fax: +1 217 244 0323.
3 Tel.: +1 859 257 3000x216; fax: +1 859 257 5671.
4 Tel.: +1 859 257 4202; fax: +1 859 257 5655.
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1. Introduction

Airstream-fed compost biofilters are used in various industrial and agricultural operations for the removal of odors and, in some cases, NH₃ or other pollutants. Aerobic nitrification is the desired biochemical pathway for NH₃ oxidation into nitrate (NO₃⁻) and nitrite (NO₂⁻). Nevertheless, uniformly distributed high levels of oxygen (O₂) rarely occur during biofilter operation. Regions of depleted oxygen in the biofilter media will favor other opportunistic microbial pathways and potentially lead to denitrification routes and greenhouse gas generation. Factors that contribute to the development of O₂ gradients include preferential flow patterns, dead zones, vessel wall effects, media aging, and natural compaction. Denitrification in nitrifying biofilters can be also beneficial if designed to (Sakuma et al., 2008) consume NO₂ and NO₃, compounds that in excess, can lead to substrate inhibition via formation of nitrous acids (Baquerizo et al., 2005) or inhibition from osmotic effects of the salt ammonium nitrate (Smet et al., 2000). Some nitrifiers activate denitrifying enzymes under microaerophilic conditions to produce nitric oxide (NO), nitrous oxide (N₂O), and nitrogen gas (Stein and Yung, 2003). Denitrifiers use the products of nitrification as substrates for denitrification. Coyne (2008) has indicated that the formation and function of denitrifying enzymes are sequential and regulated by O₂. The majority of denitrifiers are facultative aerobes that manifest their electron acceptor preference for O₂ over NO₃, NO₂, nitric oxide (NO), and N₂O (Van Spanning et al., 2007). Methanotrophs can also play important role in nitrification; for instance, in the formation of N₂O (Mandernack et al., 2000, 2009). Methanogens are not directly involved in the denitification process except for some denitrifying Archaea (Van Spanning et al., 2007). Methane generation, however, can provide reliable indication of anaerobic conditions during biofilter operation.

Oxygen profiles are also affected by media aggregate characteristics. Du Plessis et al. (1998) studied aerobic nitrifying biofilters and observed that aggregates with thick biofilms are potential sources of N₂O because of O₂ diffusion limitation. Internally, aggregates may develop mixed anoxic/anaerobic zones and potentially promote denitrification (Sexstone et al., 1985). Aggregates >10 mm in diameter often have anaerobic zones; aggregates as small as 4 mm may have anaerobic zones. Carbon, a limiting factor in denitrification (Seech and Beauchamp, 1988), is less available in smaller aggregates because of diffusion limitation and in large aggregates because of microbial respiration. High moisture content (MC) in carbon-limited environments can induce N₂O emissions (Senbayram et al., 2009).

Among many environmental factors influencing O₂ availability, moisture is determinant (Stein and Yung, 2003). In biofilters, moisture is probably the most critical operational parameter (Dorado et al., 2010; Chen et al., 2008). The formation of moisture profiles in the biofilter media structure as a whole and within inter-/intra-aggregates will sway O₂ gradients, and potentially favor conditions for greenhouse gas generation. Thus, the evaluation of moisture conditions during biofiltration is critical to help identify optimum levels of moisture for enhanced contaminant(s) removal and reduced pollutant(s) generation. The usual heuristic approach found in the literature for a large class of materials shows that MC should be between 35% and 80%, on a wet basis (Dorado et al., 2010; Nicolai and Janni, 2001; Devlinny et al., 1999; Hartung et al., 1997). On the other hand, same materials with different particle sizes (after sieving) can produce distinct media-moisture interactions (Maia et al., 2011a; Maia, 2010).

Determining media moisture conditions from media DRs is a reliable alternative (Maia, 2010), but still an underused approach in biofiltration. Foust et al. (1964) provided a general classification of the drying process for stable levels of temperature and humidity, by dividing the process into four drying stages. The initial unsteady stage (segment A of Fig. 1a) is marked by potential temperature and DR fluctuations in the solid media after first contact with the drying gas (Fig. 1a and b).

The second stage, constant DR (segment BC of Fig. 1a) occurs when the solid surface is saturated with water and the solid material does not affect the drying process. The drying controlling factors in the constant DR are: i) diffusion of water vapor across the air–moisture interface; ii) the rate at which the moisture surface is removed (Mujumdar and Menon,
2. Materials and methods

2.1. Media source and characteristics

Compost is the most used type of biofilter media in agricultural operations, prepared from recyclable organic residues found in animal feeding facilities. It complies with sustainable waste management practices using biodegradable materials that are available within the boundaries of a livestock production system. Compost material used as biofilter media was collected from a compost facility at Woodford Co., Kentucky. Active ingredients used as compost material included horse manure, cattle manure, and poultry waste. Bulking agents and carbon sources primarily included woodchips, sawdust, leaves, ground hay, tobacco stalks, and grass hay. Macro and micro nutrient availability was determined for sieved sample of particle range (PS) 8.0 mm > PS > 4.75 mm, which is the PS used in all experiments. The sieved compost was dried in an oven at 75 °C for 24 h, ground to pass a 2 mm screen, and stored at room temperature prior to analysis. Nitrogen (N) and Carbon (C) were determined via combustion.

Phosphorus (P), potassium (K), calcium (Ca), magnesium (Mg), zinc (Zn), copper (Cu), manganese (Mn), and iron (Fe) were extracted via digestion using a combination of hydrochloric and sulfuric acids. The digestate was assayed by Inductively Coupled Plasma Spectrometer for concentration determination. For the selected particle range, the nutrient content of the media as a percent was 38.74 ± 1.40 (C), 1.46 ± 0.07 (N), 0.57 ± 0.03 (P), 1.7 ± 0.1 (K), 4.2 ± 0.7 (Ca), 0.6 ± 0.1 (Mg). Trace elements included 79.4 ± 9.9 ppm (Zn), 14.0 ± 1.0 ppm (Cu), 335.8 ± 27.3 ppm (Mn), and 3057.6 ± 297.3 ppm (Fe). The O/N for the selected particle size range was 26.5. The value is in the range of stable finished compost (15–30) as suggested by Liberty (2002).

2.2. Determination of media drying-moisture relations prior to biofiltration

Media drying-moisture relations were determined prior to biofiltration and applied to biofilters in operation for estimation of MC under process conditions. Three replicates of biofilter media (947 g each) were placed inside 2.7 L containers and dried from media field capacity (61.6%, w.b.; Sales, 2008) to media equilibrium MC (8%, w.b.). Drying was performed using a flow rate of 4 L/min, at 58% relative humidity at 25 °C for 20 days. Tare containers with media were weighed every 24 h to determine media water loss (grams). Hourly DRs were then obtained from daily water loss averaged for 24 h (g H₂O h⁻¹). Drying rates and MC were then used to generate the rate-of-drying-versus-moisture-content curve.

2.2.1. Analysis

The results presented were the average and two standard deviations (±2 × SD) of 3 replicates. Drying rates were classified into three media drying stages: Constant DR, falling DR, and stable-dry system, based upon visual evaluation of the shapes of the curve at the end of the 20-day experiment (Maia, 2010). Linear-regression slopes from moving clusters of successive observation points were obtained and compared. The biggest changes between slopes were considered to be the DR stage transitions (2nd derivative method). The rate-of-drying-versus-moisture-content curve and its piecewise regression (95% prediction bands) were organized for the three DR stages, along with linear-regression parameters and inferential statistics. Curves with slopes not significantly different from zero (Null hypothesis: H₀ = 0) were considered to be constant (P > 0.05), or linearly variable (P < 0.05), otherwise.

2.3. Experimental conditions during biofiltration

2.3.1. Media preparation

Compost media was mixed with nitrifying biosolids (Lexington-Fayette County wastewater treatment plant), set at 70% MC (wet basis), and inserted into two tubular vessels (replicates with 7.62 cm internal diameter by 61 cm height). Drainage was observed after media insertion (first 30 min) but completely ceased after 1-h. Drainage caused media MC content to be reduced from 70% to 65% ± 1.6% (n = 2; ±2 × SD), which became the initial MC for the experiment (operation media field capacity). Microbial acclimation was performed...
with biofilters in full operation, representing the loading rate used during the experiment \((1.65-\text{NH}_3 \text{ g m}^{-2} \text{ h}^{-1})\). The time required for microbial acclimation was approximately twenty days: Ten days to achieve \(\text{NH}_3\) removal efficiency (RE) greater than 95% and ten more days at stable \(\text{NH}_3\) RE > 95% (Maia et al., 2011b).

### 2.3.2. Ammonia dilution and moisture application
The reactors were operated for 100-d and subjected to a continuous 6.7 L min\(^{-1}\) of humidified air at 25 ppm of \(\text{NH}_3\). Ammonia was diluted from 10,000 ppmv with humidified air at 60% relative humidity (water vapor generated through air bubbling) to produce the 25 ppmv inlet concentration. Moisture was complemented by directly daily application of water to compensate for media drying by the unsaturated air levels (RH = 60%). However, a constant water deficit was administrated during the direct application of moisture by applying less water than required to maintain 65% operation media field capacity (wet basis). With this approach, a slow and controlled drying was created, thus, moisture never recovered its initial level. The laboratory temperature range for the entire experiment varied from 23.5 to 25 °C (Fig. 2).

### 2.3.3. Gas sampling
Gas removal and emissions rates (unit: g m\(^{-2}\) h\(^{-1}\)) of water vapor, \(\text{NH}_3\), \(\text{N}_2\text{O}\), and \(\text{CH}_4\) were obtained every 2.21 h (cycle time for all sampling ports) from concentration differences between biofilter inlet and outlet (quasi-continuous method) multiplied by biofilter flowrates for determination of DRs, removal rates (\(\text{NH}_3\)), and generation rates (\(\text{N}_2\text{O}\) and \(\text{CH}_4\)). Concentrations were monitored using photoacoustic infra-red spectroscopy, IR-PAS (gas analyzer Model 1314, California Analytical, Inc., Orange, CA, USA) with detection limits of 0.2 (\(\text{NH}_3\)), 0.03 (\(\text{N}_2\text{O}\)), and 0.2 (\(\text{CH}_4\)), ppmv. Water vapor was measured for humidity interference compensation as well as to determine biofilter DRs. The gas analyzer sends gas sample concentration inputs to a personal computer (PC) via serial interface, and software on the PC receives data and provides a sequencing command to a 24 output digital USB multiplexer card (model USB1024HLS, Measurement Computing Corporation, Norton, MA, USA) for activating the solenoid valves in the multiplexer box. Sampling begins at a designated initial port and is sequentially re-started at the next sampling port location after every ten INNOVA gas sample measurements. A gas sample travels through the multiplexer’s Teflon and stainless steel

![Fig. 2 – Laboratory biofiltration unit (LBU).](image-url)
steel valve manifold and reaches the INNOVA for analysis. The time required to collect ten INNOVA readings per sampling port location was 8.3 min. Response time analysis (time to reach steady readings for a fixed and known input concentration — Maia et al., 2011b; Maia, 2010; Moody et al., 2008) indicated that NH3 had the longest response time among the monitoring gases. The time constant for NH3 to reach 98% of a known certified gas concentration was 6 min. Thus, only the last four readings (after six minutes) of the ten INNOVA readings were averaged and used for data analysis. The instrument had an overall reading scale uncertainty of around ±2% (after calibration with 2% accurate certified cylinders). Condensation on the gas sampling line wall was prevented using heat tape wrapped around the PTFE gas sampling tubes (Fig. 2).

2.4. Analysis
The results presented are the average of the two replicates for each gas (H2O, NH3, N2O, and CH4) in grams of gas removed/generated per biofilter cross sectional area per hour (g m⁻² h⁻¹). Drying rates were classified (Maia, 2010) into three media drying stages: Constant DR, falling DR, and stable-dry system, based upon visual evaluation of the shapes of the curve at the end of the 100-day experiment. The drying-rate-versus-time curve and its piecewise regression (95% prediction interval) were organized for the three DR stages, along with linear-regression parameters and inferential statistics. Sufficient data variability and noise prevented the pre-selection of stages before or during the experiment. Further, the nature of the data did not allow the use of algorithms to identify transition zones. Nevertheless, sharp changes in DRs were easily identifiable, and confirmed with the statistical analysis for the time-series. Curves with slopes not significantly different from zero (Null hypothesis: H0: slope = 0) were considered to be constant (P > 0.05), or linearly variable (P < 0.05), otherwise. Although MC was not directly monitored during biofiltration, accurate estimates of MC ranges were provided using the drying-moisture relations obtained prior to operation and described in Section 2.2.

3. Results and discussion

3.1. Determination of media drying-moisture relations prior to biofilter operation
The three different DR stages are readily apparent in Fig. 3. Moisture content varied from 43% to 61% (w.b.) in the constant DR stage, with zero slope (P = 0.8513; r² = 0.01; Table 1) and occurred over the 0–8 day-period for a mean DR of 1.9 g H2O h⁻¹. The transition between the constant DR and falling DR (critical MC) occurred in the range of 39–43% (w.b.), approximately, and the falling DR stage was identified to be in the moisture range of 13–39% (w.b.) with linear slope (P = 0.0002; r² = 0.91; Table 1). Finally, the stable-dry system stage comprised a range of 8–11% (w.b.), with small but significant slope (P = 0.0048; r² = 0.99; Table 1). A slope comparison between the first falling DR and the stable-dry system regression curves showed that they were significantly different (α = 0.05).

3.2. Moisture condition identification and its effects on gas evolution rates during biofiltration
Two gas-phase compost biofilter replicates treating NH3 were operated for 2500 h (100 days) with the set-up and analysis presented in Section 2.3. Estimates of MC for the corresponding DRs are also provided, based upon drying-moisture relations obtained prior to biofiltration (Section 3.1). Regression equations and parameters are provided in Table 2. The constant DR occurred between hour 0 and hour 1200 (H0: slope = 0; P = 0.0833; r² = 0.0086; Table 2), and corresponded to media moisture varying from 65% to 43%, respectively. The falling DR (H1: slope = 0; P < 0.0001; r² = 0.7353; Table 2) occurred in the 1200–1480 h period, falling within a moisture range of 43–13%, respectively. Finally, from hour 1500 to hour 2500, media approached dry-system stability from 13% to equilibrium MC (H2: slope = 0; P = 0.4763; r² = 0.0016; Table 2). The effects of DR on NH3 removal rates (g NH3 m⁻² h⁻¹ — grams of NH3 removed per biofilter cross sectional area per hour) and greenhouse gases generation (N2O and CH4, g m⁻² h⁻¹) are shown in Fig. 4b. Results are the average of two biofilter replicates. Process and physiological insights of NH3, N2O, and CH4 evolution rates based upon changes in moisture conditions (DR stages) are described next.

3.2.1. Moisture condition effects on NH3 removal and N2O generation
3.2.1.1. Constant DR stage. In the early constant DR stage (0–750 h; MC = 65–52%, w.b.) high levels of NH3 removal were observed but with the concurrent effect of high levels of N2O generation. Differently, at the drier region of the constant DR (750–950 h; MC = 52–48%, w.b.), high NH3 removal rates and
Statistical parameters of linear regressions (H₀: slope = 0; H₀: intercept = 0). MC = Moisture content (% wet basis); DRs = Drying rates (g H₂O h⁻¹).

<table>
<thead>
<tr>
<th>Media moisture condition</th>
<th>Regression equation with std. error</th>
<th>Coefficient of determination (r²)</th>
<th>t and P-values (alpha = 0.05) slope</th>
<th>t and P-values (alpha = 0.05) intercept</th>
</tr>
</thead>
<tbody>
<tr>
<td>Constant drying rate</td>
<td>DR = (−0.0023 ± 0.0119) × MC + (1.8754 ± 0.6323)</td>
<td>0.0063</td>
<td>t = −0.1957; P = 0.8513</td>
<td>t = 2.9658; P = 0.0251</td>
</tr>
<tr>
<td>First falling drying rate</td>
<td>DR = (0.0355 ± 0.0046) × MC + (0.1561 ± 0.1226)</td>
<td>0.9086</td>
<td>t = 7.7212; P = 0.0002</td>
<td>t = 1.2737; P = 0.2499</td>
</tr>
<tr>
<td>System dry-equilibrium</td>
<td>DR = (0.10965 ± 0.0076) × MC + (−0.7451 ± 0.0654)</td>
<td>0.9904</td>
<td>t = 14.3703; P = 0.0048</td>
<td>t = −11.3942; P = 0.0076</td>
</tr>
</tbody>
</table>

3.2.1.3. System dry-condition. Ammonia was no longer removed in the constant DR period (dry-condition) with the gas being released from hour-1250 until hour-2000, approximately. Nitrous oxide net generation was close to zero (background concentration levels).

Nitrous oxide production under aerobic conditions is well documented. A predominantly aerobic media (nearly atmospheric concentrations of oxygen) can potentially present anoxic/anaerobic zones depending on aggregate microsite conditions and its water pore filled space levels. Du Plessis et al. (1998) observed N₂O generation (9.2 ppmv) in biofilters under predominantly aerobic conditions (O₂ > 17% average) by artificially mixing the airstream with 15% of acetylene, a molecule well-known for inhibiting the reduction of N₂O to N₂. One explanation for the N₂O generation was the formation of a thick biofilm with anoxic zones. Joshi et al. (2000) studied a biofilter containing perlite as the packing material treating inlet concentrations of NH₃ of 20 and 50 ppm. In the earlier stages of the process, 100% of the N-mass balance was related to the formation of NO₂ and NO₃ from NH₃. In the later stages, the N-mass balance accountability dropped by 95% (for 20 ppm) and 75% (for 50 ppm). The explanation given by the authors for the nitrogen “loss” was provided partially from N₃O generation and NO detected but not quantified during the process. Nitrous oxide was observed in the early stages (start-up) of the process in Maia et al. (2011b). Maia et al. (2011b) also observed a high inverse correlation between NH₃ removal and N₂O generation in predominantly aerobic gas-phase compost biofilters. An explanation provided was that specific moisture conditions could have favored incomplete denitrification. In the conceptual model of Davidson (1991) nitrification is the main trigger for N₂O generation for water filled pore spaces.
(WFPS%) between 30% and 70% (approx.). The WFPS% in this work varied from 30.5% (MC = 65%, w.b) to 2.4% (MC = 13%, w.b), in the drier side of the model range, suggesting N2O was mostly generated from nitrification (enzymatic denitrification). Further, the falling DR was represented by MC < 43%, corresponding to a WFPS% < 12%, approximately. The net-generation of N2O at this WFPS level is almost zero (Fig. 4), as suggested in the Davidson model. Nitrogen biological transformations in nitrifying compost biofilters seem to be affected by moisture in a similar matter as in soils.

3.2.2. Moisture condition effects on CH4 generation

Methane generation was observed to commence at hour-1250, approximately, along with the beginning of the falling DR stage (Fig. 4) and it reached its peak (3 g CH4 m-2 h-1) in the same drying condition stage (1400-h). The peak occurred approximately at the same time period PNO was no longer generated. Methane generation ceased in the hour-2000, approximately, in the stable-dry system stage.

3.2.2.1. Period of zero net generation of CH4 (0–1250 h).

The period of zero net generation of CH4 can be explained based on two assumptions, potentially interactive: first, the assumption that CH4 was always released and/or generated but immediately consumed within the particle/aggregates and/or along the biofilter bed (overall CH4 net generation was zero); secondly, CH4 was never released/generated. In the first assumption potential sources of CH4 are microbial decay and/or formation by methanogens. The potential CH4 sinks were CH4 removal by physical sorption and/or biologically consumed by either CH4-oxidizers (methanotrophs) or NH3-oxidizers. There is enough evidence to suggest that NH3-oxidizers may oxidize CH4 owing to the affinity of CH4 with the active site of NH3 monooxygenase, the active enzyme of nitrifiers (Sylvia et al., 2005). Further, not only methanotrophs, but also NH3-oxidizers may potentially be a sink of CH4. In the second assumption, where CH4 was never generated, the higher moisture condition in the constant DR stage (65–43% MC, w.b) would actually favor anaerobiosis owing to oxygen-limited diffusion. What is left is the possibility that acclimation of the strictly anaerobic methanogens was slow enough to only be initiated after 1200 h. The retention time for anaerobic suspended-growth in standard treatment processes usually varies from 30 to 60 days (Metcalf and Eddy, 1991). Methane generation was only observed after 50 days (1200 h). A practical common occurrence in animal feeding operations is the existence of a large number of methanogens carried from the airstream to the media, or preexistent in the media. In case more methanogens were present at the beginning of the experiment, CH4 curve could have been shifted to the left and up; hence, CH4 could have been detected earlier, during the constant DR. The presence of CH4 in the early stages of the process can also negatively affect nitrification (Roy and Knowles, 1994). This is a plausible phenomenon because in the earlier stages of drying higher WFPS favors anaerobiosis, and more CH4 could have been generated and released after desorption from the biofilm/liquid. On the other hand, nitrifiers were more active in the constant DR stage (higher NH3 removal), and they can also consume CH4 before its release. That could have prevented the net-increase of CH4 in the constant DR for a situation of abundance of methanogens.

3.2.2.2. Period of positive net generation of CH4 (1250–1625 h).

The start of positive CH4 net-flux coincided with the beginning of the falling DR, which is controlled by water vapor diffusion and capillarity. Capillarity increases media osmotic potential leading to substrate limited substrate diffusion and in more critical cases, to cell dehydration. If free water is only present in the methanogenic zones, away from the particle/aggregate surface, CH4 produced from methanogens at the particle/aggregate core would not be consumed (or efficiently consumed) by methanotrophs, who are subject to stronger capillary forces with higher osmotic potentials. The unbalance of CH4 generated without being consumed may potentially be reflected in the observed positive net generation of CH4 by King and Schnell (1994) suggested that oxidation can also be inhibited by high NH3 concentrations, because CH4 and NH3 may compete for binding at the enzyme’s active site. Further, methanotrophs also oxidize NH3 and NO2, which is toxic to them. Accumulation of one of these three molecules (NH4+/NO3-/NO2-) could also have been detrimental to the CH4-oxidizers.

3.2.2.3. Return to null levels of CH4 (1400–2500 h).

The decreasing trend in the levels of CH4 (second half of falling DR) was attributed to severe drying effects with predominance of stronger capillary forces throughout the biofilter media.

Fig. 4 – (a) Drying-rate-versus-time with corresponding media moisture conditions; (b) greenhouse gas generation during NH3 removal (NH3 loading rate: 1.65 g NH3 m-2 h-1).
Potential physiological effects of drying were substrate transport limitation and cell dehydration. This condition could have caused CH$_4$-reducers and CH$_4$-oxidizers inhibition as well as increased O$_2$ penetration. Desorption was probably the primary physical–chemical mechanism of CH$_4$ decreasing rates.

3.2.3. Effects of media degradation and drying on media structure

Structural changes in media from media degradation can affect the dynamics of removal and fate of contaminants in gas-phase compost biofilters. Consequences of aging include flow channeling, porosity reduction/bulk density increase from compaction, and increased pressure drops. Nevertheless, studies have shown compost biofilter media can operate for five years without significant changes in C/N composition (Cardenas-Gonzalez et al., 1999), or for 3–10 years without large pressure drop increase (Nicolai and Lefers, 2006). Webster et al. (1997) reported that media type did not significantly affect biofilter microbial dynamics, after comparing compost and granular activate carbon biofilters for 500 days; the effects of media aging on microflora changes could not be established for the 500-d study.

For this study, 100-d was not sufficient to detect structural media changes (changes in average bulk density/porosity). Regarding the interaction of microorganisms with the media structure, the inoculated population of nitrifiers was constantly supplied with an inorganic N-source (NH$_3$), not dependent on mineralized C or N in the filter. For heterotrophic organisms, the C/N is most likely a mix of micro components: volatile fatty acids, sugars, some simple polymers (e.g. cellobiose) and acetate coming from the decomposition of the sawdust/straw. The main structural macrocomponent of the media is woodchips, hardly biodegradable in a 100-d experiment for the MC range applied. Finally, the effect of drying on media structure is likely to be minimal, because the system was intact and barely undisturbed during the experiment. A more significant effect would be the development of biofilms on new sites of the existing structure. Conversely, drying increases media capillary forces, hydrophobic zones, and substrate diffusion limitation, which, in turn, can dramatically decrease microbial activity. A reliable indication of reduced microbial activity was observed during the driest drying stage (stable-dry system), based on the very-low to null levels of NH$_3$ removal and N$_2$O/CH$_4$ generation.

4. Conclusions

Results from this study showed that media moisture conditions potentially affect the removal and fate of contaminants in nitrifying gas-phase compost biofilters. Biofilter media should remain toward the drier region of the constant DR stage and closer to the falling DR stage (MC = 52–48%, approx.) in order to maintain high levels of NH$_3$ removal, reduced levels of N$_2$O generation, and null levels of CH$_4$ generation. Ammonia was no longer eliminated at the end of the constant DR period (950–1200 h; MC estimate = 43–13%) and N$_2$O net generation decreased to near zero. The falling DR stage (1200–1500 h; MC = 13%) was marked by virtual biofilter failure with the system becoming a source of greenhouse gases. For the driest media moisture condition (stable-dry system) biofilter remained in the failure mode with no NH$_3$ removed or greenhouse gases generated. Moreover, this study suggests that the design of nitrifying gas-phase biofilters should include the generation of N$_2$O as a conflicting constraint on biofilter overall performance.

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