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Composting from a Sustainable Point of View: Respirometric Indices as Key Parameter

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ABSTRACT

Composting at an industrial scale can be performed using low technology processes, such as windrows, or by implementing more complex technologies such as tunnels or, in general, in-vessel systems. In both cases process control can be done via measurement of oxygen content in the exhaust gases (or as interstitial oxygen in the material) and/or by monitoring the temperature evolution of the material. However, the use of respiration indices (RIs) as a control parameter to obtain reliable information on the actual microbial activity is being increasingly studied. Also RIs are used to determine the biological stability of the final product or the biodegradability of the wastes intended to be composted. In this case, the RI value can be related to the amount of biodegradable organic matter content. As a new application, RIs can also be used to determine the environmental impact of composting plants. Indeed, emission factors of pollutant gases (ammonia, Volatile Organic Compounds, etc.) or consumption of resources (water, energy, electricity) can be referred to the resulting reduction of RI obtained during the entire composting process. In this case, RI might be a promising parameter for the comparison of composting technologies from the point of view of its sustainability or to define the critical phases of the process in terms of environmental impact (for instance, treatment of exhaust composting gases by biofiltration). In fact, studies on composting sustainability should consider not only the composting process but the equipment used for the treatment of its emissions, which has an important effect on the global environmental impact of waste treatment plants. This paper discusses the above mentioned topics, with RIs as the key parameter in the analysis of composting processes. The discussion will be based on the experience of our previous research on composting and gas cleaning.

Keywords: atmospheric emissions, biofiltration, composting, environmental impact, biological stability, respiration index
Abbreviations: DM, dry matter; **DRI**, Dynamic Respiration Index; **EBRT**, Empty Bed Retention Time; **GHG**, Greenhouse Gas; **LCA**, Life Cycle Assessment; **MBT**, Mechanical-Biological Treatment; **MSW**, Municipal Solid Waste; **OFMSW**, Organic Fraction of Municipal Solid Waste; **OM**, organic matter; **OUR**, Oxygen Uptake Rate; **RI**, Respiration Index; **SOUR**, Specific Oxygen Uptake Rate; **SRI**, Static Respiration Index; **VOC**, Volatile Organic Compound; **VS**, volatile solids

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Invited Review

INTRODUCTION

The role of composting in modern waste management

Solid waste management, and particularly its organic fraction, is becoming a global problem in developed countries. At present different technologies are being applied to reduce landfill destination of organic wastes (European Commission 1999a), as this management is responsible for a considerable contribution to global warming (Mor *et al.* 2006). Among the emerging technologies to treat the organic fraction of municipal solid wastes, anaerobic digestion and composting are environmentally friendly technologies that allow treating and recycling organic wastes. In fact, municipal solid wastes and sewage sludge are considered two of the main waste streams generated in Europe. According to Eurostat (2009), the generation of municipal solid waste in Europe in 2007 was 522 kg per person per year, whereas sewage sludge production from municipal wastewater treatment plants is within 25-100 kg per person per year depending on the country.

In the case of municipal solid wastes, the international policy on management of household waste has been increasingly directed towards recycling in recent years. The organic fraction of municipal solid wastes composed of kitchen wastes, yard wastes and pruning wastes can account for about one half of the totality of household waste generated. Among the available technologies, composting is presented as one of the most promising options to recycle the organic fraction into a valuable organic fertilizer popularly known as compost.

On the other hand, the amount of wastewater sludge produced is expected to increase all over the world. At present, land application is the main disposal mode used for wastewater sludge, and the unique legal restrictions for soil application are its heavy metals content and the presence of potentially toxic compounds. However, the land spreading of sludge must be carried out ensuring effective pathogen elimination and maximizing its positive agronomic aspects (Larsen *et al.* 1991; Wang *et al.* 2003). Both aspects are solved when composting is previously used before land application. In fact, wastewater sludge composting with the use of bulking agents can enhance the biological stability of organic matter, inactive pathogens and parasites and enable the production of a quality product that may be used as a soil conditioner or as an organic fertilizer.

These two examples show the huge potential of composting and compost as a key role in the management of organic solid wastes, which, of course, can be applied to other wastes generated in high amounts such as agricultural wastes, manure, animal by-products, carcasses, sludge from paper manufacturing, etc.

The composting process

Although there is no universally accepted definition of composting, Haug (1993) uses a practical definition of the process, which provides all the main points to obtain a successful process: "Composting is the biological decomposition and stabilization of organic substrates, under conditions that allow development of thermophilic temperatures as a result of biologically produced heat, to produce a final product that is stable, free of pathogens and plant seeds, and can be beneficially applied to land."

This practical definition is extremely useful for scientists, technicians and specially composting plant managers, since it states the three main characteristics of the process:

1) **It is a biological aerobic process:** this means, on the one hand, that the process conditions must be adequate for the development of microbial communities, although compost microbiology has not been studied in detail because of the inherent difficulty of cultivating microorganisms coming from heterogeneous solid samples. However, in recent publications, new microbial techniques have been successfully applied to the identification of viable compost strains (Amir *et al.* 2008). On the other hand, composting is aerobic. This implies that oxygen must be effectively transferred from air to the cells and carbon dioxide must be transported from cells to exhaust air. Several methods have been proposed in different technologies to provide oxygen to composting materials. In the passive aeration method, oxygen supply is achieved by means of the natural convective movement of the air through the pile (Mason *et al.* 2004). To achieve this, the size and porosity of the pile should be adequate to enable aeration (Szanto *et al.* 2007). Turned composting systems are passively aerated but additional turning is used to maintain the proper porosity, to provide oxygen, to mix the material and to release excessive heat, water vapour and carbon dioxide. In static forced-aerated pile composting, forced aeration is applied by means of air ducts, and aeration is provided by blowing or sucking air through the composting material, which must present an adequate level of porosity (Haug 1993).

2) **Composting deals with organic solid wastes:** this is especially important because of the inherent heterogeneity of organic substrates, which causes that sampling in composting processes is not s simple task (Barrena *et al.* 2006a). Also, the existence of several phases (gas, liquid and solid) implies that transfer mechanism for both mass and energy can limit the overall process rate. In the case of mass transfer, oxygen diffusion is the key issue (Scaglia *et al.* 2000), whereas energy is, in general, poorly transferred through organic matter (Barrena *et al.* 2006b), causing the self-heating of compost that eventually results in material sanitation and stabilization.

3) **Compost must be sanitized and stable:** compost application must be carried out in a safe manner. This implies that proper conditions of sanitation must be ensured during the process. In fact, it is considered that the high temperature reached due to the metabolic heat generated during the thermophilic phase of the composting process is effective in destroying the pathogens (Wong and Fang 2000). To regulate this point, several recipes have been proposed to ensure compost sanitation. For instance, in sludge composting, different combinations of temperature and time are indicated in order to reach the proper disinfection of the final product (temperatures over 55°C, 20 days for conventional aerobic treatments or 20 hrs for 55°C for advanced aerobic stabilization treatments) (European Commission 2000). Other international rules on sludge disinfection by composting propose similar time–temperature conditions (US Environmental Protection Agency 1995). In the animal by-products category, European legislation describes the exact sanitation conditions to be ensured for a proper composting process (Regulation (EC) No 1774/2002). In relation to biological stability, this is an important issue for composting process performance and obviously for compost quality and it will be discussed in detail in this review.

Composting technology

Haug (1993) gives an extended description of the composting systems developed at that moment. Although new systems have been implemented since then, from a practical point of view, composting plants for the processing of organic solid wastes can be still divided into two main categories:

1) **Piles or windrows:** for rural or semi-rural areas, usually plants with a capacity range of 1,000-40,000 metric tons/year. This traditional composting method was implemented in the first composting plants constructed in the world, and it is based on the use of mechanically turned or forced-aerated static piles. Typically, temperature, moisture and oxygen content inside the material are monitored during the first weeks of composting (thermophylic initial phase), and weekly during the curing phase (mesophylic final phase). Total composting time is usually about 12-13 weeks.

2) **In-vessel or reactor systems:** for urban and high-

density population areas, these plants have a capacity range of 10,000-100,000 metric tons/year. In this case, material remains for few weeks in a digester (tunnels are the most popular) with forced aeration systems and on-line monitoring of temperature, oxygen, carbon dioxide and ammonia exhaust gases. Data from different probes are computer collected and some control recipes can be applied to the system, usually in the form of temperature and oxygen set-points, allowing a rapid decomposition of organic matter. Afterwards, material is piled for the curing phase for 5-6 weeks. Typically, in-vessel plants are composed of the composting reactors and the gas collection, transportation and cleaning units (being biofilters the most popular).

Of course, there exist a lot of modifications and variations of these two main processes, whose suitability mainly depends on the properties of the feedstock to be composted. The study of the performance of these plants according to the evolution of global biological activity indicators has been the main objective of some recent research works (Barrena *et al*. 2008; Ponsá *et al*. 2008; Ruggieri *et al*. 2008), and these will be discussed in detail in this review.

Composting as a biological process. Parameters affecting biological activity

In order to control and optimize the bio-kinetics of the composting process to produce a compost of desired quality, it is important to understand the factors that influence the process. A composting matrix is an ecosystem of interdependent interactions between biotic and abiotic factors that cause degradation of organic matter. The abiotic and biotic factors playing key role in the composting process (Pietronave *et al.* 2004; Gajalakshmi and Abbasi 2008) are described next.

1. Abiotic factors

Nature of the substrate: Several kinds of organic residues susceptible to the enzymatic activities of the microorganisms can be converted into compost if necessary conditions for biodegradation are provided. As the substrate becomes the only source of food to the microorganisms in a composting matrix, the nature of the substrates is the most controlling factor in any composting process (Gajalakshmi and Abbasi 2008). The organic compounds in biowastes could be hence classified into three main categories (Komilis *et al*. 2004; Gajalakshmi and Abbasi 2008): (1) carbohydrates (polymers and simple sugars), (2) lignin, and (3) nitrogen compounds. In the beginning of the composting process, simple carbohydrates are converted to carbon dioxide and water (Bernal *et al.* 1998), and degradation of nitrogenous compounds results mainly in ammonia volatilization. In the later stages of composting, cellulose and hemicellulose are utilized by the compost microflora and eventually lignin is also subjected to slow degradation. Besides mineralization, organic matter is converted to humic substances (Tuomela *et al.* 2000; Quagliotto *et al.* 2006).

Carbon/Nitrogen ratio: The relative proportion of carbon and nitrogen is also a major controlling factor in the composting process (Hansen *et al.* 1989; Richard 1992; Ekinci *et al.* 1999; Agnew and Leonard 2003). Carbon serves primarily as an energy source for the microorganisms, while a small fraction of the carbon is incorporated to the microbial cells. Nitrogen is critical for microbial population growth (Gajalakshmi and Abbasi 2008). If nitrogen is limiting, microbial populations will remain small and decomposition rates for available carbon will be lower. Excess nitrogen is lost from the system as ammonia gas (de Guardia *et al.* 2008). According to Golueke (1992), rapid and entire humification of substrates by the microorganisms primarily depends on it initially having a C/N ratio between 25 and 35. Anyway, it must be noted that the biodegradable C/N ratio can be significantly different from typical C/N chemically determined (Sánchez 2007).

Moisture: Moisture is one of the composting variables that affects microbial activities to a considerable extent since it provides a medium for the transport of dissolved nutrients (Hamelers 2004) required for the metabolic and physiological activities of microorganisms (Richard *et al.* 2002; Agnew and Leonard 2003; Mohee and Mudhoo 2005; Iyengar and Bhave 2006).

Oxygen, temperature and aeration interaction: The microbial decomposition process enhances the interdependence and mutual control between two of the main composting parameters, oxygen levels and temperature. The temperature within a composting matrix determines the rate at which many of the biological processes take place (MacGregor *et al.* 1981; Stombaugh and Nokes 1996; Agnew and Leonard 2003; Cekmecelioglu *et al.* 2005; Mason 2006; Richard and Walker 2006) and controls the development and the succession of the microbiological flora (McKinley and Vestal 1984; Mustin 1987; Liang *et al.* 2003; Taiwo and Oso 2004). A temperature in the range of 55 to 65°C allows for considerable destruction of pathogenic organisms (Finger *et al.* 1976; Finstein *et al.* 1987; Noble and Roberts 2004; Smith *et al.* 2005).

pH: pH also significantly affects the composting process (Ekinci *et al.* 1999; Sundberg *et al.* 2004; Sundberg and Jönsson 2008). The range of pH values suitable for bacterial development is 6.0-7.5, while fungi prefer an environment in the range of pH 5.5-8.0 (Zorpas *et al.* 2003).

2. Biotic factors

Composting involves a myriad of microorganisms (Hassen *et al.* 2002; Narihiro and Hiraishi 2005). The composition and magnitude of these microorganisms are important components of the composting process. The microbes decompose the organic matter, and transform the nitrogen component through oxidation, nitrification, and denitrification (Golueke 1992; Tiquia and Tam 2000). Bacteria play the dominant role during the most active stages of composting process because of their ability to grow rapidly on soluble proteins and other readily available substrates (Strom 1985a; Epstein 1997). Strom (1985b) reports that as much as 87% of the randomly selected colonies during the thermophilic phase of composting belong to the genus *Bacillus*. The role of fungi starts when simple, easily degradable substances such as sugar, starch, and protein are acted upon by bacteria and the substrate is predominated by cellulose and lignin, which normally occurs toward the curing stage of the composting process (Bertoldi and Vallini 1983; Tiquia *et al.* 2002). Most fungi are eliminated by high temperatures (Epstein 1997), but they commonly recover when temperatures are moderate (Tiquia *et al.* 2001), and the remaining substrates are predominantly cellulose or lignin (Bertoldi and Vallini 1983). Like fungi, actinomycetes also utilize complex organic material. They tend to grow in numbers in the later stages of composting, and have been shown to attack polymers such as hemicellulose, lignin, and cellulose (Bertoldi and Vallini 1983; Epstein 1997). Actinomycetes are able to degrade some cellulose and hydrolyse lignin, and are tolerant of higher temperatures and pH than fungi. Thus, actinomycetes are important for lignocellulosic degradation during peak heating. Actinomycetes are thus well adapted to exploit the compost environment as the piles cool in the immediate post peak heat phase.

Different microbial communities predominate during the various composting phases, each of which being adapted to a particular environment (Bagstam 1978). Primary decomposers create a physico-chemical environment suited for secondary organisms, which cannot attack the initial substrates, while metabolites produced by the one group can be utilized by the other (Golueke 1992). The initial rapid increase of temperature involves a rapid transition from mesophilic to thermophilic microflora (Ryckeboer *et al.* 2003a). Often a disruption of the process is observed at temperatures between 42 and 45°C. The initial mesophilic microflora is inhibited by the high temperature, while the thermophilic populations have not yet developed and are below their temperature optimum.

Only when a sufficient number of thermophiles is generated, temperatures rise again. At temperatures exceeding 60°C, the optimum for most thermophiles is reached, and the system starts to limit itself due to the inhibitory high temperatures (McKinley and Vestal 1984). Heat may in principle inhibit organisms through enzyme inactivation (Gajalakshmi and Abbasi 2008) or may limit oxygen supply. An efficient process kinetics control thereupon provided through regular aeration, the thermophilic stage continues until the heat production becomes lower than the heat dissipation, due to the exhaustion of easily degradable substrates. High temperatures support degradation of recalcitrant organics (Tuomela *et al.* 2000) and elimination of pathogenic and allergenic microorganisms (Herrmann *et al.* 1994; Ryckeboer *et al.* 2002). During the second mesophilic (cooling) phase nutrients become a limiting factor, causing a decline in microbial activity and heat output. During the maturation phase, the substrate quality further declines and compounds such as lignin-humus complexes are formed that are not further degradable. As corollary, the inherent complexity of substrates and intermediate biochemical reactions and their products, make the microbial diversity and the succession of populations vital in ensuring an efficient bio-kinetic process control and biodegradation during the composting process. While the present review paper may not encompass the entire microbiology of the composting process, the reader is encouraged to consult the more detailed literature survey and inventory of the mesophilic and thermophilic bacteria, actinomycetes and fungi isolated during several phases of composting by Ryckeboer *et al.* (2003b).

RESPIRATION INDICES (RIs)

Respiration is directly related to the metabolic activity of a microbial population and expressed as respiration index it is a direct measure of the oxygen consumption of the microbiological communities present in an organic material. Microorganisms respire at higher rates in the presence of large amounts of bioavailable organic matter while respiration rate is slower if this type of material is scarce. In the composting process respiration activity has become an important parameter for the determination of the biological stability of compost. In addition, the biological activity can be considered a measure of biodegradability and it can also be used for the monitoring of the composting process. The implementation of this parameter can be very helpful in the design of waste treatment facilities (composting, anaerobic digestion and mechanical-biological treatment plants). Other methods based on biochemical determinations as volatile solids (VS), total organic carbon (TOC) or chemical oxygen demand (COD) have been traditionally used to monitor the organic matter evolution in biological processes as composting (Komilis and Ham 2003; Ros *et al.* 2006). However these parameters lack of precision when applied to heterogeneous organic wastes because of the small amount of sample used and the presence of non-biodegradable volatile or oxidable materials (presence of plastics, effect of bulking agent, etc.).

Hence, RIs appear as a promising tool for stability assessment, for analysis of biodegradation process efficiency, to establish which is the best biological treatment for a particular waste or as design parameter for the various technologies based on biological processes (Cossu and Raga 2008; Ponsá *et al.* 2008).

Methodology

Respirometric activity can be determined directly from the oxygen consumption or CO_2 production from a sample, and indirectly through the heat released during the process. The advantages and disadvantages of using oxygen or carbon dioxide in respirometry have been recently revised (Barrena *et al.* 2006c). Particularly, the methods that monitor CO_2 production have the disadvantage that they are unable to distinguish between CO_2 produced aerobically from that produced anaerobically and the development of local zones with anaerobic conditions could overestimate the respiration activity. On the other hand, these methods assume that the CO_2/O_2 ratio is always 1. However, this ratio can vary depending on the oxidation degree of the organic carbon. Moreover, monitoring of CO_2 evolution presents two major drawbacks: first, the solubility of CO_2 in aqueous solutions; second, this solubility is pH-dependent. This is particularly important when comparing respiration activities of different residues since their pH can vary over a wide range (Barrena *et al.* 2006c). Therefore, methods based on O_2 uptake are the most accepted for the determination of the biological activity of a material (Iannotti *et al.* 1993; Lasaridi and Stentiford 1998; Adani *et al.* 2001; Barrena *et al.* 2005; Tremier *et al.* 2005).

RIs are divided into static and dynamic methods on the basis that oxygen uptake measurement is made in absence (static respiration index, SRI) or in the presence (dynamic respiration index, DRI) of continuous aeration of the biomass (Scaglia *et al.* 2000). They can be performed either with solid or liquid conditions. SOUR (Specific Oxygen Uptake Rate) is a measure of biological activity in liquid state under static conditions using a solid sample suspended in water (Lasaridi and Stentiford 1998). SOUR has also been proposed as a suitable method for stability assessment with different materials at different processing times (Scaglia *et al.* 2007). In this sense, this methodology should not be considered a measure of the actual or potential biological activity in a solid state process, such as composting, since the organic matrix and microorganisms interactions are completely different. Thus, SOUR would be more a measure of total biodegradable organic matter under aerobic conditions and a good measure of biological stability, although that substrate could not be necessarily available in solid state. In consequence, methods in solid state are considered more suitable for solid state processes monitoring.

The SRI measures changes in O_2 concentration in the head space of a closed flask containing a compost sample of known volume and mass, at known temperature and barometric pressure. The decline in O_2 concentration over time is monitored with an O_2 electrode. This method has been used for various authors using similar procedures (Iannotti *et al.* 1993; US Department of Agriculture and US Composting Council 2001; Barrena *et al.* 2005). The dynamic respiration index is determined measuring the difference in O_2 concentration between the inlet and outlet of an air flow passing through a compost vessel. Different versions of the dynamic procedure can be found in literature (Scaglia *et al.* 2000; Adani *et al.* 2004; Tremier *et al.* 2005). Using dynamic respiration indices different authors have studied in depth the biodegradable organic matter decomposition. For instance, Barrena *et al.* (2009) have conducted a systematic study on the different chemical and biological methods to estimate the biological stability of compost samples during an aerobic process at industrial scale. These methods included dynamic respiration indices and biogas potential production tests. Dynamic respiration indices are based on the oxygen uptake rate obtained under continuous air supply by measuring the difference in oxygen concentration between the inlet and outlet air flow that passed through the material. On the contrary, biogas production tests are conducted under strict anaerobic conditions by measuring biogas or methane produced for a relatively long time. Other authors (Tremier *et al.* 2005) developed a method for characterizing the organic composition and biodegradation kinetics based in a dynamic respiration test. From the oxygen uptake profile these authors could estimate the easily and slowly biodegradable organic matter fractions in biomass. Scaglia and Adani (2008) developed an index for quantifying the aerobic reactivity of municipal solid wastes and derived waste products, called the putrescibility index. The revision of the different ways to express RI according to the existing methodologies can be found elsewhere (Barrena *et al*. 2006c; Ponsá *et al*. 2008).

Compost stability

The biological stability is defined as the measure of the degree of decomposition of biodegradable organic matter contained in a matrix (Lasaridi and Stentiford 1998). Compost requires a minimum level of biological stability to avoid problems during storage, distribution and use. During the storage and distribution a non stable material may cause reheating, odour production and deterioration quality. Once the material has been applied to the soil, there may exist a continuous decomposition of biodegradable organic matter that has negative effects on plant growth.

As discussed above, biological activity measurements have been suggested in the literature as a measure of biological stability or biodegradable organic matter content. In the European legislation drafts (European Commission 2001) 'stabilization' means the reduction of the decomposition properties of biowaste to such an extent that offensive odours are minimised and that either the respiration activity after four days (AT_4) is below 10 mg O_2/g dm or the Dynamic Respiration Index is below 1,000 mg O_2 kg^{-1} VS h^{-1}.

A number of standards for stability assessment have been already proposed (ASTM 1996; US Department of Agriculture and US Composting Council 2001; Cooper 2005) and are discussed below. Notwithstanding the amount and quality of the work referred to, there is no consensus for stability measurements within the research community in the solid waste treatment field (Barrena *et al*. 2006c).

International standards for compost

Some respirometric and biogas production methods have been considered in the European legislation drafts (European Commission 2001) and adopted in national regulations by some European countries such as Germany (Federal Government of Germany 2001), Italy (Favoino 2006) and England and Wales (Godley *et al*. 2005). Different limits have been established for the RIs for their use as a biological stability parameter. **Table 1** shows the test conditions for some of the national standards, defined for biological stability determination under aerobic and anaerobic conditions and the proposed stability limits. In this table, different indices are presented according to the nomenclature used in each national regulation. For instance, AT_4 and DR_4 are the cumulative oxygen consumption during 4 days (mg O_2/g dry matter), whereas GB_{21} and BM_{100} are the biogas and methane cumulative productions obtained during 21 and 100 days in normal litres per kg of dry matter, respectively. A detailed compilation of the limits proposed in other regulations can be found in Ponsá *et al*. (2008) and Barrena *et al*. (2006c). It is important to note that some confusion exists when applying respiration protocols probably because of lack of scientific assessment. The methodologies proposed differ in many key aspects such as the use of an inoculum, the amount of sample to be used and its preparation, the assay temperature (mesophilic or thermophilic) and the test duration. Furthermore, the way results are expressed can also be a source of confusion. For instance, RIs can be determined either from maximum values, as average of measurements made over 24 h or cumulative consumption, and the units, they can also be referred either to dry weight or to organic matter content.

Finally, a method based on respiration activity has been published recently at European level (European Committee for Standardization 2007) although it is not related to the composting field. This technical specification describes a method to determine the current rate of potential microbial self-heating of a solid recovered fuel (SRF) using the dynamic respirometric method. Spontaneous combustion can occur when SRF from municipal solid waste or biomasses are stored and/or transported. The method measures the amount of easily biodegradable organic matter of SRF and estimates the potential risk of microbial self-heating, odour production, vector attraction, etc.

Respiration indices as a measure of biological activity of composting materials

As stated in the previous section, RIs determine biological activity directly by measuring the oxygen consumed by the microorganisms in a waste sample. Other methodologies have been used for biological activity assessment. For instance, enzymatic activities have been widely used for composting process monitoring and have been proved to be a good tool both for global activity and specific substrates hydrolysis monitoring (Tiquia 2005; Gea *et al*. 2007; Barrena *et al*. 2008). Tiquia (2005) compared six different biological parameters, including OUR, as indicators of compost maturity and concluded that dehydrogenase activity was the most suitable, simple and rapid methodology. However, the methodology used by Tiquia to measure OUR was a more complicated procedure than those described in the previous section. Barrena *et al*. (2008) correlated dehydrogenase activity to SRI. In general, enzymatic procedures present several drawbacks when compared to RI such as the use of hazardous reagents, long time required for analysis, or multiple manipulation steps. Moreover many enzymatic activities can present specific inhibitions or are specific for the substrate they catalyze. On the contrary RIs are reliable methods for overall activity determination which can be assessed using relatively simple set up and procedures. ATP concentration can also be used for biological activity determination in composting processes (Tiquia 2005) and has been proved to correlate well with SRI (Montes and Sánchez 2008). However this methodology presents similar drawbacks to enzymatic activities and on the other hand is less sensitive than RIs.

Therefore, RIs are the most effective and reliable tool for biological activity measurement. Both static (Barrena *et al*. 2005) and dynamic (Adani *et al*. 2004) indices have been proved to characterized the different level of biological activity found among fresh waste samples and stabilized samples by composting, biostabilization or biodrying.

RIs are usually estimated *ex-situ* in respirometers although they also have been estimated *in-situ* at laboratory scale using pilot scale composters with complete monitoring of exhaust gas composition. In this case RI is a measure of the actual biological activity in the process and it can be consider the best parameter for process monitoring

Table 1 Stability indices proposed in some European regulations (updated from Ponsa *et al*. 2008).

Reference*	Inoculation	Water Content	Temperature	Test duration	Results expression**	Stability limit
Federal Government of Germany 2001			Abfallablagerungsverordnung – AbfAblV			
AT_4	No	40 g, saturation + vacuum filtration	20ºC	4 days + lag phase	mg O_2/g DM	5
GB_{21}	yes	50 g DM + 50 mL inoculum + 300 mL water	35ºC	21 days + lag phase	L/kg DM	20
Godley *et al*. 2005			United Kingdom Environment Agency			
DR_4	yes	400 g, 50% moisture	35ºC	4 days	mg O_2/g DM or mg O_2/g VS	No limit proposed
BM_{100}	yes	20 g VS + 50 mL inoculum + 200 mL solution	35ºC	100 days	L/kg VS	No limit proposed

* DRI, AT_4 and DR_4 are respiration indices (oxygen consumption), whereas GB_{21} and BM_{100} are anaerobic indices (biogas production)
** DM: Dry Matter; VS: Volatile Solids

as it is determined under real process conditions and it is able to reflect operational problems. When RI is determined in a respirometer under optimal conditions of moisture and aeration rate this parameter gives an indication of the potential activity of a sample or in other words its biodegradable organic matter content.

Consequently, RIs have been successfully used for process monitoring and performance analysis. It has been used as a decision tool for example to select process conditions or to compare different process systems and configurations. Barrena *et al.* (2007) used SRI to monitor biological activity in a co-composting process of sludge with hair wastes from leather industry. SRI clearly indicated a significant biodegradation of both sludge and hair wastes and was sensitive enough to detect the decline of biological activity in the process. Barrena *et al.* (2006a) also used this tool to analyze the effect of three different inoculation doses in the composting process of organic fraction of municipal solid wastes (OFMSW, mainly food and yard wastes). SRI showed a faster decrease in biological activity when using high inoculum doses and thus indicated a clear acceleration of the process. Ruggieri *et al*. (2008) used SRI to compare the performance of three different composting pile configurations for OFMSW composting: turned pile, static forced aerated pile, and turned forced aerated pile. SRI evolution demonstrated the need for turning when composting highly heterogeneous materials.

Additionally, RIs have been used to study complex installations as mechanical-biological treatment (MBT) plants. Ponsá *et al*. (2008) selected and characterized several stages of a MBT plant processing MSW and OFMSW: waste inputs, mechanically treated wastes, anaerobically digested materials and composted wastes, according to the treatment sequence used in the plant. In this work, obtained values of RIs were used not only for waste characterization but to determine the efficiency of the different operation units involved in the MBT plant operation in reducing the biodegradable matter content of wastes. These authors also used anaerobic parameters as the cumulative biogas production in 21 (GB21) and 100 days (GB100) and found these to highly correlate to SRI. The long time required for the anaerobic analysis again points to RI as the simplest and fastest procedure. Similarly, Barrena *et al*. (2009) characterized samples from a MBT plant collected at different processing times (0, 32, 42 and 63 days of process). In this work authors analyzed static and dynamic RIs, as well as GB21, among other parameters. All RIs analyzed correlated well and were proved to be a suitable and reliable tool for biodegradation process monitoring. Anaerobic indices were again not recommended due to the long analysis time required. Typical chemical parameters such as Volatile Solids (VS), Chemical Oxygen Demand (COD), Total Organic Carbon (TOC) or Dissolved Organic Carbon (DOC) (APHA, 1998) were found to be not sensitive enough to reflect differences in the organic samples due to biodegradation and did not correlate with global measures of biological activity determined by means of RIs.

Finally, one last question arises: which is the best respirometric method available? As stated before, some authors have compared different methodologies, aerobic respiration indices, anaerobic or chemical parameters (Adani *et al*. 2003; Godley *et al*. 2005; Adani *et al*. 2006; Barrena *et al.* 2009). Some of these works concluded that static procedures might lead to the underestimation of biological activity due to diffusion limitations in oxygen transfer. Later, it has been demonstrated that both static and dynamic indices provide the same information when assessed under the appropriate conditions (Barrena *et al*. 2009). However the static procedure presents one important drawback. In those samples that might present a long acclimation phase the incubation period provided for the static measure could be not long enough and SRI might result in an underestimation of the biological activity. Since DRI continuously monitors OUR until the maximum activity is reached, this parameter is recommended as the most suitable. In addition, dynamic indices allow for a longer observation period of oxygen uptake profile and thus, they provide more information about the sample biodegradability (Tremier *et al*. 2005). On the other hand, dynamic procedures allow for the determination of RIs expressed as specific rates or as cumulative consumptions. Since both parameters correlate well, specific rates are recommended, to avoid the longer times of analysis required for total cumulative consumption determinations.

ENVIRONMENTAL IMPACTS OF THE COMPOSTING PROCESS

There are some inherent impacts associated to the composting process and, in general, to organic wastes recycling in large-scale facilities that should be pointed out. Odour emissions and atmospheric pollution are the most common and could be classified as impacts on the near environment of the facility. Composting plants, as other waste treatment plants in which an aerobic biodegradation process of the waste organic fraction takes place, represent a common source of gaseous compounds that can be the cause of odour nuisance to the near living population (Eitzer 1995; Smet *et al.* 1999). In fact, problems related to odour emissions are often the limiting factor to the activity or construction of a composting plant (Sironi *et al.* 2006). Odour emissions from organic waste treatment plants are commonly a complex mixture of a wide number of organic and inorganic compounds that can be studied and determined individually or as an odour nuisance source using olfactometric techniques.

In addition to gaseous emissions, energy and water consumption coupled with leachate generation should also be considered in environmental impact determination. Energy consumption should be considered in a global perspective in environmental impact studies.

Odour nuisance

Odour emissions are the main disadvantage of the composting process, particularly during the decomposition phase. Thus, odour nuisance is a potential problem in composting facilities, which causes a significant number of social complaints. Odour annoyance is an increasingly important environmental concern for both industrial facilities trying to control these inconveniences and for the public administration trying to set standards and regulations. In consequence, efficient management of potential olfactory nuisance is based on technical know-how and relations with residents.

Along the composting process, a range of odorous compounds is produced at different levels, which generally depend on the raw materials composted, the composting technology, stage, operating conditions etc. Odour impact of composting emissions can be significantly diminished with proper design and operation of the composting process, even if collection and treatment of waste gases is usually performed, particularly when composting sites are close to residential areas. Chemicals and compounds producing odour are measured by means of analytical methods such as GC/MS, even if odour concentration measurement by dynamic olfactometry has become the standard to set regulatory limits at international level. Ammonia in composting emissions is a good example for the rationale laying behind such selection. According to Bouchy *et al.* (2008), ammonia accounted for up to 90% of the mass flow of odorous compounds at a composting facility, while ammonia only contributed with a 7% to the total odour concentration measured. Also, other compounds such as amines or reduced sulphur compounds found in very small concentrations in composting emissions cannot be reliably analyzed by GC/MS (Hobbs 2001) but they can be detected and recognized by olfactometry. In general, the low olfactory detection threshold of most of the compounds emitted during the composting process make odour impact assessment a need when designing a facility and during its operation.

Fig. 1 Example of dispersion modelling output: odour concentration mapping at an industrial facility.

Despite of limitations in sampling, stability of samples or uncertainty due to variability in sense of smell among the observers, olfactometry results are often used for predicting odour impact using dispersion modelling (Hayes *et al.* 2006), even if this does not define nuisance impact. In fact, several regulations at international level require the use of dispersion modelling to ensure a maximum odour concentration, generally between 3 and 5 OU_{EU}/m^3 at 98% percentile, in the surroundings of facilities. Additionally, Hayes *et al.* (2006) indicate that atmospheric dispersion modelling can be used to calculate approximate setback distances for new units and to locate the units appropriately and to estimate the maximum odour emission permitted and which abatement techniques to prevent odour complaints occurrence. Odour modelling output will provide an odour map similar to that of **Fig. 1** that will predict the odour concentration for a certain point under a set of particular conditions at the site.

However, a range of different models such as Gaussian, Eulerian and Lagrangian models exist, which generally produce significant differences in their predictions because of their important dependency on the meteorology of the area. Although economically inexpensive when compared to intensive field panels, important drawbacks exist in the use of a popular tool such as dispersion modelling for assessing odour nuisance. Additionally, not only odour concentration matters to assess nuisance. Nuisance can be caused by an odour stimulus which can be characterized by several factors (Frechen 2001; Jehlickova *et al.* 2008) such as strength of the odour (concentration), kind of odour, hedonic tone (offensiveness), frequency and other time-dependant factors, tolerance and expectation of the receptors, and past odour experience of the local population. Therefore tools are required to assess and anticipate odorous emissions and their impact. Although some recent attempts propose complementary tools such as the "Potential for Odour Creation" (Bouchy *et al.* 2008) or the "Odor Wheel" (Suffet *et al.* 2008) to characterize odour emissions and nuisance, neither methodology nor standards exist on how to measure olfactory nuisance (Suffet *et al*. 2008).

Composting emissions and odour nuisance

Composting facilities present numerous odour and air pollution sources, including the reception and handling of materials, forced aeration composting, stock piling, etc. Gaseous emissions in composting facilities are typically constituted by nitrogen-based compounds, sulphur-based compounds and a wide group of compounds denominated Volatile Organic Compounds (VOCs). There are also some compounds released in minor quantities that have an important environmental impact at global level, because they are powerful greenhouse gases (GHG), such as nitrous oxide (N_2O) and methane (CH_4).

Among the nitrogen-based compounds released to the atmosphere, ammonia has received much attention because it can be easily identified from other composting odours, it often represents the main nitrogen gas emitted during composting and it can be released in large amounts. In addition to the loss of fertilizing value of compost when ammonia emissions are considerable, this compound is also known to contribute to acid rain formation (ApSimon *et al.* 1987). Ammonia emissions in a composting process of organic fraction of municipal solid wastes (OFMSW) vary between 18 to 150 g NH_3/t waste (Clemens and Cuhls 2003) whereas peaks of ammonia concentrations up to 700 mg NH_3 m^{-3} have been reported in exhaust gases from wastewater sludge composting (Haug 1993). In studies performed at laboratory level with different wastes it was shown that ammonia emissions exhibit a clear correlation with process temperature, reaching maximum values during the thermophilic period (Pagans *et al.* 2006a). From an olfactive point of view, ammonia has a characteristic odour that is easily identifiable in composting emissions although its detection limit is relatively high (27 ppm). Even if emitted in lower quantities than ammonia, trimethylamine is also responsible of composting processes odour nuisance having a fishy odour and a human detection limit 100 times lower than ammonia (Goldstein 2002; Rosenfeld and Suffet 2004).

Another major group of gaseous pollutants emitted from composting facilities are VOCs, denomination used to refer to a wide group of organic compounds whose vapour pressure is at least 0.01 kPa at 20°C (European Commission 1999b). VOCs are also characterized by their low water solubility. Once in the atmosphere, VOCs participate in photochemical reactions producing photochemical oxidants. According to Eitzer (1995), most VOCs in composting plants are emitted at the early stages of process i.e. at the tipping floors, at the shredder and during the initial forced aeration composting period. Incomplete or insufficient aeration during composting can produce sulphur compounds of intense odour, while incomplete aerobic degradation processes result in the emission of alcohols, ketones, esters and organic acids (Homas *et al.* 1992). Van Durme *et al.* (1992) identified dimethyl sulphide, dimethyl disulphide, limonene and α-pinene as the most significant odorous VOCs at a wastewater sludge composting facility. According to this work, the latter two compounds were released from wood chips used as bulking agent. At laboratory scale, total VOCs concentration in exhaust gases from composting processes of different wastes has been also studied (Pagans *et al.* 2006b) and it was concluded that the higher concentrations of VOCs were emitted during the first 48 h of process. These authors also stated that VOCs emissions could not be correlated with the biological activity of the process. In addition to odorous disturbance that VOCs can cause, the presence of xenobiotic VOCs in gaseous emissions from municipal solid waste composting has also been reported (Komilis *et al*. 2004).

N_2O and CH_4 may be released when anaerobic regions are created. Compaction, insufficient aeration or turning are the main factors to induce anaerobic zones. All are considered GHG and its warming potential is higher than that produced by CO_2.

N_2O is produced during the composting process due to an incomplete oxidation of ammonia or as product of an incomplete denitrification process (Beck-Friis *et al.* 2001). According to Hellmann *et al.* (1997), N_2O emissions in composting processes are influenced by the temperature, with the most important emissions taking place at temperatures lower than 45°C. Amlinger *et al.* (2008), reported N_2O emission rates ranging from 30 to 80 g N_2O/t of manure, 120 to 180 g N_2O/t of biowaste and 190 to 450 g N_2O/t of backyard wastes. Szanto *et al.* (2007) studied N_2O and CH_4 emissions during straw-rich pig manure composting, reporting concentrations between 0 and 400 ppm N_2O.

CH_4 is generated in strictly anaerobic zones through degradation of soluble lipids, carbohydrates, organic acids and proteins (Fukumoto *et al.* 2003). Amlinger *et al.* (2008), reported CH_4 emission rates ranging from 140 to 1350 g CH_4/t of manure, 800 to 1800 g CH_4/t of biowaste and 780 to 2180 g CH_4/t of backyard wastes. While Fukumoto *et al.* (2003) reported CH_4 emission rates between 1000 and 1900 g CH_4/t of organic matter, when composting swine manure.

Minimization of composting emissions

As stated above, gaseous emissions are inherent to the composting process. However, the presence of gaseous compounds responsible of odour nuisance or considered as GHG in these emissions can be reduced. The need of gaseous emissions collection and treatment in some critical cases (short distances between composting plants or inhabited areas) lead to the construction of enclosed facilities with air collection systems and emission treatment equipment, usually wet scrubbers and/or biofilters. Gas cleaning techniques will be explained in detail further on in this paper.

In addition to these treatment techniques in enclosed facilities or in open facilities, there are some management actions to adopt for reducing emissions of some gaseous compounds. These actions can be summarized in providing optimal conditions for the composting process, including adequate water content, porosity and oxygen supply which will avoid the existence of anoxic and anaerobic zones that are the source of a number of odorants as hydrogen sulphide or GHG as methane and N_2O.

Oxygen is provided to the composting piles by natural or forced aeration. In both cases, pile size is a determining factor for a correct aeration because it determined the pile structure and porosity (Gage 2003). As pile height increases more structure is needed to maintain adequate porosity. Pile turning is recommended to keep a uniform porosity in composting materials and to reduce compaction of the composting bed (Szanto *et al.* 2007). However it should be kept in mind that gaseous compounds trapped in pile pores will be released during turning. Oxygen supply in forced aeration systems can be reached by positive and negative aeration with the former presenting the possibility of exhausted air collection and treatment even in open facilities (Nicoletti and Taylor 2005).

An adequate balance of carbon and nitrogen content in composting materials will also help in reducing the release of gaseous nitrogen compounds, mainly ammonia. The mixture of complementary feedstocks regarding these two elements should be considered (Gage 2003).

Water and energy consumption

The consumption of resources (water and energy in its different supply form, i.e. gasoil or electricity) is an important factor when studying the impact generated during the composting process. The knowledge of this consumption, together with emissions generation, could facilitate the decision with respect to which type of technology should be used in each situation.

It can be expected that different technologies should present different resources consumption. For example, low technology facilities (i.e. turned windrows in open facility) should require less energy than complex facilities (i.e. in-vessel systems with forced aeration and gas emission treatment).

In reference to energy consumption, from our own data, between 500 and 220 MJ of total energy (electricity and gasoil) are necessary for composting 1 t of OFMSW, being the lower values related to low technology facilities. Blengini (2008) reported around 385 MJ of total energy for composting 1 t of OFMSW in an aerated windrows facility.

Even though composting is a water demanding process, the general practice of watering using leachates reduces water consumption. However, to avoid pathogen contamination of the final product, leachates should not be used for watering during the curing phase. Water consumption varies from 0.02 to 0.33 m^3 of water/t OFMSW (own data). The highest value corresponds to a closed facility with water open-loop in the gas treatment system (scrubber). Then 0.33 m^3 of water/t OFMSW should be considered as the maximum water consumption for composting. Blengini (2008) reported around 0.09 m^3 of water/t of OFMSW composted in aerated windrows.

The role of respiration indices for environmental impact assessment

A commonly used functional unit in composting plant impact determination is related to the composting of 1 t of waste processed. However, this unit does not allow establishing a relationship between the impact or emission factors with the process efficiency. Then, it is interesting to refer the amounts of materials and energy entering and

Table 2 Comparison of energy consumption between two different facilities using the proposed functional unit from experimental data (OFMSW: Organic Fraction of Municipal Solid Waste; DRI: Dynamic Respiration Index, OM: Organic Matter content).

Facility	A	B	Units
Technology	Aerated windrows	Turned windrow	-
OFMSW DRI	3.7	3.5	$g\ O_2\ kg\ OM^{-1}\ h^{-1}$
Compost DRI	0.7	2.7	$g\ O_2\ kg\ OM^{-1}\ h^{-1}$
Total energy consumption	553	221	MJ/t OFMSW
Energy consumption referred to DRI reduction	184	276	(MJ/t OFMSW) / ($g\ O_2\ kg\ OM^{-1}\ h^{-1}$)

exiting a composting plant to the real performance and extent of the biological treatment process. In this sense, a possible suggestion could be to select as the functional unit the reduction in the biological activity of the material measured with a global tool such as the Respiration Index (RI) (Gea *et al*. 2004).

The need of a parameter that permits to relate the environmental impacts of a biological treatment process to the biodegradation level achieved for the organic matter (by means of O_2 consumed or CO_2 produced) has also been stated by Amlinger *et al.* (2008). These authors propose the ratio between methane produced and total CO_2 emissions as an indicator of the efficiency of the aerobic decomposition process and also the ratio between kg CO_2 equivalent (obtained by computing N_2O and methane emissions and total CO_2 produced) to relate greenhouse gases emissions to the efficiency of aerobic decomposition and organic matter transformation. However, a global aerobic activity indicator such as RI seems a more straightforward indicator to evaluate the extent and efficiency of a biological process used for organic solid wastes treatment and stabilization as similar measures are used in other environmental fields, such as the case of BOD_5 in wastewater treatment.

As an example, in **Table 2**, the effect of the proposed functional unit can be observed when comparing two real facilities, named A and B, using composting in aerated windrows and in turned windrows, respectively. If the composting of a ton of OFMSW is used as functional unit, the total energy consumption (electricity and gasoil) is more than twice for facility A (553 and 221 MJ/t OFMSW for facility A and B respectively). However, due to the low efficiency of facility B (only 23% reduction of DRI), the energy consumed to reduce one unit of DRI for each ton of OFMSW is higher than this used in facility A (184 and 276 (MJ/t OFMSW)/(g O_2 kg OM^{-1} h^{-1}) for facility A and B, respectively).

Life cycle assessment

In spite of the impacts of the composting process stated above composting and compost land application have some important positive global effects. Regarding these positive aspects Favoino *et al.* (2008) point to: i) displacement of chemical fertilizers, which implies avoidance of GHGs emission and energy consumption associated to their production ii) a reduction in the water irrigation requirements and an increase in the potential for soils to retain moisture and iii) carbon returning to soils, among others.

Life Cycle Assessment (LCA) is a methodological tool that should allow balancing between positive and negative aspects of the composting process or the compost itself as a product. LCA has as main objective to study the environmental aspects and potential impacts through the whole life of a product or service, from the extraction of raw materials, the production, the use and the final disposal. This means to develop an inventory of relevant inputs and outputs of the system (inventory analysis), assess their potential impacts (impacts assessment) and interpret the results in relation with the proposed targets (interpretation) (ISO 14040 1997).

In the last years different studies on mass and energy flows related to composting facilities have been carried out to determine the environmental impacts of this type of treatment systems. Given its importance many of the studies have been focused on MSW management systems and composting. A major concern has been the study of gases emitted during the composting process itself (NH_3, VOCs, N_2O, CH_4 and other compounds) that contribute to global warming, acid rain, human toxicity and to the promotion of photochemical oxidation reactions in the atmosphere (Hellebrand and Kalk 2001; Komilis *et al.* 2004; Pagans *et al.* 2006a). Emissions to hydrosphere have also been studied to identify impacts related to eutrophication and soil acidification (US Environmental Protection Agency 2006). Mass and energy balances, as well as economic accounts have been also performed (Diggelman and Ham 2003; Fricke *et al.* 2005).

Other authors have developed mathematical models to analyze MSW management as, for example, EASEWASTE (Kirkeby *et al.* 2005), ORWARE (Sonesson *et al.* 1997) and WASTED (Diaz and Warith 2005), which include the environmental burdens associated to waste management.

MSW treatment and/or management have also been studied by the perspective of Life Cycle Assessment (LCA). Finnvenden *et al.* (2007) analyzed the methodological aspects of LCA of solid waste management systems. Other authors studied MSW management systems from different cities or regions as Wales (Emery *et al.* 2007), Ankara (Özeler *et al.* 2006), Phuket (Liamsanguan and Gheewada 2008) or Corfu (Skordilis 2004) using LCA tool. Finally, LCA has also been applied to the study of waste treatment plants, particularly, anaerobic digestion plants (Güereca *et al.* 2006; Ishikawa *et al.* 2006).

It is important to notice that LCA on MSW management systems include a wide variety of data (necessary to perform the inventory) that, in many cases, should be deduced or directly obtained from other bibliographic sources. Since technologies, management systems or scale are not always comparable, this practice may derive in the use of erroneous data. This is even worse when the waste studied presents different properties or composting performance. Thus, a reliable LCA inventory should be the result of the study of a significant number of real facilities treating wastes with similar characteristics. Even more, as it has been shown previously, the functional unit used in LCA should reflect the efficiency of the process in terms of organic mater stabilization.

MINIMIZATION OF ENVIRONMENTAL IMPACT

All industrial processes convert raw materials in products with an added value, even if production is accompanied by the generation of liquid, solid and/or gaseous wastes. Reduction and minimization both by improved designs and by proper process operation are the first choice to minimize environmental impacts. Additionally to environmental reasons, most processes need of some type of treatment for economical reasons and to cope with regulatory limits. Composting is one of the technologies used to recover and valorise solid wastes produced in other processes, although composting is a process by itself. Wastes generated during the composting treatment, which are mainly the leachate and waste gases, need of further treatment to minimize their impact on surrounding neighbourhoods of the facilities. This is particularly important in the case of waste gases generation because of odour nuisance problems on plant vicinity.

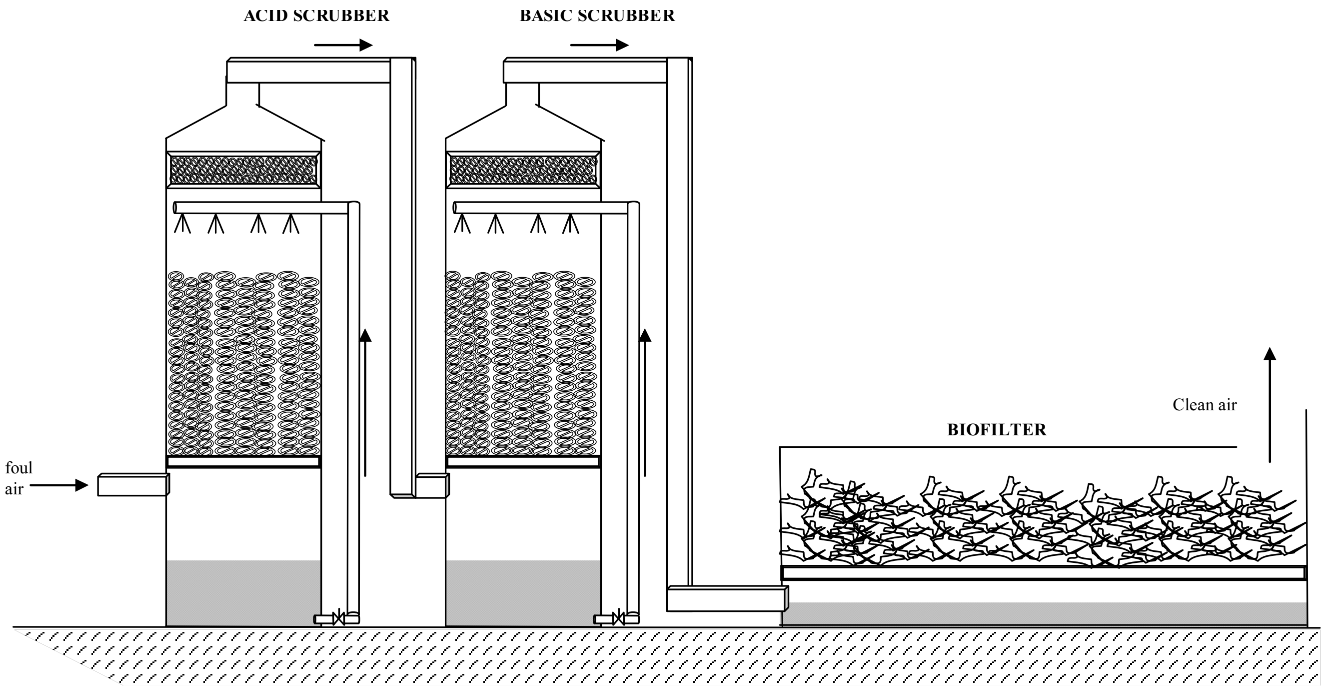

Fig. 2 Combination of chemical acid and basic scrubbers followed by a biofilter typically installed in large facilities for composting waste gases treatment.

Composting gas cleaning

Composting processes are commonly related with malodorous emissions due to the stripping of nitrogen compounds, mainly ammonia, and a range of VOCs and Reduced Sulfur Compounds (RSCs) during the aeration phase. The selection of the best available technology for composting waste gases treatment depends essentially on the characteristics of the waste gases emitted, essentially composition and air flow rate to be treated. The latter is particularly important since it directly impacts construction costs, materials and the footprint required, and, to a larger extent, operating costs of the gas cleaning system. In the case of composting, gas cleaning is not an easy task, mainly because of the complexity of the mixture to be treated, but also because of the low concentration of most of odorous compounds and the large air flow rates used during aeration.

In general, there is not a unique technology that serves for composting gas emissions treatment but a combination of these available. During last decades, several well-established technologies such as adsorption, thermal/catalytic oxidation and chemical scrubbing have been applied for the treatment of waste gases. Such technologies provide reasonable removal efficiencies in the case of composting emissions treatment, even if installation and operating costs are much higher if compared with equivalent biological processes (Devinny *et al.* 1999; Gabriel and Deshusses 2004a). Physical-chemical technologies may also produce undesirable side-effects, like the generation of different toxic compounds or the transfer of the gas pollutants from the air to another phase. In particular, thermal/catalytic oxidation results economically viable only to treat low-to-middle airflows with pollutant concentrations much higher than those found in composting (Kennes and Veiga 2001). Also, combustion may produce nitrogen and sulfur oxides if reduced N or S compounds are not previously removed. Other physical-chemical processes such as adsorption and absorption mostly transfer the pollutant from the gas phase to a liquid or a solid that will require of further treatment. Thus, the use of biological technologies for the treatment of composting waste gases is generally used for composting emissions treatment.

In spite of this, biological treatment at composting facilities is generally preceded by a single absorption step under acidic conditions, essentially to remove ammonia. The rationale behind such combination of processes lies in the large ammonia amounts produced during the composting process (Haug 1993) and the low tolerance of ammonium-oxidizing microorganisms to ammonium accumulation typically occurring in most of the bioreactors for ammonia waste gas treatment (Baquerizo *et al.* 2005; Gabriel *et al.* 2007). Additionally, large industrial facilities tend to install and additional, basic chemical scrubber after the acidic scrubber to remove other acidic character odorant. Thus, a complete schematic of a composting waste gases treatment facility is shown in **Fig. 2**. The first chemical scrubber is generally operated by adding sulfuric acid to ensure that pH is kept below 5. In consequence, ammonium sulfate is produced and removed through the drain. Similarly, the basic scrubber is operated at pH above 9 by adding caustic, thus ensuring that acid gases such as hydrogen sulfide are removed.

Since chemical scrubbers are reliable systems with the lowest cost of the chemical technologies for treatment of foul air with low concentrations of pollutants for applications over 50,000 m^3/h (Card 2001), small composting sites and medium-small facilities tend to simply clean composting gases with a single biofilter thus avoiding additional investment and operating costs.

Biological techniques and principles for waste gases treatment

In general, all biological reactors for waste gas treatment rely on the use of naturally selected microbial strains which are capable of employing the pollutant or pollutants as carbon and/or energy source. Unlike traditional technologies, biofiltration is mainly based on physical and biological principles, instead of physical-chemical ones. A number of studies have proved that biofiltration can minimize the aforementioned problems typical of physical-chemical technologies and several applications for air pollution control have been extensively described in various books and articles that cover most of the compounds typically found in composting waste gases (see e.g. Cox and Deshusses 1998; Devinny *et al.* 1999; Kennes and Veiga 2001; Deshusses and Gabriel 2005). Out of the several types of biological reactors found in the literature for waste gas treatment, biofilters and biotrickling filters have been generally chosen as most efficient and cost effective for a wide range of odorous compounds.

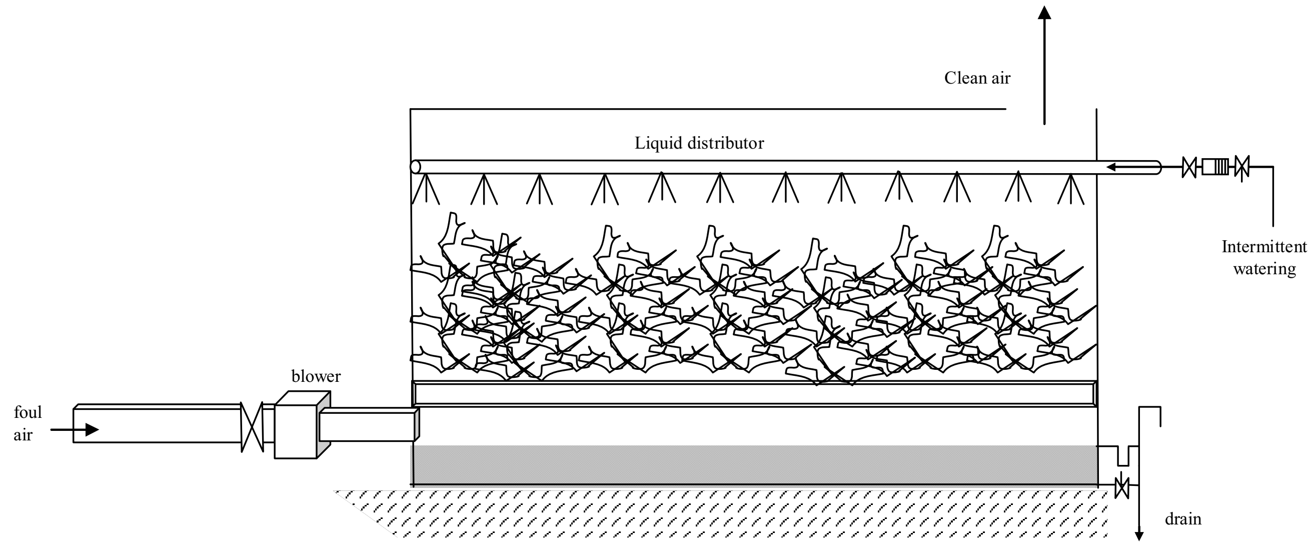

Fig. 3 Schematic of a biofilter.

Biofilters (**Fig. 3**) are those bioreactors where a humid stream of contaminated air is passed through a damp packing material – usually an organic packing material such as compost mixed with wood chips or any other bulking agent – on which pollutant degrading bacteria are naturally immobilized. The pollutant is transferred by absorption to the biofilm, where diffusion and biodegradation take place simultaneously. In the case of composting emissions, products from microbial oxidation are primarily nitrite, nitrate due to ammonia oxidation, carbon monoxide due to VOCs degradation, sulphate due to H_2S and RSCs oxidation and microbial biomass due to growth. Additionally, biofilters are simple and cost effective. They require low maintenance and are particularly effective for the treatment of odour and volatile compounds that are easy to biodegrade, and for compounds that do not generate excessive acidic by-products. Biofilters are increasingly used in industrial applications. As an example, municipal solid waste treatment facilities in Europe generally use full-scale biofilters packed with coconut fiber for the treatment of warehouse and composting gases emissions (van Groenestijn 2005).

Biotrickling filters (**Fig. 4**) work in a similar manner to biofilters, except that an aqueous phase is continuously trickled over the packed bed, and that the packing is usually made of some synthetic or inert material, like plastic rings, open pore foam, lava rock, etc. The trickling solution contains essential inorganic nutrients such as nitrogen, phosphorus, potassium, etc. and is usually recycled. Biotrickling filters are more complex to built and operate than biofilters but are usually more effective, especially for the treatment of compounds that generate acidic by-products, such as H_2S, or toxic or inhibitory ones, such as NH_3. Biotrickling filters can be built taller than biofilters thus saving footprint requirements. However, biotrickling filters are more recent than biofilters, and have not yet been fully deployed in industrial applications for composting gas treatment because of the nature of the emissions.

Although most industrial applications for composting emissions treatment rely mainly on biofilters, biotrickling filters have demonstrated that may be used for successfully retrofitting chemical scrubbers (Gabriel and Deshusses 2003, 2004b) thus being a promising complement of biofilters for VOCs, H_2S and NH_3 removal from composting emissions (Sakuma *et al.* 2008; Prado *et al*. 2009).

Although biofilters and biotrickling filters are relatively simple systems from a conceptual point of view, the interactions between physical, chemical and biological processes that take place in the reactor are extremely complex. However, operation and performance of biological reactors for air pollution control is generally reported in terms of simple parameters such as the removal efficiency, or pollutant elimination capacity as a function of the pollutant loading, or the gas empty bed retention time (EBRT). These terms are defined in the next four equations.

$$\text{Removal} = \text{RE} = \frac{C_{in} - C_{out}}{C_{in}} \times 100 \quad (\%)$$

$$\text{Pollutant Elimination Capacity} = \text{EC} = \frac{(C_{in} - C_{out})}{V} \times Q \quad (\text{g m}^{-3}\ \text{h}^{-1})$$

$$\text{Empty Bed Retention Time} = \text{EBRT} = \frac{V}{Q} \quad (\text{s or min})$$

$$\text{Pollutant loading} = \text{L} = \frac{C_{in}}{V} \times Q \quad (\text{g m}^{-3}\ \text{h}^{-1})$$

where C_{in} and C_{out} are the inlet and outlet pollutant concentrations (usually in g m^{-3}), respectively, V is the volume of the packed bed (m^3) and Q is the air flow rate (m^3/h). It should be stressed that the EC, the L and the EBRT are calculated using the volume of the packed bed and not the total volume of the reactor. Depending on the reactor design and packing material selected, the volume of the packed bed will be about 40-90% of the total reactor volume. Thus, the actual gas residence time will be lower depending on the porosity of the packing, the dynamic liquid hold-up and the amount of biomass attached to the packing. Usually, the removal efficiency of a bioreactor operating under proper conditions is close to 100% at low inlet loads of pollutant, with ECs equal or slightly lower to that of the inlet load. If the inlet load is increased, the bioreactor will attain its maximum EC (EC_{max}), which will keep constant at higher inlet loads. If the pollutant has a toxic effect on microorganisms, then the EC will diminish instead.

As described in Devinny *et al.* (1999), there is a list of key parameters and operating conditions that need to be kept within typical values for proper biofilters operation:

1) Composition of the gas: compounds present in the composting gas must be biodegradable and soluble in water to a certain extent and do not present toxic effects on microorganisms. Substances such as a variety of alcohols, aldehydes, ketones, and certain simple inorganic compounds such as hydrogen sulfide or ammonia can be easily treated in biofilters. Another important factor to consider is concentration since biological techniques are generally viable for pollutants concentrations below 1-1.5 g/m^3 (Devinny *et al.* 1999).

2) Packing material: Proper packing material selection is a key factor in the reactor performance and stability. Main characteristics to consider upon the selection of an appropriate packing material are its specific surface area, density, porosity, pH, water holding capacity, buffering

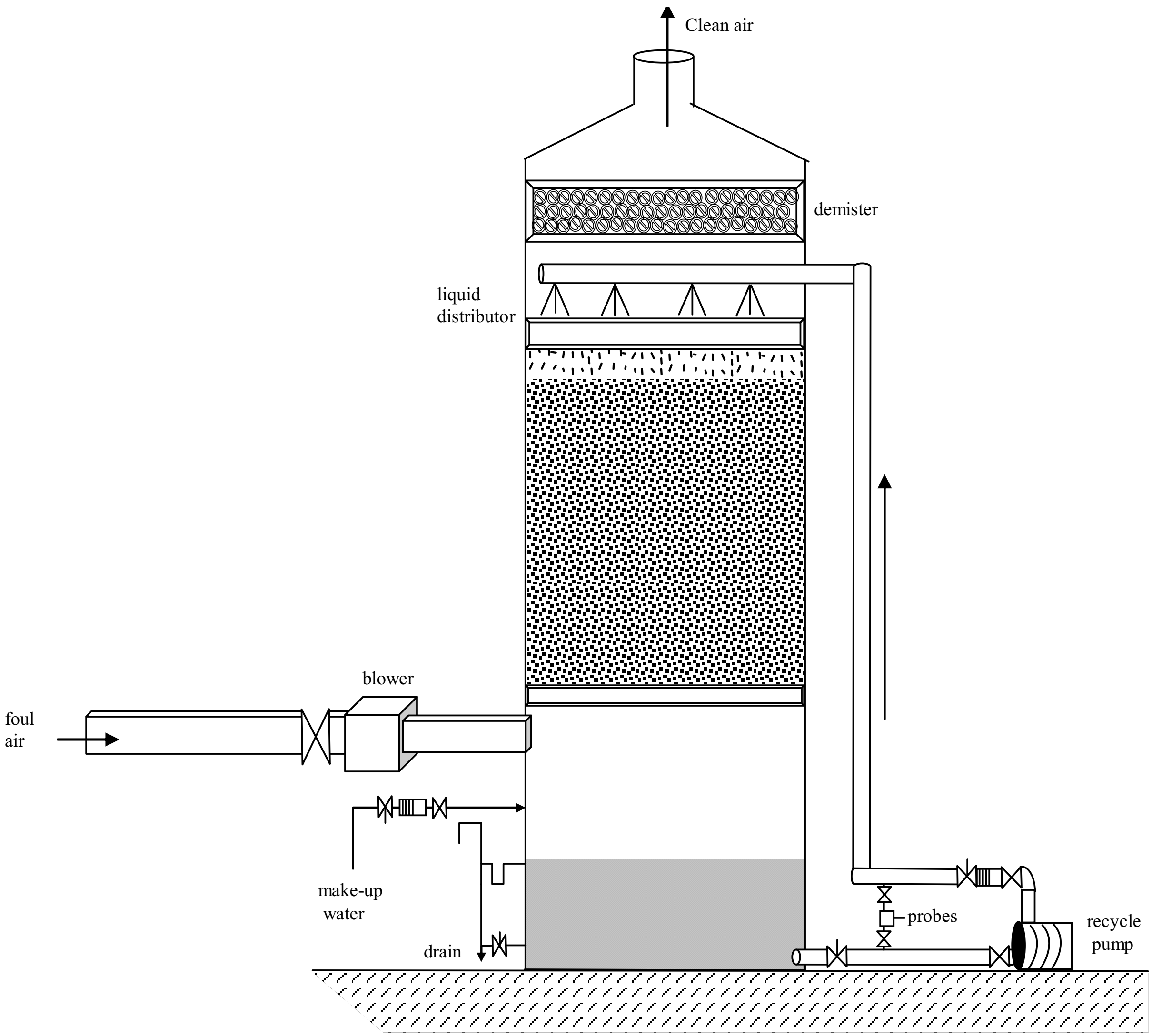

Fig. 4 Schematic of a biotrickling filter.

capacity and elemental composition (Bohn 1996). Also, physical and chemical characteristics must be accompanied with testing of operational conditions in lab and pilot-scale reactors before moving to full-scale systems. Suitable materials to act as biomass support must be economic and have a high specific surface area, lightness, high chemical and mechanical resistance and large durability. Once packed, material must allow a good adhesion of the biomass and generate a low pressure drop.

3) Nutrient supply: As in other biological processes, nutrients requirements for microorganisms include sources of macronutrients as nitrogen and phosphorus and micronutrients. In biotrickling filters nutrients need to be supplied externally, while in biofilters microorganisms grow using nutrients presents in the packing material (Leson and Winer 1991). Nevertheless, extra-nutrient addition, and particularly nitrogen, may be necessary in order to treat high loading rates (Morgenroth *et al.* 1996; Maestre *et al.* 2007). However, an excess of nitrogen can lead to a rapid clogging of the system due to an excessive growth of microorganisms.

4) Temperature: Biofilters and biotrickling filters generally operate in the mesophilic range. However, some cases of successful treatment have been carried out at temperatures of 0°C (Lehtomaki *et al.* 1992) and 70°C (Kong *et al*. 2001).

5) pH: Generally biofilters and biotrickling filters operate in a pH range between 6 and 8 for the treatment of VOCs from composting emissions since heterotrophic microorganisms grow properly in such pH range. Also, since pH has to be stable periodical supply of buffers is often carried out for the regulation of the pH. One of the problems in biofilters is the loss of efficiency due to a pH drop due to the production of acid by-products, which is particularly important in the treatment of halogenated hydrocarbons, nitrogenous or sulphur compounds, some VOCs, etc.

6) Pressure drop: Pressure drop is a key parameter since operating costs in biofilters are mainly due to the airflow supply (Gabriel and Deshusses 2004a). Thus, a pressure drop increase implies a large electrical consumption to maintain the airflow. Pressure drop is mainly influenced by the porosity of the reactor, which depends essentially on the packing material, and the biomass and water content. Generally, a pressure drop of 1-2 cm water column/m bed height is considered appropriate.

7) EBRT: The optimum value of the gas residence time depends on the solubility of the pollutant treated, and the rule the larger the solubility the lower the EBRT applies. Typical EBRTs in biofilters for odour treatment are between 15 and 60 seconds, while EBRTs below 10 seconds have been reached with successful removal of NH_3 or H_2S from waste gases (Gabriel and Deshusses 2003; Duan *et al.* 2005; Sakuma *et al.* 2008).

Biological treatment of composting emissions

Out of the two biological technologies commonly used at industrial level for waste gases treatment, biofilters have been more extensively used than biotrickling filters for composting emissions treatment. Such fact is mostly related with the solubility of the compounds found in composting emissions and the different water content of biofilters and biotrickling filters. In a biofilter the water content comes from the condensation of a fraction of the water present in the gas current entering the biofilter. Relative humidity close to saturation is generally reached at the biofilter entrance after a humidification step of the waste gas in humidification towers. In consequence, biofilters do not have a continuous water phase over the surface of the packing material except for short periods of time if external watering is performed. Instead, biotrickling filters have a continuous recirculation of water over the packed bed that leads to a flowing water layer on the surface of the packing material. Thus, removal of low soluble compounds in water such as a large list of VOCs present in composting emissions is superior in biofilters. In fact, Kennes and Thalasso (1998) established a range of air/water partition coefficients for biofilters and biotrickling filters. Biofilters are the recommended technology for compounds with air/water partition coefficient below 1 while biotrickling filters are recommended for compounds with air/water partition coefficient below 0.1.

Based on a typical composition of composting emissions, ammonia is one of the key compounds to be removed. A wide number of applications have proven that biofiltration results in good NH_3 removal efficiencies (Busca and Pistarino 2003) and several works have shown that biofilters are able to remove ammonia from composting emissions under a wide range of biofilter designs and operating conditions (Hong *et al*. 2002; Park *et al*. 2002; Chung *et al.* 2003; Pagans *et al.* 2007). However, some studies have questioned the efficiency of ammonia biofiltration at relatively high inlet concentration because of low efficiencies on ammonia removal, sudden reactor failure during the long run or poor performance due to the sensitivity of nitrifying bacteria (Demeestere *et al*. 2002; Chen *et al*. 2005; Gabriel *et al*. 2007). Most of the references reporting proper biofilter performance for ammonia removal only consider process efficiency in terms of gas cleaning without taking into account by-products generation in the drain of the biofilter. Several authors have reported that around 50% of the ammonia removed in a biofilter is simply absorbed and recovered as ammonium in the drain (Smet *et al*. 2000; Chen *et al*. 2005; Gabriel *et al.* 2007), while the oxidized fraction is only partially nitrified and accumulated as nitrite due to nitrifying microorganisms inhibition (Baquerizo *et al.* 2005). Highly variable performances reported in classical biofilters for NH_3 removal are largely influenced by the sum of several factors, but proper biofilter watering, pH control and biomass acclimation may play a key role in improving nitrification of the ammonia absorbed in the biofilm (Baquerizo *et al*. 2009).

In general, ammonia removal efficiencies close to 100% are easily found for constant ammonia inlet loads up to 50-60 g NH_3 m^{-3}/h, which imply an inlet ammonia concentration of up to 260-320 ppm_v of NH_3 at an EBRT of 15 seconds. Also, reactors can treat sudden inlet peak loads of up to 800-1000 ppm_v without much impact on their removal efficiency (Pagans *et al*. 2005). However, sustained operation at inlet concentrations above 300 ppm_v makes biological treatment rather limited since reactors can fail because of ammonia toxicity to the microbial community (Gabriel *et al*. 2007). Under suboptimal operating conditions, as in example because of poor bed watering or non uniform water distribution, poor performance because of nitrification inhibition can be found at inlet concentrations below 100 ppm_v NH_3 (Gabriel *et al.* 2007).

Because of their configuration, biotrickling filters are a much proper reactor configuration for composting emissions containing high loads of NH_3 since the water phase in the biotrickling is continuously purged, which avoids accumulation of inhibitory by-products in the bed. Ammonia elimination capacities above 100 g NH_3 m^{-3}/h can be easily found in biotrickling filters avoiding inhibition problems found in biofilters (Sakuma *et al.* 2008). In consequence, such data indicates that biotrickling filters can serve for retrofitting chemical scrubbers at these composting facilities that treat composting waste gases in a schematic as that depicted in **Fig. 2**. However, one should mention that a larger EBRT than this typical of chemical scrubbers must be warranted for the biotrickling filters to ensure acceptable removal efficiencies for VOCs from composting emissions (Prado *et al.* 2009).

Although not much has been published regarding biofiltration of VOCs from composting emissions, a vast literature exists regarding biofiltration of model VOCs usually contained in composting emissions (Devinny *et al.* 1999). According to Deshusses and Johnson (2000) that developed a method for assessing biofilters performance for a wide range of VOCs with different air/water partition coefficients, maximum elimination capacities in typical biofilters are around 100-120 g m^{-3} h^{-1}, which are generally found in biofilters mainly colonized with bacteria. However, ECs close to 300 g m^{-3} h^{-1} have been latterly reported in fungal biofilters (Aizpuru *et al*. 2005) since some authors hypothesise that fungi are able to improve the solubility of hydrophobic compounds compared to bacterial biofilms due to the direct contact between the fungal mycelia and the gaseous pollutant (Van Groenestijn and Liu 2002). However, the large range of VOCs emitted in composting make difficult to precisely indicate how efficient this technology for composting emissions treatment is. Several authors report proper biofilters efficiencies for composting off-gases biofiltration (Sironi and Botta 2001), even if complete odour removal is difficult to achieve in such reactors (Pierucci *et al.* 2005), not only because of the technology itself but also because maintenance tasks in these reactors are often scarce.

Opposite to what is generally though, biofilters need of further maintenance than biotrickling filters to keep the packing material healthy. This is mostly because there is a lack of automation in the former type of reactor and because some necessary tasks must be performed manually by plant personnel. The packed bed needs of frequent attention such as watering frequency adjustment or turn-over and periodic replacement to keep performance. Such tasks are simply not performed or extended in time which endangers biofilter efficiency. Proper biofilter maintenance would make biofiltration a more reliable, robust and trustable technology at industrial level.

CONCLUSIONS

Composting is nowadays one of the emerging technologies for the treatment of organic solid wastes in developed societies. Although the technology is well known, established and easy to understand, several aspects remain still unclear from the scientific point of view. RIs are the most suitable technique to monitor the process and to minimize the environmental impact of the composting process, especially when gaseous emissions are considered and need to be treated. As it is demonstrated in this work, RIs can be used to determine the environmental impact of composting plants. Emission factors of pollutant gases or resources consumption can be referred to the resulting reduction of RI obtained during the entire composting process, to enable a scientifically based comparison among proposed technologies or input wastes.

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