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REVIEW

Greenhouse gas emissions from organic waste composting

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Abstract There is actually a common consensus in using biological technologies for the treatment of organic wastes. For instance, composting is used for aerobic biological stabilisation of organic wastes. The amount of materials and the variety of wastes composted are increasing. However, composting is inherently a process generating gaseous emissions. Greenhouse gases (GHG) such as carbon dioxide, methane and nitrous oxide from compost are of special relevance for global warming. Although a part of these gases is inherent to the process, another important part can be abated by low-cost biological technologies such as biofiltration and its variations. This article reviews the emission of GHG from composting gases, from detection and measurement to minimisation and abatement. Special emphasis is given to the measurement of GHG to obtain reliable emission factors for the different composting technologies. These factors will help to compare different waste treatment options based on overall analysis tools such as life cycle assessment. A specific chapter is related to carbon and nitrogen dynamics in the composting matrix, and their consequences on the production of carbon and nitrogen gases. Finally, we present a review of the best available practices to minimise the GHG emissions from composting and the final treatment of composting off-gases.

Introduction

The sustainable use of resources and wastes, including waste minimisation and valorisation, is a common objective of the plans, directives and rules published in the last few decades. One example is the Sixth Programme of Community Action in the field of Environment ("Environment 2010: the future is in our hands") published by the European Union for the period 2001–2012 (European Union 2008a). The Sixth Program of Action includes the implementation of seven thematic strategies and, among them, specifically waste prevention and recycling, with the objective to reduce the negative environmental impacts during the whole life cycle of wastes, from their production to their elimination, including their recycling. One of the results of all these legislation efforts was the publication of the Waste Framework Directive in 2008. This Directive considers waste not only as a potential source of pollution, but also as a resource that can be used. Specifically, in the case of biodegradable wastes, the Directive 1999/31/CE on landfilling of wastes encourages the diversion of these wastes to other treatment technologies involving the recycling and energy recovery from wastes, where composting will have a great importance (European Union 2008b; Commission of the European Communities 2008).

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Nevertheless, the environmental impact assessment during the whole life cycle of wastes lacks of data obtained directly at full-scale waste treatment facilities operating in different locations, thus limiting the quality and reliability of these analyses necessary for the decision-making process.

A direct consequence of the above-mentioned plans and directives has been the proliferation of a large number of new waste treatment plants installed in Europe and all over the world in the last years, as well as the modification and adaptation of the existing ones. In particular, composting and anaerobic digestion are the more widely accepted processes for organic waste treatment. Composting plants are typically operated either in piles or tunnels, whereas anaerobic digestion can take place either in wet or dry digesters, typically followed by composting of the digested sludge with the aim of ensuring its sanitation and stabilisation (Ponsá et al. 2008).

Waste treatment facilities can be the origin of public complaints, most of them associated with annoyances caused by odour emissions generated during the process. Biological treatment plants are a clear example of this problem. Odours generated from this type of treatment plants are mainly associated with the emissions of volatile organic compounds (terpenes, alcohols, ketones, sulphur compounds, amines, etc.) and ammonia (Goldstein 2002; Komilis et al. 2004). Some of the annoyances caused by these emissions are often magnified because of the lack of real data from operating plants that would contribute to have an objective and scientific base to analyse these problems. Such lack of data also represents a problem for the design of mitigation measures such as the use of biofilters. In addition to this, greenhouse gases (GHG) emission inventories evidence the increase in the amount of these compounds that are emitted from waste treatment facilities. Emission of CO_2, CH_4 and N_2O is the main responsible of this increase (Colón et al. 2012).

Emissions generated in waste treatment plants, in particular those based on biological treatments, are related to the type of technology, the type of wastes treated and the operational conditions of the plant. For this reason, it is very important to relate the emissions to the performance of the biological treatment plants and also to the wastes being treated, since each treatment technology and waste will give rise to different end products quality and organic matter stabilisation degrees. The use of respirometric indices to monitor the stability of the organic matter has been one of the main research topics in the last years (Barrena et al. 2005, 2006, 2009a, b; Ponsá et al. 2008).

Although ammonia is not considered a GHG, its emissions during composting are usually studied because it causes acid rain and from the point of view of the conservation of nitrogen in the end product because of the potential use of compost in agriculture, as well as for the determination of the efficiency of the systems for gas emission treatment, such as scrubbers and biofilters. Ammonia emissions are affected by the C/N ratio of the initial composting mixture, by the temperature reached during the process and by the aeration (Pagans et al. 2006b; Raviv et al. 2002; Sánchez-Monedero et al. 2001). Biofilters have shown to be an efficient equipment for the reduction in ammonia emissions in enclosed waste treatment plants (Hong and Park 2004; Pagans et al. 2006b), although for long periods ammonia tends to reduce the efficiency of this technology (Baquerizo et al. 2005).

An important part of the published literature in the field of gaseous emissions is related to odours, mainly by means of dynamic olfactometry, in both composting plants and mechanical–biological treatment plants (MBT). As already mentioned, a number of laboratory-scale experiments have been performed with the aim of determining the compounds that more significantly contribute to odour pollution. Thus, Goldstein (2002) identified terpenes, alcohols, aldehydes, fatty acids, ammonia and a range of sulphur compounds as the main responsible of odour emissions at composting plants. Other authors have studied the effect of some operational conditions, such as ventilation and turning, in these emissions (Szanto et al. 2007). Gage (2003) proposed a number of managing practices aimed at reducing the annoyances generated by odour emissions; for instance, preparation of an optimal initial mixture and the maintenance of high levels of porosity to assure aerobic conditions in the pile (Ruggieri et al. 2009). Enclosing the composting operations and the use of biofilters are among the main mitigation strategies for both odours and GHG.

The importance of GHG emissions generated during the biological treatment of wastes has been also stated by several authors. CO_2 emissions coming from biological process are not considered to contribute to global warming since this carbon has a biogenic origin, i.e. this carbon has been previously fixed biologically. Regarding other gases, He et al. (2001) measured the emissions of N_2O and CH_4 during the composting of food wastes under laboratory conditions in a closed system with forced aeration. Although generated in small amounts, N_2O and CH_4 have a great contribution to global warming since they have a warming potential 25 (CH_4) and 235 (N_2O) times higher than that of CO_2.

There are some scientific publications that provide gaseous emissions data generated during the biological treatment of organic wastes, mainly for manures and sewage sludge. However, the number of published papers dealing with municipal solid wastes is scarce (Colón et al. 2012). The works carried out by Eitzer (1995) and Staley et al. (2006) are very important for the characterisation of the emissions generated during the biological treatment of

wastes and the identification of specific compounds. In 1995, Eitzer performed a comprehensive characterisation of the volatile organic compounds (VOC) generated in composting plants treating municipal solid wastes and its possible relation to the process performance. On the other hand, Staley et al. (2006) studied the VOC emissions originated during the aerobic treatment of wastes and also during the anaerobic biodegradation process. These works highlighted the importance that forced aeration, used in the biological processes, had on the total emissions (Delgado-Rodriguez et al. 2012). Terpenes and ketones are shown to be the most abundant compounds. These experimental works were performed under laboratory conditions, which would limit their extrapolation to full-scale plants. Pagans et al. (2006a) also evaluated the effect of the type of waste (industrial and municipal solid waste) on VOC emissions under laboratory conditions. Komilis et al. (2004) identified the main VOC emitted during composting of pruning residues (mainly terpenes, alkyl benzenes, ketones and alkanes) and also during composting of food wastes (sulphides, organic acids and alcohols), as well as the stages of the process that generated the highest emissions (thermophilic phase).

The determination of emission factors for different wastes and different treatment technologies will be a useful tool for the calculation of global emissions at facilities operating with a technology already studied in other treatment plants. Emission factors for VOC, NH_3 or GHG are usually expressed per ton of treated waste or per amount of obtained compost (Amlinger et al. 2008).

Sampling and measurement protocols for the determination of emissions have also been studied (Sironi and Botta 2001). Even though there are several published papers about this topic (Sommer et al. 2004), there is a lack of information on the measurement of emissions from surface sources, in both non-aerated (composting piles with natural aeration by convection) and those with a common source that will be later spread in an outlet surface (biofilters).

The main factors controlling a composting process are those characteristics of an aerobic biological process such as oxygen concentration, temperature, moisture, pH and C/N ratio. The optimum values for the C/N ratio range from 15 to 30, even though it is possible that composting takes place in a wider range of values (Haug 1993). For this reason, adjusting the optimum C/N ratio of the starting mixture is recommended. The use of different organic wastes or some selected additives could also be satisfactory (Charest and Beauchamp 2002). Nevertheless, the amounts of carbon and nitrogen used for the calculations should be referred to the amounts that are readily available for the microorganisms when considering the C/N ratio as a parameter to be optimised (Puyuelo et al. 2011). This specific point is very important for the potential practical

implications in the preparation of starting composting mixtures. In relation to pH, recent studies have demonstrated its effect on the emissions of odours (Sundberg et al. 2013).

In this context, respirometric methodologies have been shown to be suitable and reliable for the determination of the amount of biodegradable organic matter in wastes of different origin and characteristics. There are two types of respirometric analysis for this purpose: dynamic and static determinations, being the dynamic methods the most widely accepted and recommended (Adani et al. 2004; Barrena et al. 2006; Gea et al. 2004). The measurement of the CO_2 produced during the respirometric test is also used as a measurement of the biodegradability of the organic matter (Cooper 2004) and, consequently, of the biodegradable organic C.

Other researchers have worked on the emissions generated during the composting process of agricultural wastes (Komilis et al. 2004; Cayuela et al. 2006; Mondini et al. 2006, 2007; Sánchez-Monedero et al. 2008; Szanto et al. 2007). In the USA, other studies are focused on VOC and NH_3 emissions during the composting of biowaste (Büyüksönmez and Evans 2007).

This review is a compilation of the different works dealing with the measurement, detection, minimisation and treatment of the GHG emitted during the composting process of a wide variety of organic wastes. This article is an abridged version of the chapter by Smith et al. (2010) [Chapter 2] published in the book series Environmental Chemistry for a Sustainable Word (http://www.springer.com/series/11480).

Composting

The specific role of composting in GHG emissions

Composting is an environmentally friendly waste treatment process where organic matter is biologically degraded. Although the benefits of composting are evident, GHG can be generated and emitted to the atmosphere during this process contributing to global warming.

In this context, composting of organic waste contributes (composting process) and avoids (compost application) at the same time to GHG emissions. GHG are released from composting facilities due to degradation of organic matter and the use of electricity and fuels in management waste operations. The use of compost in agriculture has a positive effect in GHG emissions since its application as an organic amendment provokes that carbon stays bound to soil, although the content of other nutrients (N, P, etc.) is typically low. GHG emissions from composting processes depend on the waste type and composition, the technology

systems used (static and dynamic process, open and closed systems, presence or not of gas treatment units) and the final use of compost.

Benefits of compost application have to be assessed together with a real knowledge about the amount of GHG such as N_2O and CH_4 generated during the composting process. The relation of GHG with some operational conditions and the technology used must also be considered. Data on GHG emissions from full-scale composting facilities are necessary to improve the knowledge about the contribution to the composting in GHG emissions. In the last years, there has been an increase in the number of scientific publications studying GHG emissions during composting (Amlinger et al. 2008; Boldrin et al. 2009; Sánchez-Monedero et al. 2010; Cayuela et al. 2012; Colón et al. 2012; Deportes 2012).

GHG emissions from composting processes are highly dependent on the waste type and composition. The composition and characteristics of the feedstock are key parameters for the design and operation of the composting facilities and for the final quality of the compost (Haug 1993).

Wastes with a low C/N ratio and high water content have a great potential for generating GHG emissions both during the storage and the composting process. In fact, wastes lacking of nutrients, porosity and structure, or presenting low biodegradability can hamper the correct evolution of the process, increasing the GHG emission. In order to minimise these emissions, optimal conditions for the initial mixture are required.

For some wastes, pre-treatment operations before composting are required. This is the case of municipal solid wastes, especially when a source separation system is not implemented. The production of high-quality compost from MSW may require a lot of energy because of the use of heavy machinery that makes GHG emissions unavoidable (Lou and Nair 2009). Other materials, such as biosolids or manure, have a poor structure and an excess of water content and require the use of a bulking agent. Grinding and mixing this bulking agent are operations that require energy that again contribute to GHG emissions.

Composting technologies can be open and closed systems. In open systems, composting is performed in facilities where, in general, gaseous emissions are neither collected nor treated. However, when the composting process takes place in an enclosed system, usually the exhaust gases are treated. As expected, concentrations of GHG reported in facilities when the gas treatment systems are well-implemented were lower (Colón et al. 2012) than those of open systems. Effects of forced aeration and turning in GHG emissions have been also studied. Szanto et al. (2007) observed lower N_2O and CH_4 emissions in turned piles than in static systems. They related these emissions to the

prevalence of anaerobic regions in the static systems, as other similar studies (Parkinson et al. 2004). Amlinger et al. (2008) proposed that high aeration and effective stripping of NH_3 during the early stages of composting can reduce N_2O formation. Ermolaev et al. (2012) studied the effects of different aeration and temperature settings on the emission of CH_4, N_2O and CO_2 during windrow composting with forced aeration following three different control strategies. However, they found that the emissions of CH_4 and NO_2 were low regardless of the amount of ventilation. The oxygen concentration, temperature profile and moisture content are factors controlling GHG emissions. Nowadays, in the composting field, the technology that allows the control of these parameters is available.

Regarding CO_2, its emissions in composting derived from the organic matter biodegradation are not taken into account in their contribution to global warming since this carbon has a biogenic origin. The CO_2 that contributes to GHG emissions is generated by composting facilities as a result of operational activities. In composting, the main GHG that can contribute to global warming are CH_4 and N_2O. Both are related to a lack of oxygen during the composing process and consequently they depend on the management of the composting process (Cayuela et al. 2012; Colón et al. 2012). These gases, although they are generated in small amounts, have a great contribution to global warming since they have a warming potential of 25 (CH_4) and 235 (N_2O) times higher than that of CO_2.

Several authors reported that even in well-aerated process, CH_4 was emitted (He et al. 2000; Clemens and Cuhls 2003) while Beck-Friis et al. (2000) observed a rapid decrease when the oxygen supply was increased. The production of N_2O can be due to an incomplete ammonium oxidation or incomplete denitrification (Beck-Friis et al. 2000). Emissions of N_2O have been reported at different stages of the process. Some authors reported high emissions at the beginning of composting (He et al. 2000, Parkison et al. 2004). Other studies reported the production of N_2O during the mesophilic and maturation phases (Beck-Friis et al. 2000; Hao et al. 2004) when the readily available carbon sources have been depleted (He et al. 2000). According to Cayuela et al. (2012), N_2O formation will be hampered if there are conditions to inhibit nitrification (such as low available NH_4^+ in the pile or high pH). Beck-Friis et al. (2000) and Fukumoto et al. (2003) related N_2O emissions to the temperature of the process and CH_4 emissions to the size of the pile (both works were performed at full scale, using windrows and forced aeration systems, respectively), the structure of the material and the time of the process. Higher emissions were measured in larger piles, with a poor structure and longer composting times. Monitoring of CH_4 emissions showed a large experimental fluctuation in all works.

Several authors have reported the GHG emissions generated during the biological treatment of several typologies of wastes. Most of them were calculated from laboratory and pilot scale processes, although interesting data at industrial scale have been also reported (Boldrin et al. 2009; Colón et al. 2012; Ermolaev et al. 2012). There are an important number of studies that quantify CH_4 and N_2O emissions from animal manures (Fukomoto et al. 2003; Hao et al. 2004; Szanto et al. 2007). However, less published works dealing with municipal solid wastes can be found, and even less works studying the GHG emissions of different composting systems have been published.

Colón et al. (2012) evaluated four different full-scale facilities treating the source-separated organic fraction of municipal solid wastes (OFMSW). They reported a range of CH_4 and N_2O emissions between 0.34 and 4.37 kg CH_4 Mg OFMSW^{-1} and 0.035 and 0.251 kg CH_4 Mg OFMSW^{-1}, respectively. Regarding CH_4, the highest values were found in facilities without gas treatment units. Also, Boldrin et al. (2009) presented a study where several technologies for municipal solid waste treatment were evaluated. They reported CH_4 and N_2O emissions ranging from 0.02 to 1.8 kg CH_4 Mg OFMSW^{-1} and 0.0075 and 0.252 kg CH_4 Mg OFMSW^{-1}.

As previously commented, although ammonia is not considered a GHG, its emission during composting plays an important role. Ammonia emissions are affected by the C/N ratio of the initial composting mixture, by the temperature reached during the process and the aeration (Pagans et al. 2006b). High loads of ammonia can reduce the optimal use of the biofilter system in enclosed facilities (Amlinger et al. 2008). Moreover, the conservation of nitrogen in the end product improves compost use in agriculture as organic fertiliser. Consequently, from a global warming point of view, less use of chemical fertilisers will be required (Favoino and Hogg 2008).

In the role played by composting in GHG emissions, it is important to bear in mind the role of compost as an end product. The use of compost as an organic amendment can contribute to mitigate GHG in several forms.

Compost utilisation can reduce the need of chemical fertilisers and pesticides, which implies the reduction in GHG emissions associated with their production and application. Also, a positive effect in soil structure is produced with compost application by improving tillage and workability. Improved structure of soils associated with the application of organic matter can help to reduce requirements for water irrigation in periods of drought and to increase the potential of soils to retain moisture (Favoino and Hogg 2008).

One of the aspects associated with compost utilisation that more attention has received in the last years is the potential for sequestration of carbon in agricultural soils (Mondini et al. 2007; Favoino and Hogg 2008). By applying compost, biogenic carbon is held in soils for a period of time before carbon is released, increasing carbon uptake and storage within the plant and removing CO_2 from the atmosphere.

GHG emitted during composting and their relationship to C and N dynamics

Microbial transformations involved in the formation of CH_4 and N_2O in composting piles are similar to those taking place in other environments such as soil, water bodies, wastewater treatment plants. However, the microbial gas production and the final emission to the atmosphere will be affected by the particular environmental conditions of composting piles (such as temperatures up to 70 °C, high organic matter content, easily available organic compounds, rich and active microbial population and limited amount of oxygen) and composting management operations (turning, watering, pile size and geometry, etc.). All these variables represent a characteristic environment affecting not only the microbial gas production in the pile but also its transport within the pile and the final emission to the atmosphere. In the following sections, the impact of the C and N dynamics on GHG emissions during composting will be also discussed.

Carbon dioxide (CO_2)

As previously mentioned, there are two main sources of CO_2 emissions from composting facilities, biogenic and non-biogenic CO_2. Biogenic CO_2 emissions derive from the biological degradation of the organic matter, mostly as a consequence of aerobic decomposition and, to a lesser extent, from anaerobic processes or the oxidation of CH_4 by aerobic methanotrophic bacteria. This emission accounts for the highest amount of gas generated during the process, since between 40 and 70 % of the original organic matter can be degraded during composting (Haug 1993). However, the global warming potential of these emissions is not taken into account in the environmental impact of composting operations since this biological CO_2 is considered to be carbon neutral (IPCC 2006). The exclusion from the inventories has reduced the number of papers studying CO_2 emissions, and this gas is only studied from the point of view of establishing mass balances of composting operations (Boldrin et al. 2009) or as an index of the overall microbial activity of the pile, reflecting the progress of the process (Hobson et al. 2005; Sánchez-Monedero et al. 2010) and the evaluation of the stability of the end product (Barrena et al. 2006).

Non-biogenic CO_2 from composting includes the emissions associated with energy and fuel consumption in the

composting facility. These emissions are dependent on the technology of the plant and the machinery used such as shredders, front-loaders, turning equipment, screenings and other processing activities. These emissions are beyond the scope of this review but updated information can be found elsewhere (Boldrin et al. 2009; Scheutz et al. 2009; Lou and Nair 2009; Brown et al. 2008).

Methane (CH₄)

Methane emissions derived from organic waste composting have attracted the attention of researchers as a considerable contributor to global warming since this greenhouse gas has a global warming potential 25 times greater than carbon dioxide over a time horizon of 100 years (IPCC 2006).

The optimum growing conditions for methanogenic bacteria are a lack of oxygen (strict anaerobic microorganisms), a redox potential below -200 mV, neutral pH and the presence of nutrients and substrates rich in organic matter (Kebreab et al. 2006). These conditions can be temporally found at the early stages of the composting process, where large amounts of nutrients and available sources of organic compounds stimulate microbial growth, depleting the oxygen levels in the pile. Accordingly, most of CH_4 emissions have been recorded during the initial weeks of the process, at the beginning of the thermophilic phase (Beck-Friis et al. 2000; Sánchez-Monedero et al. 2010). The high temperatures reached at the beginning of the process reduce oxygen solubility (Pel et al. 1997), facilitating the creation of anaerobic spots within the pile. However, there are other variables such as high concentration of ammonia, which may inhibit the activity of methanogens at pH > 9 (Kebreab et al. 2006), or the presence of electron acceptors such as sulphates, which reduce their activity by competition with sulphate-reducing bacteria (Hao et al. 2005). Sánchez-Monedero et al. (2010) reported that the high ammonia levels generated by the hydrolysis of urea, used as nitrogen source, inhibited the production of CH_4 in olive mill waste composting piles.

The emission of CH_4 from composting piles is governed by the biological activity of the pile (Hao et al. 2001) and also by other factors affecting gas transport from the anaerobic spots to the pile surface, such as gas diffusion within the pile and the presence of methanotrophic bacteria. Methanotrophs are aerobic microorganisms colonising the surroundings of anaerobic zones and pile surface, which are able to oxidise between 46 and 98 % of the CH_4 generated in the pile (Jäckel et al. 2005). Methanotrophic bacteria also play an important role in the production and consumption of other relevant GHG emitted during composting, such as N_2O and CO (Topp and Hanson 1991). Sánchez-Monedero et al. (2011) performed a four-year interannual evaluation of the GHG emissions from a composting plant treating olive mill wastes and found a reduction in CH_4 emissions associated with the improvement of the management of the composting plants (watering and turning frequencies).

Kebreab et al. (2006) and Brown et al. (2008) reviewed the topic of GHG emissions from livestock and composting operations and they highlighted the importance of the composting feedstock, the height and shape of the pile, the control of moisture content and turning frequency as the main factors governing CH_4 emissions during the process, since these variables will affect both the oxygen availability and gas diffusion in the composting pile. The presence of manure can also increase the methane emissions due to the incorporation of anaerobic microorganisms, as observed by He et al. (2000) and Sánchez-Monedero et al. (2010) in composting piles treating food and olive mill wastes, respectively.

Nitrous oxide (N₂O)

There is an increasing awareness about the emission of N_2O from composting operations due to the high global warming potential of this gas (296 times higher than that of CO_2 over a 100 year horizon, IPCC 2006) and its impact on the ozone layer (Smith et al. 2010). Despite the relatively small amounts of N_2O released during composting, its contribution to the global N_2O budget in waste management or livestock agriculture cannot be discarded due to the impact of composting operations treating manures or other N-rich organic wastes (de Klein et al. 2010).

The biological production of N_2O during composting is a complex process since there are different microbial pathways involved in the formation of N_2O (nitrification, nitrifier denitrification and denitrification among others), which may simultaneously occur at different locations within the pile (Czepiel et al. 1996; Kebreab et al. 2006; Maeda et al. 2011). For this reason, the identification of N_2O sources as well as the microorganisms involved in these processes still remains a key research topic (Maeda et al. 2011).

Nitrification is one of the main microbial processes leading to the emission of N_2O during composting. Aerobic nitrification involves the initial transformation of ammonia to nitrite by different genera of ammonia-oxidising bacteria (AOB), such as *Nitrosomonas* and *Nitrososporas*, according to the following equation:

$$2NH_3 + 3O_2 \rightarrow 2NO_2^- + 2H^+ + 2H_2O$$

and the oxidation of nitrite to nitrate by nitrite-oxidising bacteria, such as *Nitrobacter* (Kowalchuk et al. 1999; Maeda et al. 2010):

$$2NO_2^- + O_2 \rightarrow 2NO_3^-$$
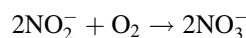
NH_4^+ is the main precursor of nitrification. NH_4^+ is generated by ammonification of OM at early stages of the process (Sánchez-Monedero et al. 2001). Typical alkaline pHs found in composting matrices favour the transformation of this soluble NH_4^+ into NH_3, which is then initially oxidised by AOB into NO_2^- and then transformed to NO_3^- by nitrite-oxidising bacteria (NOB). N_2O is produced during the initial step of the oxidation of NH_4^+, as an intermediate between NH_2OH and NO_2^- (Czepiel et al. 1996). Ammonia-oxidising *archaea* (AOA) have been recently suggested to be actively involved in nitrification in composting piles but the contribution of AOA to the total amount of N_2O still remains unclear (Yamamoto et al. 2010; Zeng et al. 2012).

Denitrification has traditionally represented the main source of N_2O, especially in the case of manures (Kebreab et al. 2006). Denitrification is an anoxic process carried out by denitrifiers, which are heterotrophic microorganisms that can use NO_3^- as the electron acceptor, causing the reduction of NO_3^- to N_2 according to the following steps:

$$NO_3^- \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2$$

In absence of O_2, NO_3^- is reduced to N_2 without appreciable N_2O production, but N_2O production can increase as the concentration of O_2 increases in the pile (Czepiel et al. 1996). In this case, nitrifier denitrification (denitrification coupled to an incomplete nitrification at low O_2 concentrations) can be responsible for the generation of N_2O during the initial step of ammonia oxidation and also as a consequence of NO_2^- reduction. This mechanism has been studied in agricultural soils (Wrage et al. 2001), but there is only limited information during composting (He et al. 2001; Hobson et al. 2005). Fukumoto and Inubushi (2009) observed that the addition of NOB reduced the emission of N_2O during composting of pig manure, suggesting that the accumulation of NO_2^- in the pile could be a significant source of N_2O, due to the reduction of NO_2^- to N_2O (under limited O_2 conditions) rather than the final oxidation to NO_3^- (with no O_2 limitation). Under these conditions, when available C was depleted, nitrifier denitrification would be the main mechanisms leading to N_2O emissions, as observed by He et al. (2000), who found an increase in the N_2O emission when the ratio between water-soluble C and dissolved N was lower than 5.

Nitrifiers and denitrifiers show their optimal growth under different environmental conditions. Nitrifiers require aerobic conditions, mesophilic temperatures (below 40 °C), pH values above 5 and the presence of NH_4^+, whereas denitrifiers need anaerobic conditions, or at least low O_2 concentration, the presence of sources of available C and the presence of NO_3^-, NO_2^- or NO as electron acceptors (Kebreab et al. 2006). Due to the heterogeneity of the composting materials, both environmental conditions (aerobic and anaerobic zones) can coexist simultaneously in the composting mass, since different oxygen concentration gradients are created along the pile (Beck-Friis et al. 2000; Hao et al. 2001). Denitrifiers may colonise the inner part of the pile, whereas nitrifiers, which require oxygen concentrations in the range within 1 and 10 % (Béline et al. 1999), may colonise the aerobic pile surface. The relative contribution of nitrifiers and denitrifiers to the N_2O emission was governed by the oxygen concentration and moisture of the pile (Hwang and Hanaki 2000). These authors reported that denitrification was the main source of N_2O at moisture levels between 40 and 60 % and oxygen concentrations around 10 %, whereas nitrification became more dominant at higher oxygen concentrations.

Similar to those of CH_4, N_2O emissions can be affected not only by the biological activity of the composting mixture but also by the N availability and gas diffusion within the pile (Hao et al. 2001). Several authors reported peak N_2O emissions either at early stages of the process or after the thermophilic phase of composting, when the environmental conditions of the pile (temperatures below 40 °C) favour the growth of nitrifying bacteria (He et al. 2001; Kebreab et al. 2006; Sánchez-Monedero et al. 2010). Once NO_3^- has been generated, the mixing of the composting matrix facilitates the transport of nitrates from the surface to the interior of the pile where they can be reduced to N_2 and N_2O by denitrifiers. The use of urea as N source can enhance N_2O emissions up to levels similar to those found in N-rich manure heaps due to the increase in available N from the hydrolysis of urea (Sánchez-Monedero et al. 2010). Vermicomposting also increases the N_2O emissions by stimulating denitrification and nitrification processes, due to the increase in N availability and the transport of N facilitated by the activity of earthworms (Frederickson and Howell 2003; Hobson et al. 2005).

Gas exchange within the pile also plays an important role since the generation of N_2O by both nitrifiers and denitrifiers is enhanced at low oxygen concentrations (Czepiel et al. 1996). N_2 is obtained as the final product of denitrification in the absence of O_2, but significant amounts of N_2O are generated as the concentration of O_2 increases in the pile. In addition, pure cultures of *Nitrosomona* bacteria responsible for the initial step of ammonia oxidation have been shown to significantly increase the production of N_2O under limited oxygen conditions (Goreau et al. 1980). Since these factors are highly dependent on the composting material and the process performance, the specific characteristics of the starting materials will determine the environmental conditions for N transformation during composting.

Other relevant greenhouse gases

There are other gases generated in small amounts during organic waste composting that have been studied due to

their impact on global warming. Carbon monoxide (CO) and nitrogen oxides different than N_2O (NO_x) have small direct global warming potential but they both lead to indirect radiative effects by increasing CH_4 lifetime and elevating concentrations of tropospheric O_3 (IPCC 2006). The calculation of their contribution to global warming is subject to large uncertainties due to the short lifetime and reactivity of these gases in the atmosphere. According to IPCC (2006), the global warming potential, over a 100-year horizon, is likely to be 1–3 for CO, and in the order of 5 for surface NO_X emissions.

The emission of CO occurs during the aerobic decomposition of the organic wastes during composting by a mixture of physical processes and biological activity (Hellebrand and Halk 2001; Hellebrand and Shade 2008). These authors found the maximum CO flux rates at the beginning of the composting process, probably due to physicochemical generation, and then the levels decreased during periods of high biological activity, reflecting the temperature dependence of CO emissions and also the impact of oxygen availability and the oxidation to CO_2. CO emissions only represent a minor GHG source in green waste and livestock waste (Hellebrand and Shade 2008) and in urban wastes, where CO–C emissions varied from 0.07 to 0.13 kg Mg^{-1} of wet feedstock, which represents approximately about 0.04–0.08 % of the total C emitted (Andersen et al. 2010a; 2010b). CO emissions have also been investigated as a potential health risk to workers in enclosed facilities treating municipal solid wastes (Phillip et al. 2011).

From the two gases composing NO_x ($NO + NO_2$), only NO is generated during composting, either as by-product or intermediate of microbial nitrification and denitrification (Del Prado et al. 2006; Hao et al. 2001). Fukumoto et al. (2011) studied the NO emissions from swine manure composting and observed a similar trend to that of N_2O, characterised by a peak after the thermophilic phase of composting (coinciding with the activity of nitrifiers) and a decreasing trend towards the end of the process. Total NO emissions only represented one-tenth of the magnitude of N_2O emission, approximately 3 % of total N losses.

Greenhouse gas production for different typologies of organic wastes

There is a wide range of organic wastes that can be used as composting substrates such as manures, municipal solid wastes, garden and yard wastes, agricultural crop residues, sewage sludge and other industrial sludge. The characteristics of these starting materials will affect the physicochemical properties of the pile and, consequently, will govern the microbial processes leading to the formation of

GHG and also their diffusion and transport within the pile. As already discussed in the description of the main pathways of CH_4 and N_2O generation, the main variables affecting GHG emissions are the moisture content and porosity, which control the oxygen availability and gas diffusion, and the presence of nutrients and organic compounds to be used as substrates for the microorganisms participating in gas production. The composting technology used for the aeration (forced aeration or windrowing), the size of the piles and pile temperature also represent key variables affecting GHG generation and emission.

Manures

Manures represent one of the most important and studied substrates for composting (Kebreab et al. 2006). Manures are N-rich organic materials characterised by high moisture contents that make them to be considered as wet feedstock for composting (Haug 1993). The treatment of manures through composting permits the reduction in volume and moisture, their sanitisation and organic matter stabilisation, giving rise to a valuable end product that can be safely used in agriculture. However, manure characteristics favour GHG emission during composting. The large amounts of easily available N compounds enhance the microbial activity of the pile and can serve as substrates for the nitrification and denitrification processes leading to the emission of N_2O. Furthermore, high moisture together with enhanced microbial activity at early stages of the process can lead to the creation of anaerobic spots for the formation of CH_4.

A summary of the amounts of CH_4 and N_2O generated during manure composting is shown in Table 1. The amounts of CH_4 emitted during composting are within 0.1 and 8.93 kg of CH_4 per ton of treated manure. This wide range may be affected by the pre-treatment of manure prior to composting (manure storage can represent an important source of CH_4) and also by the aeration system, windrow versus forced aeration (agitation favours CH_4 emissions) (Kebreab et al. 2006). The levels of N_2O emitted from manure composting piles varied from 0.046 up to 0.370 kg N_2O–N per ton of treated manure depending on the composting system. Aerated static piles usually increase the emissions of N_2O by preventing ammonia losses, which can be later oxidised to nitrates generating N_2O. The emission of N_2O–N from manure composting can represent up to 9.8 % of the initial N. These experimental results have been used by IPCC (2006) to propose that default emission factors are 4 kg CH_4 ton^{-1} and 0.3 kg N_2O ton^{-1} (Table 1) from the biological treatment of organic wastes (for different types of feedstock and composting operations).

Table 1 Summary of CH_4 and N_2O emissions reported in the literature for different typologies of wastes and composting technologies

Feedstock	Bulking agent	System	N_2O	CH_4	Reference
Manures					
Swine	Cardboard	Windrow	0.1 % of initial N	–	Kuroda et al. (1996)
Swine	Bedding (straw)	Static	9.8 % of initial N	12.6 % of initial OM	Szanto et al. (2007)
		Windrow	0.5 % of initial N	0.5 % of initial OM	
Swine	Sawdust	Static	3.7–4.7 % of initial N	1.0–1.9 kg CH_4 ton^{-1}	Fukumoto et al. (2003)
Swine	Sawdust	Static	3.0–9.3 % of initial N	–	Fukumoto and Inubushi (2009)
Swine	Barley straw	–	0.058 kg N_2O–N ton^{-1}	0.19 kg CH_4–C ton^{-1}	Sommer and Moller (2000)
Swine	Sawdust	Static	1.6 µg m^{-2} s^{-1}	5.2 µg m^{-2} s^{-1}	Park et al. (2011)
		Windrow	7.9 µg m^{-2} s^{-1}	7.5 µg m^{-2} s^{-1}	
Cattle	Bedding (straw)	Static	0.11 kg N_2O–N ton^{-1}	6.3 kg CH_4–C ton^{-1}	Hao et al. (2001)
		Windrow	0.19 kg N_2O–N ton^{-1}	8.1 kg CH_4–C ton^{-1}	
Cattle	Bedding (straw)	Windrow	0.077 kg N_2O–N ton^{-1}	8.92 kg CH_4–C ton^{-1}	Hao et al. (2004)
	Bedding (wood chips)		0.084 kg N_2O–N ton^{-1}	8.93 kg CH_4–C ton^{-1}	
Cattle (dairy)	House wrap	Static	0.370 kg N_2O–N ton^{-1}	1.14 kg CH_4–C ton^{-1}	Pattey et al. (2005)
Cattle (beef)			0.103 kg N_2O–N ton^{-1}	0.11 kg CH_4–C ton^{-1}	
Cattle (dairy)	Bedding (straw)	Static	0.046 kg N_2O–N ton^{-1}	–	El Kader et al. (2007)
	Bedding (straw)	Windrow	0.070 kg N_2O–N ton^{-1}	–	
Turkey	Wood shaving and straw	Static	0.091 kg N_2O–N ton^{-1}	–	Czepiel et al. (1996)
Cattle and horse (50:50)	Bedding (hay)	Windrow	0.32 kg N_2O–N ton^{-1}	–	Leytem et al. (2011)
Cattle (dairy)	Bedding	Windrow	0.90 g N_2O m^{-2} day^{-1}	13.5 g CH_4 m^{-2} day^{-1}	
MSW					
Food waste	Biochip	Static	1.36 kg N_2O–N ton^{-1}	–	He et al. (2001)
MSW	–	Static	0–0.24 kg N_2O–N ton^{-1}	4.5–9 kg CH_4–C ton^{-1}	Clemens and Cuhls (2003)
Separated organic household waste		Static	0.02–0.11 kg N_2O–N ton^{-1}	0.04–0.8 kg CH_4–C ton^{-1}	Amlinger et al. (2008)
		Windrow			
Organic fraction	Wood chips or pruning	In vessel	0.048 kg N_2O–N ton^{-1}	0.26 kg CH_4–C ton^{-1}	Colón et al. (2012)
Source-separated MSW	Wastes	Confined windrow	0.048 kg N_2O–N ton^{-1}	1.26 kg CH_4–C ton^{-1}	
		Turned windrow	0.160 kg N_2O–N ton^{-1}	3.28 kg CH_4–C ton^{-1}	
		Home composting	0.430 kg N_2O–N ton^{-1}	0.12 kg CH_4–C ton^{-1}	
Source-separated MSW	Yard waste	Windrow	261 mg N_2O–N m^{-2} day^{-1}	35 g CH_4 m^{-2} day^{-1}	Beck-Friis et al. (2000)
Other wastes					
Cattle mortalities	Barley straw	Windrow	0.97 % of initial N	0.55 % of initial C	Hao et al. (2005)
	Cattle manure and barley straw	Windrow	0.85 % of initial N	0.13 % of initial C	

Table 1 continued

Feedstock	Bulking agent	System	N$_2$O	CH$_4$	Reference
Animal meals	Straw and cotton	Windrow	0.07–0.11 % of initial N	–	Cayuela et al. (2012)
Biosolids	Wood ash	Static	0.32 kg N$_2$O–N ton^{-1}	–	Czepiel et al. (1996)
Grass	Soil	Static	0.054 kg N$_2$O–N ton^{-1}	5 kg CH$_4$–C ton^{-1}	Hellbrand (1998)
Hens mortalities	–	Windrow	0.003–0.004 kg N$_2$O–N ton^{-1}	52–120 kg CH$_4$ ton^{-1}	Dong et al. (2011)
Garden waste	–	Windrow	0.05 kg N$_2$O–N ton^{-1}	1.9 kg CH$_4$–C ton^{-1}	Andersen et al. (2010a)
Olive mill waste	Diverse manures and olive pruning	Windrow	0.01–3.36 g N–N$_2$O m^{-2} d^{-1}	1–147 g CH$_4$–C m^{-2} d^{-1}	Sánchez-Monedero et al. (2010)
Default emissions factors for CH$_4$ and N$_2$O emissions from waste composting from IPCC (2006)					
On a dry basis	0.6 g (0.2–1.6) N$_2$O per kg waste treated	10 g (0.08–20) CH$_4$ per kg waste treated	IPCC (2006)		
On a wet basis	0.3 g (0.06–0.6) N$_2$O per kg waste treated	4 g (0.03–8) CH$_4$ per kg waste treated			

Municipal solid wastes (MSW)

Municipal solid wastes also represent a major source of organic wastes suitable for composting. This group includes not only mixed MSW but also other materials such as the organic fraction of the source-separated MSW, garden and yard wastes, food wastes. This type of composting substrates is characterised by lower organic matter, nitrogen and moisture content than manures. For this reason, the impact on GHG emissions is expected to be different, since lower amounts of organic C and N in the feedstock would lead to reduced GHG emissions (Brown et al. 2008; Büyüksönmez 2012).

Amounts of CH$_4$ emitted during MSW composting varied from 0.12 up to 9 kg CH$_4$ per ton of treated waste (Table 1). This large variability in gaseous emissions reflects the impact of the feedstock, the composting system and the efficiency of the composting facility on GHG emissions (Colón et al. 2012). The levels of N$_2$O emitted from MSW composting ranged from 0 to 0.430 kg N$_2$O–N per ton of treated waste, which represents values generally lower than those registered from the biological degradation of manure. In the case of MSW, where most of the composting piles are operated with little amounts of water, the small amount of CH$_4$ generated in the pile is most likely oxidised when it reaches the aerobic surface, considering CH$_4$ emissions to be essentially zero from a practical point of view, as far as life cycle assessments are concerned (US EPA 2006).

Other organic wastes

Table 1 shows the CH$_4$ and N$_2$O emissions for a range of organic wastes used as feedstock for composting. The impact of the different wastes will depend on their physical–chemical composition. Organic wastes such as biosolids, characterised by high N and moisture contents, are expected to have a similar behaviour than manures, whereas other wastes such as cattle and hens mortalities or olive mill wastes can have different behaviour depending on their physical–chemical characteristics. Sánchez-Monedero et al. (2010) studied GHG emissions from composting piles prepared with olive mill wastes and different N sources and bulking agent observing that the peculiar characteristics of these wastes, characterised by a low degradation rate and low N levels, reduced the emission of GHG.

Reduction in greenhouse gas emitted from composting

Best practices for the minimisation of greenhouse gas emissions

GHG emissions from composting can be minimised through diverse actions undertaken from different points of

view: the material to be composted and the process performance.

Feedstock and initial mixture

The effect of the composition of the mixture of wastes to be composted is critical in the amount and type of emissions derived from the process. High moisture content and high bulk density have been related to higher GHG emissions. An excess of water reduces free air space (FAS) and creates anaerobic sites where methane can be formed (Tamura and Osada 2006). A correct level of FAS ensures the proper aeration of the composting material both in forced and natural aerated systems and prevents anaerobiosis (Ruggieri et al. 2009).

The biochemical composition of the material to be composted also plays an important role in gaseous emissions, especially the C/N ratio. However, the bioavailability of these nutrients determines the carbon and nitrogen dynamics along the process and the derived emissions (Cayuela et al. 2012). Consequently, the C/N ratio assessment should be based on the biodegradable content (Puyuelo et al. 2011). Co-composting of complementary wastes to obtain a balanced initial mixture with a balanced porosity and biodegradable C/N ratio should significantly reduce the GHG emissions of the subsequent composting process.

Composting process

The composting process can be undertaken in different industrial systems. A general classification is made as open and closed systems. Contrary to open systems, closed systems present the collection of the exhaust gases to a gas treatment system.

Closed systems include closed reactors such as rotatory drums and composting tunnels but also confined piles (with textile cover) or composting piles inside closed buildings with a gas management system. Plants with gas treatment systems present much lower environmental impact because process emissions are not released to the atmosphere (Colón et al. 2012). Discussion on how to treat GHG emissions is presented below. In this sense and according to published conclusions (Colón et al. 2012), a first technical recommendation to minimise GHG emissions would be to include gas treatment systems wherever possible.

Another important process parameter to consider is process temperature. Higher temperatures enhance volatile compounds volatilisation resulting in higher gaseous emissions (Pagans et al. 2006b; Cayuela et al. 2012).

Open systems as static piles, turned piles and aerated windrows at open air have been studied to better understand gaseous emissions dynamics related to aeration strategies: airflow and pile turning. Different authors have highlighted the importance of airflow in gaseous emissions

from composting. It is considered that a high airflow increases oxygen availability, avoiding anaerobic pockets and consequent methane formation, and dilutes gaseous emissions. On the negative side, a high airflow strips gaseous compounds present in the composting mass facilitating their volatilisation (Pagans et al. 2006a). Jiang et al. (2011) reported that an increase in the aeration rate reduced methane emissions but increased NH_3 and N_2O emissions. Pile turning enhances the composting process by providing matrix homogenisation (moisture and microorganisms redistribution) and particle size reduction. It also provides punctual oxygenation of the solid material and compaction correction. From a biodegradation point of view, turning is recommended to enhance the process. However, pile turning has been shown to have a negative effect on gaseous emissions, including GHG (Colón et al. 2012). The turning itself releases the entrapped gases within the pile. Ahn et al. (2011) reported that CO_2, CH_4 and N_2O fluxes increased after turning due to greater gas diffusion rates resulting from porosity increased after turning. They recommend avoiding pile turning in the first stage of the process if the oxygen concentration and temperature of the pile are in an appropriate range. In a second stage, when oxygen levels within the pile increase, the formed methane is oxidised to CO_2. These authors suggest considering a turning plan to minimise CH_4 emissions and maximise CH_4 oxidation within the pile. Park et al. (2011) also reported higher emissions in turned systems than in aerated systems. When considering methane and nitrous oxide as CO_2 equivalents, the non-aerated system provided the higher process emissions, followed by the turned system, the system aerated by natural convection and finally the forced aerated system, which presented the lowest process emissions. However, as pointed by the authors, when approaching the problem from an overall impact assessment, the energy consumed to aerate the pile contributes to total CO_2 non-biogenic emissions. The operational activities can contribute to GHG of composting process more than the decomposition process itself (Lou and Nair 2009).

LCA tools impute the impact of both process emissions and emissions related to energy consumption (operational activities, aeration, turning and mass displacement within the plant) to assess the comparison of different waste management systems. In this sense, turned pile composting systems resulted in an overall higher impact than aerated systems (confined aerated windrows and tunnel) because of fuel consumption and turning that implies the above-mentioned increase in gas emissions (Colón et al. 2012; Kong et al. 2012).

Final recommendations to minimise GHG emissions

From the text above, it can be stated that a critical point for the success of the composting process with minimal

gaseous emissions is the disposal of the material in piles with a suitable size and porosity to favour homogeneous oxygen distribution. In non-aerated systems, this would enhance natural convection. In aerated systems, it is recommended to adjust forced aeration to ensure aerobic conditions without providing air in excess. High air flows beyond oxygen needs can be justified to avoid the emissions increase due to high temperatures. To overcome these problems, new advanced controllers have been proposed to base the aeration of the oxygen uptake rate measured on-line (Puyuelo et al. 2010).

Besides the physical structure of the matrix, the mixture to be composted should present appropriate moisture content and a balanced biodegradable C/N ratio.

Despite of whether the composting system is open or closed, the operational activities that imply electricity or fuel consumption must be optimised to reduce the overall environmental impact of the process.

Finally, gas treatment (by biofiltration or other technologies) is recommended when possible as the final solution to minimise gaseous emissions to the atmosphere.

Treatment of greenhouse gas emissions

A variety of technologies are available nowadays for treating emissions from composting processes. Selection of the best available technologies depends essentially on the composition and gas flowrate to be treated. Among such technologies, chemical scrubbing combining acidic plus caustic scrubbers coupled to biological processes such as biofilters are the most common technologies installed in full-scale facilities (Artola et al. 2009). However, current reactors design and operation focuses on treatment of VOCs and ammonia as main pollutants contained in composting emissions while low attention has been paid to GHG treatment. In any case, biological systems still are the preferred alternative from an economical and environmental point of view for GHG removal since the low concentrations of GHG in composting emissions make existing physical–chemical technologies non-viable from an economical point of view.

Acidic scrubbers preceding biofilters are installed to reduce the large ammonia loads often generated during composting. Otherwise, ammonia may inhibit AOB and/or NOB that, concurrently, would hinder the performance of the biofilters (Gabriel et al. 2007). Caustic scrubbers are often installed to remove acid gases such as hydrogen sulphide and to absorb highly soluble VOCs emitted such as alcohols. Biofilters have demonstrated to work well as end-of-pipe systems to treat a variety of odorant compounds found in off-gases from composting systems.

Design and operating conditions of chemical scrubbers and biofilters do not provide suitable conditions for the treatment of GHG. Dimensionless gas–liquid Henry coefficients for N_2O, NO, CH_4 and CO of 1.7, 21.5, 29.2 and 43.1 (Sander 1999), respectively, indicate that GHG are sparingly soluble in water. Except for N_2O, which can be considerate as moderately soluble in water, large gas contact times in the treatment system are required to solubilise significant amounts of NO, CH_4 and CO which, consequently, leads to large reactor volumes and investment costs. In addition, the relatively low concentrations of GHG in the gas phase provide low driving force for GHG mass transfer from the gas to the liquid/biofilm phase. Chemical scrubbers generally operate at gas contact times below 2–3 s and retain large amounts of water within the packed bed to facilitate the absorption of soluble compounds (Gabriel and Deshusses 2003). Instead, biofilters are generally operated at gas contact times between 20 and 40 s for the treatment of composting off-gases with low to no external supply of water to improve sorption of poorly soluble compounds (Gabriel et al. 2007; Pagans et al. 2006a, b). Altogether it leads to reduced elimination capacities of GHG in both systems in conventional chemical scrubbers and biofilters.

A short number of references exist about GHG treatment capacities in biofilters from composting emissions, even if several references exist about CH_4 removal by biofiltration. The latter has been addressed by several authors and shown as an effective technology for biofiltration of landfill biogas or gaseous emissions from the piggery industry (Nikiema et al. 2007; Girard et al. 2012). Moderate-to-large CH_4 concentrations of such gases are partly responsible for such effectiveness and treatment capacities. In composting facilities with biofilters, where much lower methane concentrations are found, removal efficiencies between 33 and 100 % have been reported for CH_4 (Boldrin et al. 2009).

In the case of N_2O emissions, Amlinger et al. (2008) reported that additional N_2O may be synthesised during the oxidation of NH_3. Also, Maia et al. (2012) found a clear correlation between the NH_3 load and the N_2O production in a compost biofilter, demonstrating that NH_3 removal was a trigger for N_2O production. Clemens and Cuhls (2003) studied the emission of direct and indirect greenhouse gases in a MBT facility. They also found that biofilters had no net effect on CH_4 and approximately 26 % of the NH_3 that was removed in the biofilter was transformed into N_2O when NH_3 was the exclusive nitrogen source due to nitrifier denitrification.

Regarding NO emissions, Barnes et al. (1995) showed that removal efficiencies up to 90 % can be achieved in a compost biofilter for NO concentrations of 500 ppm_v at a gas contact time of 60 s if an external carbon and energy source were added. Similarly, Yang et al. (2007) found that NO concentrations in the range of 200–500 ppm_v can be treated in aerobic and anoxic biofilters with a strong

influence of the O_2 percentage on NO removal. Even if hardly difficult to implement in composting facilities, anoxic conditions were reported to almost double NO removal compared to aerobic biofilters.

It is interesting to notice that almost no studies exist concerning CO biofiltration. Prado et al. (2008) reported CO removal efficiencies higher than 80 % for low (40 ppm$_v$) CO concentrations from synthetic resin-producing industries in a biofilter operated at above 30 s gas contact time. Further optimisation showed that a maximum elimination capacity of 33 g CO m^{-3} h^{-1} could be obtained with a mixture of lava rock and peat as packing material with more than 85 % removal efficiency at gas contact times of 3 min or more, suggesting that biofiltration offers potential for the biological removal of CO from polluted gas streams (Jin et al. 2009).

Reported data on GHG removal in chemical scrubbers are inexistent. However, one can infer from biofilters design, operating conditions and performance that GHG removal efficiencies in chemical scrubbers are probably very close to zero mostly due to the extremely reduced gas contact time of the gas in the scrubber coupled to the reduced solubility of most GHG.

Most of the research efforts on biological processes for GHG removal have been directed towards the use of existing bioreactor configurations (bioscrubbers, biotrickling filters or biofilters) while improving methane solubility using other solvents different to water. As reviewed by Muñoz et al. (2007), two-phase partitioning bioreactors (TPPBs) provide a non-aqueous phase (e.g. hexadecane, silicone oil) to an aqueous phase that contains the microorganisms responsible for degrading CH_4. Larger CH_4 mass transfer coefficients are encountered in TPPBs compared to conventional biofilters. Thus, improved solubilisation of hydrophobic compounds and, concomitantly, CH_4 elimination capacities are found. Rocha-Rios et al. (2009) reported increases of 131 and 41 % in the specific and volumetric CH_4 elimination capacity, respectively, in a biotrickling filter when silicone oil was added compared to the elimination capacities without silicone oil addition. However, silicon oil is expensive and difficult to manipulate which may hinder its use in full-scale systems. Alternatively, non-ionic surfactants do not pose the above-mentioned problems and have shown to improve CH_4 elimination capacities in biofilters, even if some growth problems may exist leading to decreased biomass accumulation in the packed bed due to their detergent character (Ramirez et al. 2012). Similarly, ionic liquids have shown to largely improve non-methane-VOCs absorption in biological reactors without much toxicological issues (Quijano et al. 2010; Darracq et al. 2012). Such ionic liquids can be specifically designed based on the characteristics of the gaseous compound to be selectively separated (Carvalho and Coutinho 2011), which provides potential application for improving CH_4 absorption in biofilters and biotrickling filters.

Overall, there are a number of opportunities to improve GHG removal by means of biological reactors. While CH_4, CO and NO can be treated to a certain extent in conventional biofilters already installed in full-scale composting facilities, N_2O has been shown to be generated rather than removed in biofiltration systems. Thus, research efforts should be directed towards reducing N_2O generation during the composting process and improving biofiltration conditions to reduce its production. Also, proper characterisation of current biofiltration systems installed in composting facilities in terms of GHG treatment capacities is necessary to gain specific knowledge. Finally, design and operating conditions of end-of-pipe systems should not be based only on odours and ammonia removal but also on GHG loads.

Conclusions

GHG from composting are an important issue for research and for improvement in real-scale composting facilities. From this review, it is evident that now GHG can be accounted, measured and properly characterised. However, it is clear that the disparities of emissions factors for the different GHG that can be found in scientific literature are due to several factors:

1. The diversity of wastes and technologies used for the composting of organic wastes.
2. There is wide margin to minimise the GHG emissions from composting, by changing or updating the current facilities and by improving the performance of the treatment technologies.
3. The beneficial uses of compost must be also investigated, since it is not clear if the GHG emitted during the process are compensated by this compost utilisation in the long term.
4. From a life cycle assessment perspective, it is necessary to have experimental data both on the GHG emissions and the efficiency of the process, to have a fair evaluation of the environmental impacts of composting.

Further research is necessary to solve these limitations and to provide reliable emissions factors for composting processes and, in general, for any biological technology for waste treatment.

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