Biofilter: a promising tool for mitigating methane emission from manure storage

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Abstract: Liquid manure storage may contribute to methane (CH_4) emission and this emission can be greatly reduced if appropriate management practices are applied. Biofiltration has been used in other fields for mitigating greenhouse gas (GHG) emission (e.g., landfill) and shown promise for mitigation CH_4 emission from liquid manure storage. It has been reported that biofilter was capable of reducing 80% of CH_4 emissions from manure storage. The CH_4 removal efficiency is influenced by many factors, including CH_4 and O_2 concentrations, temperature, moisture, composition of the filter bed, nutrient, and empty bed residency time (EBRT). Biological conversion of methane of a biofilter is a slow process due to the low water solubility of methane. The residence times (EBRT) between 5 min and 5 h have been used, whereas a typical EBRT of 25 s is used for common biofilter applications. Temperature at which methanotrophic bacteria are active ranges from 10ºC to 45ºC. The maximum activity is found at around 30ºC. The optimal filter bed water content depends on both the gas flow rate and the type of filter bed (soil, compost, etc.) and ranges from 30%–70% of the water holding capacity. Compost is the best material for filter bed. The optimal pH for methanotrophic bacteria is neutral to slightly acidic. Copper and nitrogen compounds especially nitrate are important nutrients to methanotrophic bacteria but their optimal concentrations have not been founded. Phosphorus and other elements such as potassium and manganese are reported to affect the performance of methanotrophic bacteria but need further confirmation.

Keywords: biofilter; greenhouse gas; methane; manure storage

1 Introduction

For many countries, agricultural industry is responsible for a significant amount of total greenhouse gases (GHG) emission. In Canada, for example, nitrous oxide (N_2O) contributes 61% of all N_2O emission, methane (CH_4) contributes 38% and carbon dioxide (CO_2) contributes less than 1%. In the agricultural sector, the main sources of emissions are from the bacterial activity in a ruminant's digestive system (55%), from soil (24%) and from manure (21%) (Massié, 2006). Therefore, manure management is one of primary sources of greenhouse gas (GHG) emissions in the agricultural industry (Cole *et al*., 1997; Patty *et al*., 2005). It is possible to reduce methane (CH_4) emission from manure storage systems by up to 25% to 80% using appropriate manure management and treatment practices (Cole *et al*., 1997).

Many livestock operations, especially for hog and dairy farmers, use liquid manure storage systems. Those liquid manure storage facilities were considered to be responsible for significant portions of the overall CH_4 emission from manure storages. In recent years, covering liquid manure storage or manure storage tank is advocated to be an effective way of preventing odor and GHG emission. The beneficial management practices of Manitoba Agriculture states: "Covering liquid manure storage facilities can be an effective way to reduce GHG emissions. Stored manure emits methane (CH_4), a potent GHG. An impermeable storage cover traps CH_4 and prevents its release. The CH_4 can be flared off to produce carbon dioxide (CO_2), a less potent GHG, or used as a source of heat for the farm". Flaring CH_4 may require expensive equipment and skills to operate. Additionally the CH_4 content of the emitted gas needs to be high enough to allow effective combustion. An alternative is using biofiltration (biofilters). A biofilter uses microorganisms (bacteria)

Received 2010-09-07, accepted 2010-10-09
doi: 10.3724/SP.J.1227.2011.00061
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to break down various compounds in the air when the air passes the biofilter media. The bacterial oxidation of methane is a common phenomenon that occurs in nature, such as tropical forest, grassland; landfills cover soil, peatland, and rice paddy soil. Removing methane by biofiltration is basically aerobic conversion of methane to carbon dioxide (CO_2) and water by methanotrophic (methane consuming) bacteria grown in the biofilter media. Despite the vast body of literature detailing the oxidation of methane in natural environments, relatively little research has been conducted on the application of methane biofilters for manure storage emissions (Massé, 2006). This paper summarizes the information on the use of biofilters in mitigating GHG emissions, assesses the technical feasibility of using biofilters in mitigating GHG emissions from covered hog manure storage based on the literature review.

2 The release rate of methane from liquid manure storage

For covered liquid manure storages, methane concentration in the headspace depends on air exchanges between the headspace and outside air. In theory, at an air-exchange rate of zero, the headspace concentration equals to undiluted biogas that contains about 425 g/m^3 (65%, v/v) methane (Safley and Westerman, 1988, 1989; DeSutter and Ham, 2005). In practice, however, the cover contains some openings that permit ventilation that results in methane concentrations ranging from 0.1 g/m^3 to 20 g/m^3 (0–3%) (Roland *et al*., 2005).

The published gas fluxes varied greatly and were expressed in various units. For the data expressed in volumetric flux unit, the measured fluxes of CH_4 ranged from 0.0007 to 0.64 $m^3/(m^2 \cdot day)$. For studies that expressed flux results on weight-based units, the measured fluxes ranged from 0.2 to 202.74 $g/(m^2 \cdot day)$. The measured fluxes usually have great spatial, daily, and seasonal variations (Safley and Westerman, 1988; DeSutter and Ham, 2005). Spatial variation was observed both within the same liquid manure storage and among storages and was closely related to substrate distribution (Safley and Westerman, 1988; Wagner-Riddle *et al*., 2006). Greater CH_4 flux was observed near inlet portion of storages, where more manure was distributed than in other places. The flux variation observed among several storages in one area was a result of loading rate differences among storages (Safley and Westerman, 1988). Higher loading rates often resulted in higher biological activity and therefore higher gas flux.

The flux of biogas from storages showed both daily and seasonal variations. Higher flux rates were usually observed during the day and summer, while lower flux rates were observed during the night and winter (Sharpe and Harper, 1999). The daily and seasonal variations in biogas emission rates resulted from storage temperature and wind speed variations. Higher temperature and wind speed during the day and summer would result in higher emission rates, and lower temperature and wind speed during the night and winter would result in lower emission rates (Sharpe and Harper, 1999). However, the pattern of seasonal flux variation might be altered by substrate availability for methanogenesis. DeSutter and Ham (2005) found the highest emission rate in spring, which declined during summer due to substrate limitation. The peak emission rate occurred in early June and nearly 50% of annual gas losses occurred within 30 days.

3 The principles of biofiltration

Biofiltration uses microorganisms to degrade and oxidize gaseous pollutants to un-harmful gases. This technique has been used for reducing odors, hydrogen sulfide and ammonia emissions from farms (Massé, 2006). It has also been employed to treat air streams contaminated with many harmful gases (Tahraoui and Denis Rho, 1998; Li *et al*., 2002; Yoon and Park, 2002). However, the application of bio-filtration in the treatment of CH_4 is relatively new. The experiment conducted by Massé (2006) indicated that biofiltration is one of the promising techniques capable of reducing GHG emissions from livestock operations, particularly methane.

Methane biofilters use methanotrophs living in porous media to oxidize CH_4 to CO_2. Methanotrophic bacteria (or methanotrophs) use methane as their energy and carbon source whereby methane is degraded to carbon dioxide and water (Hanson and Hanson, 1996). Methane oxidation by methanotrophs occurs in natural environments and can be found in many natu-

ral aerobic anaerobic interfaces. For example, methane oxidation has been reported in tropical forests, grasslands and meadows, landfill cover soils, deserts, and agricultural soils (Nikiema *et al*., 2007).

A man-made biofilter is designed and constructed with the goal of exerting maximum efficiency for gas treatment. A good biofilter is a three-phase bioreactor: the filter bed constitutes the solid phase, the biofilm the liquid phase and the gaseous pollutants the gas phase (Nikiema *et al*., 2007). The solid biofiltration media and the liquid phase offer microorganisms' surface for immobilization, nutrients and water for growth, and the space for gas exchange. The gaseous phase offers microorganism the necessary gases (CH_4 and O_2 for methane biofilter) for their survival. Therefore, biofilters favor some specific microorganisms' activities.

According to the way that gases circulate in the biofilter, the biofilter can be classified as a closed system or open system (Nikiema *et al*., 2007). The majority of biofilters, as used in lab-scale experiments, are closed systems, in which air supply is ensured by a forced ventilation system. Gas circulation in the biofilter can be from either top to bottom or conversely. In a closed biofilter, maintaining steady operational parameters is also a relatively easy, resulting in good performance. Nikiema *et al*. (2007) summarized the performance of biofilters used for mitigation of CH_4 emission from landfills and reported methane conversion values as high as 90%.

Open systems are mostly found in landfill sites. In this case, the flow of the polluted gas in the bed proceeds upwards, while the O_2 diffuses from the ambient air into the bed (passive ventilation) (Nikiema *et al*., 2007). The open system is difficulty in controlling the operational parameters, such as temperature and moisture levels. Moreover, the transfer of O_2 to the bed's lowest layers limits the performance of open system (Kjeldsen *et al*., 1997; Gebert *et al*., 2001). For example, an open biofilter installed on a landfill site only had a removal efficiencies of up to 60% if the empty bed residence times (EBRT) was at least an hour (Du Plessis *et al*., 2003; Gebert and Groengroeft, 2006a, b).

A biofilter should have at least 1 m^3 of filter bed for achieving flow rates of CH_4 in the range of 0.01–2.5 m^3/h (Straka *et al*., 1999; Streese and Stegmann, 2003; Haubrichs and Widmann, 2006). The height of the open biofilters with passive ventilation, used for CH_4 elimination, must also be lower than 1 m (Kjeldsen *et al*., 1997; Boeckx and Van Cleemput, 2000; Stein and Hettiaratchi, 2001). Open systems are usually less expensive (by at least 15%) than closed systems.

4 Microorganisms

4.1 Methanotrophs

Three basic steps were indentified in the process of decomposing CH_4 by methanotrophs. The first step is the oxidation of CH_4 to methanol through utilizing the enzyme methane monooxygenase, MMO (Hanson and Hanson, 1996; Auman and Lidstrom, 2002). Then the methanol is further transformed into formaldehyde. In the final step, formaldehyde produced from the previous step is used in a dissimilatory pathway (i.e. being oxidized to CO_2, with formate as an intermediate) or via several types of assimilatory pathways, leading to the synthesis of cell components, which is necessary for the growth of methanotrophs (Hanson and Hanson, 1996).

Generally, the specific bacteria responsible for the decomposition of CH_4 are named as methanotrophs. However, depending on their roles in the CH_4 decomposition process, the genera of methanotrophs are grouped into three main types: type I, type II, and type X. Type I methanotrophs assimilate formaldehyde by the ribulose monophosphate pathway and their cellular membranes are mainly made up of fatty acids with 16, or sometimes 14 atoms of carbon (Nikiema *et al*., 2007). Type II methanotrophs assimilate formaldehyde through the serine pathway and their cellular membranes contain fatty acids of 18 carbons. Type X has both properties of types I and II. Its cellular membranes fatty acids have 16 carbons and the assimilation of formaldehyde is through both the ribulose monophosphate cycle and the serine pathway (Nikiema *et al*., 2007).

4.2 Methane monooxygenase enzyme (MMO)

A specific enzyme known as methane monooxygenase or MMO was reported to be the key enzyme allowing methanotrophs to perform the decomposition of CH_4 (Hanson and Hanson, 1996). This enzyme exists in two forms: particulate MMO (pMMO) and soluble

MMO (sMMO). The pMMO enzyme can be found in and synthesized by all methanotrophs, except Methylocella, but the sMMO is almost always present in bacteria of types II and X. Methanotrophs containing pMMO grow more rapidly than those having the sMMO (types II and X) (Nikiema *et al*., 2007).

4.3 Other bacteria

In addition to methanotrophs, some other bacteria can also decompose CH_4 under some situations. For example, nitrifying bacteria, that are responsible for the decomposition of ammonia (NH_3), can also degrade CH_4. However, their performance rate is less than 5% of the pure methanotrophic populations (Hanson and Hanson, 1996; Bodelier and Frenzel, 1999). Also, some bacteria involved in the decomposition of methanol are capable of degrading CH_4 if the CH_4 concentrations remain below 10% (v/v).

5 The performance of methane biofilters

The performance of a methane biofilter is measured by a removal efficiency parameter as defined in the following equation:

$$\text{Removal Efficiency: } RE = \frac{C_{in} - C_{out}}{C_{in}} \times 100, \quad (1)$$

where, C_{in} is the methane concentration of the gas at the inlet of the biofilter in ppm, and C_{out} is the methane concentration of the gas at the outlet of the biofilter in ppm. This Removal Efficiency was sometimes given different names, Conversion (*X*), for example (Nikiema *et al.* 2007). Elimination capacity (*EC*) is another parameter that can be used to access the performance of a biofilter. It was calculated using the following equation:

$$EC = IL \times \frac{X}{100}, \quad (2)$$

where *EC* is elimination capacity (g/(m^2·d) or g/(m^3·d); *IL* is the inlet load (g/(m^3·d)); and *X* is Conversion (%). The inlet load (*IL*) is calculated according to the following equation:

$$IL = \frac{C_{in} \times Q}{S}, \quad (3)$$

where C_{in} is the CH_4 concentration of the biogas flowing into the methane biofilter in g/m^3; *Q* is the flow rate of the biogas in m^3/d; and *S* is the biofilter bed cross section in m^2.

The performance of methane biofilters depends on various factors, including the biofilter type, the empty bed residence time (EBRT), and operating conditions. In mitigating CH_4 emissions from landfill sites, the closed biofilter was reported to show good performance, with CH_4 X-values as high as 90% (Dammann *et al*., 1999; Gebert *et al*., 2001; Streese *et al*., 2001; Du Plessis *et al*., 2003; Nikiema *et al*., 2005), in contrast to a X-value of 60% that was obtained from the open methane biofilter with an EBRT of at least an hour (Du Plessis *et al*., 2003; Gebert and Groengroeft, 2006a, b). The best *EC* obtained with a laboratory-scale closed biofilter was in the range of 325–400 g/(m^2·d) (Hettiaratchi and Stein, 2001). In general, the biofilter eliminates some 10%–100% of the CH_4 escaping from the upper layers of landfills, depending on local climatic conditions (Nozhevnikova *et al*., 1993; Kightley *et al*., 1995; Czepiel *et al*., 1996).

Compared with many cases of biofilters used for mitigating CH_4 emission from landfill sites, few cases were reported for using biofilters for mitigating CH_4 emission from manure storages. Masse (2007) reported that the removal efficiency reached up to 80% for the four types of media material used in the biofilter. This technology was identified as a promising method for controlling methane emissions from manure storages. Venugopal *et al*. (2003) conducted a lab-scale experiment using a methane biofilter for mitigating CH_4 emission from a gas meter station. They report that the conversion varied with temperature, reaching 90% during the summer and dropping to 20% during the winter. Melse *et al*. (2005) reported a CH_4 removal rate up to 85% in a lab-scale biofilter used in the mitigation of CH_4 emissions from a liquid manure storage.

6 Factors affecting the efficiency of a methane biofilter

6.1 Oxygen, methane, and carbon concentrations

Methane degradation by methanotrophs requires CH_4 and O_2 as substrates. In fact, methanotrophs can be found in small quantities in any environment exposed simultaneously to significant amounts of CH_4 and O_2 (Borjesson *et al*., 1998; Dammann *et al*., 1999). However, the optimal concentrations of CH_4 and O_2 for CH_4 degradation have not been determined. Lit-

eratures showed a variation in CH_4 and O_2 optimal concentrations. It was reported that type I bacteria grew better in an environment with O_2 concentration of 21% (v/v), associated with a CH_4 concentration less than 1,000 ppm, while type II bacteria develop better in an environment with CH_4 concentration above 1% (v/v) and O_2 concentration at about 1% (v/v) (Hanson and Hanson, 1996; Henckel *et al.*, 2000). However, some type I bacteria have their growth stimulated only in the presence of an appreciable concentration of CH_4 (> 1%, v/v), and correspondingly, a low amount of O_2 (< 1%, v/v) (Henckel *et al*., 2000; Erwin *et al*., 2005). Bender and Conrad (1994), Czepiel *et al*. (1996) and Stein and Hettiaratchi (2001) have shown that, by increasing the O_2 concentration from 3% to 20% (v/v) in the gas mixture, the CH_4 conversion varies only slightly (less than 10%). However, a decrease of O_2 concentrations from 3% to 1% causes a decrease of CH_4 oxidation of more than 50%. In the experiments of Stein and Hettiaratchi (2001), maximal CH_4 elimination was obtained at O_2 concentrations between 0.75% and 1.6%. The presence of CO_2 in a biofilter modifies the behavior of the microorganisms. Acha *et al*. (2002) reported that the activity of the methanotrophs, using the serine pathway for the assimilation of formaldehyde obtained during the decomposition process of CH_4, requires some CO_2 input (partial pressure of CO_2 around 11.6 kPa).

6.2 Temperature

The temperature in which methanotrophic bacteria are active ranges from 10°C to 45°C. The maximum activity was found at around 30°C (Whalen *et al*., 1990; Bender and Conard, 1994; Boeckx and Cleemput, 1996). However Priemé and Christensen (1997) observed methane oxidation to be active in temperatures as low as 1°C in the field and 2°C in soil cores experiments. King and Adamsen (1992) observed methane consumption at –1°C and they suggested that methane consumption might occur at low temperatures as long as the soil water remains liquid. Summerfield *et al*. (1993) showed that the soil microflora was active even when the soil was covered by snow and near 0°C, and that methane consumption was taking place under these conditions.

It was reported that the conversion (X) fell by around 50% when the temperature was reduced from 30°C to 20°C or from 29°C to 24°C (Dammann *et al*., 1999; Streese *et al*., 2001). Between –5°C and 10°C of ambient temperature, the biological elimination of CH_4 in an open biofilter system (landfill cover soil) considerably decreased, i.e. more than 80%, compared with the value at 15°C (Christophersen *et al*., 2000; Le Mer and Roger, 2001).

6.3 Moisture content

The optimum moisture content varies depend on what kind of material used to build biofilter. It ranges from 30%–70% of the water holding capacity, which is clearly lower than the optimum for odor and trace gas treatment (Whalen *et al*., 1990; Bender and Conrad, 1995; Boeckx and Cleemput, 1996). Very high soil water content may impede gas diffusion and thus restrict the supply of CH_4 to the methanotrophs. The low solubility of CH_4 in water enhances this effect especially at low CH_4 concentrations (Bender et Conrad, 1995).

The optimal filter bed water content depends on both the gas flow rate and the type of filter bed (soil, compost or other material employed) (Christophersen *et al*., 2000). Optimal moisture content of soil materials (from the upper layers of landfills) ideally lies between 13% and 15.5% (wt/wt), on a dry basis (Chiemchaisri *et al*., 2001b; Jackel *et al*. 2001; Stein and Hettiaratchi, 2001; Park *et al*., 2002). For composts or biological residues, optimal bed moisture lies between 25% and 50% (wt/wt) (Humer and Lechner, 1999). See Table 1 for a summary of these studies.

6.4 pH

It is generally suggested that a neutral or slightly acidic medium is maintained for a methane biofilter (Bender and Conrad, 1995). Nikiema *et al*. (2007) suggested that the pH of the filter bed is a parameter of less importance because the biodegradation of CH_4

Table 1 Optimal water content of Biofilter beds used for methane elimination (Nikiema *et al.*, 2007)

Filter bed	Water content (% (wt/wt))	Sources
Compost	25–50	Humer and Lechner (1999)
Landfill cover soil	13–30	Boeckx and Van Cleemput (1996), Visvanathan *et al*. (1999), Stein and Hettiaratchi (2001), Giani *et al*. (2002), Park *et al*. (2002)
Meadow soil	30–50	Wang and Li (2002)
Woodland soil	18–33	Wang and Li (2002)
Various soils	11–35	Bender and Conrad (1995), Christophersen *et al*. (2000)

does not generate intermediate or final products capable of significantly influencing the pH. The optimal pH values for the oxidation of CH_4 are in fact the same as those promoting the growth of the majority of methanotrophic bacteria.

According to Hanson and Hanson (1996), methanotrophs are neutrophiles but can tolerate pH from 5.5 to 8.5. Bender and Conrad (1995) suggested that the optimum pH ranges between the values of 6.7 and 8.1 for the soil-based filter beds, while Le Mer and Roger (2001) suggested that the range lies between 5 and 6.5 for peat.

6.5 Filter bed

The filter bed is the solid phase of the biofilter. A good filter bed provides sufficient space for the development of microorganisms and also has a texture providing a great moisture holding capacity, in addition to appropriate bacteriological and mechanical properties (Nikiema *et al*., 2007). Various materials that are generally classified as soils, composts, and other materials or combinations, have been used as the filter bed media and tested by researchers (Nikiema *et al*., 2007). Compost was known as the most efficient filter bed with the Conversion rate (X) ranging from 90% to 100%. Its conversion rate also varied depending on what materials the compost made from (Hettiaratchi and Stein, 2001; Wilshusen *et al*., 2004; Haubrichs and Widmann, 2006). In addition, mature compost is more efficient for the biofiltration of CH_4 than freshly generated compost (Humer and Lechner, 1999). Soil is another important material for filter bed, and various soils such as agricultural soils, soils derived from mountains, forests and rice plantations, peat bogs and swamps, have also been tested for CH_4 biofiltration (Dobbie and Smith, 1996; Hutsch, 1998; Del Grosso *et al*., 2000). The most effective soils for CH_4 elimination are those taken directly from the upper layers of landfill covers with a reported X value of greater than 80%. Usually the filter bed is made by mixing soil with organic materials, such as vegetable residues (beet leaves, wheat straw), clarifier sludges or composts, and this can enhance CH_4 elimination (Borjesson *et al*., 1998; De Visscher *et al*., 1999; Humer and Lechner 1999; Park *et al*., 2002). The soils should have appropriate particle size which preferably lies between 0.5 and 2 mm (Borjesson *et al*., 1998; Hettiaratchi *et al*., 2000; Min *et al*., 2002). When particle sizes are less than 0.02 mm, the bed tends to become packed, preventing the effective diffusion of pollutants in the gas phase and then negatively affecting the conversion (Bender and Conrad, 1995; Le Mer and Roger, 2001; Min *et al*., 2002). There were also some other synthetic or inert filter materials having been used for CH_4 biofiltration. The high X values as more than 95% were also achieved by using those filter bed materials (Sly *et al*., 1993; Nikiema *et al*., 2004b). However, they limited for being used in field scale due to their cost.

6.6 Nutrients

Nutrients such as copper, nitrogen and phosphorus are necessary for the growth of microorganisms and therefore are factors that affect the performance of a biofilter (Trotsenko and Khmelenina, 2002). These nutrients are supplied to the microorganisms through the mixture with water that is used to humidify the filter bed (Nikiema *et al*., 2005).

6.6.1 Copper

Copper affects bacterial growth, however, the threshold concentrations have not yet been determined. Hanson and Hanson (1996) demonstrated that while copper inhibits the sMMO enzyme at concentrations above 1 μmol/L, it supports the synthesis of the pMMO at concentrations between 1 and 5 μmol/L. The CH_4 oxidation has been reported to increase by 5% after adding 0.02 g $CuCl_2$ per kg of paddy soil (Mohanty *et al*., 2000).

6.6.2 Nitrogen compounds

Nitrogen is usually provided to microorganisms in inorganic forms: e.g. nitrate (NO_3), ammonium (NH_4) or nitrite (NO_2) ions. Many studies have been performed to determine the effect of each of these compounds on methanotrophs. The sources of NH_4 most frequently tested are ammonium chloride, ammonium sulfate and urea. For NO_3, sodium nitrate and potassium nitrate are the most studied. On some occasions, ammonium nitrate was used as a nitrogen source (Kightley *et al*., 1995; Hettiaratchi *et al*., 2000). No final conclusions on nitrogen effect have been obtained up to now. Some studies reported the improvement on CH_4 oxidation by adding some specific nitrogen sources at certain concentrations while others reported different results using same nitrogen sources at similar concentration ranges (Table 2).

Table 2 Studies of investigating the effect of N on the work of CH_4 biofilters (Nikiema *et al*., 2007)

Sources	Filter beds	N forms & Concentration	Effect
Hettiaratchi *et al*. (2000)	Soil	25 mg N/kg soil in the form of NH_4^+ or NO_3^-	improve CH_4 elimination by 100%
Chiemchaisri *et al*. (2001a)	Soil	⩾30 mg N/kg soil in the form of NH_4^+ or NO_3^-	Inhibit CH_4 elimination
Bronson and Mosier (1994); Cai and Mosier (2000); Hettiaratchi *et al*. (2000); Novikov and Stepanov (2002); Park *et al*. (2002)		10–200 mg N–NH_4^+/kg soil	Inhibit CH_4 elimination, however, its extension depends on the type of soil
Nikiema *et al*. (2005)	Inorganic filter material	Sodium nitrate, from 0.14 to 0.75 g N/L	5 times increase in the EC (from 130 to 700 g/(m^2·d).
		Sodium nitrate > 0.75 g N/L	Decrease the CH_4 oxidation
Boeckx and Van Cleemput (1996); Park *et al*. (2002)	Soil	25–100 mg N–NO_3^-/kg soil	No CH_4 elimination effect
Kumaraswamy *et al*. (2001)	Soil	2,500 mg N–NO_3^-/kg soil	Inhibit CH_4 elimination

The NH_4^+ was considered to have a competition with CH_4 when it was provided as a nitrogen source (Mancinelli, 1995; Boeckx and Van Cleemput, 1996; Humer and Lechner, 1999; Sitaula *et al*., 2000; Novikov and Stepanov, 2002). In addition to oxidizing methane, methanotrophs can convert NH_4^+ to NO_2^-. Novikov and Stepanov (2002) reported that 12%–28% of the methanotrophic population was dedicated to a nitrification step instead of the CH_4 oxidation. The inhibitory effect of NH_4 could be minimized if higher CH_4 concentrations were continuously provided to the filter media.

While several people reported that NO_3^- has stronger inhibitory effect on methanotriphs than NH_4^+, most people suggested that NO_3^- is the preferred source of fixed nitrogen for methanotrophs (Mancinelli, 1995) and can improve CH_4 elimination (Le Mer and Roger, 2001).

6.6.3 Phosphorus

Generally speaking, phosphorus is of universal importance in promoting the growth of bacteria. However, few documents have been found in clarifying P effect on CH_4 elimination. Several scientists (Kightley *et al*., 1995; Hettiaratchi *et al*., 2000; Le Mer and Roger, 2001) reported that the addition of 0.1 g P–K_2HPO_4 nutrient per kg soil did not result in any noticeable effect on promoting CH_4 elimination. Thus, the role of phosphorus in CH_4 elimination remains unclear and further investigations would be needed (Nikiema *et al*., 2007).

6.6.4 Other elements

Potassium sulfate and manganese oxide have been reported to increase the oxidation of CH_4 (Kumaraswamy *et al*., 2001), while the excessive concentrations of sodium chloride and potassium chloride were reported to be CH_4 elimination inhibitors (Cai and Yan, 1999; Kravchenko, 2002; Gebert *et al*., 2003).

7 Empty bed residence time (EBRT)

Biofilters have been developed and operated for landfill gas treatment at various methane inlet concentrations up to 260 g/m^3 (40%, v/v) at empty bed air residence times (EBRT) between 5 min and 5 h, whereas typical EBRT is 25 s to over a minute for common biofilter applications (Melse and Van De Werf, 2005).

Biological conversion of methane in a biofilter is a slow process due to the low water solubility of methane (Henry's law constant is 1.5×10^{-3} M/atm), and this is why such long EBRT are applied. Previous work by Streese and Stegmann (2003) showed first-order removal kinetics for methane inlet concentrations up to 16 g CH_4/m^3 in an operated biofilter.

8 Conclusions

For covered liquid manure storage, the methane concentration in the headspace varied depending on the sealed condition of the headspace. In theory, the headspace concentration equals undiluted biogas that contains about 425 g/m^3 (65%, v/v) if the headspace is completely sealed. In practice, however, the cover seldom completely airtight, which results in methane concentrations ranging from 0.1 to 20 g/m^3? Biofiltration is one of the promising techniques that are capable of reducing 80% of CH_4 emissions from manure storage. The CH_4 removal efficiency is influenced by many factors, including CH_4 and O_2 concentrations,

temperature, moisture, composition of the filter bed, nutrient, empty bed residency time (EBRT). Biological conversion of methane in a biofilter is a slow process due to the low water solubility of methane. The residence times (EBRT) between 5 min and 5 h have been used, whereas a typical EBRT of 25 s is used for common biofilter applications. The temperature in which methanotrophic bacteria are active ranges from 10°C to 45°C. The maximum activity was found at around 30°C. The optimal water content of filter bed depends on both the gas flow rate and the type of filter bed (soil, compost or other material employed) and ranges from 30%–70% of the water holding capacity. Compost is the best material for filter bed. The optimal pH for methanotrophic bacteria is neutral to slightly acidic. Copper and nitrogen compounds, especially nitrate, are reported as important nutrients to methanotrophic bacteria but their optimal concentrations have not been found. Phosphorus and other elements such as potassium and manganese were reported to affect the performance of methanotrophic bacteria but need to further confirmation.

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