ENVIRONMENTAL IMPACT ANALYSIS AT FULL-SCALE OFMSW BIOLOGICAL TREATMENT PLANTS. FOCUS ON GASEOUS EMISSIONS

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by

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CERTIFIQUEM:
Que el Llicenciat en Enginyeria Ambiental Erasmo Cadena Martínez ha realitzat sota la nostra direcció el treball que amb títol "Environmental impact analysis at full-scale OFMSW biological treatment plants. Focus on gaseous emissions", es presenta en aquesta memòria, la qual constitueix la seva Tesi per optar al Grau de Doctora per la Universitat Autònoma de Barcelona.

I perquè en prengueu coneixement i consti als efectes oportuns, presentem a l'Escola Tècnica Superior d'Enginyeria de la Universitat Autònoma de Barcelona l'esmentada Tesi, signant el present certicat a

Bellaterra, Novembre de 2009

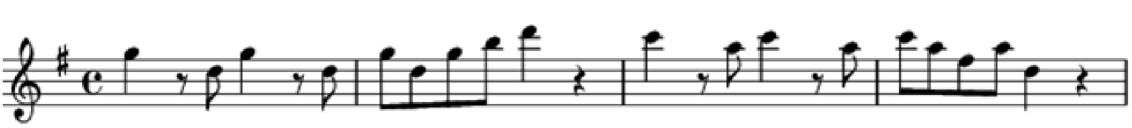
Dr. Xavier Font Segura

Dra. Adriana Artola Casacuberta

For Aurora & Cadena, my parents

I firmly believe that, a few centuries hence, the history of what we call the scientific activity of this age will be a prolific subject for the hilarity and pity of future generations.

Lev Nikoláyevich Tolstói, My religion (1885)



Eine Kleine Nachmusik (Serenade), K. 525; 1st Movement, W. A. Mozart

“Ich glaube diese Welt müßte groß genug, weit genug, reich genug, für uns alle sein”

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Overview of the dissertation

In the last decades the treatment of municipal solid waste (MSW) throughout the use of biological treatment systems such as composting and anaerobic digestion, has been widely used at European level as an alternative to landfilling. However this type of treatments involves positive and negative environmental aspects that should be studied. In this sense, this dissertation grew up of the interest in the determination of the environmental impacts from the biological treatment of the organic fraction of municipal solid waste (OFMSW) at representative full-scale plants in Catalonia. The selected plants, using different treatment technologies (composting in turned windrows system, composting in tunnel system, composting in confined windrows system and anaerobic digestion plus composting system) have been studied from the life cycle analysis (LCA) perspective. To identify the different environmental impacts and determine the values presented in this work, it was necessary to develop a specific methodology and analytical procedures. The main axis of this study was directly focused on the analysis of gaseous emissions such as NH_3 and volatile organic compounds (VOCs), and the materials flow inside the studied plants.

The results obtained provide real data on the biological treatment plants currently working at industrial scale in Catalonia on the gaseous emissions field and its potential environmental impact.

On the other hand, part of the results presented in this Thesis were obtained in collaboration with the RICICLA group of the Università degli Studi di Milano (Italy) as part of a research stay where gaseous emissions from a full-scale biological treatment plant were also studied but mainly focused in the odours coming from the emitted compounds.

This Thesis represents the beginning of a new research line in the Composting Research Group, at the Department of Chemical Engineering of the Universitat Autònoma de Barcelona on the gaseous emissions and environmental impacts identification at full-scale biological treatment plants. This project was developed in cooperation with the *Agència de Residus de Catalunya* (ARC) since 2005, who provided financial support and facilitated the access to the installations analyzed.

Resumen

En las últimas décadas el tratamiento de los residuos municipales mediante el uso de sistemas de tratamiento biológico tales como el compostaje y la digestión anaeróbica, ha sido ampliamente utilizado a nivel Europeo como alternativa al uso de los vertederos. Sin embargo, este tipo de tratamientos se asocian a una serie de aspectos ambientales positivos y negativos que deben ser estudiados. En este sentido, el objetivo de esta Tesis surge del interés relacionado con el estudio de los impactos ambientales procedentes del tratamiento de la Fracción Orgánica de los Residuos Municipales (FORM) en plantas reales y representativas en Cataluña. Las plantas seleccionadas, que utilizan distintas tecnologías (compostaje en pilas volteadas, compostaje en túneles, compostaje en pilas confinadas y digestión anaeróbica más compostaje), se han estudiado mediante el uso de la herramienta de análisis del ciclo de vida (ACV). Para obtener los valores que se presentan en este trabajo, fue necesario desarrollar una metodología y unos procedimientos analíticos específicos. La línea principal de este trabajo se enfocó principalmente en el análisis de las emisiones de amoniaco y compuestos orgánicos volátiles (COV) y en el estudio del flujo de los materiales dentro de las plantas estudiadas.

Los resultados obtenidos proporcionan datos reales de emisiones gaseosas y sus correspondientes impactos ambientales de plantas de tratamiento biológico a escala industrial actualmente en funcionamiento en Cataluña.

Por otro lado, es importante destacar que una parte de los análisis presentados en esta Tesis se llevaron a cabo en colaboración con el grupo de investigación RICICLA de la Università degli Studi di Milano (Italia) como parte de una estancia de investigación durante la que se analizó el impacto ambiental producido por las emisiones gaseosas en instalaciones industriales de tratamiento biológico de la FORM, en este caso enfocado principalmente al impacto odorífero.

Esta Tesis representa el inicio de una nueva línea de estudio en el Grupo de Investigación en Compostaje del Departamento de Ingeniería Química de la Universitat Autònoma de Barcelona en la determinación de emisiones gaseosas e identificación de impactos ambientales en plantas reales a escala industrial. Este proyecto fue desarrollado en cooperación con la *Agència de Residus de Catalunya* (ARC) iniciándose en el año 2005. La *Agència* financió el trabajo y facilitó el contacto con las instalaciones analizadas en esta Tesis.

Resum

En les darreres dècades el tractament dels residus municipals mitjançant l'ús de sistemes de tractament biològic com el compostatge i la digestió anaeròbica, s'ha estès àmpliament a nivell Europeu com alternativa a l'ús dels abocadors. Tot i això, aquest tipus de tractaments estan relacionats amb una sèrie d'aspectes ambientals positius i negatius que caldria estudiar. En aquesta línia, l'objectiu d'aquesta Tesi sorgeix de l'interès relacionat amb l'estudi dels impactes ambientals procedents del tractament de la Fracció Orgànica dels Residus Municipals (FORM) en plantes reals representatives a Catalunya. Les plantes seleccionades, que utilitzen diferents tecnologies (compostatge en piles voltejades, compostatge en túnels, compostatge en piles confinades i digestió anaeròbica més compostatge), s'han estudiat mitjançant l'eina d'anàlisi del cicle de vida (ACV). Per obtenir els valors presentats en aquest treball es va haver de desenvolupar una metodologia i uns procediments analítics específics. L'estudi es va enfocar principalment cap a l'anàlisi de les emissions d'amoníac, compostos orgànics volàtils (COV) i dels fluxos de materials en de les plantes avaluades.

Els resultats obtinguts proporcionen dades reals d'emissions gasoses i els seus corresponents impactes ambientals de plantes reals de tractament biològic a escala industrial actualment en funcionament a Catalunya.

D'altra banda, és important destacar que una part de les anàlisis presentades en aquesta Tesi es van dur a terme en col·laboració amb el grup d'investigació RICICLA de la Università degli Studi di Milano (Itàlia) com a part d'una estada de recerca on es va analitzar l'impacte ambiental produït per les emissions gasoses en instal·lacions industrials de tractament biològic de FORM, però en aquest cas enfocat principalment a l'impacte per males olors.

Aquesta Tesi representa l'inici d'una nova línia d'estudi en el Grup d'Investigació en Compostatge del Departament d'Enginyeria Química de la Universitat Autònoma de Barcelona en la determinació d'emissions gasoses i identificació d'impactes ambientals en plantes de tractament biològic a escala industrial. Aquest projecte, iniciat l'any 2005, va ser desenvolupat en cooperació amb l'Agència de Residus de Catalunya (ARC) que el va finançar. L'Agència també va facilitar l'accés a les instal·lacions analitzades en aquesta Tesi.

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Chapter 1

Introduction

"Waste itself is a human concept; everything in nature is eventually used. If human beings carry on in their present ways, they will one day be recycled along with the dinosaurs"
Peter Marshall

The massive growth of industrial activities, population and urban planning has lead to an increase in waste generation. There is an international concern, which is reflected in current legislation, about the importance of achieving adequate waste management and treatment in order to ensure the protection of human health and the prevention of environmental impacts (Eriksson et al., 2005). Solid waste management, and particularly, the management of the organic fraction, is an unsolved issue in developed countries around the world. At the European Union level, a total of 76.5-102 Mt/year of organic fraction of municipal solid waste (OFMSW) is generated, which represent 30-40% of the total annual municipal waste generated (European Commission, 2008). In Catalonia for example, 1.64 kg per capita per day of municipal solid wastes was generated in 2006, from which 45-55% corresponded to the organic fraction (Agència de Residus de Catalunya, 2008).

As a result of pressure from European Union legislation (Directive 1999/31/EC), the overall amount of municipal solid waste (MSW) disposed in landfills has gradually decreased in Europe—from 293 kg per capita in 1995 to 221 kg per capita in 2005. (Rades and Tilesman, 2008).

Nowadays, different technologies that improve the recycling of organic matter and nutrients mainly by means of biological treatments such as composting and anaerobic digestion are being applied to divert organic wastes from entering landfills (European Commission, 1999a). These improvements result in lower greenhouse gases emissions from landfills, which contribute considerably to global warming (Mor et al., 2006). However, in addition to these benefits, biological treatment processes also have the potential to contribute to number of negative environmental impacts such as atmospheric emissions and the consumption of resources. Among the existing technologies to treat the OFMSW, composting and anaerobic digestion are widely considered as environmentally friendly technologies.

Returning to legal issues, in the last years, the emerging international environmental policies are increasingly showing support for sustainable development and recycling and recovery technologies. This is the case in the European Union where strict requirements to prevent and reduce the waste sent to landfill exist. European Directive 99/31/CE on

Landfills includes the key for bio-waste land filling. The aim of this Directive is to reduce, in 15 years, the amount of waste landfilled by more than 65% from the production in 1995. This reduction would decrease CH_4 production at landfill sites (which represents one of the highest contributions to global warming from MSW management), allow OFMSW to be used to obtain renewable energy (via biogas) and allow for stabilized material (compost) to be obtained (Favoino, 2006).

Another focus of the international policies discussed above, is to selectively collect wastes in order to guarantee that biological treatment and solid waste valorization systems operate effectively.

This chapter aims to introduce the reader to biological treatments of MSW (composting and anaerobic digestion) and its diverse implications for the environment, summarizing the benefits and limitations involved in these processes.

1.1 Biological treatments of Municipal Solid Waste

In light of the discussion above, the principle question to answer is: What is the most environmentally friendly way to manage our wastes? Biological treatment processes such as composting and anaerobic digestion have been widely studied around the world (Ahring, 2003; Haug, 1993), and nowadays they are the main biological treatments applied to biowaste, the organic fraction of MSW. This is especially true for source-selection collection systems in developed and developing countries. Biological processes are known to have several advantages over landfilling. These advantages include the reduction of waste volume, waste stabilization, destruction of pathogens and production of biogas for energy use in the case of anaerobic digestion. Depending on its quality, the final product of these processes can be used as fertilizer and/or for soil amendment (Haug, 1993). In the following sections both biological treatment systems will be described in detail.

1.1.1 Composting

Biological decomposition is an age-old, natural process. As vegetation falls to the ground, it slowly decays, providing minerals and nutrients needed by plants, animals, and microorganisms. Composting is often used synonymously with biological decomposition. However, composting in this paper refers to the purposeful and controlled decomposition of organic matter by microorganisms into a stable humus material known as compost. The process is controlled and managed with the aim of accelerating decomposition, optimizing its efficiency, and minimizing any potential environmental problems or other nuisances that may result during the process (Rynk, 1992).

Composting is defined as the biological decomposition of organic substrates, under conditions that allow thermophilic temperatures to be attained as a result of biologically produced heat, to generate a final product that is stable, free of pathogens and plant seeds, and can be beneficially applied to land (Haug, 1993).

Composting is a widely used technique. For instance, Spain has experience exponential growth in waste composting installations. Although it is difficult to find reliable up-to-date data on plants in operation at the national level, in 2005, 82 composting plants and 9 anaerobic digestion plants were operating in Spain to treat 7824 Mt of source-separated and mixed municipal solid waste (Ministerio de Medio Ambiente y Medio Rural y Marino, 2009). Among these installations, 23 composting plants and 4 anaerobic digestion plants are located in Catalunya (ARC, 2009).

1.1.1.1 Composting phases

In the composting process it is important to maintain the biological, chemical and physical requirements of microorganisms to obtain the optimum degradation levels throughout the stages of the process. The composting process usually occurs in two phases, a high-rate composting phase and a curing stage.

- **Decomposition phase**

The first stage is called decomposition or high-rate phase because during this stage the decomposition activity of the feedstock into simpler compounds by microorganisms is intense and, as a result of the metabolic activities, heat is produced (Hang, 1993). This stage is also characterized by high oxygen uptake rates. The microorganisms present in this stage are classified as mesophilic and thermophilic. Mesophilic microorganisms (those that grow best at temperatures between 25 and 45°C) are dominant throughout the composting mass in the initial phases of the process when temperatures are relatively low. These organisms use available oxygen to transform carbon from the composting feedstock to obtain energy and organic materials to build new biomass and, in the process they expel carbon dioxide (CO_2) and water. As a consequence of the degradation of the organic matter, heat is produced as the microorganisms metabolize the composting feedstock. When the temperature approaches 45°C, the mesophiles die or become dormant, a-waiting more suitable conditions (Gray et al., 1971a). At this point, thermophilic microorganisms or thermophiles become active, consuming the materials that are readily available and multiplying rapidly replacing the mesophiles in most sections of the material. Thermophiles generate greater quantities of heat than mesophiles, and if the temperature is sufficiently high (between 45 and 70°C) the organic matter becomes sanitized as pathogens are killed and weed seeds become unviable. In many composting facilities, temperature of 55°C is maintained for 72 hours in order to guarantee pathogen destruction and weed seed destruction (Gray et al., 1971a). When sources of readily available carbon are depleted, thermophiles die and the temperature drops. Mesophiles begin to dominate the decomposition process once again until all of the readily available energy sources have been consumed. In full-scale composting facilities, the high-rate phase takes a few weeks depending on the materials treated and the composting technology used.

- **Curing phase**

The second stage is called the curing phase because the product is “cured” or finished. This phase is characterized by slow degradation because the nutrients available to microorganisms have been depleted (Adani et al., 1997). As a consequence of the slow activity during this phase, temperature decreases and the texture of the material becomes dry and powdery. At the end of this phase the material is considered stabilized or mature, which is the reason that this phase is also known as the maturation stage (Haug, 1993). At this time the nitrogen obtained from the dead biomass is incorporated into high molecular weight compounds, which act as nitrogen reserves because they are resistant to microbial decomposition (Tchobanoglous et al., 1994; Haug, 1993). This phase is less demanding in terms of oxygen and humidity than the first phase and results in a net loss of total organic matter (OM) and inorganic constituents. The principal products of the composting process are fully mineralized such as CO_2, H_2O, mineral ions, ash and stabilized organic matter (i.e. humic substances) (Haug, 1993).

- **Pre and post processing**

Process operations prior to the first stage of composting are termed pre-processing and those at intermediate stages or after the curing phase are termed post-processing.

Since municipal solid waste is heterogeneous and only a percentage is acceptable for composting, pre and post-processing instruments are needed to separate compostable fractions from unwanted materials (refuse). Hand-sorting, trommel screening, magnetic separation of iron, separation of aluminum and froth separation of glass are pre-processing activities that are commonly employed. After the compostable fraction of waste is separated, one of the key activities in the pre-processing stage is structuring the organic matter by means of a bulking agent, which is usually organic (i.e. wood chips, pallets refuse and tree trimming) or inorganic (i.e. small perforated plastic cylinders) to provide structural support and maintain air space within the composting matrix (Larsen and McCartney, 2000). This step is known as conditioning. In other cases, moisture and nutrients may also be added during the pre-processing phase (Haug, 1993).

As in pre-processing, post-processing requires equipment to separate glass and plastic. The principal aim of post-processing is to produce a high quality final product by removing the small refuse materials and the remaining fraction of the bulking agent. Trommel screening and ballistic separators are usually used for this purpose (Haug, 1993). Physicochemical characteristics of compost depend on the nature and composition of the composted matter (Tchobanoglous et al., 1994).

1.1.1.2 Principal composting variables

Several parameters determine the chemical environment for composting, principally: carbon and nutrient balance, moisture, oxygen, temperature, pH and particle size.

- **Carbon balance and nutrients**

Microorganisms require specific nutrient balances in an available form, proportion and proper concentration to perform composting efficiently. The essential nutrients that microorganisms require in large quantities include carbon (C), nitrogen (N), phosphorus (P) and potassium (K). Chemoheterotrophic microorganisms require carbon as energy source, C and N to synthesize proteins, build cells and to reproduce. P and K are necessary for the reproduction of cells and for metabolism. In composting systems, C and N are usually the limiting factors for efficient decomposition (Richard, 1992). An initial C:N ratio of 15 to 30 is recommended as an optimum value for composting materials (Hang, 1993).

Organisms also need micronutrients to properly digest all the nutrients. The principle micronutrients required are boron, calcium, chloride, cobalt, selenium, sodium and zinc (Boyd, 1984). Although these nutrients are present in sufficient quantities, they may be present in a form, which makes them unavailable for some microorganisms (Gray et al., 1971a).

High C:N ratios (i.e. high C and low N) inhibit the growth of decomposing microorganisms, while low C:N ratio (i.e. low C and high N) initially accelerates microbial growth and decomposition. However, with this accelerated decomposition, the available oxygen is quickly consumed and proper aeration of the material is not maintained. Excess N causes a high release of ammonia and can result in a toxic environment for the microbial population, inhibiting the process (Haug, 1980; Gray et al., 1971b).

- **Moisture**

Microorganisms require moisture to absorb nutrients, metabolize and produce new cells because they can only use organic molecules if they are dissolved in water. Under conditions of low humidity, the composting process slows down. High moisture conditions can reduce and even stop the transfer of oxygen air-filled pores. Microorganisms also produce water as part of the decomposition process. Water is removed via aeration or evaporation (Gray et al., 1971b). The recommended range of moisture during the process is between 40-60%. The moisture content is usually maintained at this level by watering with leachate during the decomposition stage (high-rate phase) and rainfall water or other in the maturation stage (curing phase) to avoid contamination of the sanitized material if leachate was used (Haug, 1993). Below 20% humidity, very few bacteria are active (Haug, 1980).

- **Oxygen**

According to Haug (1993), the main functions of aeration in composting processes are to supply the oxygen needed by aerobic microorganisms, to facilitate the regulation of excess moisture by evaporation and to maintain the proper temperature. To support microbial activity, there must be many available pores in the material to serve as air chambers. Oxygen can be provided throughout the turning and mixing of the material, or by using forced aeration systems.

An oxygen concentration of 10 to 15% is considered adequate in a composting matrix, although a concentration as low as 5 percent can be sufficient for microbial activity. While a higher oxygen concentration will not negatively affect the composting process, it may indicate that an excessive amount of air is circulating through the material, which can be problematic. For example, excess air removes heat and can promote excess evaporation, both of which slow the composting rate. Moreover, excess aeration is also an added expense which increases production costs (Rynk et al., 1992).

It is important to highlight that oxygen consumption, measured using respiration techniques, is used as an indicator of the biological activity of composting microorganisms. These techniques measure the oxygen consumed in the composting process. The first stage is characterized by high oxygen consumption and, due to a decrease in the activity of microorganisms, the maturation stage is characterized by low oxygen consumption. Respiration techniques allow for the measurement of the oxygen uptake rate (amount of oxygen consumed per kg of organic mater and time unit, mg O_2 /kg OM h) or cumulative oxygen consumption (oxygen consumed per kg of organic matter, mg O_2/kg OM) (Barrena et al., 2006). According to the literature (Adani et al., 2004; California Composting Quality Council, 2001), materials with respiration index values between 0.5 and 1.5 mg O_2 /g OM h are considered stable, while respiration index values greater than 1.5 mg O_2/g OM h correspond to unstable materials.

- **Temperature**

When aeration is controlled, the temperature in the compost pile is determined by the level of activity of the heat-generating microorganisms (Richard, 1992b). The effective temperature in the process is between 45 and 59°C (Richard, 1992a). Temperatures below 20°C, inhibit the activity of microorganisms lowering their decomposition capacity (Finstein et al, 1986; Strom, 1985). The regulatory machinery of cell metabolism is affected by temperatures below the minimum requirement for a group of organisms. A Typical temperature profile of a composting process is shown in Figure 1.1 along with the active microorganisms in each temperature range. Microorganisms tend to decompose materials most efficiently within their temperature tolerance range. Although composting occurs within a range of temperatures, the optimum temperature range of thermophilic microorganisms is preferred for two reasons: it promotes rapid composting and it destroys pathogens and weed seeds. The rates of microbial decomposition therefore increase when the temperature rises to an absolute upper limit. For this reason, it is important to maintain high temperature ranges without reaching process inhibition during the decomposition phase (Richard, 1992a; Rynk et al., 1992). As shown in Figure 1.1 at the beginning of the composting process, materials are at room temperature and then, as explained above, temperature gradually begins to rise due to the activity of microorganisms. As can be seen in this figure, psychrophilic and mesophilic microorganisms are present throughout the process and thermophilic microorganisms are active when at higher temperatures. When the temperature approaches 45°C, mesophilic microorganisms die or become dormant. When this occurs, thermophilic microorganisms become active consuming available materials and multiplying rapidly. The activity of thermophiles generates greater quantities of heat than that of mesophiles leading to higher temperatures in the composting mass (Figure 1.1). Thermophilic microorganisms continue decomposing as long as energy and nutrients are plentiful. When sources of energy and nutrients become depleted, thermophilic microorganisms die. As the temperature drops mesophilic microorganisms become active once again and consume the remaining nutrients (Richard, 1992a; Rynk et al., 1992).

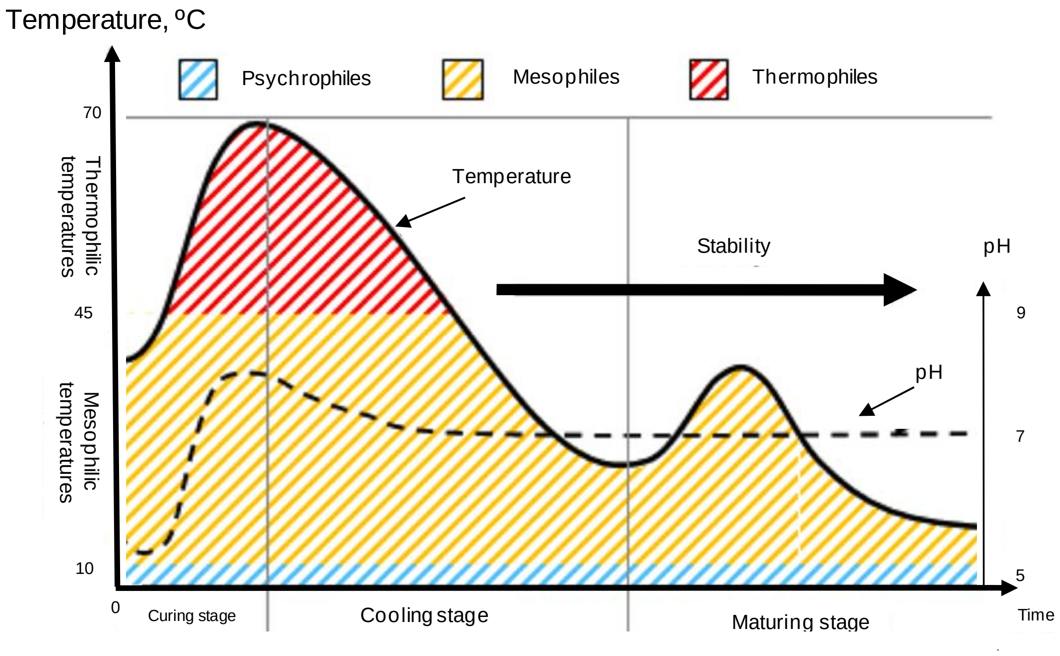

Figure 1.1. *Typical temperature and pH profiles through the composting process.*

- **pH**

The optimal pH for a biological process is normally in the range of 6 to 7.5 for bacteria and 5.5 to 8 for fungi (Boyd, 1984). If the pH is below 6, microorganisms, particularly bacteria, die off and decomposition slows down (Wiley, 1956). If the pH rises above 9, ammonium becomes ammonia, which is toxic for microorganisms (Rynk et al., 1992).

Like temperature, pH follows a typical profile throughout the composting process. As shown in Figure 1.1, most of the decomposition phase occurs at a pH between 5.5 and 8 (Rynk et al., 1992, Gray et al., 197lb). In the beginning of the composting process, organic acids are formed and the materials often become acidic with a pH slightly over 5. Thus, the acid tolerance of fungi plays an important role in decomposition. Microorganisms break down the acids; hence pH levels increase gradually to a more neutral (8.5). The role of bacteria in the composting process becomes more important when the pH rises again. If the pH does not increase, this may indicate that the final product is not fully mature.

- **Particle size and air filled porosity**

Particle size significantly affects the composting process. In general, small particles have a greater surface area to volume ratio. This means that much of the particle surface is exposed directly to decomposition by the microorganisms in the early composting stages (Gray et al. 197la). The optimum particle size is that providing enough surface area for rapid microbial activity, but also enough void space to allow air to circulate for microbial respiration and material decomposition (EPA, 1994). The particles should be large enough to prevent compaction, thus excluding the oxygen in the voids. For yard trimmings or municipal solid wastes, the desired combination of void space and surface area can be achieved by particle size reduction.

Air filled porosity (AFP) is defined as the ratio of air volume to total volume of the sample (Haug, 1993). This parameter depends on particle size and water content (Agnew and Leonard, 2003). An optimum value recommended for AFP is 25-30%, but this depends on the type of material being composted.

Table 1.1 shows a typical quality standard for finished compost with respect to the parameters discussed above as well as some other.

Table 1.1. *Quality standard for finished compost (CIWMB, 2004).*

Indicator	Quality standard for finished compost	
Physical	Moisture: 30-40%, fine texture	
Odor	Smell like rich humus from the forest floor	
Nutrient	C:N ratio	<17:1
	Organic matter	20-35%
	Total nitrogen	1-2%
	Nitrate nitrogen	250-350 ppm
	Sulfide	0 ppm
	Ammonium	0 or trace
	pH	6.5-8.5
	Humic acid content	5-15%
Microbiological	Compost maturity	>50% on maturity index at dilution rate appropriate for compost application
	Compost stability	<100 mg O_2/kg dry matter-hour
	E. coli	<3 CFU/g

1.1.1.3 Composting technologies

There are different technologies used in composting to treat solid waste. The most commonly full-scale composting methods currently employed are:

- Passive piles,
- turned windrows,
- aerated static piles and
- in-vessel systems.

- **Passive piles**

Passive piles, as the name suggests, are piles that remain static without alteration and may occasionally be turned during the process (EPA, 1997). Passive piles have a delta or trapezoidal cross section with length exceeding width and height. For most materials the ideal height is between 1.5 to 2 meters with a width of 4.3 to 4.8 meters. Although this method is simple and generally effective, it is not applicable under all conditions or for all types of waste. Composting under these conditions is very slow and it is satisfactory or sufficient for materials that are relatively uniform with respect to particle size. Passive piles can theoretically be used for composting vegetable waste or MSW. In the case of MSW or large quantities of vegetable waste, odor may be a problem (EPA,1997).

Passive piles require low investment and technology. The piles should be constructed so they are large enough to conserve sufficient heat but not so large that they overheat (EPA,1997). If the temperature of the material is very high (above 65-70ºC) the microorganisms needed for decomposition may die. Passive piles have the advantage of low operating costs, however the time required to obtain the finished product is much longer than it is for more intensive composting methodologies. For more intensive techniques, a finished product is obtained within a few weeks to a few months, whereas for passive piles over a year is needed for the composting process to be complete. In addition, the minimal turning of piles results in formation of anaerobic conditions and thus odor emissions (Finstein and Strom, 1989).

- **Turned windrow**

Turned windrows are elongated composting piles that are turned frequently to maintain aerobic composting conditions. The frequent turning promotes uniform decomposition of the materials as cooler outer layers of the compost pile are moved to inner layers where the material is exposed to high temperatures and intensive microbial activity. This method achieves a finished product in 3 months to a year (UConn CES, 1989).

As with passive piles, the ideal height for turned windrows is between 1.5 and 1.8 meters (CRS, 1989). However, the windrow height varies depending on the materials being composted, the season, the region where it is composted, the tendency of the material to compress and the type of machinery used. The width of the windrow is usually twice its height.

- **Aerated static piles**

Aerated static piles or forced aeration windrows are high technology approaches to composting MSW, yard trimmings and other organic feedstock. Aerated static pile composting implies forcing or pulling air through a trapezoidal compost pile, which minimizes the need for turning. To better manage odors, piles are often covered with a textile layer.

On average aerated piles are 2 to 2.6 meters in height. To facilitate aeration, wood chips are placed over the aeration pipes at the base of the windrow (Rynk et al., 1992). The composting process using this method takes 3 to 6 months.

- **In-vessel system**

In-vessel system is a high-technology method in which the composting process is conducted and controlled within a fully enclosed structure. Composting parameters such as aeration, moisture and temperature are mechanically controlled.

The in-vessel method can be developed for container composting systems (individual containers) or tunnel composting systems. Composting materials are retained in the system for 6 to 28 days and then cured in windrows for 1 o 2 months, but the range of time may vary depending on the composition of the material (EPA, 1997).

To summarize this dissertation on composting, some benefits and limitations of the composting process are presented in Table 1.2.

***Table 1.2**. Benefits and limitations of composting (Fullana, 1996).*

Benefits	Limitations
- Diminishes the solid waste quantity in the final disposal systems. - Reduces leachate generation and methane emissions in landfill. - Allows the recovery of organic waste. - Facilitates the extraction of inorganic materials to be recycled and increases its calorific value. - Reduces waste volume due to water loss and organic matter decomposition to CO_2. - Stabilizes and sanitizes solid waste. - Compost produced can be used as fertilizer in agriculture or gardening, recovery and restoration of eroded areas.	- May present adverse environmental impacts depending on the pollutant load. - The use of low quality compost can contaminate water, resulting in bioaccumulation of heavy metals in plants and animals. - The inappropriate application of compost can alter soil properties (acidification, salinisation). - To avoid impact, strict controls on operating conditions and emissions should be applied. - Low economic returns due to the considerable investment in technology are required. - The final product has a relatively low price, partly for the negative perception of the composting process. - Presents environmental impacts mainly due to gaseous pollutants emissions such as volatile organic compounds (VOCs), NH_3, CH_4 or N_2O, among others.

1.1.1.4 Gas emission control systems for composting

As discussed at the beginning of this chapter, the composting process results in environmental impacts including the emissions of gaseous pollutants such as NH_3, volatile organic compounds (VOCs), N_2O, CH_4 and odors. Therefore, many composting facilities employ the gaseous emissions mitigation systems shown in Table 1.3.

Table 1.3. *Composting emissions control systems (EPA, 1997).*

Technology	Description	Effectiveness
Biological oxidation (biofilter, bioscrubber, trickling filters)	Gases are biologically degraded. Contains filter media that supports pollutant-degrading microorganisms (i.e. bacteria and fungi) support.	More than 95% removal
Absorption (Wet scrubbers with packed tower or mist scrubbers)	Gases are absorbed into a liquid media.	Up to 70% per stage in case of packed tower and <90% in case of mist scrubber.
Adsorption	Gases are removed by passing gases over an inert medium to which the gas-causing compounds become attached.	Effective for removal and control of VOCs
Combustion	Gases are captured and compounds are burned.	99% removal

1.1.2 Anaerobic digestion

Anaerobic digestion is another biological process that has been used for over 100 years to stabilize materials such as wastewater sludge, municipal solid waste and other industrial refuses (Ferrer et al., 2008; Burke, 2001). Anaerobic digestion is a biological process in which the biodegradable matter is degraded or decomposed in the absence of oxygen using specific microorganisms that produce biogases (composed mainly of methane and carbon dioxide) that can be used for electricity generation (Lissens et al., 2001).

In recent years, anaerobic digestion has become an interesting technology because it provides a clean fuel from a renewable feedstock and thus leads to the partial replacement of fossil fuels for energy production (Adani et al., 2001; Chynoweth et al., 2001).

1.1.2.1 Anaerobic digestion stages

Anaerobic digestion involves four main stages: hydrolysis, acidogenesis, acetogenesis and methanogenesis.

- **Hydrolysis**

In the first stage, undissolved and complex organic molecules are fragmented into simpler compounds (amino acids, sugars, fatty acids, alcohols and CO_2) through the controlled action of extracellular enzymes excreted by fermentative bacteria (Ponsá et al., 2008; Pavlostathis and Giraldo-Gómez, 1991).

- **Acidogenesis**

This phase involves the transformation of hydrolyzed compounds into volatile fatty acids (mainly acetate, propionate and butyrate), alcohols and other products including ammonia, hydrogen and carbon dioxide.

The bacteria in this stage are facultative and proteolytic bacteria, which are abundant in nature. Acidogenic bacteria are fast growing compared to other groups used in anaerobic digestion. *Clostridium*, *Bacillus*, *Pseudomonas* and *Micrococcus* are some examples of those bacteria (Madigan et al., 1998).

- **Acetogenesis**

In acetogenesis, alcohols, fatty acids and aromatic compounds are degraded to produce acetic acid, carbon dioxide and hydrogen-substrates that will be used by methanogenic bacteria in the final anaerobic digestion stage (Archer, 1983).

- **Methanogenesis**

During methanogenesis, anaerobic methanogenic microorganisms transform organic products obtained in the earlier stages (acetate, carbon dioxide, methanol, hydrogen and some methylamine) into methane (Madigan et al., 1998).

- **Pre and post processing**

Besides the main digestion stages, it is necessary to perform pre- and post-processing operations to condition the OFMSW for the biological process.

- Pre-processing

As for the composting process, for anaerobic digestion it is necessary to pre-process OFMSW or MSW to increase its digestibility. One of the most common pre-treatment methods is sorting. Depending on the quality of the waste, manual separation of unsuitable materials such as plastics, metal and large materials may be required. Following manual sorting, if the material is treated under wet conditions, the next separation step usually involves a pulper, which also reduces the particle size. For dry anaerobic digestion, trommel screening is normally employed. Another indispensable pre-processing activity is particle reduction to render an appropriate to obtain a homogeneous size distribution inside the reactor. Size reduction is developed by means of screw cutting, milling, drumming, pulping or shredding, among others (Braber, 1995).

- Post-processing

In wet anaerobic digestion, depending on the quality and stabilization of the material, it can either be applied directly to farmland or may be separated into solid and liquid phases. Solids can be composted and the liquid phase can be treated in a wastewater treatment plant.

Wastes treated by dry process are sometimes composted with other farm or MSW materials or may be centrifuged and then composted (Braber, 1995).

1.1.2.2 Operating parameters in the anaerobic digestion process

As in composting processes, there are some main operating parameters in anaerobic digestion systems. These parameters determine the microbial activity and thus influence/affect the anaerobic degradation efficiency. The principal parameters are discussed in the following paragraphs.

- **Temperature**

Anaerobic biological activity can be developed for temperatures ranging from 5 to 70°C. However, there are generally two temperature ranges used at the full-scale level that provide optimum digestion conditions for methane production: the mesophilic and thermophilic ranges. The mesophilic range is between 20-40°C but the optimum temperature is considered to be 30-35°C. The thermophilic temperature range is between 50-65°C (Cecchi et al., 1993), but the processes are normally undertaken at 50-55°C. It has been observed that temperatures in the thermophilic range reduce the required retention time to obtain a given level of organic matter degradation and that substrate removal increases with temperature (National Renewable Energy Laboratory, 1992), but the energetic requirements are also higher.

- **pH**

The pH has a large influence on the biodegradation process because it affects the process velocity and the selectivity of the microorganisms that can be developed in the medium. Methanogens in particular are very sensitive to acidic conditions and unsuitable pH levels can inhibit their growth (Ratledge, 1991). pH affects the activity of microbes (Clark & Speece, 1989) by: changing the state of ion groups of enzymes such as carboxyl or amino group, altering ionisable components of the system such as substrates, and denaturing the protein structures of enzymes.

A pH between 6.5 and 7.5 is considered an optimum and stable value for anaerobic digestion (APAT, 2005) but no general consensus on this issue exists. Some authors have concluded that the optimal for methanogenic activity ranges from 6.7 to 7.4 because they have found that outside this range the methanogenesis velocity decreases considerably (Bitton, 1994).

Degradation products such as CO_2 and volatile fatty acids can lower the pH, while cations such as ammonium and sodium ion can increase the alkalinity (Speece, 1996). The retention time of the digestate affects the pH value and in a batch reactor acetogenesis occurs at a rapid rate.

- **Volatile fatty acids**

Volatile fatty acids (VFA) concentration in the digester is one of the most important parameters for anaerobic digestion reactors because instability of the system is often marked by a rapid increase in the VFA concentration, which signals methanogenic phase inhibition (Ianotti and Fisher, 1984). According to Angelidaki (1992) carbohydrate and protein hydrolysis are limited by high VFA concentrations. VFAs are expressed as concentration of acetic acid (AcOH) in the feedstock and, depending on the type of material treated, this value can range from 200 to 2000 mg AcOH/L.

- **Retention time**

The retention time (RT) for completion of the anaerobic digestion reactions varies in relation to feedstock composition, anaerobic digestion technology and process temperature. The retention time for biomass treated using mesophilic digestion ranges from 10 to 40 days, while biomass treated in thermophilic reactors requires a minimum of 14 days (Bello-Mendoza and Shararrat, 1999).

- **C:N ratio**

The optimal C:N ratio in anaerobic digestion is approximately 20-30. A high C:N ratio is an indicator of rapid consumption of nitrogen by methanogens and results in lower gas production while a lower C:N ratio results in the accumulation of ammonia and pH values exceeding 8.5, which is toxic for methanogenic bacteria (Boone et al., 1987). To achieve an optimum C:N ratio, co-digestion of different materials, for example combining OFMSW with sewage or animal manure may be done.

- **Mixing**

Some experiments have shown that in order for the process to operate smoothly an adequate degree of mixing in the reactor must be achieved (Campos, 2001). The purpose of mixing is to blend the fresh material with the digestate that contains microorganisms. The mixture reduces the phenomena of matter transfer limitation of substrate or nutrients within the liquid phase to the microbes. A 60% reduction in the degree of mixing may cause as much as a 50% decrease in treatment efficiency (Bello-Mendoza and Shrarratt, 1999). Mixing systems vary in terms of reactor type and solid content in the digester (Campos, 2001).

- **Biogas production**

Biogas is the most important product obtained during the anaerobic digestion process. This product is mainly composed of methane and CO_2, but inert gases (N_2) and sulfur compounds (H_2S) are also present but at lower concentrations. Habitually, 100 to 200 m^3 of biogas is produced per ton of OFMSW digested (Braber, 1995). The composition of wastes affects the yield and biogas quality as well as the digestate quality. A typical biogas composition is presented in Table 1.4.

Table 1.4. *Typical biogas composition (Barber, 1995).*

Energy content	5.6-7 kWh/m^3
Methane (CH_4) by vol.	55-70%
Carbon dioxide (CO_2) by vol.	30-45%
Hydrogen sulfur (H_2S)	200-4000 ppm

1.1.2.3 Anaerobic digestion systems

Anaerobic digestion systems can be classified into three categories depending on different parameters:

- By number of stages, single-stage or multistage,

- dry and wet systems, and,
- by temperature, mesophilic or termophilic processes.

A summary of anaerobic digestion systems is given in the following paragraphs.

- **Number of stages: single or multistage**

Single stage systems are those in which the entire biological process occurs within a single reactor or digester.

Two stage and multistage systems are those in which hydrolysis and acidification normally take place in the first reactor while methane fermentation occurs in the second reactor. These systems are particularly reliable and stable for waste with high organic matter content such as fruit and vegetables wastes (Pavan et al., 2000). The high capital cost of installing a multistage system has a resulted in a reduction in the number of these types of facilities. Currently, the installed capacity of multistage systems in European countries is 7% (De Baere and Mattheeuws, 2009). According to De Baere and Mattheeuws (2009) in the period between 2006 and 2010, multistage systems will represent only 2% of the European capacity, thus most of the capacity will be derived from single stage systems.

- **Dry and wet systems**

In wet processes, waste is first diluted. Conventional slurry systems usually function by co-digestion with animal manure, MSW and another feedstock with a dry matter content of approximately 10% (Braber, 1995).
In dry systems, the waste is treated without dilutions. This process is fed materials with a dry matter content of 20-40% (Poggi-Varaldo, 1997). Biogas production in these systems ranges from 90-150 m^3/t of waste digested. The lowest value corresponds with garden waste, whereas and the highest, with food waste (Pavan et al., 2000).

Between 2000 and 2005 a large number of full-scale wet treatment plants were installed in European countries, but since 2005, more dry anaerobic digestion plants were installed (De Baere and Mattheeuws, 2009). In recent years, dry anaerobic digestion has provided nearly 54% of the European capacity while wet anaerobic digestion represents 46% of total current capacity (De Baere and Mattheeuws, 2009).

- **Mesophilic and termophilic processes**

There are three conventional operational temperature levels for anaerobic digesters:

1. Mesophilic system, where mesophilic microorganisms are active, takes place between 20°-45°C and operates optimally between 37-41°C (Song et al., 2004).
2. Thermophilic system, where thermophilic microorganisms are active, operates optimally between 50-52°C, but can reach temperatures as high as 70°C (Song et al., 2004).

3. Microorganisms can also grow at psychrophilic temperatures (15-19°C). However, low biogas production is achieved for anaerobic digestion at these temperatures and thus industrial anaerobic digestion processes do not normally operate in the psychrophilic range (Lettinga, 1995).
4. Until the early nineties, all plants operated under mesophilic conditions. Nowadays, mesophilic systems continue to be used more than thermophilic systems (70-30°C), since mesophilic bacteria are more tolerant to changes in environmental conditions than thermophilic bacteria. However, operating at higher temperatures facilitates greater hygienisation of the materials and gas production is more rapid (Huyard et al., 2000). In 2004, 77% of the treatment capacity in Europe was provided by mesophilic plants (De Baere and Mattheeuws, 2009). However, a large number of thermophilic plants were built in 2005 and 2006 and it is estimated that between 2006 and 2010, 41% of the plants in Europe will operate under thermophilic conditions (De Baere and Mattheeuws, 2009).

Table 1.5 describes some benefits/opportunities and limitations/barriers of the anaerobic digestion process that must be considered.

Table 1.5. *Benefits and limitations of anaerobic digestion (Braber, 1995).*

Benefits	Limitations
- Net production of energy provided by the use of biogas. - When the biogas is valorized, there is a reduction on CO_2 emissions by displacement of fossil fuels. - Potential to treat the wet fraction of MSW which is less amenable to incineration. - Volume reduction of the waste for subsequent disposal. - Some recalcitrant xenobiotic compounds can be biodegraded. - Biomass acclimatization allows a broad type of organic compounds to be transformed. - If agro-waste is processed, it can contribute to the economic viability of farms by keeping both the costs and benefits within the farm if products are used on-site. - End products can potentially be sold (biogas, soil conditioner and a liquid fertilizer).	- Anaerobic digestion facilities have capital and operation costs higher than composting facilities. - Digested waste may require further treatment with an aerobic treatment process to meet discharge requirements. - Input waste may need heating (often by utilization of process gas) to achieve adequate reaction rates. - The process is sensitive to temperature, pH, loading rate and changes of feedstock type. - Anaerobic digestion facilities often have problems of corrosion due to the presence of H_2S in the biogas. - Explosion risk due to the production of biogas. - The process is very sensitive to the presence of toxic substances.

1.1.3 Combined biological treatments

If the collected MSW is high quality, it can be treated directly using composting processes. In cases where low quality MSW is collected (i.e. it contains a large amount of inappropriate materials), mechanical biological systems (MBT) are needed to treat the material. MBT systems involve mechanical pre-processing stages to sort out recycled materials such as metals, plastics and glass, and biological stages to reduce and stabilize the organic matter using controlled anaerobic and/or aerobic (composting) conditions (Juniper, 2005).

1.2 Environmental impacts from biological treatment processes

Although the objective of a biological treatment plant is to safely transform wastes into less polluting and/or dangerous substances as well as, when possible, to obtain useful products, the organic wastes recycling processes employed in large scale treatment facilities inherently involve some potentially negative environmental impacts. Odor emissions and the release of atmospheric pollutants such as volatile organic compounds (VOCs), ammonia (NH_3), methane (CH_4) and nitrous oxide (N_2O), among other gases, are the most common emissions produced in these facilities (Komilis et al., 2004; Smet et al., 1999; Epstein, 1997). Energy and water consumption as well as leachate generation are other impacts that must be considered.

Two main groups of studies which deal with the environmental performance of organic waste treatment plants exist in the literature: those exclusively focused on the atmospheric emissions of the treatment process itself and those which perform Life Cycle Assessments (LCA) to evaluate total impacts (i.e gaseous emission, energy consumption, wastewater generation and waste transportation).

Important conclusions can be obtained from atmospheric emissions studies. There are many sources of odor and pollution in composting plants. These might include the reception and handling of materials, forced aeration composting, stock piling, etc. Gaseous emissions in composting facilities typically contain nitrogen-based and sulphur-based compounds as well as a wide variety of VOCs. Anaerobic metabolism produces a number of odorous compounds including inorganic molecules (ammonia and hydrogen sulphide) and organic molecules such as VOCs, aromatic compounds, mercaptanes and alkyl sulfides (Komilis et al., 2004).

Among the nitrogen-based compounds released to the atmosphere, ammonia has received a great deal of attention because, as it is released in large quantities, it causes a large amount of odor compared to other compounds and is the main nitrogen gas emitted during composting (Beck-Friis et al., 2001).

The reported ammonia emissions for composting the organic fraction of municipal solid wastes (OFMSW) vary between 18 to 1150 g NH_3/t waste (Clemens and Cuhls, 2003) whereas concentrations as high as 700 mg NH_3/m^3 have been detected in the exhaust gases of wastewater sludge composting installations (Haug, 1993). Laboratory studies have shown that ammonia emissions exhibit a clear correlation with process temperature, reaching maximum values during the thermophilic period (Pagans et al., 2006a). Aeration rate, pH and initial total ammonia nitrogen content are other factors that directly influence the emission of NH_3 during the composting process (Cronje et al., 2002; Beck-Friis et al., 2001; Grunditz and Dalhammar, 2001).

Another major group of gaseous pollutants emitted from composting facilities are VOCs, which are a group of organic compounds whose vapor 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), who undertook an exhaustive characterization of the different VOCs emitted during different stages of the composting process, most VOCs in composting plants are emitted at the early stages of the process i.e. at the tipping floors, at the shredder and during the initial forced aeration-composting period. Other authors relate the presence of some VOCs and their concentration to the odor nuisance level (Mao et al., 2006; Defoer et al., 2002). 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 dimethylsulphide, 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. Laboratory studies have shown that the highest concentration of VOCs in the exhaust gases from composting different wastes occurs during the first 48 hrs of the process (Pagans et al., 2006b). These authors stated that VOCs emissions could not be correlated with the biological activity of the process. In addition to the odorous disturbance that VOCs may cause, the presence of xenobiotic VOCs in gaseous emissions from composting municipal solid waste has also been reported (Komilis et al., 2004).

As stated earlier, composting is an aerobic process and a large fraction of the biodegradable organic carbon in the waste material is converted into biogenic CO_2. According to the IPCC (2006), the CO_2 produced as a consequence of a biological treatment is not considered to be a source of environmental impacts because it is biogenic in origin (it belongs to the short CO_2 cycle, the biogenic carbon is sequestered from the atmosphere by biomass and reported as carbon stock). On the other hand, although composting is an aerobic process, zones of anaerobic decomposition appear within the material during the composting process and can be intensified due to mismanagement. Methane is produced in strict anaerobic conditions due to incomplete degradation of soluble lipids, carbohydrates, organic acids and proteins (Husted, 1994). The warming potential of methane is 25 times higher than that of CO_2 (IPCC, 2001). Notwithstanding the fact that methane is degraded in the troposphere by reacting with OH^- radicals, the continuous emissions of other pollutants such as CO, NOx, SO_2 and organic compounds, which also react with these radicals, decreases the purifying capacity of the atmosphere. For this reason, methane accumulates and due to its high residence time in the atmosphere (about 10 years) a fraction of this methane (approximately 10%) diffuses to the stratosphere where it is involved in processes that deplete stratospheric ozone (IPCC, 2001). A number of studies have estimated that the amount of CH_4 released into the atmosphere during composting is less than 1% of the initial carbon content of the organic matter (Detzelet al., 2003; Beck-Friis, 2001).

Incomplete oxidation of ammonium and incomplete denitrification during the composting process results in the emission of N_2O (Beck-Friis, 2001). The global warming potential of N_2O is 296 times that of CO_2 (IPCC, 2001). Furthermore, as it is a very stable gas, it is travels high up in the atmosphere where it causes ozone layer depletion. Estimations of N_2O emissions from composting processes vary from less than 0.5% to 5% of the initial nitrogen content in the organic matter (Detzel et al., 2003; Beck-Friis, 2001; Hellebrand, 1998; Petersen et al., 1998; Vesterinen, 1996). N_2O emissions are influenced by temperature, with the greatest emissions occurring at temperatures below 45°C (Hellebrand, 1998).

In anaerobic digestion, the CO_2 generated is of biogenic origin as thus, as in composting, these emissions are not considered contributors to global warming (IPCC, 2006). CH_4 emissions generated in anaerobic digestion plants due to disruption of the process or technical problems are normally between 0 to 10% of the total CH_4 produced. However, under optimal operation, CH_4 emissions are near zero (Pipatti and Manso, 2006). On the other hand, N_2O emissions in anaerobic digestion plants are considered negligible but N_2O data are scarce (Pipatti and Manso, 2006).

In addition to the impacts discussed, all of the aforementioned gaseous emissions have repercussions on human health and quality of life (Tsai et al., 2008); issues, which have affected recent environmental policy for solid waste treatment plants (APAT, 2003; IPCC, 2001).

Table 1.6 shows reported quantities of gaseous compounds (NH_3, VOCs, CH_4 and N_2O) emitted during the composting process of MSW from several works.

Table 1.6. *Gaseous emissions on the MSW composting process.*

Substance	Value	Emission unit	Reference
NH_3	0.17	kg/t OFMSW	Pagans (2006a)[1]
	0.38	kg/t OFMSW	Baky and Eriksson (2003)[1]
	0.02 to 1.15	kg/t OFMSW	Clemens and Culhs (2003)[3]
	0.38	kg/t OFMSW	Díaz (2005)[2]
	2.30	kg/t OFMSW	Muñoz and Rieradevall (2002)[2]
	0.15	kg/t OFMSW	Smet (1999)[1]
	4.70	kg/t OFMSW	Tchobanoglous (1994)[4]
VOCs	1.70	kg/t OFMSW	Baky and Eriksson (2003)[1]
	1.69	kg/t OFMSW	Díaz (2005)[2]
	4.30	kg/t OFMSW	Diggelman (2003)[2]
	0.59	kg/t OFMSW	Smet (1999)[1]
	0.59	kg/t OFMSW	Muñoz and Rieradevall (2002)[2]
	61.70	mg/C m^3	Pagans (2006b)[1]
CH_4	1.54E-04	kg/t OFMSW	Díaz (2005)[2]
	1.2E-02	kg/t OFMSW	Clemens and Culhs (2003)[3]
	3E-02 to 8	kg/t OFMSW	Beck-Friis et al. (2001)[1]
	5.8E-08	kg/t OFMSW	Diggelman (2003)[2]
N_2O	6E-02 to 6E-01	kg/t OFMSW	Hellebrand (1998)[1]

[1] Data from laboratory study
[2] Bibliographic data
[3] Real data of a full-scale mechanical-biological treatment plant
[4] Emission factor estimated from $C_{60}H_{94.3}O_{37.8}N$ aerobic decomposition reaction

As shown in Table 1.6, gaseous emissions reported for the composting process vary among authors. These variations can be attributed to the differences in the scale of composting, the composition of the MSW as well as whether real or bibliographic data was used. Values in

Table 1.6 show how the data obtained in real systems or in laboratory studies contrast with the theoretical values used in many studies. For example, in the case of ammonia emissions, values in Table 1.6 range from 0.02 to 4.70 kg/t OFMSW. In the full-scale study by Culhs and Clemens (2003) these values range from 0.018 to 1.15 kg/t OFMSW whereas the values obtained in a theoretical study by Tchobanoglous (1994) are much lower (4.70 kg NH_3/t OFMSW). VOCs emissions range from 0.59 to 4.3 kg/t OFMSW, with the highest reported values corresponding to bibliographic data. Values reported for methane emissions vary widely as well (5.8E-08 to 1.2E-02 kg/t OFMSW) and in this case, the highest values were obtained in full-scale study. Finally, with respect to N_2O emissions, the only values found were from laboratory studies of OFMSW composting and these ranged from 6E-02 to 6E-01 kg/t OFMSW.

It is clear from this comparison that the results of environmental impact studies vary considerably depending on the type of study, theoretical or real, and depending on the size and type of system studied, thus underlining the importance of being careful when examining such results.

1.3 Life Cycle Assessment

A life cycle assessment (LCA) is a methodological tool for studying the environmental aspects and potential impacts of a product or service throughout its lifecycle, from extraction of raw materials, production, its use and finally, its disposal. An LCA involves the development of relevant inventory inputs and outputs of the system (inventory analysis), the assessment of their potential impacts (impacts assessment) and the interpretation of the results within the context of proposed targets (interpretation) (ISO 14040, 1997). In brief, LCA comprises mass and energy balances applied to the product systems, plus an assessment of the environmental impacts related to the inputs and outputs.

The first official definition of LCA was provided by the Society for Environmental Toxicology and Chemistry (SETAC on 1991):

> “Life Cycle Assessment is an objective process to evaluate the environmental burdens associated with a product, process or activity by identifying and quantifying energy and materials used and wastes released to the environment, to assess the impact of those energy and material uses and releases to the environment, and to evaluate and implement opportunities to affect environmental improvements. The assessment includes the entire life cycle of the product, process or activity, encompassing extracting and processing raw materials, manufacturing, transportation and distribution, use, re-use, maintenance; recycling and final disposal” (Consoli et al., 1993).

Several tools to support decision-making have been developed in environmental management such as cumulative energy analysis (CERA), environmental impact assessment (EIA), environmental risk assessment (ERA), input-output analysis (IOA), material flow accounting/substance flow analysis (MFA/SFA) and material intensity analysis (MIA) (Wrisberg et al., 2002). These tools include the term life cycle in their definitions but the LCA differs because its principle goal is to reduce the use of resources and the volume of waste in order to optimize the environmental performance of the process being studied. The life cycle analysis is a useful for making informed decisions because it allows the environmental impacts of different products and activities to be compared (Fullana et al., 1997). According to Sonnemann (2005), LCA methodology is the most widely accepted, it has been standardized (ISO, 2006) and it is the only tool undergoing the process of harmonization.

1.3.1 Life cycle assessment phases

Life Cycle Assessments can be divided into four steps: goal and scope definition, inventory analysis, impact assessment and interpretation. These steps and their relationships are presented in Figure 1.2.

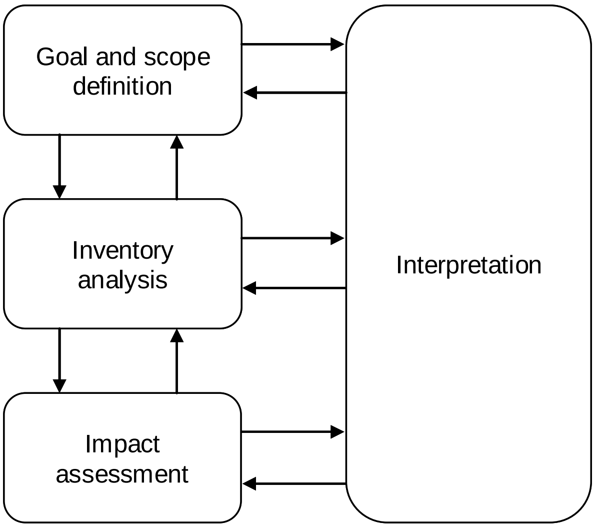

Figure 1.2. *General methodological framework of LCA (ISO 14040, 1997).*

1.3.1.1 Goal and scope definition

Defines and describes the product, process, service or activity to be studied. Establishes the context in which the assessment will be conducted and identifies the boundaries and environmental effects to be evaluated. This section is key for the development of the study, because it implies the definition of the functional unit and the system boundaries, which are linked to the entire analysis.

The functional unit is defined as the quantification of the function(s) of the process under study and its primary purpose is to provide a reference related to input and output data (ISO, 2000). For comparative studies, the choice of functional unit becomes critical and can have an important impact on the results obtained. For this reason, some works such as that by Haas et al. (2000) have performed their assessments using more than one functional unit, in order to observe how the choice of the functional unit impacts the results.

Generally, system boundaries are defined with respect to the following parameters (Tillman and Ekvall, 1994):

- Natural system, which means, the border between the technical system and natural environment;
- Geographical boundaries, which means, the area to which system is limited;
- Time boundaries, refer to the time perspective of the study, and,
- Technical boundaries relate to the activities that are considered in the study or relate to the life cycle of another product (if several systems share the same process, the environmental load will be shared between them).

1.3.1.2 Inventory Analysis

This phase involves the identification, collection and quantification of the data necessary to meet the defined goal and scope of the study. The level of detail of the inventory depends on the objectives outlined in the study (ISO, 2000). This phase is usually the most time and resource consuming step of a LCA (ISO, 2000).

1.3.1.3 Impact assessment

This step is devoted to the assessment of the potential human and ecological effects of energy, water and materials used and discharged to the environment. In an impact assessment, it is mandatory to do the following:

- Selection of impact categories, category indicators and models.
- Classification, in which inputs and outputs identified in the inventory analysis are grouped into impact categories or indicators (i.e. CO_2, CH_4 and N_2O emissions are classified as contributing to the global warming potential category; NH_3 emissions, NOx and SOx, and other emissions contribute to acidification potential, and VOCs, CH_4, NOx, benzene, hexane among other substances contribute to the photochemical oxidation potential).
- Characterization involves the quantification of the potential contribution of the inputs and outputs to the environmental impacts, which allows them to be aggregated into a single value by weighting. The following are some examples: Global warming potential, expressed in kg equivalent of CO_2, receives contributions from CO_2, CH_4, N_2O and other emissions. Ammonia, H_2S, HCl, NOx and SOx and other emissions contribute to the acidification potential, which is calculated in kg equivalent of SO_2. VOCs, CH_4, NOx, benzene, hexane and other emissions contribute to the photochemical oxidation potential which is expressed as kg equivalent of C_2H_4.

1.3.1.4 Interpretation

Involves the evaluation of the results of the inventory analysis and impact assessment to select the product, process or service with the best performance within the context of the goal and scope of the study (ISO, 2000).

It is important to highlight that only the first three phases are considered mandatory for impact assessment under ISO 14040 regulations (ISO, 2006).

1.3.2 Limitations of LCA

The holistic nature of the LCAs is simultaneously a major advantage and disadvantage of employing this methodology. Some of the main LCA limitations are listed below (Guinée, 2002):

- LCA cannot address localized impacts. This methodology does not provide the framework for a full-fledged local risk assessment study.
- LCA model focuses on physical characteristics of the industrial activities and some other processes but does not include the secondary effects of technological development.
- LCA regards all processes as linear, both in the economy and in the environment. Moreover, LCA focuses on the environmental aspects of products, and does not take into account economic or social effects.
- Many databases are being developed in several countries, and the format for databases is being standardized, however, in practice, data are frequently obsolete, incomparable, or of unknown quality.

1.3.3 LCA applied to waste management and treatment systems

Some literature can be found on waste management systems modeling, for example, EASEWASTE (Kirkeby et al., 2006), ORWARE (Sonesson et al., 1997) and WASTED (Diaz et al., 2005), are simulation tools which include the environmental burdens associated with waste management. LCAs have also been applied to study generic waste management systems (De Feo and Malvano, 2009) and MSW management systems in different cities and regions such as Wales (Emery et al., 2007), Ankara (Özeler et al., 2006), Phuket (Liamsanguan et al., 2008), Corfu (Skordilis, 2004) and Delaware (Ozge Kaplan et al., 2009). Other authors have focused their research on the environmental impacts of waste collection systems (Iriarte et al., 2009).

A common limitation in the majority of published works is a lack of field data corresponding to real waste treatment processes working under real conditions. In LCA analysis of waste management systems including a large number of processes, it is obvious that general assumptions must be made: gaseous emissions of the vehicles used for waste transportation, distances between collection points and waste treatment installations, waste composition, quantities and types of waste generated during treatments, etc. (Emery et al., 2007; Eriksson et al., 2005). However, special care should be taken when using of bibliographic data from different scenarios as this data may have been collected for a different purpose. If bibliographic data is used incorrectly, the final results may contain considerable error because technology type, waste composition and quality of process management strongly influence the results obtained in a study (Fricke et al., 2005). For waste management, theoretical LCA studies that include different waste treatment and handling operations cite different data sources to overcome the difficulty of obtaining reliable data from studies for the same waste in the same area (Güereca et al., 2006). In addition to these limitations, the use of data obtained in laboratory studies is not recommended as in most cases the process conditions in these studies do not correspond to those of a full-scale treatment plant (Szanto et al., 2007). Moreover, LCA studies focused on a particular organic solid waste treatment facility are scarce. An example of this type of study is that performed by Blengini (2008). Blengini (2008) used real data obtained from a plant in Italy to perform the facility's LCA.

To determine the environmental impacts generated by biological treatment installations using a LCA, data obtained in rigorous field studies of these installations are necessary to:

- Calculate emissions factors to allow for the comparison among different treatment plants,
- contribute atmospheric emissions data for industrial activities to regional, national and international databases databases such as the European Pollutant Emission Register (EPER) (European Commission, 2000), and,
- obtain on-site indicators of the environmental performance of different waste treatment processes to improve the design and operation of these treatment plants in order to minimize environmental impacts.

Chapter 2
Objectives

The principal aim of this work is to study the environmental impact associated with the biological treatment process and the organic fraction of municipal solid waste (OFMSW). Focusing on ammonia, volatile organic compounds (VOCs) and odor emissions determination at full-scale biological treatment facilities.

The following partial objectives have been established:

- Develop a systematical methodology as a tool to analyze environmental impacts of biological treatment facilities with a focus on gaseous emissions.
- Apply the methodology on full-scale biological treatment plants.
- Compare different OFMSW full-scale treatment facilities currently existing in Catalonia, in order to identify their yield from an environmental impact perspective.
- Incorporate tools such as life cycle assessment (LCA) as a complementary process in order to assess the environmental impacts of each analysis.
- Obtain a reference of the environmental impacts produced at the OFMSW full-scale plants based on LCA.
- Study the gas emission mitigation systems as a means of environmental impact reduction on the OFMSW full-scale plants.
- Identify the critical points and possible improvements of the studied biological treatment systems.
- Calculate the environmental impacts at OFMSW full-scale plants in relation to the efficiency of the biological treatment applied by means of respiration techniques.

Chapter 3
Materials and Methods

3.1 Solid sampling procedure

All the studied cases included in this dissertation (Chapters 5 to 8) correspond to full-scale OFMSW treatment plants. It was thus important to take samples at strategic points in the facilities to obtain a complete analysis of their processes. The sampling points considered in all the studied cases were the following:

- OFMSW input (fresh matter),
- decomposition stage,
- maturation stage and,
- final product, compost or digestate.

Due to the heterogeneity of the materials at the first three sampling points, the samples were prepared from 5 kg subsamples of the material extracted. For each pile of compost analyzed, 7 samples were taken. On each side of the pile, 3 lateral points were sampled approximately 1/4, 1/2 and 3/4 of the total length of the pile and one sample was taken from the top middle of the windrow, at a depth of approximately 60 cm. The sample of the final product (compost), comes from two subsamples. For all four sampling points the total sample volume was manually mixed in the laboratory. Physico-chemical and biological analyses were carried out from an aliquot of 1 kg of this representative sample.

All the samples analyzed in Chapter 8 came from an anaerobic digestion plant (digestate). Sampling specifications and material manipulations are outlined in Chapter 8.

3.2 Analytical methods for solid samples

Triplicates analysis were performed for heterogeneous samples and duplicates were performed when the samples were homogenous.

3.2.1 Physico-chemical parameters

The physico-chemical parameters of solid samples measured were selected in order to observe the behavior of the materials throughout the treatment process. Moisture content, organic matter present, acid and base concentration (pH), density, porosity, and total nitrogen formation (due to microorganism's activity) were measured for each solid sample. Physico-chemical parameters were determined according to the standard procedures outlined in *Test Methods for the Examination of Composting and Compost* (TMECC) (US Department of Agriculture and US Compost Council, 2001).

3.2.1.1 Dry matter and moisture content

Dry matter (DM) and moisture content (MC) were analyzed by calculating the water losses of the solid samples using Equations 3.1 and 3.2. The sample was dried at 150°C to a constant weight (18-24 hours) using a Theroven J.P. Selecta S.A oven. (Barcelona, Spain).

$$MC = \frac{(P_i - P_f)}{(P_i - P_0)} \cdot 100 \quad \text{(Equation 3.1)}$$

$$DM = 100 - MC(\%) \quad \text{(Equation 3.2)}$$

where,

MC = sample moisture content (%),

P_i= initial wet weight of the sample (g),

P_f= final dry weight of the sample (g), and,

P_0= beaker weight (g).

3.2.1.2 Organic matter

The organic matter (OM) equivalent to volatile solid content (VS) was determined by sample ignition at 550°C for 2.5 hours using a Hobersal muffle furnace (model 12 PR/300 Serie 8 B, Barcelona, Spain). OM was calculated using Equation 3.3.

$$OM = \frac{(P_i - P_f)}{(P_i - P_0)} \cdot 100 \quad \text{(Equation 3.3)}$$

where,

OM= organic matter of the sample (%),

P_i= initial dry weight of the sample (g),

P_f= final dry weight of the sample (g), and,

P_0= beaker weight (g).

3.2.1.3 pH

The pH was determined on slurry of compost. Deionized water at a ratio of 1:5, w/w basis was used.

The mixture was shaken for 20 minutes at room temperature to allow salts to solubilize in the deionized water. pH was measured with an electrometric pH meter (Crison, micropH200, Barcelona, Spain) on the compost/water solution.

3.2.1.4 Total nitrogen

This method included three principal steps:

- Sample digestion. This process converts all the organic nitrogen into ammonia. This change is achieved by exposing the sample to concentrated sulfuric acid in the presence of a catalyst at a high temperature.
- Distillation. The N-NH_4^+ from an aliquot is transformed into NH_3 by distillation in the presence of excess of base into a test tube containing an excess of boric acid at a known concentration.
- Titration. The difference between the equivalents of acid initially present and those remaining after distillation equal the equivalent of acid neutralized by ammonia, i.e. the equivalent of ammonia from both the N-organic and the N-NH_4^+ existing in the initial sample. Unlike the N-NH_4^+ content of the sample, the amount of organic nitrogen can be determined.

Total nitrogen Kjeldhal (TNK) was determined using 0.5 g of the sample. The sample was digested for 1.5 hrs at 400°C using 25 mL of concentrated sulphuric acid in 100 mL Kjeldhal tubes using a Bloc Digester 6 (with six tubes capacity) (J.P. Selecta S.A., Barcelona, Spain). To speed up the digestion, a catalyst (Kjeltab®) was added. Each digestion block contained two blank tubes that contained the standard amount of acid described above and a catalyst tablet (Kjeltab®). After allowing the sample to cool, the sample was diluted using deionised water. A Büchi Distillation Unit K-355 (Flawil, CH) was used for sample distillation with an excess of NaOH (35%). The condensate was placed in a conical flask with 100 mL of boric acid (4%) with mixed indicator. A colorimetric assay was used to measure the amount of nitrogen formed by adding, HCl and an acid indicator. TNK was calculated using Equation 3.4.

$$TNK = \frac{(V_i - V_0) \cdot N \cdot 14}{W_{wb}} \qquad \text{(Equation 3.4)}$$

where,

TNK= total N-Kjeldhal (%),

V_i= HCl volume consumed (mL) in sample titration,

V_0= HCl volume consumed (mL) in control titration,

N= normality of the HCl used in determination, and,

W_{wb}= sample weight in wet basis (g).

3.2.1.5 N-NH$_4^+$

Soluble N-NH$_4^+$ determination is based on the Kjeldahl distillation process (excluding digestion) as follows:

N-NH$_4^+$ was determined by distilling 10 mL of the same extract used for pH determination mixed with NaOH (35%) in excess. The N-NH$_4^+$ formed was measured using a colorimetric assay with the addition of HCl (0.25N). Soluble N-NH$_4^+$ was calculated as follows (Equation 3.5):

$$N - NH_4^{\ +} = \frac{(V_i - V_0) \cdot N \cdot 14 \cdot V_w}{10 \cdot P} \quad \text{(Equation 3.5)}$$

where,

N-NH$^+_4$= ammonium nitrogen of the sample (%),

V_i= HCl volume consumed (mL) in sample titration,

V_0= HCl volume consumed (mL) in control titration,

V_w= volume of water used for the extraction (mL),

P= sample weight (g),

N= normality of the HCl used in determination.

3.2.1.6 Bulk density and porosity

Bulk density (BD) is defined as the weight per unit volume of sample. BD was calculated on wet basis dividing the sample weight by the sample volume (Equation 3.6).

$$BD_w = \frac{W_s}{V_s} \quad \text{(Equation 3.6)}$$

where,

BD_w= wet bulk density (kg/L),

W_s= sample weight (kg),

V_s= sample volume (L).

Porosity (P) represents the free volume of the sample (not occupied by solid particles) and is expressed as a percentage. To determine P, the solid sample was transferred to a graduated beaker and was slowly saturated with deionized water until it no longer absorbed water. By measuring changes in the volume of the sample and the ratio of water retained, porosity volume was estimated (Equation 3.7

$$P = \frac{V_s}{V_w} \cdot 100 \quad \text{(Equation 3.7)}$$

where,

P= porosity (%),

V_s= sample volume (L),

V_s= volume of water-saturated sample (L).

3.2.2 Biological activity parameters

Respiration indices (static and dynamic) were used in this dissertation (Chapter 5 to 7) as a measure of biological activity and material stabilization, which are related to the amount of biodegradable organic matter present in the sample. Specific oxygen uptake rate (SOUR) test is another biological aerobic assay used in this thesis (Chapter 8) to measure the oxygen uptake rate in an aqueous solution that is a result of microbial respiration during the digestion of the suspended solid matrix.

Biogas production is also related to biological activity and organic matter degradation. In this study, anaerobic biogasification potential (ABP) was determined for digestate samples (Chapter 8).

3.2.2.1 Static respiration index

The static respiration index (SRI) was determined by using a custom built respirometer (Barrena et al., 2005). The static respirometer was built according to the original model previously described by Ianotti et al. (1993), using the modifications and recommendations described in TMCC (US Department of Agriculture and US Composting Council, 2001). A detailed description of the respirometer can be found in Barrena (2005). Approximately 250 mL of sample were placed in a 500 mL Erlenmeyer flask on a nylon mesh screen to allow for air movement beneath and throughout the solid sample. The setup included a water bath to maintain the temperature at 37°C during the respiration test. Prior to the assays, samples were incubated for 24 hrs at 37°C. During the incubation period, samples were aerated with previously humidified air at the sample temperature. The drop of oxygen content in a flask containing a sample was monitored with a dissolved oxygen meter (Lutron 5510, Lutron Co. Ltd., Taiwan) connected to a data logger. The rate of sample respiration (Oxygen Uptake Rate, OUR, based on dry matter content or organic matter content) was then calculated from the slope of the O_2% versus time curve, using Equation 3.8 (Ianotti et al., 1993). Results of the static respiration index are expressed in g O_2/kg OM h and are presented as a mean of three replicates.

$$SRI = \frac{V \cdot P \cdot 32 \cdot m \cdot 60}{R \cdot T \cdot X \cdot DM \cdot OM} \quad \text{(Equation 3.8)}$$

where,

SRI= respiration index (g O_2/kg OM h),

V= volume of air in flask (mL),

P= atmospheric pressure at elevation of measurement (atm),

m= slope of change in percent O_2 saturation per minute divided by 100,

R= ideal gas constant (0.082061 atm L/mol K),

T= temperature (K),

X= wet weight of material test aliquot (g),

DM= fraction of total solid of a parallel sample aliquot (g DM/g X), and,

OM= fraction of organic matter of a parallel sample aliquot in dry basis (g OM/g DM).

3.2.2.2 Dynamic respiration index

The dynamic respiration index (DRI) was determined following the methodology proposed by Adani (2006) adapted for an Erlenmeyer flask. DRI was determined by measuring the difference in oxygen concentration (mL/L) between the inlet and outlet airflow of the respirometer and calculated following Equation 3.9. In this respirometric method, approximately 100 g of sample are placed in a 500 mL Erlenmeyer flask and incubated in a water bath at 37°C. Air is continuously supplied to the samples using a mass flow-meter (Bronkhorst Hitec, The Netherlands) to ensure aerobic conditions (oxygen concentration higher than 10%). Oxygen content in the exhaust gas from the flask is measured by using a specific probe (Xgard Crowcon, UK) and recorded in a personal computer equipped with commercial software (Indusoft Web Studio, version 2008, USA). The incubation period ends when the oxygen concentration is equal to that in atmospheric air.

$$DRI_h = Q \cdot \Delta O_2 \cdot Vg^{-1} \cdot 31.98 \cdot OM^{-1} \quad \text{(Equation 3.9)}$$

where,

DRI_h= hourly dynamic respiration index (mg O_2/kg OM h),
Q= airflow (L/h),
ΔO_2= difference in oxygen concentration between the inlet and outlet airflow of the respirometer (mL/L),
Vg= volume occupied by 1 mol of gas at inlet air temperature (L/mol);
31.98= molecular weight of O_2 (g/mol), and,
OM= total organic matter (kg).

DRI is expressed as an average of 3 measurements of the oxygen uptake rate taken during the 24 hrs of the respiration assay when the highest activity was measured.

3.2.2.3 Specific oxygen uptake rate and oxygen demand

Specific oxygen uptake rate (SOUR) was performed as described in Lasaridi and Stentiford (1998) and modified by Adani (2003). 0.4 g of dried matter were set in flask with 500 mL of deionized water, 12 mL of phosphate buffer solution (KH_2PO_4, K_2HPO_4, $Na_2HPO_4 \cdot 7H_2O$) and 5 mL of nutritive solution ($CaCl_2$, $FeCl_3$ and $MgSO_4$) made up according to the standard biochemical oxygen demand (BOD) test procedures (APHA, 1992). Oxygen uptake potential is the oxygen demand during a 20-hour test (OD_{20}, mg O_2 /g DM). It is calculated using Equation 3.10.

$$OD_{20} = \frac{V}{(m \cdot TS \cdot 100)} \cdot \int_{t=0}^{t=20} \left|S\right|_t \cdot dt \quad \text{(Equation 3.10)}$$

where,

m= mass of the sample (g, wet weight),

V= sample volume (L),

TS= total solids content in the sample (in % over wet weight), and,

$|S|_t$ = rate of oxygen consumption at time t (mg O_2/L min).

3.2.2.4 Anaerobic biogasification potential

Anaerobic biogasification potential (ABP) was estimated as described in Schievano (2008). 0.62 g of dried matter, 37.5 mL of inoculum and 22 mL of deionized water were combined in 100 mL serum bottle. A control blank was prepared with 60 mL of inoculum. The reactors were sealed with Teflon hermetic caps, flushed with N_2, and incubated at 37 ± 1°C for 60 days. Biogas production was periodically measured. Quantitative biogas production was estimated by withdrawing extra-pressure gas with a 60 mL syringe. Biogas production of the control batches was subtracted from biogas production in each sample. Duplicates were performed for each test.

The inoculum was obtained using the output digestate of a full-scale anaerobic digestion plant. The digestate was incubated at 37 ± 1°C for 15 days before being applied.

3.3 Tools and analytical methods for determining NH_3 and volatile organic compounds emissions

Some sensors and instruments were used to sample and analyze gaseous emissions. These tools and methods were previously tested both in the laboratory and for a full-scale plant (Cadena, 2007) in order to ensure they meet the sensitivity requirements of this study.

3.3.1 Ammonia sensor

Ammonia emissions were analyzed by using an Industrial Scientific multigas sensor (iTX-T82) (Oakdale, PA, USA) with an ammonia detection range of 0 to 200 ppmv and a temperature range of 20 to 50°C.

3.3.2 Anemometer

To determine the exhaust gas velocity at each emission point, a thermo-anemometer (VelociCalc Plus model 8386, TSI Airflow Instruments, UK) was used. This anemometer has a measurement range from 0 to 50 m/s with a resolution of 0.01 m/s.

3.3.3 Gas pump and sample bags

Gaseous samples were collected in 1 L Tedlar® bags using a SKC Universal Pump model TX8 Deluxe (Eighty four, PA, USA).

3.3.4 Volatile organic compounds

Volatile organic compounds (VOCs) were determined using gas chromatography. VOCs content from gaseous samples were analyzed as total carbon content (C-VOCs) using a gas chromatograph (Agilent Technologies 6890N, Madrid, Spain) equipped with a flame ionization detector (FID) and a 2 m x 0.53 mm x 3.0 μm cdimethylpolysiloxane column (Tracsil TRB-1, Teknokroma, Barcelona, Spain). The column allows for the determination of the C-VOCs as a unique peak (Figure 3.1). The volume injected was 250 μL and the analysis time was 1 min.

Operating gas chromatography conditions were as follows:

- Oven temperature: isotherm at 200°C.
- Injector temperature: 250°C.
- FID temperature: 250°C.
- Carrier gas: helium at 1.5 psi pressure.

Data were acquired and quantified by Empower 2 software (Waters Associates Inc., Milford, USA). Calibration was performed using n-hexane (99% purity) (Spingo et al., 2003). The calibration curve was obtained by injecting different amounts of liquid n-hexane (pattern gas) in a clean and sealed 5 L glass bottle and then the gas sample was analyzed (Torkian et al., 2003). The volume of the pattern gas needed to obtain the desired concentration was calculated using Equation 3.11.

$$\mu l \; patterngas = X \; mgC / m^3 \cdot \left(\frac{MW \; patterngas \; V}{MW \; carbonN \cdot \rho} \right) \quad \text{(Equation 3.11)}$$

where,

X mg /Cm^3= pattern gas concentration expressed as mg C/m^3,
MW pattern gas= molecular weight of n-hexane (86.18 mg),
MW carbon= molecular weight of carbon (12.01 mg),
N= number of carbon atoms that contains one molecule of n-hexane (6 atoms),
V= air volume where the patter gas (liquid) is introduced (5.75E-03 m^3 in this case),
ρ= n-hexane density (0.659 g/mL).

For each VOCs concentration, three measurements were performed and the observed error of the method was below 10%. The detection limit of this technique was 1 mg C-VOCs/m^3.

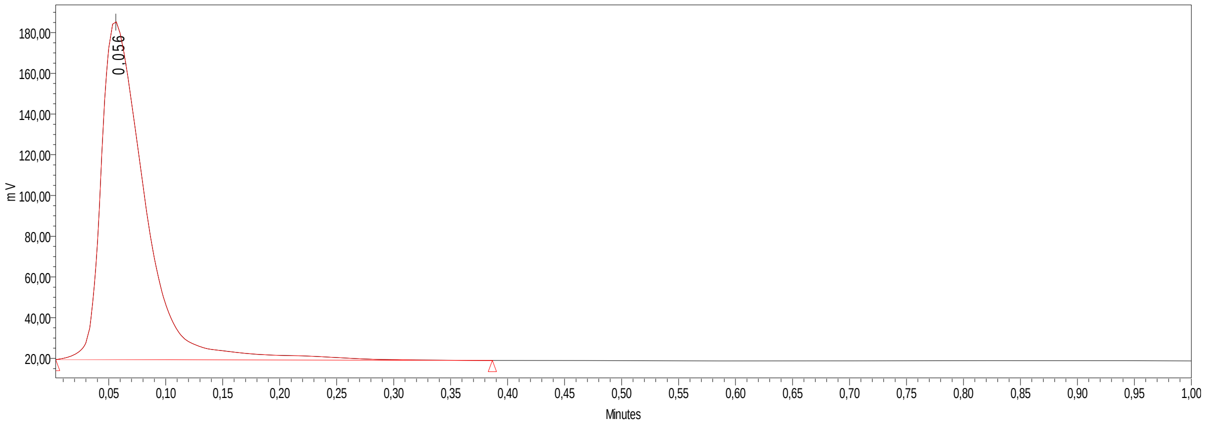

***Figure 3.1**. Example of a C-VOCs chromatogram, the peak corresponds to the total VOC emission.*

3.4 LCA software

SimaPro® 7.07.1, a software program was used to evaluate the environmental impact potentials (by PRé Consultants) using the Center of Environmental Science-Leiden University (CML) baseline 2000 V2.02 method (Guinée, 2001). This analytical tool is in accordance with ISO14040 standards (ISO 14042, 2000). SimaPro® is widely used in LCA studies (Rigamonti et al., 2009; Scipioni et al., 2009; Brambilla Pisoni et al., 2006; Bonoli et al., 2004; Morselli et al., 2002), and was chosen for the inventory and impact assessment phases because of its reliability, interactive potential and facility to adapt and change. It is important to note that only the first three LCA phases (goal and scope definition, inventory analysis and impact analysis) were considered in this, in this sense, interpretation phase was not evaluated.

The environmental impact potentials considered (concurring with several works discussed before and in Chapter 1) to perform the LCA perspective in all cases studied were:

- Global warming potential (GWP100).
- Acidification potential (AP).
- Photochemical oxidation potential (POP).
- Eutrophication potential (EP).
- Ozone depletion potential (ODP).
- Human toxicity potential (HTP).

3.4.1 Global warming potential

Global warming potential (GWP100) is defined as the impact that human emissions have on the radiative forcing of the atmosphere (i.e. heat radiation absorption), which is best known as the "greenhouse effect". Most of these emissions enhance radiative forcing, causing the earth's temperature to rise. Radiative forcing may have adverse impacts on ecosystem and human health as well as material welfare. The model developed by the Intergovernmental Panel on Climate Change (IPCC) defines the global warming potential of different greenhouse gases: CO_2, CH_4 and N_2O. The global warming potentials of greenhouse gases are given in kg CO_2 equivalent/kg emission for a 100-year time period (GWP100) (Guinée, 2002). According to the IPCC (2001), over a 100-year time period, CH_4 and N_2O are 21 and 296 times respectively more effective at trapping heat in the atmosphere than CO_2.

3.4.2 Acidification potential

Acidifying pollutants have a wide variety of impacts on soil, groundwater, surface waters, biological organisms, ecosystems and materials. Some examples of these impacts include fish mortality, forest decline and the crumbling of building materials. The major acidifying pollutants are SO_2, NOx and NHx. Acidification is reflected in deposition and/or acidification critical load. Acidifying Potential (AP) for each acidifying emission is calculated in kg SO_2 equivalents/kg emission (Guinée, 2002). According to the CML baseline method, NH_3, NOx and SOx correspond to 1.6, 0.5 and 1.2 kg SO_2 eq/kg, respectively (Guinée, 2002).

3.4.3 Photochemical oxidation potential

Photochemical oxidation potential (POP), which is also known as summer smog or secondary air pollution, is the formation of reactive chemical compounds from certain primary pollutants due to the action of sunlight. These reactive compounds have the potential to be harmful for human health and ecosystems and may also damage crops. POP may be formed in the lower layer of the atmosphere under the influence of ultraviolet light, through photochemical oxidation of VOCs and carbon monoxide (CO) in the presence of nitrogen oxides (NOx). Ozone is considered the most important of these oxidizing compounds, along with peroxyacetylnitrate (PAN). Emissions of VOCs or CO to the atmosphere are reported in kg ethylene equivalents/kg emission (Guinée, 2002). In this sense, CH_4, CO, NOx and no methane VOCs represent 0.007, 0.04, 0.028 and 0.416 kg C_2H_4 eq/kg respectively (Guinée, 2002).

3.4.4 Eutrophication potential

Eutrophication covers all potential impacts of the release of high levels of macronutrients—most importantly nitrogen (N) and phosphorous (P) to the environment. Nutrient enrichment can cause an undesirable shift in species composition and an elevated level of biomass production in both aquatic and terrestrial ecosystems. In aquatic ecosystems an increase in biomass production may lead to reduced oxygen levels as increased oxygen is consumed to decompose biomass(measured as BOD). In short EP is the N/P deposition equivalent in biomass. EP for each eutrophying emission to air, water and soil is expressed in kg PO_4^{3-} equivalent/kg emission (Guinée, 2002). For example, the EP of ammonia is equal to 0.35 kg PO_4^{3-} eq/kg and that of NOx is 0.13 kg PO_4^{3-} eq/kg (Guinée, 2002).

3.4.5 Ozone depletion potential

Ozone depletion potential (ODP) measures the potential of anthropogenic emissions to deplete or thin the stratospheric ozone layer. The thinning of the ozone layer allows greater amounts of UV-B radiation to reach the earth's surface. Increased UV-B levels can threaten human and animal health, terrestrial and aquatic ecosystems, biochemical cycles and materials. The World Meteorological Organization (WMO) developed this model to define the potential of different gases to deplete the ozone layer. The characterization factor of ODP for each emission to the air is in kg CFC-11 equivalent/kg emission (Guinée, 2002).

3.4.6 Human toxicity potential

Human toxicity potential (HTP) measures the potential of toxic substances present in air, water and soil to impact humans. HTP is expressed in kg 1,4-dichlorobenzene equivalent/kg emissions. The category indicator is the acceptable/predicted daily intake of toxic substances (Guinée, 2002). In this case ammonia and NOx have equivalences of 0.1 and 1.2 kg 1,4-dichlorobenzene eq/kg respectively (Guinée, 2002).

Chapter 4

Development of a methodology to assess environmental impacts on biological treatment plants

Part of the results presented in this chapter have been published under the name of: Erasmo Cadena, Joan Colón, Antoni Sánchez, Xavier Font, Adriana Artola, "A methodology to develop an inventory of gaseous emissions in a composting plant". Waste Management, 29, 2799-2807, 2009.

The study of different technologies and/or different methodologies of plants can determine the best processes that would result in a higher yield and produce less environmental impacts. As stated in the Introduction of this dissertation (Chapter 1) there is a lack of real data from currently operating full-scale biological treatment plants. In order to obtain these values, establishing a working methodology is necessary when performing a systematic study of the facilities. Basically it is essential to collect data on indicators, find optimal or ideal values for these indicators and then to establish criteria in order to identify whether or not the impact of a waste treatment facility has gone beyond the ideal values. Environmental impacts will be assessed from two points of view: impact factors and impact potentials.

The methodology developed to determine the environmental impacts associated with OFMSW biological treatment plants is composed of four different steps:

- Data collection of plant characteristics and operation,
- determination of atmospheric emissions,
- laboratory analysis, and
- calculation of emission factors.

Not only could environmental impacts be determined by this methodology, but also their relationship with plant operations.

4.1 Data collection on plant characteristics and operations

Data collection is systematically carried out using a specifically designed questionnaire. Plant characteristics are classified into general data, historical data and socio-economical data. Information such as plant capacity, soil occupation, characteristics of the final product obtained (compost and/or energy), including amounts, main product destinations, the quantities of waste treatment, energy and water consumption, as well as other plant operations are included in this data. Reliable data on plant operation could allow parallels and relationships to be drawn relating to the emissions obtained and the proposal for emission minimization options if necessary. Plant operation data is then grouped and related to process main operations, i.e. reception, pre-treatment, waste decomposition phase, curing phase and post-treatment, including process (technology) types, turning and/or aeration periodicity, watering, amount of rejected materials obtained from pre and post-treatment operations, type and amount of energy used (electric or fuel), equipment characteristics and machinery used. The questionnaire is included in Appendix 1 (English version). Due to confidential information from each installation included in the plant questionnaires, only a select few are included in this dissertation.

4.2 Atmospheric emissions from the biological treatment process

The sampling methodology proposed can be applied to determine the overall emission of any chemical compound in a full-scale biological treatment facility. Moreover, the pollutants studied in this dissertation include total VOCs and ammonia (NH_3). Ammonia emissions are analyzed *in situ* using the multigas sensor described in materials and methods of this thesis (Chapter 3). Total VOC emissions are determined in the laboratory by means of the gas chromatographic method also explained in Chapter 3.

The sampling methodology has been developed under the assumption that:

- Gaseous emissions from the biological treatment process, taking place in open facilities, are those released from composting windrows external surface, and,
- gaseous emissions from the biological treatment process that take place in enclosed installations with air collection and treatment are reduced to the emissions released from the external surface of the gaseous emission treatment equipments (usually biofilters).

A systematic data collection on air velocity and gaseous compound concentrations in external surfaces of biofilters and composting windrows is undertaken with minimal variations in the sampling methodology applied to these two types of emitting surfaces. The procedure followed is explained bellow.

4.2.1 Measurement of gaseous emissions from composting windrows

- The first step in measuring the gaseous emissions is determining the emitting surface dimensions of composting windrows, including: height, length, width and perimeter of the composting windrow, all of which are measured using a measuring tape.
- The second step is to define a matrix of sampling points covering the entire emitting surface: The number of sampling points for the windrow is based on its dimensions and are determined by dividing it into sampling profiles and considering at least three sampling points in each profile, shown in Figure 4.1. Out of the three sampling points, one point is considered for each profile and is located at the top of the windrow while the other two are located at the windrow sides (Figures 4.1a and 4.1b). For example, the distance within sampling profiles for a 20 m long composting windrow can be seen in Figure 4.1c. At each sampling point the temperature at the surface (at 10 cm depth) and inside the windrow (at 50 cm depth) was also determined.

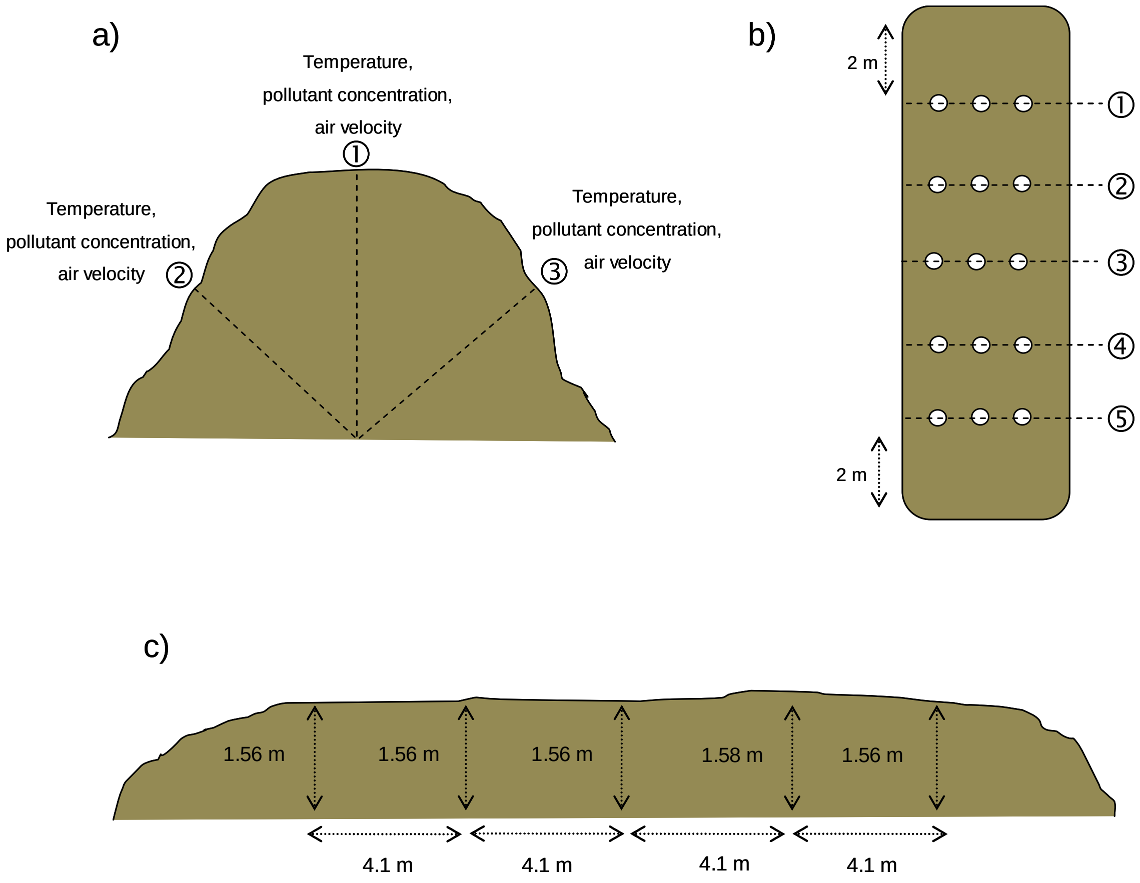

Figure 4.1. *Example of sampling points in a composting windrow: a) sampling on frontal view profile, b) sampling points represented in a windrow upper view, c) sampling profiles on a side. .*

- Exhaust air velocity at each sampling point is determined by means of a thermo-anemometer, described in Chapter 3, and a homemade Venturi system (Veeken et al., 2002). The Venturi was used to increase the air velocity in order to achieve more reliable data. The homemade Venturi built for the experiments conducted in this thesis (Figure 4.2) allowed an increase of air velocity by 41.5 times. This Venturi has a diameter entry of 21 cm, where at the narrowest part of the system the diameter is 0.5 cm (Figure 4.2a). To avoid potential interferences (wind and air turbulences) the Venturi was placed inside a protective plastic container that was open at bottom and top (Figure 4.2b).

- Concentration of ammonia, VOCs and air velocity were measured simultaneously at each of the sampling points. The three parameters were measured on the surface of the composting material without disturbing the gas flow pattern throughout the windrow. The product of compound concentration (mg/m^3) and air velocity (m/s) results in the mass flow of a given compound (ammonia or total VOCs) released per windrow surface area unit (mg/s m^2). Measures of gaseous emissions were repeated on different days throughout the composting process in order to determine the evolution of the emission of each compound. The sampling periodicity was established as a function of plant operation and the development of the composting process.

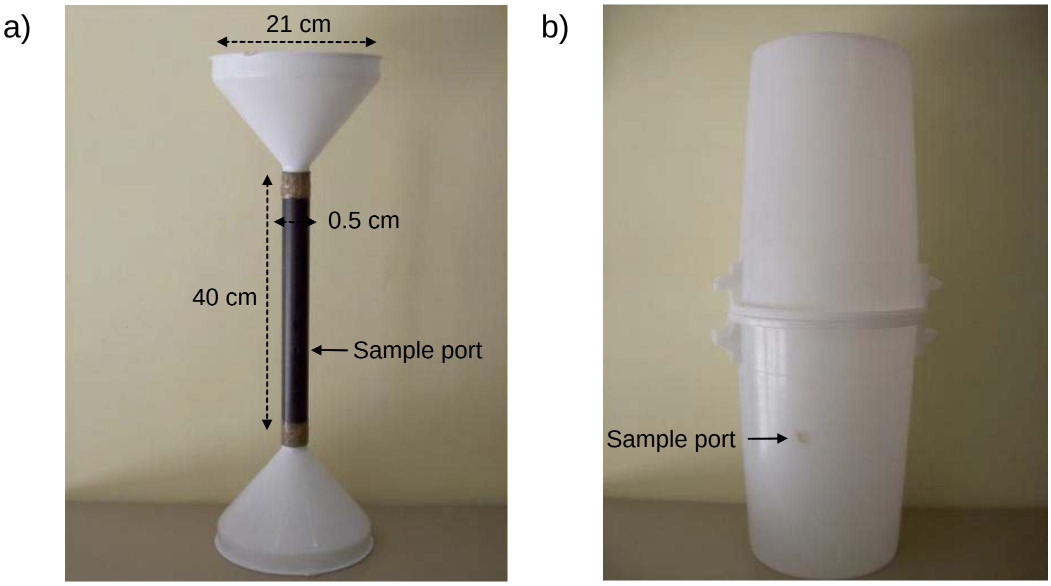

Figure 4.2. *a) Venturi system; b) Venturi protection.*

- Data obtained from emission measurements during a single sampling day are represented in a three-dimensional graph (Figure 4.3a) with the windrow length and perimeter on the x and y axes respectively. The center of the windrow is taken as the (0,0) point in the graph. Mass flow value of ammonia or VOC per square meter are placed on the z-axis to obtain an emission surface for each pollutant. The three-dimensional emission surface is then projected into a two-dimension graph (windrow perimeter on the x-axis and windrow length on the y-axis) where per area unit of emissions for each pollutant is presented as iso-emission surfaces (Figure 4.3b). To calculate the iso-emission surfaces area, obtained in the two dimensional representation, ImageTool version 3.0 (developed by the University of Texas Health Science Center, San Antonio, TX) was used. Multiplying the pollutant mass flow per area unit by the corresponding area in the graph resulted in the compound mass flow where the sum of the different quantities obtained corresponds to the total mass flow of a pollutant (kg/s).

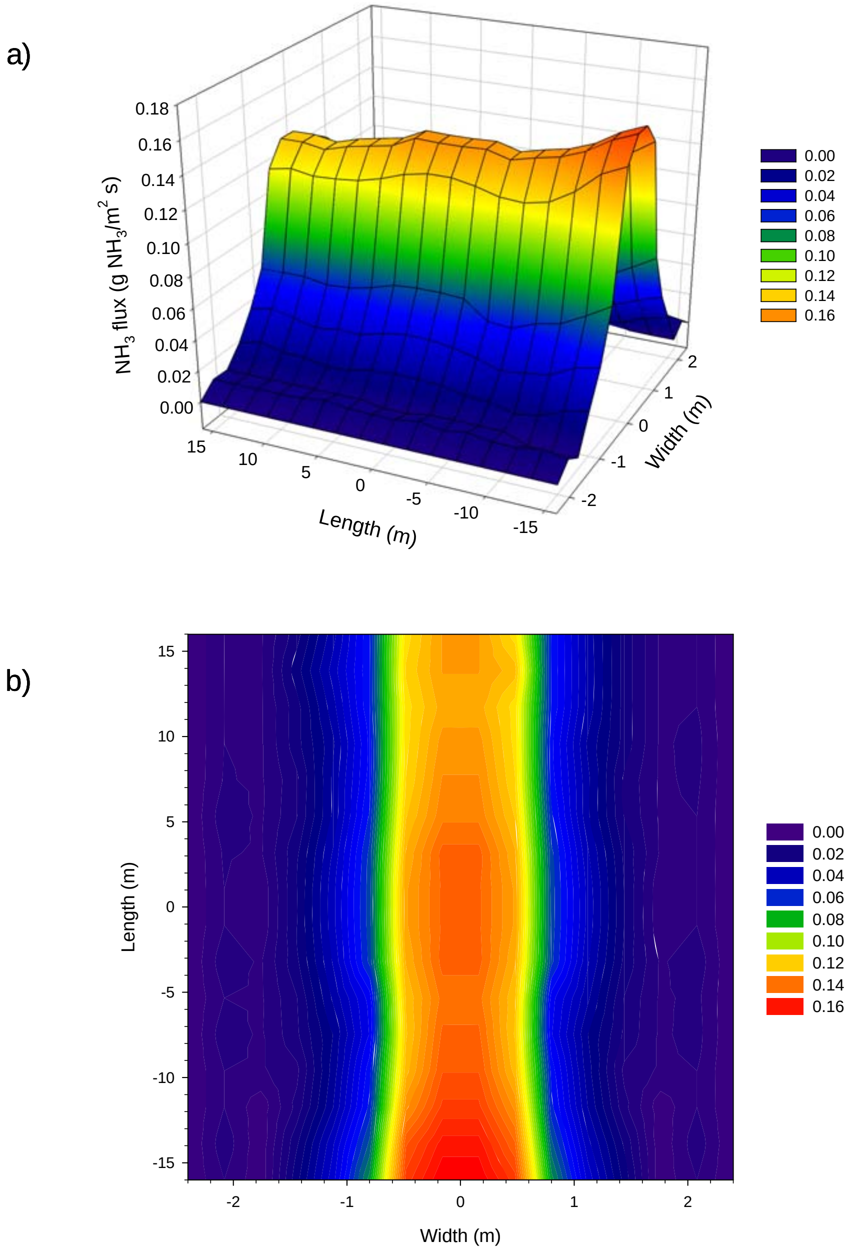

Figure 4.3. *Example of NH_3 emissions in a composting windrow: a) Three-dimension emission profile (g NH_3/m^2 s); b) two-dimension emission profile, iso-emission surfaces (g NH_3/m^2 s).*

- Finally, the values of pollutant mass flow that were obtained for each sampling day are represented in front of process time, as shown in Figure 4.4. The area below the obtained curve corresponds to the total mass of a given pollutant emitted throughout the composting process analyzed.

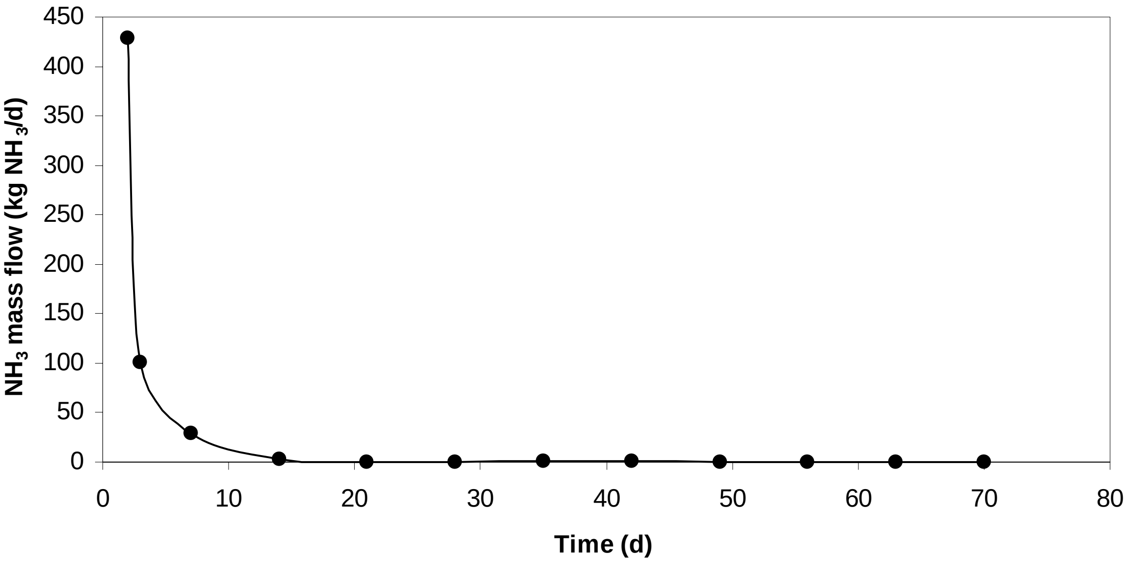

Figure 4.4. *Example of NH_3 emission profile, kg NH_3/d.*

4.2.2 Measurement of gaseous emissions collection in a biofilter

To determine the gaseous emissions that have been collected in a biofilter, the first step is to identify the possible existence of preferential pathways on the surface of the biofilter. The following step were used for this determination:

- Measurement of the external mission surface of the biofilter.
- Creation of a matrix of sampling points that cover the whole surface.
- Determination of the air velocity at each point, by using the anemometer described in Chapter 3 and the homemade Venturi system, previously commented.
- Simultaneously measure air velocity, NH_3 and VOCs concentration at each sampling point. Pollutant emission values for each sampling point are obtained and defined per unit area and time (g or kg pollutant/m^2 s).
- The difference between air velocity values, NH_3 and VOC concentrations at the different sampling points are determined. If these values are not significantly different, it can be concluded that there are no preferential pathways, thus the gas samples taken at any point on the surface is representative of the biofilter emission. If the values of velocity and/or pollutants concentration have a difference greater than 1% at each point, it is necessary to continue measuring points that cover these differences along the biofilter surface in order to gather data representative of the behavior of this systems. Once the number of sampling points is determined, the measurement process is repeated periodically to establish the evolution of the emissions over time (studied period).
- The data gathered in the above measures are used to determine the total amount of ammonia and VOCs emitted and are plotted in 3D graphics and projected in 2D graphs, as those in Figure 4.3. This information is used to obtain an emission surface for composting windrow emissions, following the same pattern explained above. Figure 4.5 represent the NH_3 emission surfaces (kg NH_3/m^2 s) obtained in a biofilter.

The emissions value per unit area and time gives the amount of pollutant that has been emitted per unit time. The total emissions per unit of time are obtained by adding all values gathered from the different zones of the biofilter.

The emissions of pollutant per unit time (kg/s) are plotted verses process time in the 2D graphic. The area under the curve (Figure 4.4.) of this plot represents the total mass of pollutant emitted during the studied period.

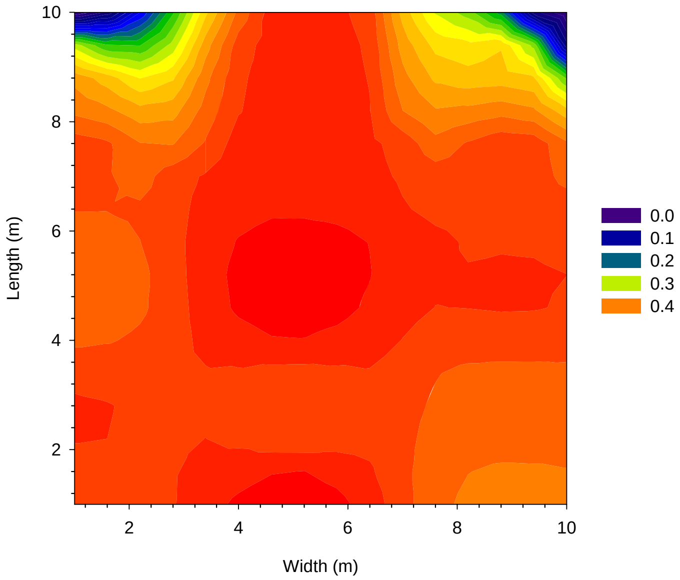

Figure 4.5. *Example of NH_3 emissions profile (iso-emission sufaces) on a biofilter surface (kg NH_3/m^2 s).*

4.2.3 Determination of atmospheric emissions from energy consumption

Emissions from fuel (diesel) and electricity consumption in the biological treatment plants were gathered from the BUWAL 250 database (BUWAL, 1998). The electricity models considered in this dissertation, including production and transport of primary energy sources, included the electrical power generation in Spain (Spain B250) for the cases studied in Chapters 5 to 7 and the Italian power generation model (Italy B250) for the case studied in Chapter 8. The fuel consumption model used was the Heat Diesel B250, from 1 kg of diesel, and includes the primary energy source for the emission data of the primary energy sources (Goedkoop, 2004). The values used are summarized in Table 4.1.

As stated in the Materials and Methods chapter of this thesis (Chapter 3), carbon dioxide emissions from the biological decomposition process are considered neutral. This emitted carbon has been previously fixed from the atmosphere by decomposition of organic matter (IPCC, 2006).

Emissions data of carbon dioxide and non-methane VOCs associated with energy consumption are provided by the plant managers (section 4.1) in the form of fuel and electricity. It is important to highlight that both methane and non-methane VOCs are individually determinate (Table 4.1), while total VOC emissions (including methane) from biological processes are determined. If the composting process is managed properly (as stated in the Introduction), CH_4 emissions should be very low and the differences between total VOCs and non-methane VOCs will not be significant.

Proposed by the Queensland Government Environmental Protection Agency (2008), diesel consumption is converted into kWh in order to calculate the total energy impacts, where 1 L diesel is equivalent to 10.67 kWh.

***Table 4.1**. Emission factors from energy sources considered on the studied cases in this dissertation.*

Pollutant	**Emission factor (kg)**		
	1 kWh electricity (Spain B250)	**1 kWh electricity (Italy B250)**	**1 L diesel (Heat Diesel B250)**
CO_2	0.53	0.59	2.85
NH_3	1.96E-06	8.84E-06	7.74E-08
Non methane VOCs	2.40E-04	9.86E-04	1.78E-02
NOx	1.18E-03	1.31E-03	5.13E-02
SOx	2.77E-03	4.52E-03	4.29E-03
CH_4	1.46E-03	1.12E-03	3.47E-03

4.3 Laboratory analysis

To determine the performance of biological processes of the facilities studied, analytical methods described in Chapter 3 are employed.

The solid sampling procedure required to analyze moisture, organic matter (OM), nitrogen content (as N-Kjeldahl) and pH was also described in Chapter 3.

Biological activity and material stabilization are determined by means of a respiration index. Static respiration index (SRI) was used at the plants studied in Chapters 5 and 6, while dynamic respiration index (DRI) was used at the plants studied in Chapters 6 and 7. Oxygen demand at 20 hours (OD_{20}) was applied at the plant discussed in Chapter 8. The methodologies used to determine these indices are described in Chapter 3.

4.4 Calculation of emission factors

Gaseous emissions generated in the biological treatment process are related to one ton of OFMSW treated in the plant. This functional unit allows the comparison of the plant capacities and will be used in this dissertation for the comparison the results obtained (Chapter 9).

In addition to gaseous emissions the functional unit also relates to energy consumption, its associated emissions, as well as water consumption (section 4.1). The results obtained for these emissions and operating factors are summarized in the table following the pattern in Table 4.2.

Table 4.2. *Input and output flows framework corresponding to the functional unit (1 t OFMSW).*

Inputs	Raw materials	t OFMSW/y t bulking agent/y
	Resources	kWh electricity/t OFMSW L fuel/t OFMSW m^3 water used in gas treatment/t OFMSW m^3 water in composting process/t OFMSW Total m^3 water/t OFMSW Total kWh energy/t OFMSW
Outputs	Atmospheric emissions (energy)	kg CO_2 electricity/t OFMSW kg CO_2 fuel/t OFMSW kg VOCs electricity/t OFMSW kg VOC fuel/t OFMSW
	Atmospheric emissions (biological process)	kg NH_3/t OFMSW kg VOCs/t OFMSW
	Product	t compost/y t compost/t OFMSW
	Refuse	t refuse/t OFMSW

Studied indicators are divided into inputs and outputs, and can be observed in Table 4.2. Inputs are subdivided into raw materials and resources. Raw materials are important because they have a significant repercussion throughout the study (OFMSW treated/year) and on the final mass balance. They represent, mainly, the amount of waste treated as well as the bulking agent and other amendments and chemicals if used. Fuel and water are also essential factors necessary to carryout a comprehensive analysis of the total impacts of resource consumption, such as electricity (kWh). Many times, resources consumption represents the largest environmental impact of the facilities, contributing greatly to potential impacts such as global warming or ozone layer depletion due to gaseous emissions.

The outputs section is categorized into atmospheric emissions from energy use, atmospheric emissions from biological processes, product generation (tons of compost produced) and refuse. Regarding atmospheric emissions (including process and energy consumption), only compounds (NH_3 and VOCs) are considered in this study. CO_2 emissions are accounted for in the case of energy consumption but are not evaluated in the case of biological treatment, as has been previously discussed. Product and refuse are also relevant indicators of the waste input and represent the real performance of the plant. On the other hand, the final balance between the input tons and those that are rejected reflects, among other things, the quality of the incoming organic matter.

Chapter 5

Environmental impact analysis of a confined windrow composting plant

Part of the results presented in this chapter have been published under the name of: Erasmo Cadena, Joan Colón, Adriana Artola, Antoni Sanchez, Xavier Font. "Environmental impact of two aerobic composting technologies using Life Cycle Assessment", International Journal of Life Cycle Assessment, 14, 401-410, 2009.

As discussed in previous chapters, in order to achieve an objective representation of the environmental impacts of a full-scale composting plant, it is necessary to obtain data on real emissions from this plant. This chapter describes the results obtained in a full-scale environmental impacts analysis of an OFMSW composting plant using confined windrow module technology.

5.1 Objectives

The principal aim of this chapter is to characterize and to analyze the environmental impacts of a full-scale composting plant that treats OFMSW under confined windrow technology focusing on the following:

- Impact indicators,
- impact potentials,
- process quality, and
- process improvements.

5.2 Composting plant description

The studied plant is located in Barcelona province (Catalonia, Spain). This plant uses a confined windrows composting technology that treats approximately 91 tons per year of OFMSW obtained from door-to-door collection. The study period began in the second week of August 2007 and ended in the first week of January 2008.

It is important to note that this facility is a low cost, small-scale plant that lacks complex equipment and is open to the atmosphere. During the course of this study, the plant had a single confined module, but they plan to expand their treatment capacity by installing two more confined modules in the near future.

The process steps of this plant are shown in Figure 5.1. Input materials are transported from the reception area to the decomposition using of an industrial tractor. Input materials remain for a maximum of 12 hours in the reception zone before they are moved to the decomposition area. OFMSW is conditioned to enhance its composting properties by mixing with pruning waste as bulking agent at a volume ratio of 6:4 (OFMSW: bulking agent) using a mixer operated by the same tractor. OFMSW does not need pre-processing in this plant because of its high quality in terms of organic matter and improper materials content. According to the OFMSW input characterization developed by the Agència de Residus de Catalunya (ARC, 2007), the improper material content of this waste during the study period was less than 0.64%.

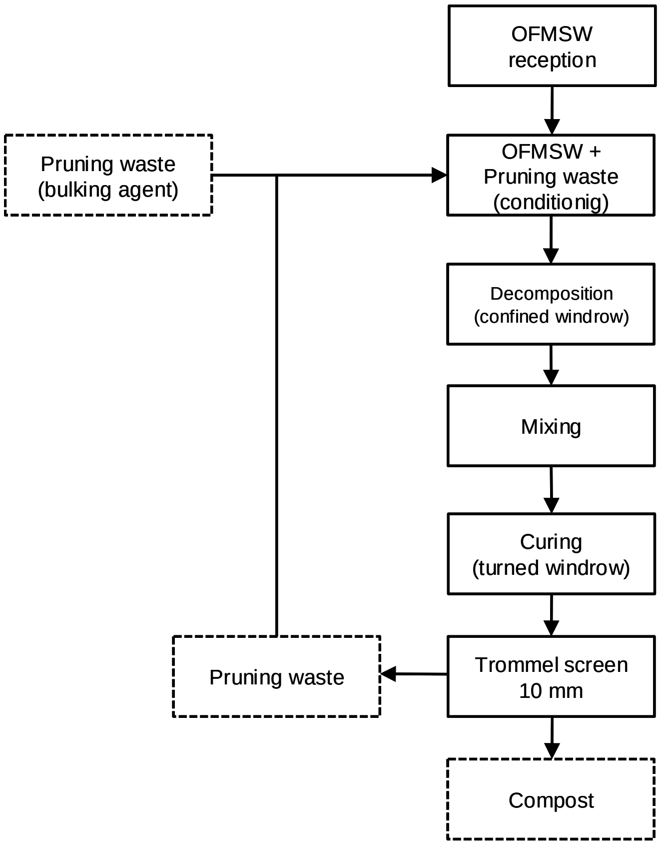

Figure 5.1. *Flowchart of confined windrow processes.*

After the OFMSW is conditioned, the mixture enters the decomposition stage, which is performed in a 2 x 6 m confined windrow module (Agrotech technology, model BC-F4, Vic, Spain, Figure 5.2) covered with a textile liner (Figure 5.3). The confined windrow module is made of concrete and has 3 ventilation tubes at its base. Its capacity is approximately 3.6 t and it operates under programmed forced aeration conditions. The aeration frequency follows a 2:8:20 minutes cycle. For 2 minutes the electric-valves open to drain the leachate generated during the process, for 8 minutes aeration occurs alternating between the 3 ventilation tubes and this is followed by a 20-minute rest period. Two pairs of temperature sensors (sensor group A and B) are strategically placed to monitor the temperature changes within the confined module that occur as a result of the decomposition stage.

Figure 5.2. *Composting material in the confined windrow module.*

Figure 5.3. *Confined windrow module with the textile liner cover.*

Leachate generated during decomposition is collected in a 500 L underground tank. The leachate collected in the tank is used to water the composting material in the confined windrow by means of perforated tubes that are positioned between the top of the windrow and the textile liner. These tubes are manually repositioned depending on the observed moisture content of the material. The decomposition period within the confined windrow lasts for 35 days.

After the decomposition stage, the materials are transported to the maturation area that takes place in a paved zone of the plant. The maturation stage is performed by means of turned windrows (Figure 5.4). The pile is 2 m high and 7 m long. The same tractor used in the mixing operations before the decomposition stage turns the material periodically. If the material needs to be watered during this stage, it is sprayed with tap water. The maturation stage in turned windrows is approximately 6-8 weeks long.

When the composting process is complete, the material undergoes post-treatment, which is achieved throughout a 10 mm grid trommel screen. Most of the final compost obtained is used in gardening and agriculture.

As stated earlier, this plant is open to the atmosphere, meaning that there is not treatment of gaseous emissions.

Figure 5.4. *Appearance of a windrow in the maturation area.*

5.3 Materials and methods

Analytical parameters were determined in the laboratory from representative material samples taken from the confined windrow and from the maturation pile. Routine analytical parameters such as moisture content, dry and organic matter, bulk density, porosity, pH, biological stability (static respiration index, SRI) were determined as explained in Chapter 3. Bulk density and porosity were determined *in-situ* as is also explained in Chapter 3.

The methodology described in Chapter 4 for the determination of the gaseous emissions of NH_3 and VOCs was used. According to the confined windrow module and the maturation pile dimensions in this plant (4 m profile and 6 m length and 3 m profile and 7 m length, respectively), 4 equidistant profiles were set in each case. As stated in the methodology section, 3 samples were taken for each one of the four profiles. One was taken from the top and one from each side of the windrow. Gaseous samples for VOC analysis were taken at the top of each emission profile.

The composting process in this plant was monitored twice, i.e. two confined windrows and two maturation piles were analyzed. During the first trial VOCs could not be analyzed due to technical problems with the gas chromatograph, for that reason the second trial was devoted to the analysis of VOC emissions. It is important to note that the results for ammonia emissions in both confined windrow and maturation pile were obtained in the first trial and VOC emissions were calculated were obtained from the second trial.

The first trial began the second week of August 2007 and ended the first week of November 2007, the second trial began the second week of September and ended the third week of December 2007.

The routine analytical parameters discussed above were determined from material samples taken during the first trial.

5.4 Results and discussions

5.4.1 Inventory analysis

Data obtained during the inventory analysis of the plant are presented for each stage of the composting process: decomposition and maturation or curing. As mentioned, the OFMSW residence time in the reception area before being loaded on the confined module was 12 hours. At this stage, NH_3 and VOC emissions were negligible.

5.4.1.1 Decomposition stage

Figure 5.5 shows an example of the ammonia emission profiles obtained from the composting confined windrow. Figure 5.5a and 5.5c represent the NH_3 emission on the second day at the decomposition stage, while Figure 5.5b and 5.5d represent the emissions in the last day of this stage (day 34). As shown in Figure 5.5, ammonia emissions in the last days of decomposition are higher than in the initial days of process. This makes sense because at this corresponds to the thermophilic stage where decomposition activity is high. In both cases, the highest emissions were measured at the top of the confined windrow and were lower on the sides on sides following the pattern of air movement during aeration.

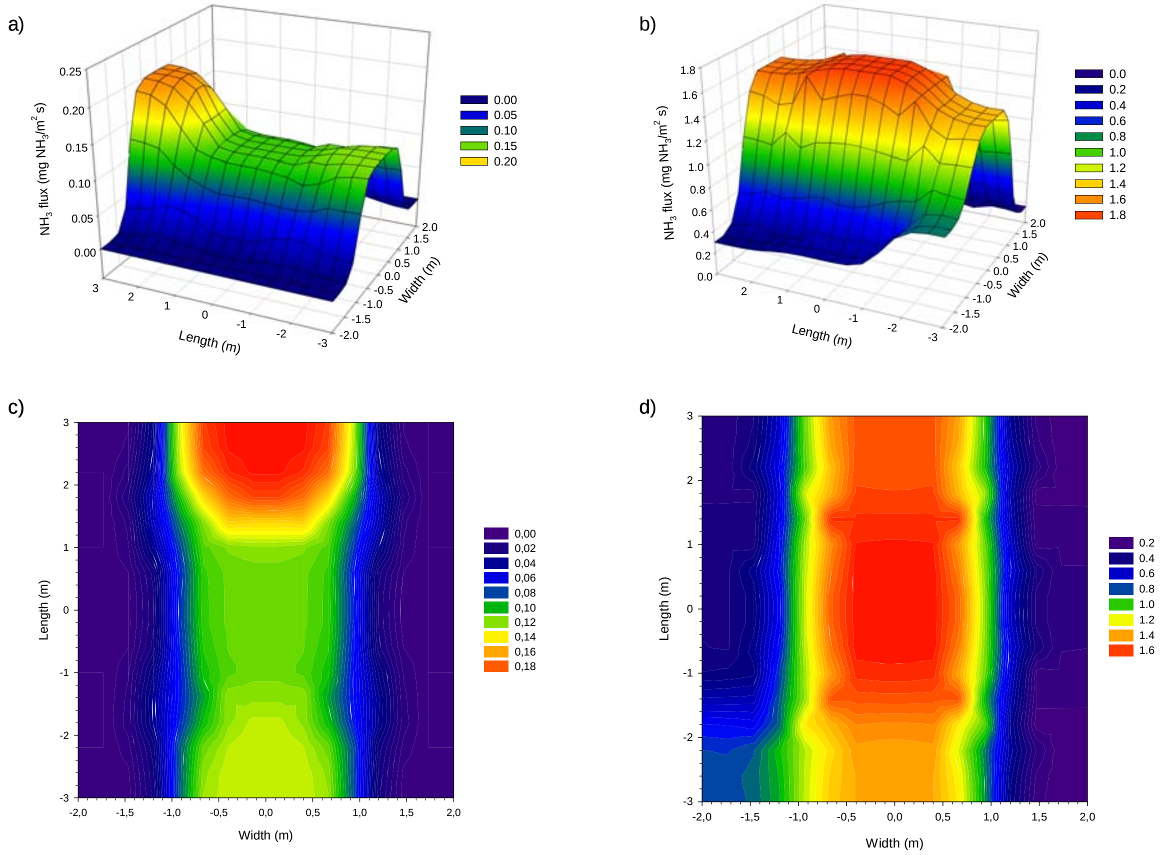

Figure 5.5. *Ammonia emissions from the confined windrow a) Ammonia emissions profile on the second day of decomposition, b) Ammonia emissions profile on the 34th day of decomposition, c) Two dimension projection of the ammonia emission profile on the second day of decomposition (mass flow mg NH_3/m^2 s), d) Two dimension projection of the ammonia emission profile on the 34th day of decomposition (mass flow mg NH_3/m^2 s).*

Figure 5.6 shows the evolution of the ammonia emissions of the confined module during the whole decomposition phase. The values obtained for ammonia emissions with or without aeration were the same throughout the studied period. In Figure 5.6 it can be observed that ammonia emissions increased from the beginning of the process, peaking at day 7 and decreasing by day nine. From day 9 to day 23, ammonia emissions remained within 0.55-0.62 kg NH_3/d but after 23 days, the emissions increased to more than 1.85 kg NH_3/d by the last day of the decomposition stage. Ammonia concentration in the gaseous emissions from this composting stage ranged from 0 to 140 ppmv.

The measured area under the curve corresponds to the total emissions of ammonia during the decomposition stage. In this case, ammonia emissions during this phase were 25.7 kg NH_3.

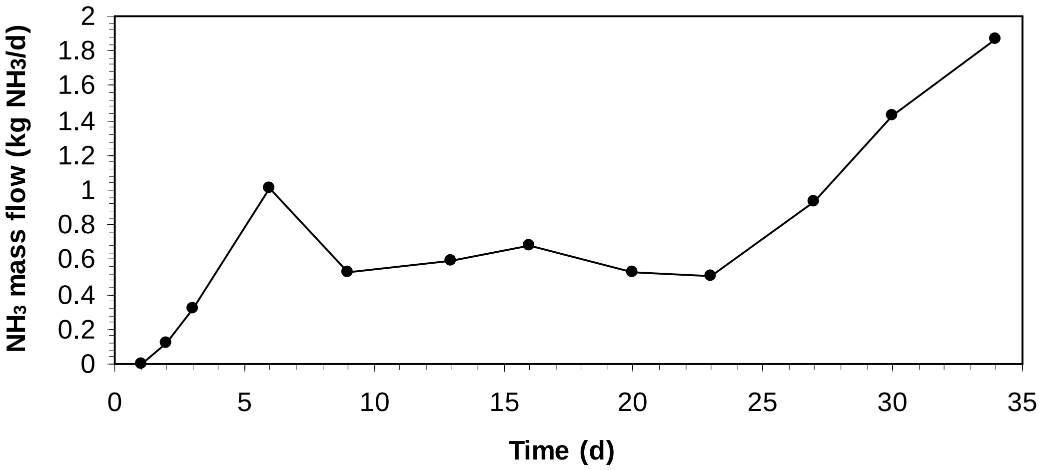

Figure 5.6. *NH_3 mass flow in decomposition stage (at confined module) obtained on each sampling day.*

Figure 5.7 shows the temperature values in the confined module during the decomposition stage. The sensor groups A and B were positioned 1 meter from the either side of the composting windrow. The temperature of the group A sensor group remained at an average of 67°C, while sensor group B showed fluctuations beginning with a temperature of 69°C and declining to reach 40°C by day 27. What caused this fluctuation was the excessive watering of the material in this zone. Following the 27th day, the temperatures increased and at the end of this stage the temperature was consistent throughout the confined module (66°C).

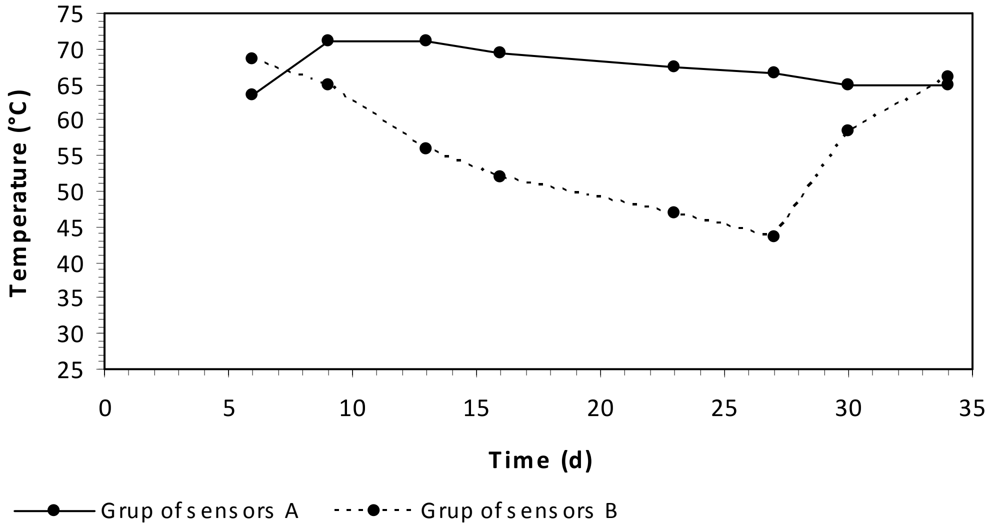

Figure 5.7. *Temperature pattern during decomposition stage (at the confined module) obtained on each sampling day.*

Figures 5.6 and 5.7 show that during the decomposition phase, there is some correlation between ammonia emissions and temperature. NH_3 emissions peak on the 6th day when the temperatures were high through the material but these emissions decrease when the temperature begins to decline in one of the composting module sides. NH_3 emissions increased at the end of the decomposition process and this coincided with a rise in temperature detected at the B sensor group. Pagans et al. (2006a), in a laboratory study of OFMSW composting found that there was a strong relationship between ammonia emissions and temperature. As temperature increased, an exponential increase of ammonia emissions was observed.

Figure 5.8 shows the VOC emissions during the decomposition stage with and without aeration of the materials. A trend in the VOC emissions can be seen. Beginning on the first day, total VOC emissions were zero and then throughout the process, these emissions fluctuated between 2 to 6.3 kg/d, and by the end of the decomposition phase the emissions returned to zero.

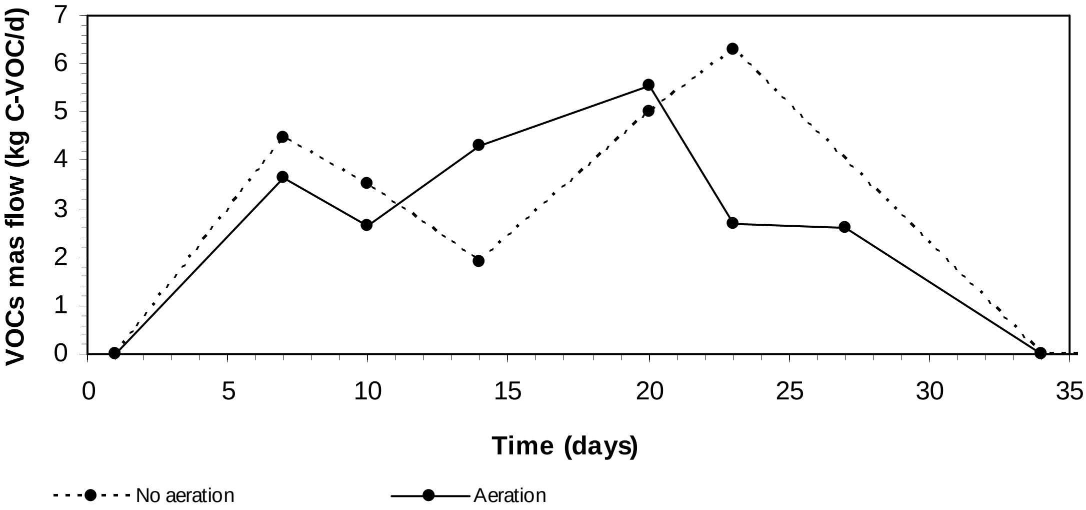

Figure 5.8. *Total VOCs mass flow in decomposition stage (in the confined module) obtained on each sampling day.*

Due to differences in VOC emission with and without aeration (8 minutes with aeration followed by 20 minutes without aeration), the two curves were used to calculate the total emission during this stage. The VOCs concentration in the emissions during this phase ranged from 0 to 118 mg C-VOCs/m^3. The weighted average of the total VOC emissions throughout the decomposition period was estimated to be 104.6 kg C-VOCs.

By comparing VOCs and temperatures within the confined module, it is clear that these two parameters do not follow the same pattern. Unlike ammonia, there is no relationship between VOC emissions and temperature. This result is consistent with the results in a study by this fact was also observed in Pagans et al. (2006b). However, another observation by Pagans et al. (2006b) that does not agree with the results of this study is the timing of the maximum VOC emissions. While they observed maximum VOC emissions at the beginning of the decomposition process, as shown in Figure 5.8, this result was not obtained here.

In addition to gaseous emissions other inputs and outputs (materials and energy) of the decomposition stage were considered:

- Electricity was consumed by the aeration system in the confined module. The energy consumption calculated during this phase was 1540 kWh.

- Fuel consumption in the decomposition stage is due to the operation of the tractor used to mix the materials and transport them from the reception to the decomposition area. It was not possible to obtain fuel consumption from the plant's operational data. Instead, the fuel consumption of the tractor was calculated using the equipment technical data. The theoretical diesel consumption for this tractor (New Holland TL100A) corresponds to 23.7 L/h. Since the tractor was used for 4 hours (obtained from the plant operation data) to perform the maneuvers in this phase (loading and unloading the material to the confined module and mixing), fuel consumption was equal to 94.68 L.
- Although leachate was used to water the material during the decomposition phase, 0.47 m^3 of tap water were also consumed during the study period (34 days in this phase).

As discussed above, the leachate generated is collected in an underground tank and re-circulated to the same decomposition process to maintain adequate moisture levels. pH and total nitrogen content (TKN) of the leachate were determined to be 5.34 and 45.4 ± 3.8 g/L respectively. These values agree with those obtained by other authors examining OFMSW composting leachate (Faday et al., 2007). According to the plant's operational data, leachate is fully consumed to water the material and there is therefore no need to treat the leachate.

5.4.1.2 Maturation phase

Figure 5.9 shows two examples of ammonia profiles obtained during the maturation stage (days 1-37). Figure 5.9 shows that the emissions were higher at the top of the pile during this phase, which was also seen in the decomposition stage.

A comparison emissions profiles on day 1 and day 37 (Figure 5.9a and 5.9c) shows that in the beginning of the process, the release of emissions was relatively consistent throughout the pile and as the process proceeds, emissions became concentrated in the bottom-middle of the pile. This is due to the chimney effect within the composting windrows (which was also seen in the confined windrow) (Figure 5.9b and 5.9d). This effect implies that there is a flow of water vapor and other exhaust gases within the windrow. Termed natural draft ventilation, this flow is analogous to the hot gases leaving a chimney (Haug, 1993). Normally, elevated temperatures would also be observed near the center of the windrow (Haug, 1993), but temperature was not measured during the maturation phase since a proper temperature probe was not available.

The evolution of ammonia emissions in the maturation stage is shown in Figure 5.10. Total emissions of ammonia during the studied maturation period (57 days) were 21.32 kg NH_3. Ammonia emissions decreased during the maturation process, from 2.6 kg NH_3/d in the beginning to nearly zero emissions by the end the end of week 5 (day 37). Ammonia concentration in the gas emissions from this composting phase ranged from 0 to 37 ppmv.

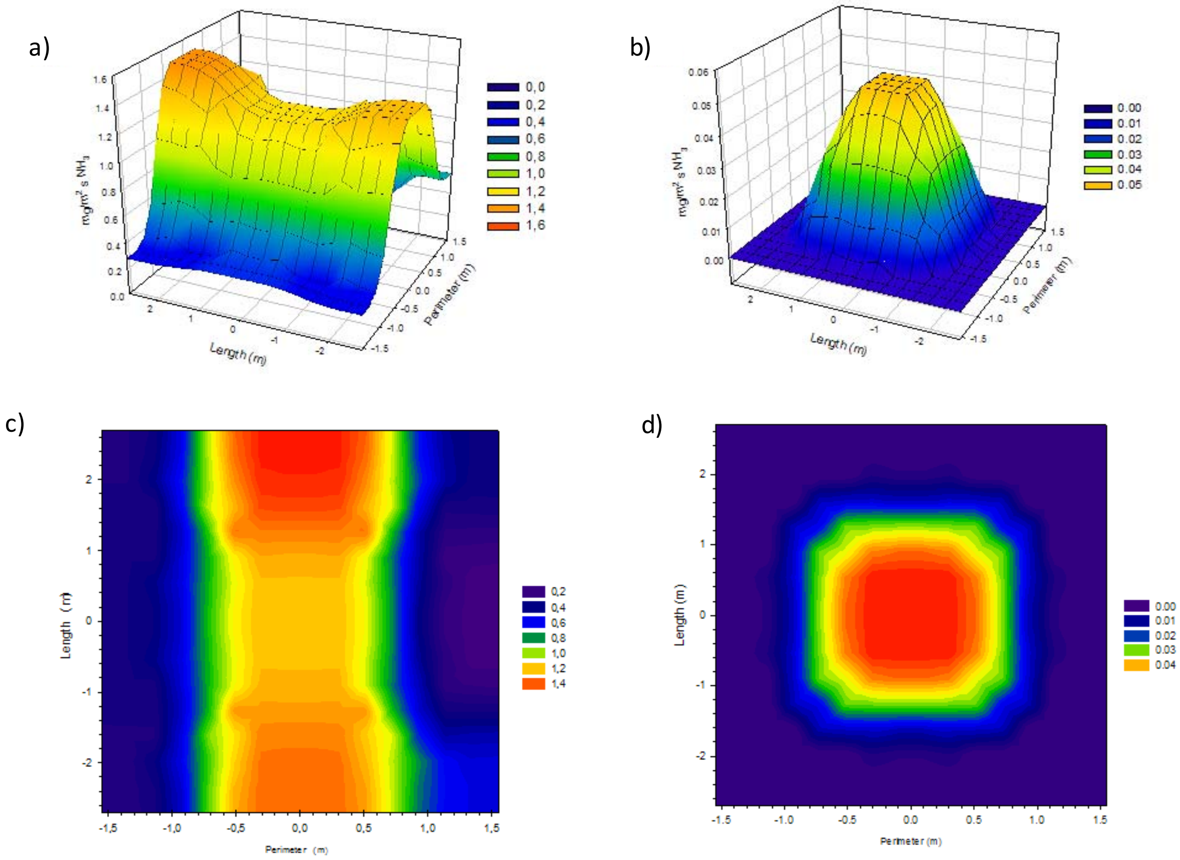

Figure 5.9. *Ammonia emissions from the maturation pile a) Ammonia profile at the first day of maturation, b) Ammonia profile at day 37 of maturation, c) Two dimension projection of the ammonia emission profile on the second day of maturation (mass flow mg NH*$_3$*/m*^{2} *s), d) Two dimension projection of the ammonia emission profile on day 37 of maturation (mass flow mg NH*$_3$*/m*^{2} *s).*

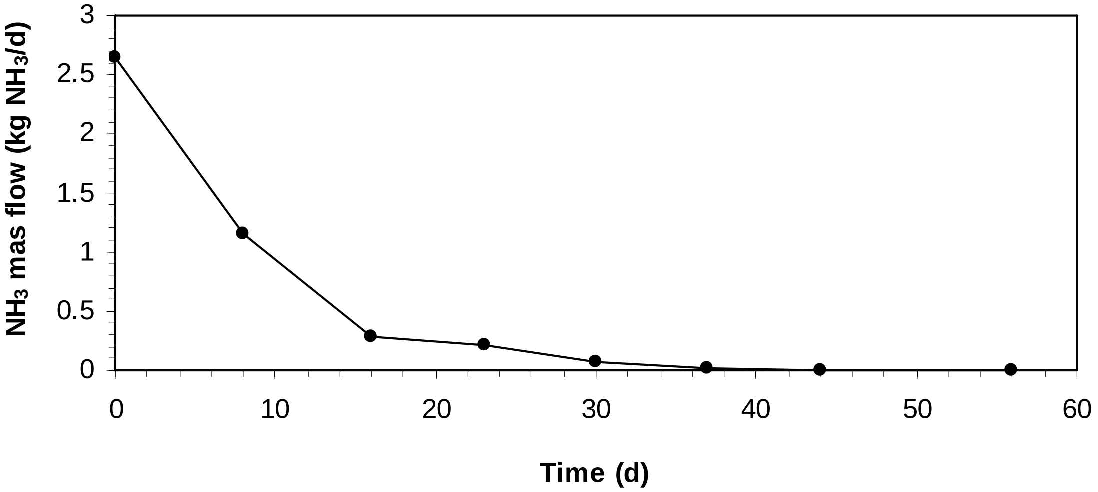

Figure 5.10. *NH*$_3$ *mass flow during the maturation stage (turned pile) obtained on each sampling day.*

VOC emissions in this stage (Figure 5.11) were between 0.5-1 kg C-VOCs/d, except for week 3 (day 23) when these emissions peaked to 3.3 kg C-VOCs/d. Therefore, unlike the case with ammonia, a clear trend in VOC emissions cannot be established. As shown in

Figure 5.11, the timing and number of samples of VOCs do not correspond with what was done for ammonia. As mentioned earlier, the reason for this is that VOCs samples were obtained during the second monitoring trial (September-December). It is clear that to determine VOC emissions for maturation piles, sampling must be done more regularly than what was possible in this study. The availability of analysis equipment in this study permitted for just four VOCs measurements. Given that the value obtained on day 21 is twice the values obtained on other days, it is clear that more samples need to be taken in order to obtain reliable VOC emissions data. The concentration of VOC emissions during this stage ranged from 10 to 66 mg C-VOCs/m^3.

Total VOC emissions measured for maturation stage were 67.06 kg C-VOCs.

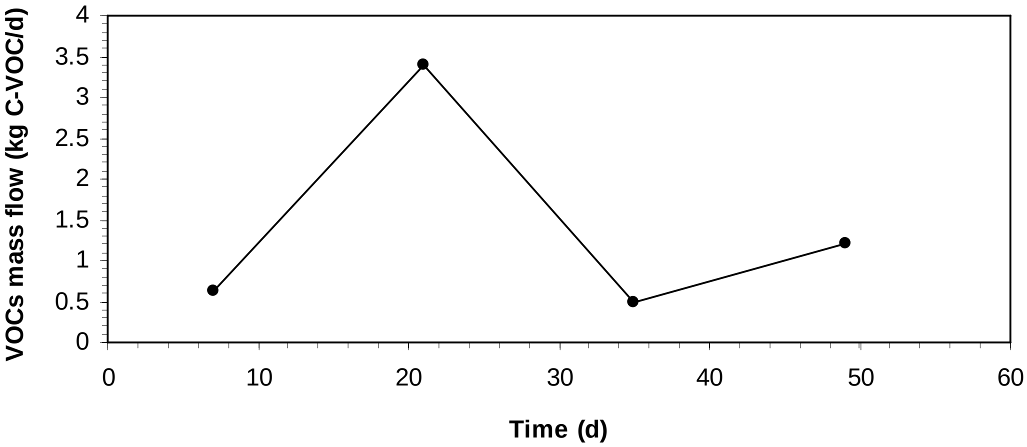

Figure 5.11. *Total VOCs mass flow in maturation stage (turning pile).*

Regarding the other outputs and inputs during the studied period of this stage (37 days):

- The maturation pile was not watered, thus water consumption was zero.
- No electricity was consumed during maturation (piles are not aerated).
- Fuel consumption during the manipulation of the maturation pile using the tractor for 3 hours (pile construction and turning) was 71.2 L of diesel (estimated as explained in the decomposition stage).

5.4.1.3 Post-treatment

Emissions of NH_3 and VOCs were not measured in this stage, which lasts 1.5 hours. With respect to resources consumption in the post-treatment activities (trommel screening):

- The tractor consumed 31.80 L of diesel while transporting the material from the maturation area to the post-processing (screening) area.
- As for the tractor's fuel consumption calculation, trommel (RosRoca model CS TR13-25) consumption was estimated from the equipment technical sheets and the number of hours the trommel was used. The theoretical consumption of the trommel was 7 L/h of

diesel and given it was used for two hours, the diesel consumption of the trommel was found to be 14 L.

5.4.1.4 Overall process evolution

The changes in the static respiration index (SRI), moisture content and organic matter content of the composting material are presented in Figure 5.12. As shown, it was not possible to sample the composting material in the confined windrow during the decomposition period. Thus, the values in Figure 5.12 correspond to the initial material (day 0), the material at the end of the decomposition phase (day 40) and the material during the maturation phase.

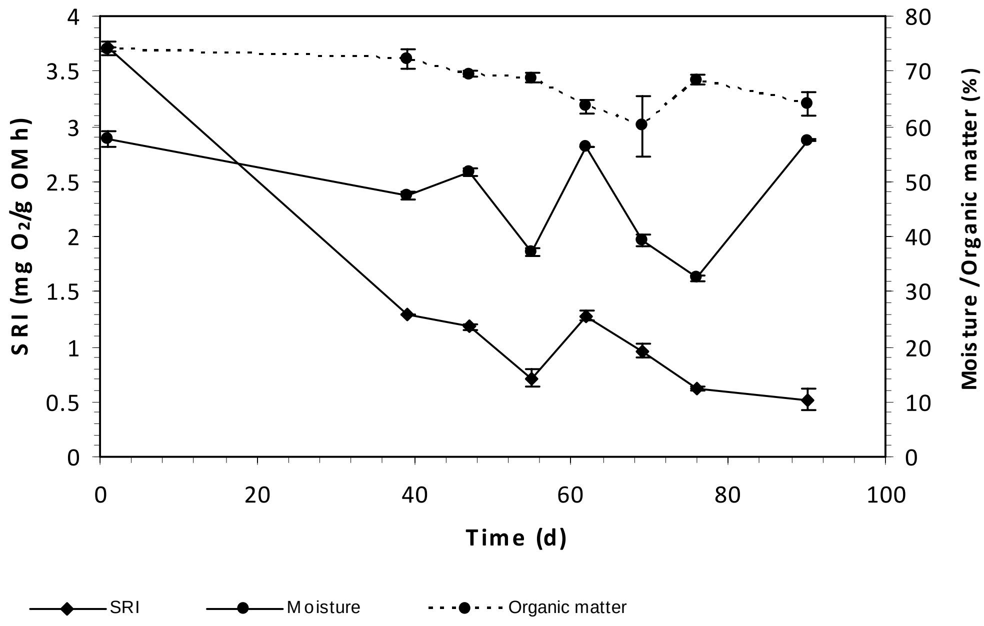

Figure 5.12. *Static respiration index (SRI, mg O_2/g OM h), moisture (%) and organic matter (%) of materials obtained during the composting process.*

In the initial mixture, SRI was 3.7 mg O_2/g MO h and in the final compost, SRI was 0.5 mg O_2/g MO h. At the end of the decomposition stage, the SRI value was 1.3 mg O_2/g MO h. According to the literature (California Composting Quality Council, 2001), materials with SRI values between 0.5 and 1.5 mg O_2 /g OM h can be considered as stable, while a SRI greater than 1.5 mg O_2/g OM h indicates the materials are unstable. Thus, given its SRI value, the final compost material in this study reached a high degree of stabilization. The SRI value obtained for the initial material in this plant represents a normal index for fresh OFMSW (Barrena et al., 2009).

In terms of moisture content and total organic matter (Figure 5.12), appropriate initial conditions were provided in the confined windrow, with 57.7% MC and 74% OM (Haug, 1993). In the beginning of the maturation stage these values were 47.5% and 72.3% of MC and OM respectively. MC fell below 40% (a low value in the recommended range for proper process evolution) during the maturation phase because it was insufficiently watered during this phase. Moreover, as shown in Figure 5.12, on the 62nd day of the process (maturation pile), both SRI and moisture content increased because it rained a few days prior to sampling. This fact had an effect on the composting material that has not been watered in the previous weeks reflected in the value of these two parameters. The OM pattern was similar to that of the SRI except on the first day of analysis; both decreased on day 55 and then increased on day 66 of process.

The changes in bulk density and porosity during the maturation stage (Figure 5.13) followed reversed patterns as was expected: bulk density decreased on at the 56th day (0.34 kg/L) of the process and increased at the end of the process (0.50 kg/L), while porosity increased on the same 56th day (62%) and finally decreased to a value of 44%. The final values can be attributed to organic matter degradation and material compaction. The porosity values obtained for this plant were within the optimal range (35-40%) described by Haug (1993).

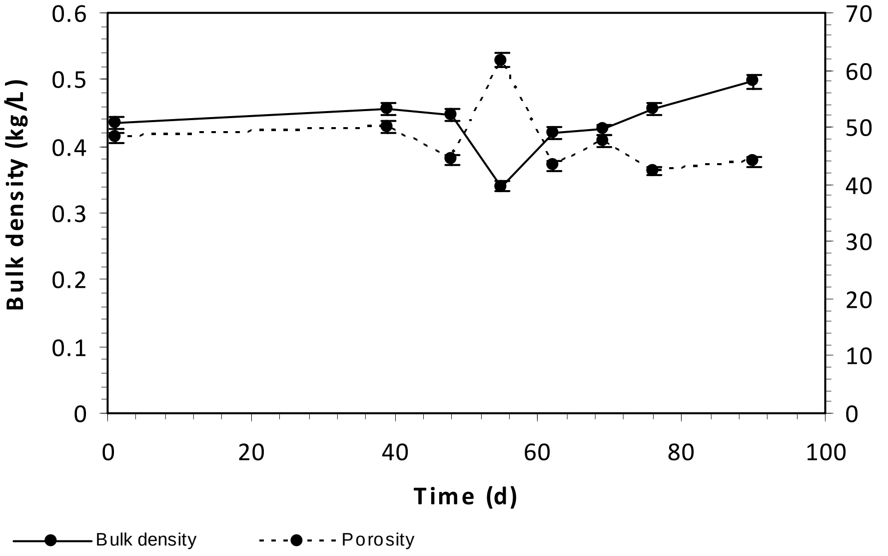

Figure 5.13. Bulk *density (kg/L) and porosity (%) of the materials obtained during the composting process.*

The input and output flows for each stage of the process discussed above are summarized in Table 5.1.

Table 5.1. *Consumption and emissions summary of the confined windrow plant during the studied period.*

Resource	**Decomposition stage**	**Maturation stage**	**Post-treatment**
Diesel (L): Tractor	47.34	71.22	31.80
Mixer	47.34	0.00	0.00
Trommel	0.00	0.00	14.00
Electricity (kWh)	1540	0.00	0.00
NH_3 emissions (kg NH_3)	25.70	21.32	0.00
Total VOC emissions (kg C-VOCs)	104.6	67.06	0.00
Water (m^3)	0.47	0.00	0.00

5.4.2 Plant performance indicators

According to the methodology described in Chapter 4, the next step after assessing the resources consumed and emissions released throughout the process is to calculate the following plant performance indicators (impacts related to the operating process): process indicators, resource consumption indicators and emission indicators relative to tons of waste treated in the facility during the study period. During the study period, 23.5 tons of OFMSW were treated in confined windrow plant.

5.4.2.1 Process indicators

Table 5.2 shows the calculated process indicators. As shown, no refuse was generated. The main reason for this is that the OFMSW contained a low level of inappropriate materials. Even though the obtained refuse was separated with the bulking agent from compost during post-treatment and recycled to the beginning of the process.

Table 5.2. *Process indicators in confined windrow plant.*

Process indicators	
t compost/t OFMSW	0.52
t bulking agent/t OFMSW	0.63
t refuse/t OFMSW	Negligible*
t refuse/t OFMSW[1]	6.31E-03[1]

* Refuse recycled into the process.

[1] OFMSW characterization carried out by ARC (2007).

The compost generation per ton of OFMSW treated during the studied period in this plant was quite a bit higher than that reported in other works performed on full-scale plants. For example, the value obtained in Blengini (2008), was 0.28 t compost/t OFMSW. This issue is mainly due to the high quality of the input materials in this facility and part of the bulking agent passing thought the trommel screen (10 mm) (no ballistic separator exists).

5.4.2.2 Resources consumption indicators

The resource consumption indicators of the plant are presented in Table 5.3

Table 5.3. *Resources consumption indicators in confined windrow plant.*

Resources consumption indicators	
kWh/t OFMSW	65.50
L diesel/t OFMSW	9.00
m^3 water/t OFMSW	0.02
Total energy kWh/t OFMSW	161.53

Some authors have reported composting plant electricity consumption values ranging from 35 to 60.83 kWh/t OFMSW treated (Razza et al., 2009; Blengini, 2008). The electricity consumption value of the plant in this study is normal compared to values reported for full-scale composting plants. The fuel consumption in the confined windrow plant was much higher than the value (2.06 L diesel/t OFMSW) reported by Blengini (2008). The major reason for this is the configuration of the confined windrow plant. The tractor must cover long distances because the decomposition and maturation areas are far apart. The water consumption of the plant in this study can be considered negligible when compared to the 89 L water/t OFMSW reported by Blengini (2008). Though the material in the confined windrow plant was initially water with tap water, water consumption was low because leachate was used to maintain the material moisture level and no water was added during the maturation phase.

5.4.2.3 Emission indicators

The emissions indicators presented in Table 5.4 and Figure 5.14 correspond to the NH_3 and VOCs emitted during the composting process itself and the NH_3, VOCs and CO_2 emissions due to energy consumption (electricity and diesel).

Table 5.4. *Emissions indicators in confined windrow plant.*

	Emission indicators (kg/t OFMSW)		
	NH_3	**VOCs**	**CO_2 (energy)**
Process	2.00	7.30	—
Electricity	1.28E-04	0.02	34.58
Diesel	6.97E-07	0.16	25.65
Total	2.00	7.48	60.23

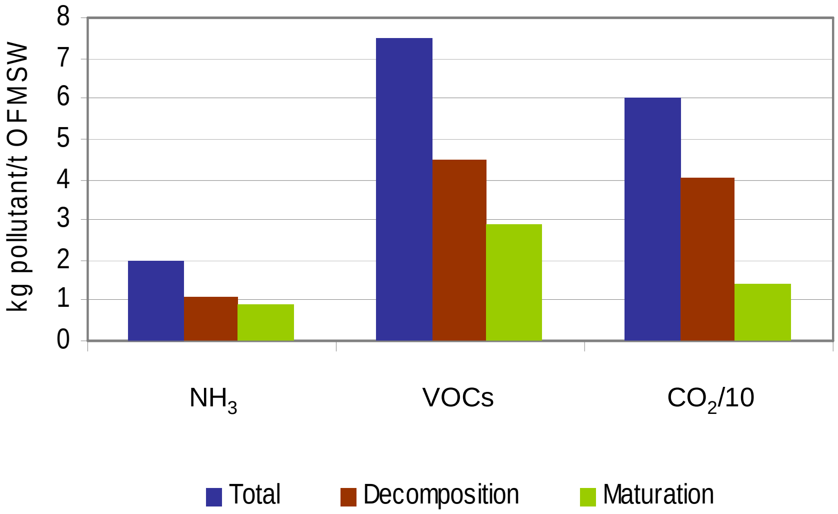

Figure 5.14. *Total emissions indicators in confined windrow process (CO_2 values are divided by 10 for improved graphical representation).*

A more detailed, graphical representation of Table 5.4 values is presented in Figure 5.14, which shows the contribution of each stage to the emissions indicators. NH_3 and VOCs are emitted principally during the decomposition phase when microbial activity is higher. Electricity consumption made an important contribution to CO_2 emissions during the decomposition stage, whereas during the maturation stage, CO_2 emissions were due to diesel consumption. The high consumption of diesel during maturation was mainly due to the configuration of the plant, since the tractor must travel a long distances between the decomposition and maturation area. As stated above, this setup is temporary as there are plans to concentrate the entire process in one area, thus reducing diesel consumption and its associated atmospheric emissions.

It is difficult to find literature that reports ammonia or VOC emissions from full-scale composting processes because to obtain reliable values requires an exhaustive emissions monitoring campaign. Care should be taken when comparing values between studies because results will vary depending on the characteristics of the waste and the process conditions. For instance, Clemens and Cuhls (2003) reported variable ammonia emissions (from 0.018 to 1.15 kg NH_3/t waste) when analyzing the composting process of OFMSW in different mechanical-biological treatment plants. Other values of ammonia emissions have been reported in pilot plant scale experiments. In their study, Beck-Friis et al. (2001) examined on how temperature affects ammonia emissions during OFMSW composting in 200 L aerated reactors. They obtained a value of 2.12 kg NH_3/t waste, which corresponds to 24-33% of the nitrogen content in the initial compost material. Under similar conditions (125 L aerated reactors), Elkind and Kirchmann (2000) reported an ammonia emission of 9.6 kg NH_3/t waste (70% of the initial nitrogen).

Table 5.5 summarizes the results of the entire inventory analysis: the efficiency of the process (t compost/t OFMSW), generation of refuse (t refuse produced/t OFMSW) and resource consumption efficiency ratios (kWh/t OFMSW, L diesel/t OFMSW and m^3 water/t OFMSW) that were discussed above.

Table 5.5. *Inventory analysis for confined windrow plant.*

Inputs	Raw materials	t OFMSW/studied period	23.50
		t bulking agent/studied period	14.70
	Resources	kWh/t OFMSW	65.50
		L diesel/t OFMSW	9.00
		m^3 water/t OFMSW	0.02
		Total energy kWh/t OFMSW	161.53
Outputs	Liquid emissions	m^3 leachate/t OFMSW	0.00[a]
	Atmospheric emissions	kg CO_2 energy/t OFMSW	60.23
		kg NH_3/t OFMSW	2.00
		kg VOCs/t OFMSW	7.48
	Product	t compost/studied period	12.2
		t compost/t OFMSW	0.52
	Refuse	t refuse/t OFMSW	Negligible*

[a] Leachate is recycled into the process.
* Refuse is recycled into the process.

5.4.3 Environmental impact analysis

LCA methodology using inventory data was used to assess the environmental impacts that results when OFMSW is treated in a confined windrow composting plant. Emissions from electricity consumption, fuel consumption and organic matter degradation (NH_3 and VOCs) during the composting process were used to calculate the different impact potentials. The impact categories analyzed are described in Chapter 3.

5.4.3.1 Functional Unit

The functional unit chosen to assess the environmental impact of the confined windrow composting plant was the treatment of 1 ton of OFMSW.

5.4.3.2 System boundaries

The LCA was performed on the composting process, excluding OFMSW transportation, compost and refuse to their final destinations as well as wastewater treatment (Figure 5.15). Fuel, electricity and water consumption as well as atmospheric emissions were studied in depth.

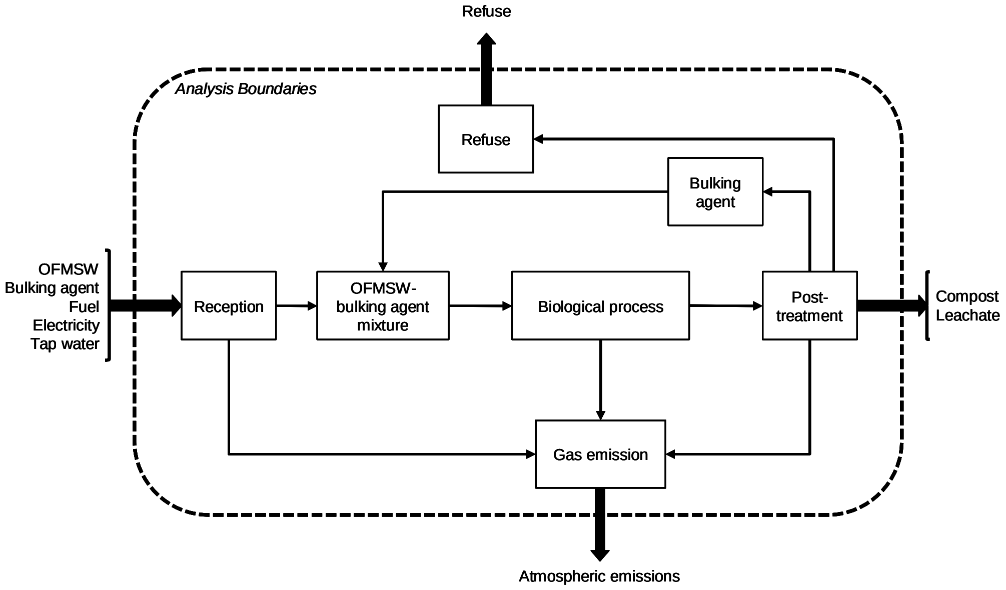

Figure 5.15. Input and output flows analyzed in the confined windrow plant.

5.4.3.3 Life cycle impact assessment

Inventory data was collected and analyzed for the functional unit in order to calculate the impact categories.

The environmental impacts are presented in Figure 5.16, and Table 5.6 summarizes the percent contributions of each emission source to the total impact potential.

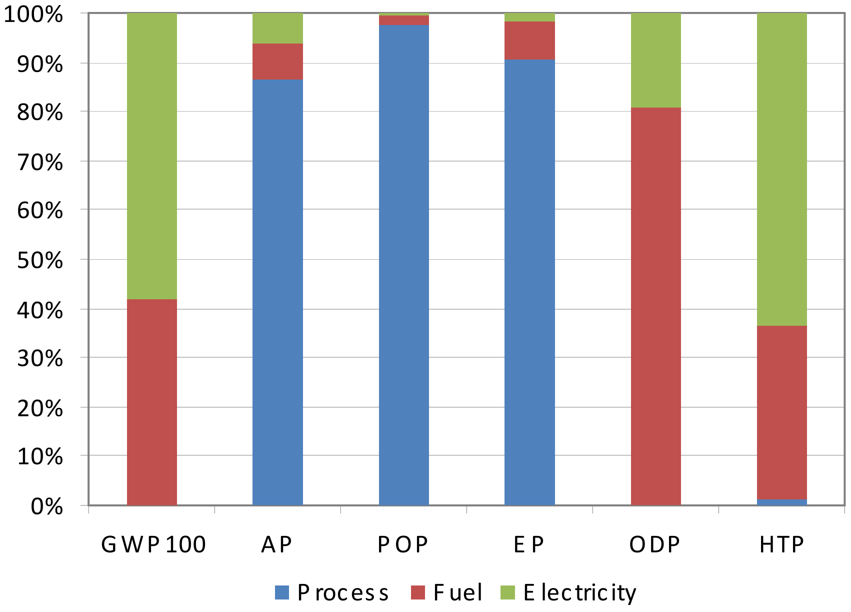

Figure 5.16. *Environmental analysis for the confined windrow plant, showing the contribution of composting, fuel and electricity to the total impact potential*

Table 5.6. Result of the impact characterization of the confined windrow plant including the contribution of composting, fuel and electricity consumption to the total impact potential (the percent contribution of each item to the total value of the plant is shown in brackets).

Impact potentials	Process	Fuel	Electricity	Total
Global Warming (kg CO_2 eq/t OFMSW)	0.00 (0%)	26.37 (41.8%)	36.78 (58.2%)	63.15
Acidification (kg SO_2 eq/t OFMSW)	3.20 (86.7%)	0.27 (7.3%)	0.22 (6%)	3.69
Photochemical Oxidation (kg C_2H_4 eq/t OFMSW)	3.04 (97.6%)	0.07 (2.2%)	0.007 (0.2%)	3.11
Eutrophication (kg PO_4^{3-}eq/t OFMSW)	0.70 (90.7%)	0.06 (7.8%)	0.01 (1.5%)	0.77
Ozone layer depletion (kg CFC-11eq/t OFMSW	0.00 (0%)	2.24E-05 (80.9%)	0.53E-06 (19.1%)	2.77E-05
Human toxicity (kg 1,4-DBeq/t OFMSW)	0.20 (1.4%)	5.10 (35%)	9.24 (63.6%)	14.54

- **Global warming potential**

The impacts of electricity, fuel and compost emissions on global warming potential are shown in Figure 5.16. The value obtained was 63.15 kg CO_2 eq/t OFMSW. The contribution of composting to GWP100 was null since CO_2 produced during the composting process was not considered as it comes from a biogenic source (IPCC, 2006). The relative contribution of electricity was 58.2% (36.78 kg CO_2 eq/t OFMSW) (Table 5.6). The principal substance responsible of GWP value for the plant studied is CO_2.

- **Acidification potential**

The major contribution to the acidification potential was produced by process emissions, particularly NH_3 emissions. Ammonia emissions generated during the composting process have been found to be important in other studies as well (Pagans et al., 2006; Hellebrand et al., 2001). Plant impact is 3.69 kg SO_2 eq/t OFMSW (Table 5.6). Ammonia emitted during the composting process represents 86.7% (3.2 kg SO_2 eq/t OFMSW) of the total AP. Thus these data highlight the importance of ammonia as a contributor to AP.

- **Photochemical oxidation potential**

The emissions from the composting process (Table 5.6) in the confined windrow plant were found to be 3.04 kg C_2H_4 eq/t OFMSW, which represents 98% of the total POP value. VOCs, which were mostly generated during the decomposition phase in the confined windrow, were the main contributors to POP.

- **Eutrophication potential**

As observed for AP, ammonia emissions during the composting process were the main contributors to the eutrophication potential (Figure 5.16). The contribution of these emissions was of 90.7% (0.7 kg PO_4^{-3} eq/t OFMSW) (Table 5.6). The contribution of energy consumption to the eutrophication potential was practically insignificant.

- **Ozone Layer Depletion**

The major contributor to the ODP was energy consumption (2.77E-05 kg CFC^{-11} eq/t OFMSW). The greatest contribution came from fuel consumption, which was responsible for 80.9% of total ODP (Table 5.6).

- **Human toxicity potential**

Fuel and energy consumption were the main contributors to the total human toxicity potential (Figure 5.16). Electricity consumption caused 63.6% while fuel consumption accounted for 35% of the total HTP. Thus, if human toxicity is the only impact category considered plants, which consume fuel, are expected to contribute less to HTP than plants, which use electricity. The contribution of the composting process itself to human toxicity is minimal for this plant.

In summary, the LCA performed revealed that the composting process in this plant was responsible for the greatest impact in three of the six impact potentials analyzed (AP, POP and EP). This fact is reflected in the percent contributions of the different emission sources to the total values of AP and EP, where ammonia accounts for more than 85% of these potentials (Table 5.6). Treatment of exhaust gases generated during the composting processes is a clear opportunity to reduce the values of AP and EP and thus diminish the impacts of the plant within these categories. The impact of energy resource consumption (fuel and electricity) on AP, POP and EP was minimal compared to the impacts of the composting process.

For the remaining of impact potentials studied, GWP100, ODP and HTP, both diesel and electricity consumption represent almost 100% of the total impact potential. Electricity is consumed mainly during decomposition by forced aeration, while diesel is consumed throughout the process by the tractor and trommel screening. It can thus be stated that electricity will have a greater contribution to total energy consumption than fuel in two (GWP100 and HTP) of the six impact potentials analyzed. Therefore, plants consuming fuel should contribute less to HTP than plants using electricity. In the case of ODP the situation is reversed, diesel is a less favorable source of energy within this impact potential category.

5.4.4 Composting plant improvement proposals

The results obtained during the present study highlight some aspects of plant operation that offer opportunities for process optimization and reduced environmental impacts. Some suggestions for plant performance improvements are listed below.

- Irrigation of the composting module should be improved to prevent leachate accumulation in some parts of the confined module, which occurred during this study (zone with excess moisture that registered lower process temperatures).
- Moisture content in the maturation windrow should be monitored at least twice a week. If necessary, the windrow should be watered.
- Material in the maturation windrow should be turned at least once a week to ensure they are adequately aerated and homogeneous.
- Reconfiguring the plant to reduce the distance traveled by machinery would result in decreased diesel consumption and its associated impacts.
- Possibilities for aeration pattern optimization at the confined module should be studied to reduce electricity consumption and its associated impacts.

Chapter 6

Environmental impact analysis at full-scale composting plant using tunnels system

Part of the results presented in this chapter has been published under the name of: Erasmo Cadena, Joan Colón, Adriana Artola, Antoni Sanchez, Xavier Font. "Environmental impact of two aerobic composting technologies using Life Cycle Assessment", International Journal of Life Cycle Assessment, 14, 401-410, 2009.

Continuing with real field data collection at full-scale plants, discussed in the previous chapter, this chapter discusses a medium-to-large OFMSW composting plant. Tunnel (in-vessel) technology is used for the decomposition stage and aerated windrows are used in the maturation stage. The working methodology described in Chapter 4 has also been used. This plant was studied during two different periods (February-March 2007 and June-September 2009), wherein some operative and structural changes were made within the plant.

The original structure and operation characteristics of the plant will be explained in section 6.2, while the results obtained during the first period of study are reported in section 6.4.1 to 6.4.4. The changes introduced and results obtained from the second period of study are summarized in section 6.4.5.

6.1 Objective

The purpose of this chapter is to analyze the environmental impacts of a full-scale composting plant, which treat OFMSW under tunnel technology, while concentrating on the impact indicators, potential impacts and process improvements from a LCA perspective.

6.2 Composting plant description

Located in the province of Girona (Catalonia, Spain), this facility treats around 6000 t OFMSW/year coming from a street bin collection system (source selection) and 4000 t sewage sludge/year. The study undertaken in the plant was focused only on the OFMSW treatment line. The common points within the two lines (OFMSW and sewage sludge) will be highlighted and considerations will be made for the calculated impacts.

The first study of this plant was undertaken from February to March 2007. The specific operations of the plant, relating to OFMSW treatment, are presented in Figure 6.1 and discussed below.

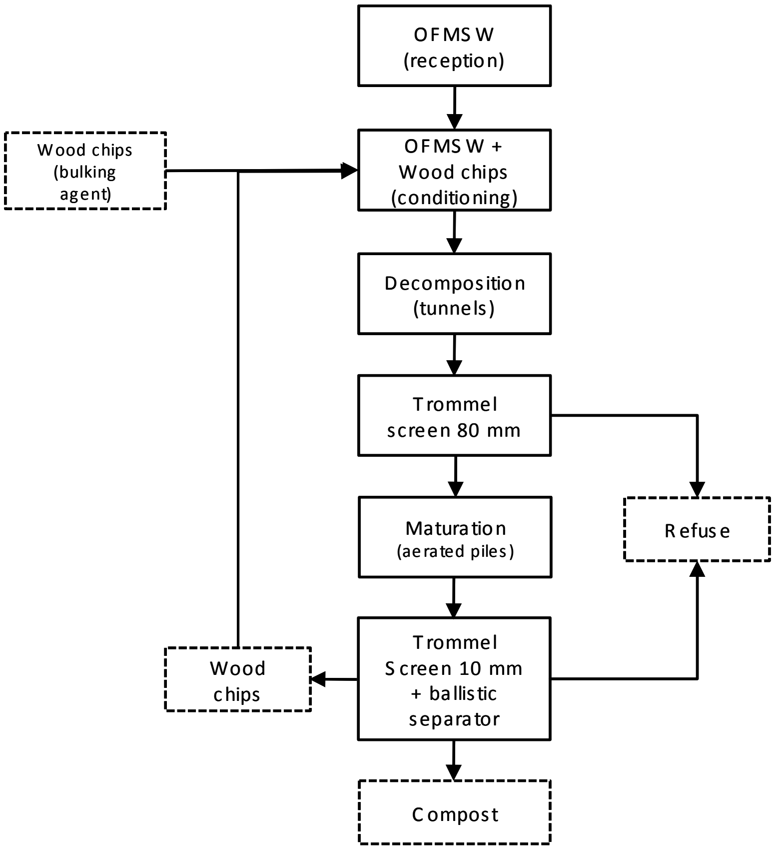

Figure 6.1 *General flowchart of the tunnel technology composting plant.*

The first step in this plant is the mixing (conditioning) of the input material (OFMSW) with wood chips, used as bulking agent, at a volume ratio of 2:1 (OFMSW:bulking agent). This activity is performed in an industrial homogenizer and is powered by a tractor. After mixing, the materials are introduced directly into the composting tunnels by the same tractor used for decomposition.

The decomposition phase is carried out in closed reactors (tunnels) under controlled conditions of aeration and watering. Tunnel dimensions are 15 x 5 x 5 (length, width, height) and each tunnel is filled to 80% capacity (around 107 t OFMSW). The estimated residence time of the mixture inside the tunnel is approximately 14 days. The process parameters, temperature, humidity and oxygen, are controlled automatically. After the decomposition in tunnels, the resulting materials are screened with a mesh trommel of 80 mm before the materials are sent to the maturation area. The layout of the plant is shown in Figure 6.2.

The gaseous emissions coming from the composting tunnels and the main hall, which includes the reception and pretreatment area (trommel screen and mixing), are treated with a wet scrubber followed by a biofiltration system. This system consists of two biofilter units, each with an external surface of 480 m^2 and a height of 1.5 m. Biofilters are watered periodically and their packing materials is made of pine bark. This equipment also treats exhaust gases coming from the tunnels where sewage sludge is composted.

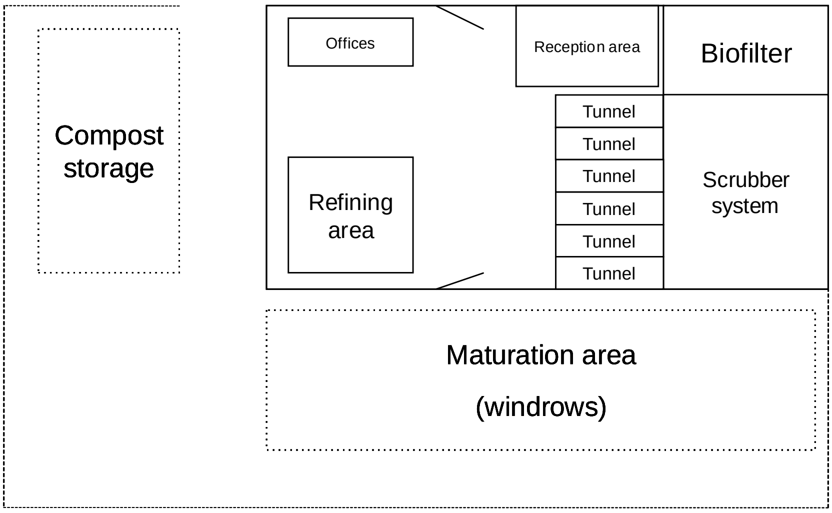

Figure 6.2. *Layout of the tunnels plant. Maturation area (dashed square) is open to the atmosphere.*

Prior to building the maturation pile (Figure 6.3), an absorbent layer mainly made from wood chips is placed over the pavement and is perforated for windrow aeration. The absorbent layer partially retains leachate, which is used to control material moisture content and also ensures a correct distribution of air thorough the windrow. Next, the composting materials are made into long piles with a trapezoidal section (2 m high, 2 m width and 30 m length approximately). These piles remain static and are watered according to the temperature and humidity present throughout the process. A ventilation system located in the base of the maturation area is used to aerate the piles. Figure 6.4 shows a diagram of the aerated windrow. According to the technical data, the aeration system has a pressure drop of 15% from the air inlet point to the air outlet section. A single fan is designed to aerate 2 piles. In the first three days of maturation there is continuous aeration. Following these three days, the pile aeration system is intermittent where there is 15 minutes of applied aeration followed by 15 minutes without aeration. The total maturation stage lasts for approximately 6-8 weeks. Gaseous emissions generated during the maturation phase are emitted to the atmosphere without any mitigation treatment.

To obtain the final compost (post-processing), the matured materials are screened to 10 mm by means of a trommel and refined using a ballistic separator. The final compost produced is used for agricultural practices and civil works.

All of the water used in the composting process of this facility comes from the Municipal Wastewater Treatment Plant (WWTP), due to its close proximity. The leachate produced by the composting process is collected and treated at the nearby WWTP, as is the wastewater produced in the exhaust gas treatment (wet scrubber) process. Further, the sewage sludge created by the WWTP is treated by the biological treatment plant.

In general, this composting plant is of medium capacity, with high investment cost and air cleaning.

Figure 6.3. *Maturation area at tunnel composting plant. Detail of the absorbent layer and perforated pavement.*

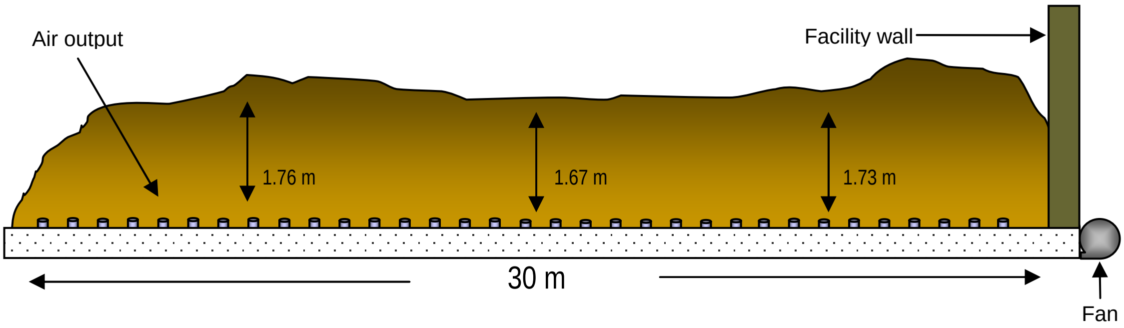

Figure 6.4. *Aeration system distribution at maturation windrows in tunnel composting plant.*

6.3 Materials and methods

Routine analytic parameters i.e. moisture, dry and organic matter, pH, biological stability (static respiration index) and refuse characterization were determined at the laboratory (following the methods described in Chapter 3) from representative samples taken weekly from the maturation windrow during the whole stage and from the final compost. Bulk density and porosity were determined *in-situ.*

The systematical methodology described in Chapter 4 was used to measure the gaseous emissions coming from different areas in this plant. As commented before, this plant is divided in two emission zones:

- Biofilter area, where emissions from the reception area, decomposition in tunnels and pre-treatment sector are treated. Gaseous emissions are extracted and addressed to the wet scrubber before going to biofiltration, and,
- maturation area, which is partially covered to protect the windrows from rainfall and protects the gaseous emissions from being directly released to the atmosphere without treatment.

As discussed in the composting plant description, this facility treats two types of waste: OFMSW and sewage sludge. To identify the impacts of OFMSW treatment, it has been assumed that the amounts of impact were proportional to the amounts of each impact that was treated. In this case OFMSW inputs account for 60% of the income in this plant. This proportion was applied to determine electricity and diesel consumption. It should be noted that water consumption was not considered, as sewage sludge treatment requires no water. Relating to gaseous emissions from biofilters, which treat gases from all the composting tunnels (OFMSW and sewage sludge tunnels) and common areas in sludge and OFMSW treatment, data provided by Pagans et al. (2006a and 2006b) was used to allocate the impact of each waste. According to their research, ammonia emissions during the decomposition stage of OFMSW are 5 folds higher than in sewage sludge and those corresponding with VOCs are 7 folds lower for OFMSW than for sewage sludge.

6.4 Results and discussions

6.4.1 Inventory analysis

During 2007, 6082 t OFMSW and 4091 t of sewage sludge were treated, 1419 t of wood chips were used as bulking agent and 567 t of compost were produced in this plant. In accordance with the assumptions explained before, electric energy (577790 kWh/y) was used during in-vessel composting of OFMSW and windrow maturation was used to provide the desired aeration. Fuel (21895.2 L/y) was necessary for the machinery used for pre and post-treatment operations as well as for material transportation to the different areas of the plant.

Water from the WWTP that was located next to the composting plant was necessary for moisture adjustment (2007.2 L/y), during the composting process, this includes biofilter watering and water used in the wet scrubber came from the WWTP. Water from the scrubber and leachate from composting reactors are re-sent to the WWTP for treatment.

Refuse materials are collected daily from the plant and transported to a landfill. These refuse materials correspond to impurities such as plastics, glass and metals in OFMSW separated from organic matter during pre-treatment operations and from final compost during material post-treatment (1520.5 t/y). Three representative samples of 5 kg each were used to determine the refuse composition of these materials after screening with the 80 mm trommel. Results obtained are graphically reported in Figure 6.5. According to this characterization, the bulking agent was the fraction with the highest percentage in the separated materials (34%). followed by plastic with 21%. A significant percentage of organic matter (17%) was also present. This value coincides with other studies performed at different biological treatment plants in Catalonia, where large quantities of biodegradable organic matter (43% for OFMSW and 28% for MSW) were found in the refuse that was sent to the landfill (Ponsá et al., 2008). Other materials of copious amounts present in refuse from this plant consisted of metals (11%) and paper (17%).

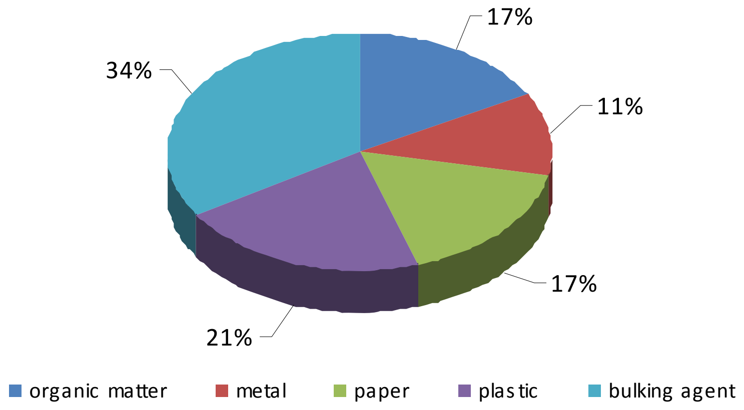

Figure 6.5. *Refuse characterization in the tunnel technology plant during the studied period (weight basis).*

6.4.1.1 Decomposition stage (biofilter emissions)

As discussed before, all the gaseous emissions coming from the decomposition stage in tunnels and from the reception and pre-processing zones are treated by means of a wet scrubber and a biofilter system.

In order to determine gaseous emissions from a biofilter surface, required by the methodology described in Chapter 4, the first step is to identify possible preferential pathways trough the biofilter packing material. This step allows the consideration of the sampling points that need to be determined for the study. In this regard, a matrix of 18 sampling points was established on the biofilter surface, thus covering the entire emission area. The values of gas velocity and contaminant concentration at these 18 sampling points were not different under the criteria established in Chapter 4 (less than 1% difference within values); therefore, the existence of preferential pathways for the gas passing through the biofilter media was not considered and VOCs, ammonia concentration, and flow were determined in one of the sampling points established.

Ammonia emissions were not detected in the biofilter system during the entire measurement period. Further, the wet scrubber and the biofilter removed ammonia entirely from gaseous emissions. Ammonia concentrations were measured before the biofilter (after the wet scrubber) obtaining a mean value of 41.3 ± 16.59 mg NH_3/L. This value represents an ammonia load to the biofilter of 123 g NH_3/m^3 biofilter d. It was not possible to measure ammonia concentration at the entrance of the wet scrubber due to the lack of accessibility in the zone. Colón et al. (2009) studied ammonia removal in a biofilter system at a full-scale OFMSW composting plant (without scrubber system) reporting NH_3 removal efficiencies up to 90%, their values ranges between 11.8 to 193.4 g NH_3/m^3 biofilter d.

Regarding total VOC emissions, Table 6.1 presents the airflow and concentrations of VOCs at the biofilter surface. Figure 6.6 represents mass flow of VOCs before and after entering the biofilter for the period studied. Every sampling day two gas samples were taken before and after the biofilter stage. As can be observed in Table 6.1 and Figure 6.6a clear difference exists between day 21 and the end of the study period. This difference was due to operational changes and tests carried out in the wet scrubber, thus the quantity of VOCs emitted from the biofilter was calculated taking into account only data collected from day 0 to 21. Nevertheless, this fact demonstrates the suitability of the methodology proposed in Chapter 4 to detect changes occurring in gaseous emission sources.

Table 6.1. *Total VOCs emissions from the biofilter surface in tunnel technology plant.*

Day of measurement	**Air velocity (m/s)**	**VOCs concentration (mg C-VOCs/m³)**	**VOCs emission rate (kg C-VOCs/d)**
0	0.08	0.07 ± 0.04	0.71 ± 0.39
3	0.09	0.04 ± 0.01	0.41 ± 0.11
7	0.07	0.05 ± 0.02	0.47 ± 0.18
14	0.06	0.06 ± 0.00	0.42 ± 0.01
21	0.08	0.44 ± 0.08	4.35 ± 0.78
28	0.09	0.41[a]	4.63[a]*
35	0.08	0.22 ± 0.06	2.21 ± 0.56*
42	0.07	0.26 ± 0.02	2.26 ± 0.21*
49	0.08	0.51 ± 0.08	5.10 ± 0.81*
56	0.06	0.39 ± 0.23	2.93 ± 1.70*
63	0.06	0.75 ± 0.41	5.56 ± 3.26*

*Scrubber system modifications period.

[a] Only one value was correct.

Biofilters treat gaseous emissions from all the composting tunnels (OFMSW and sewage sludge); however, it was not possible to determine the amount of waste simultaneously composted in the studied plant. Instead, the annual quantity of OFMSW treated during 2007 (6082 t) was used to determine weight when referring to VOC emissions. The proportion between VOC emissions in OFMSW and sewage sludge composting was applied by calculating the emissions of the studied period and extrapolating them on an annual data. According to this, VOCs mean mass flow calculated at the biofilter surface during the studied period was 0.002 kg C-VOCs/t OFMSW (with VOCs concentration ranging from 0.04 ± 0.01 to 0.07 ± 0.04 mg C-VOCs/m^3). VOC emissions before the biofilter (after wet scrubber) were 0.057 ± 0.026 mg C-VOCs/L, which represent 0.58 g C-VOCs/m^3 biofilter d (Figure 6.6). As was explained in Chapter 5, there are few data from the emissions of biofilter systems for full-scale OFMSW composting plants. Colón et al. (2009), for example, found values between 103.2 to 1749.6 g C-VOCs/m^3 d in a full-scale OFMSW composting facility that did not use a scrubber system, as can be seen, the values found were considerably different. This fact may be related to the scrubber removal efficiency.

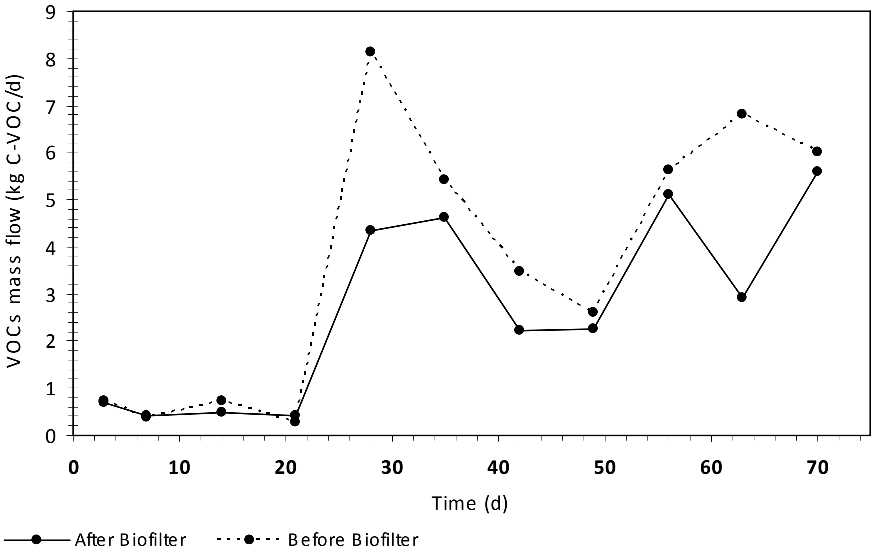

Figure 6.6. *Total VOCs mass flow before and after the biofilter obtained on each sampling day.*

6.4.1.2 Emissions from the maturation area

To analyze the emissions from the maturation area, one aerated windrow was monitored after decomposition in tunnels until the materials were cured. It is important to highlight that all the emissions generated in this area come from the OFMSW treatment; sewage sludge, on the other hand, can not be cured. According to the data supplied by the plant, 106.5 t OFMSW were composted in the tunnel and the material resulting from the decomposition stage was piled, ultimately forming the maturation windrow that was monitored. The maturation windrow was evaluated the first 3 days (since there was continuously aeration) and then once a week. This windrow had a trapezoidal shape with 1.6 x 30 x 4.8 m (height, length, contour) dimensions (Figures 6.7a and 6.8). In the tridimensional representation of the windrow (Figure 6.7a) it can be observed that the right side of this windrow is linear, being placed at the facility wall (Figure 6.8). The fans that provide aeration to the material are located at the other side of this wall. Figure 6.7b shows the 3D representation of the air velocities at the windrow surface that were gathered the first day of the maturation stage. The highest surface air velocity (0.4 m/s) was obtained at the top of the windrow, close to the area where the fan is located, which can be seen in this figure. Air velocity decreased at the surface areas located furthest from the fan, including the surface area at the top of the windrow. A total decrease of 15% in air velocity was detected; however, this value is within the range considered during the design of the aeration system of the maturation windrows (according to the data provided by plant supervisors).

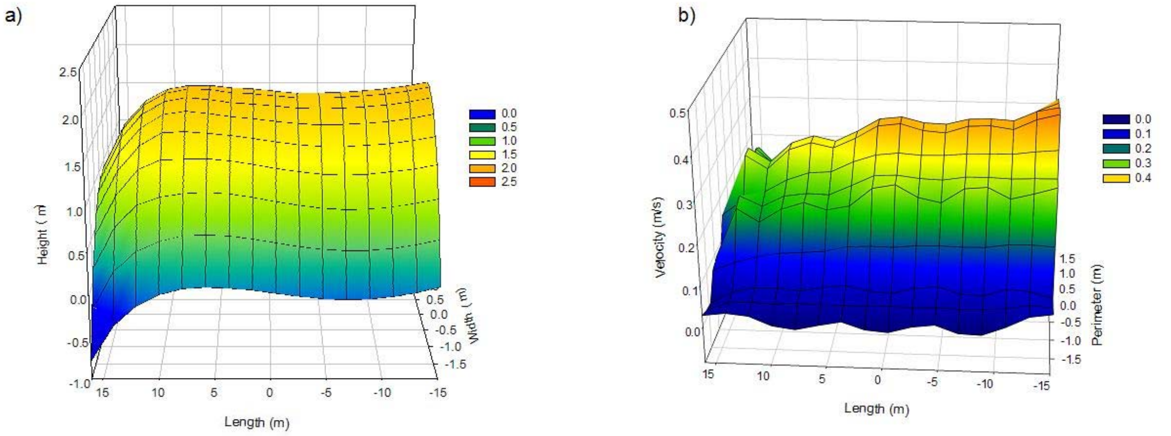

Figure 6.7. *a) tridimensional representation (m) of the studied maturation windrow, b) tridimensional representation of the output air velocity (m/s) obtained at the maturation windrow (first day).*

Figure 6.8. *Maturation windrow monitored in tunnels technology plant.*

According to the windrow dimensions, five equidistant gas emissions sampling profiles were defined. It is important to mention that during the first 3 days forced aeration was constant (24 h/d). After these 3 days intermittent periods of 15 min with aeration and 15 without aeration were used. During these periods of on and off aeration, measures of air velocity, NH_3 and VOCs remained constant throughout the study period (the differences were less than 1%). Thus, measures carried out during active aeration were considered for this study.

Figure 6.9 shows two examples of the emission surfaces used to obtain ammonia in the maturation windrow. Figures 6.9a and 6.9b correspond to the ammonia emission at days 2 and 14 of the maturation period respectively. As can be seen in these two graphs, maximum emissions were found on the top of the pile, and are congruent with observations reported in the previous chapter. The prevalence of emissions location at the top of the windrow increases as process proceeds, when global emissions significantly decrease. For the first three days of the study, ammonia concentration at the maturation windrow was similar for the different sampling points (between 3 and 6 ppmv on the sides and between 50 and 199 at the top of the windrow). Following these three days, the surface velocity of the exhaust gases affected the ammonia emissions. On day 14 the highest concentration of ammonia was found where the fan is located, near the facility wall, where the air velocities were higher (0.15 m/s) (Figure 6.9b). Figures 6.9c and 6.9d correspond to the two-dimension representation of Figures 6.9a and 6.9b respectively, from which the final value for ammonia mass flow is obtained for a sampling day.

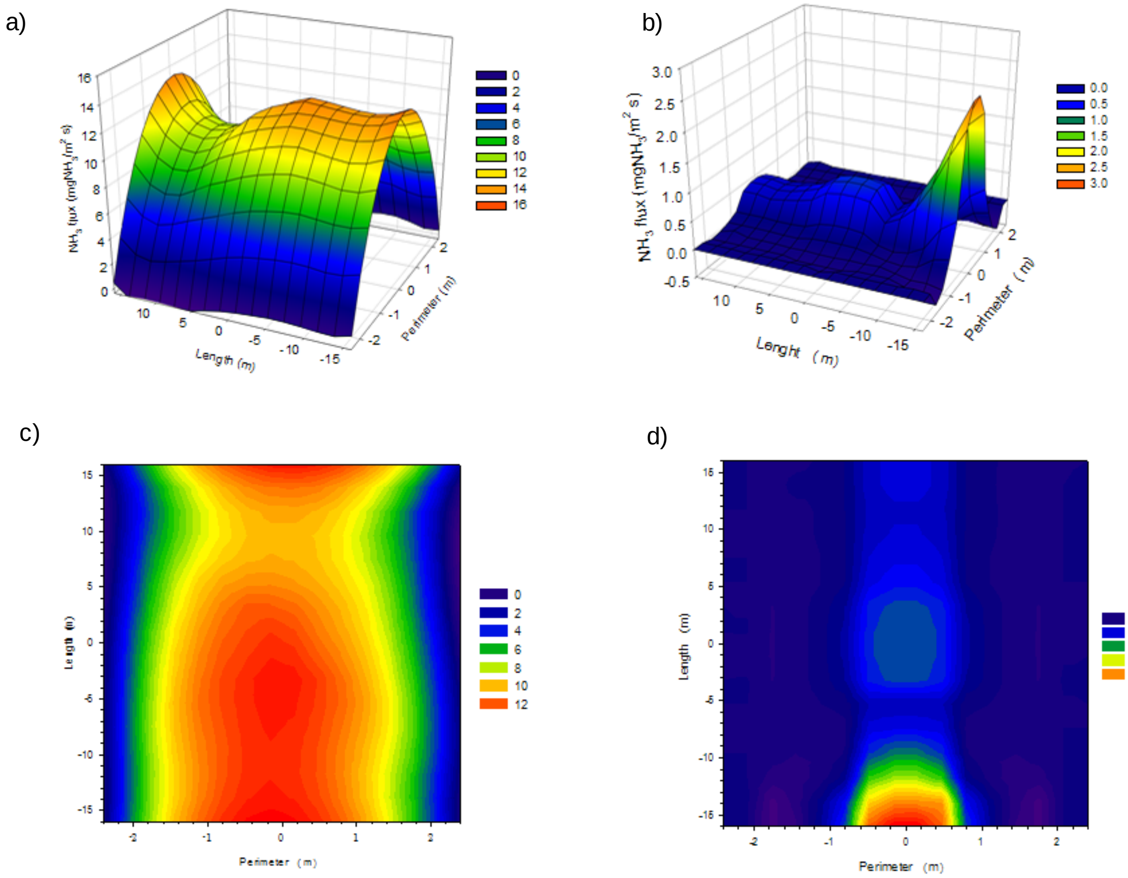

Figure 6.9. Ammonia emissions from the forced aerated maturation windrow a) Ammonia profile at the second day of maturation, b) Ammonia profile at the day 14 of maturation, c) Two dimension projection of the ammonia emission profile at the second day of maturation (mass flow mg NH_3/m^2 s), and d) Two dimension projection of the ammonia emission profile at the day 14 of maturation (mass flow mg NH_3/m^2 s).

Ammonia mass flow values determined for each day of sampling are represented versus process time, and can be seen in Figure 6.10. By integrating the curve in Figure 6.10, the total amount of ammonia emission was calculated as 416 kg. Ammonia emission concentrations were between 0 and 199 ppmv (0 to 428 kg NH_3/d). Considering the initial amount of waste treated in the monitored windrow (106.5 t of OFMSW) the ammonia emission factor for the maturation phase was calculated resulting in 3.9 kg NH_3/t OFMSW.

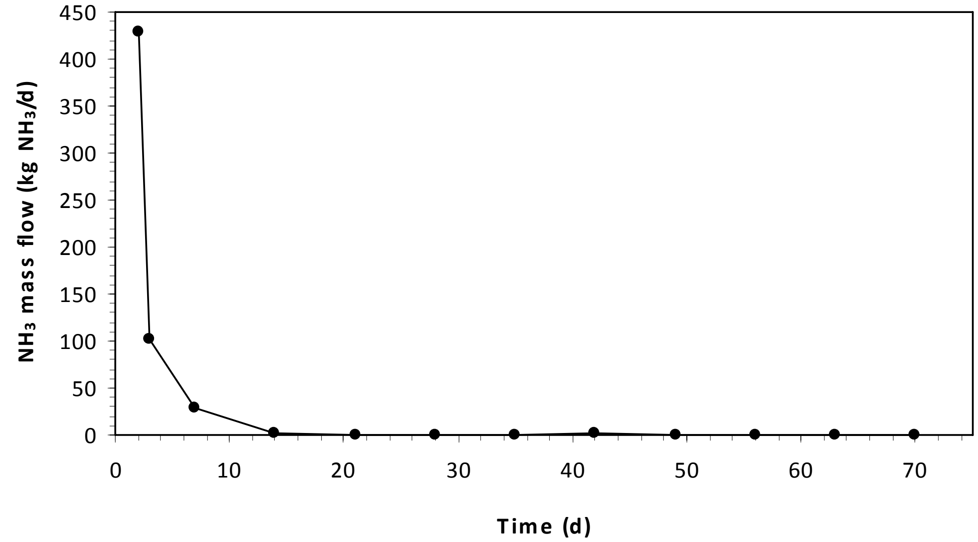

Figure 6.10. *Ammonia mass flow in the forced aerated maturation windrow obtained on each sampling day.*

Average temperatures obtained at the maturation windrow are represented in Figure 6.11. During the first days of maturation, the temperatures (in the center of the windrow) were around 70°C, then the temperatures declined drastically. After day 21 the temperature stayed within 16°C throughout the windrow. At day 42 the temperatures slightly increased (18°C) when the windrow was watered. The main causes of the rapid cooling of the materials could be the constant aeration at the beginning of this stage followed by the subsequent intermittent aeration of the windrow in addition to the ambient temperature during the study period (February to March 2007).

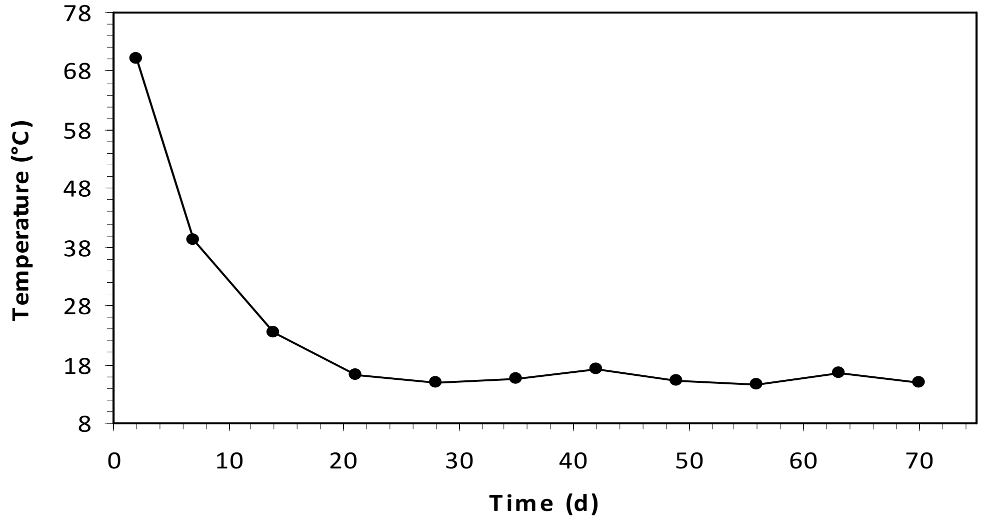

Figure 6.11. *Average temperatures in the forced aerated maturation windrow obtained on each sampling day.*

As was observed in the previous chapter, temperatures in the composting process have a close relationship with the ammonia emissions, and present the same pattern throughout the maturation phase (Figures 6.10 and 6.11) (Pagans et al., 2006a).

Total VOCs emission profiles during the maturation phase followed the same procedure explained for ammonia and is presented in Figure 6.12. The total quantity of VOCs emitted during the curing period was of 21.2 kg, where VOCs emissions were between 0.03 and 1.6 mg C-VOCs/m^2 s (0.08 to 0.77 kg C-VOCs/d). When this amount is related to the quantity of OFMSW treated, an emission factor of 0.20 kg VOCs/t OFMSW is obtained for the maturation phase.

VOC emissions were not affected by the temperature pattern presented along the maturation stage. VOC emissions declined drastically during the first three days of this stage (from 0.77 to 0.11 kg C-VOCs/d), where emissions were then erratic the rest of the maturation phase.

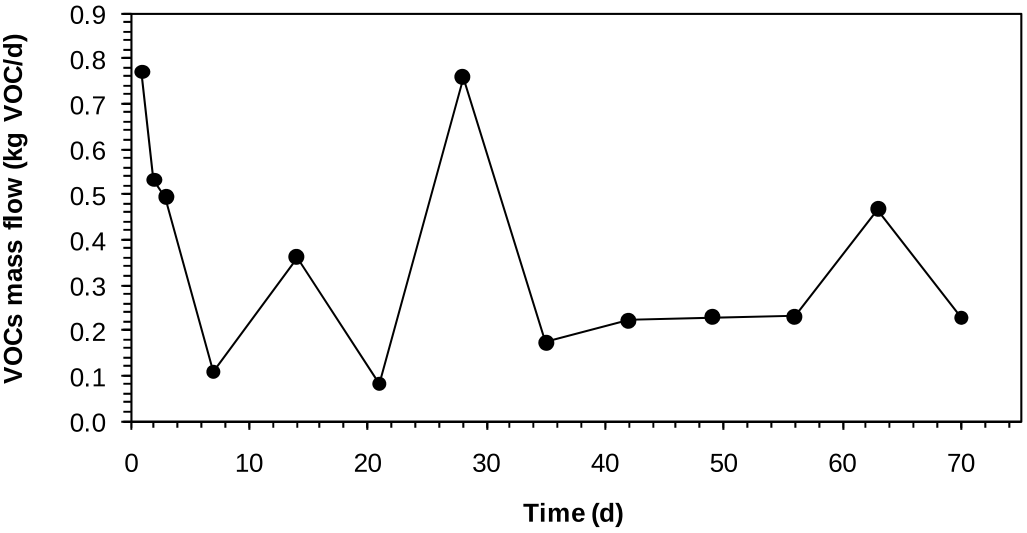

Figure 6.12. *Total VOCs mass flow in the forced aerated maturation windrow obtained on each sampling day.*

Adding values obtained during the decomposition and the maturation phases determined the overall emissions factor for the entire composting process. Total emission factors were 0.2 kg VOCs/t OFMSW and 3.9 kg NH_3/t OFMSW. As commented in Chapter 5, it is difficult to find literature on ammonia or VOC emissions from full-scale composting plants. Ammonia emissions found by other works, such as Clemens and Cuhls (2003), show an emissions variation between 0.018 to 1.15 kg NH_3/t waste.

6.4.1.3 Overall process evolution

Routine analytical methods were applied on the weekly samples taken at the maturation windrow, as was commented on in the Materials and Methods section of this chapter. Once in operation, it is impossible to enter the composting tunnel, therefore, the decomposition stage was not sampled.

Figure 6.13 presents the evolution of moisture, organic matter content, and SRI values during the maturation stage. Values obtained from days 35 to 49 were discarded from this figure because data associated with sampling errors was found. Moisture content along the maturation phase (within 48 and 57%) was in the range proposed as appropriate (40-60%) for microorganism's activity (Haug, 1993). The material lost moisture without reaching critical values for the process during the first two weeks of the curing process (until day 14). In the case of the values obtained for organic matter, moisture content remained between 60-70%. Moisture and organic matter content were within optimum ranges for the materials degradation; however, SRI values remained around 1 mg O_2/g OM h (Figure 6.13), creating a reduction in biological activity rate that was practically null. The degradation that the material suffers during the decomposition stage does not progress during maturation. Initial SRI calculated in this phase was 0.94 mg O_2/g OM h and final compost SRI corresponds to 1.03 mg O_2/g OM h. These values can be related to the temperature profile that showed the material during the maturation stage. As already mentioned, high temperatures were only registered at the beginning of this phase, after day 21 temperatures along the windrow were around 16°C, revealing that the composting process had stopped. This phenomenon of accelerated cooling, as stated, could be due to intense windrow aeration and the environmental conditions during this period. When material samples were taken, it was observed that the materials in the upper parts of the windrow were dry and it was difficult to sample the central part of the windrow where the materials were wetter. Analysis of MC and OM the materials are mixed; therefore, the difficulty of taking samples and the differences between material layers could have had several repercussions reflected in sampling error.

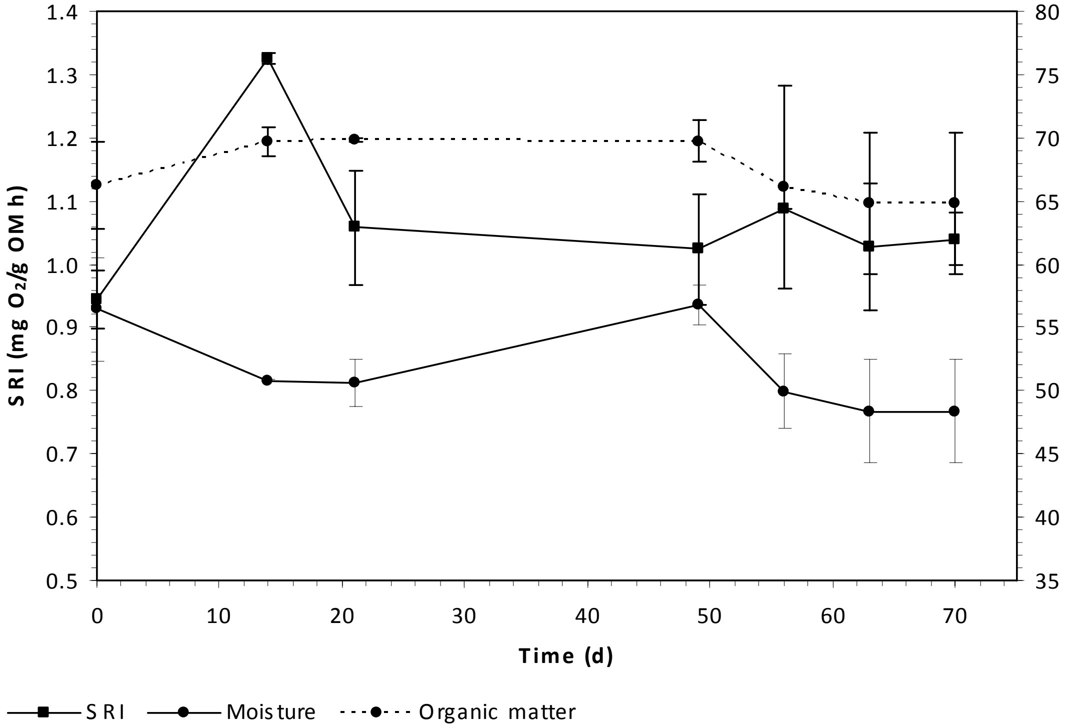

Figure 6.13. *Static respiration index (SRI) (mg O_2/g OM h), moisture (%) and organic matter content (%) evolution at the maturation windrow.*

The porosity and bulking density analysis could have been impacted by these possible sampling errors. As shown in Figure 6.14, the samples present an adequate density, 0.39 to 0.49 kg/L (Haug, 1993). However, porosity values show an inverse pattern to those obtained for the bulk density for only some of the sampling points. Further, it should be stated that when the maturation stage proceeds, and due to a lack of material homogenization, it was difficult to obtain a representative solid sample of the whole windrow.

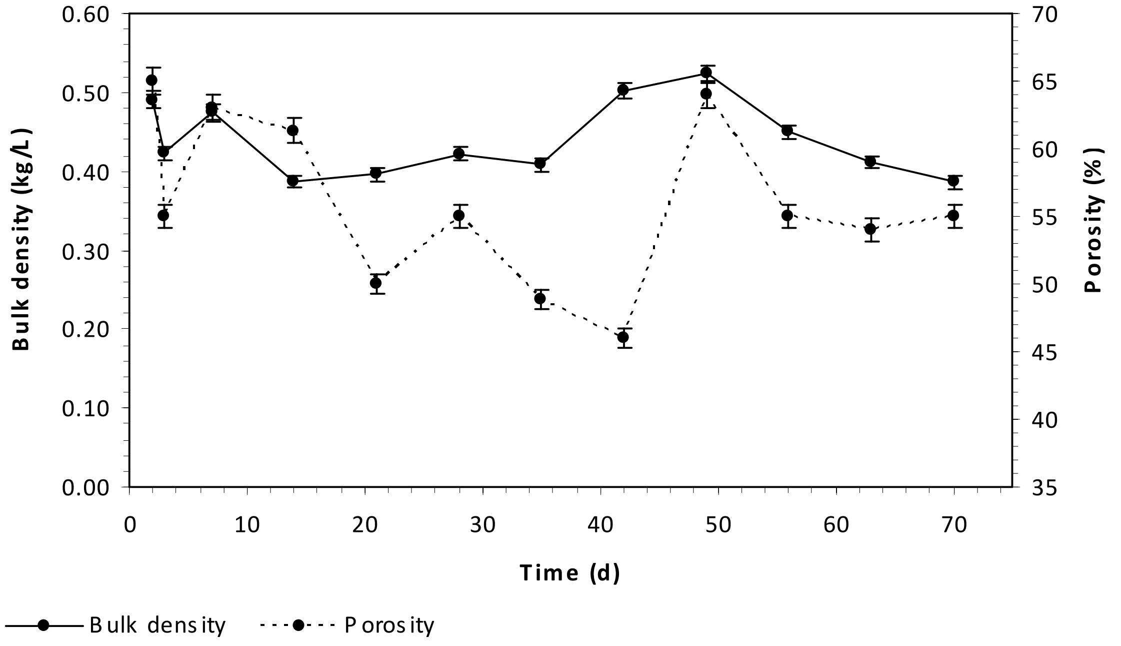

Figure 6.14. *Bulk density (kg/L) and porosity (%) evolution at the maturation windrow.*

6.4.2 Plant performance indicators

Plant performance indicators and confined windrows plant indicators (Chapter 5) were calculated, including: process indicators, resources consumption indicators and emissions indicators.

6.4.2.1 Process indicators

Table 6.2 shows the process indicators calculated in this plant.

Table 6.2. *Process indicators in tunnel technology plant.*

Process indicators	
t compost/t OFMSW	0.09
t bulking agent/t OFMSW	0.23
t refuse/t OFMSW	0.25
t refuse (theoretical)/t OFMSW[1]	0.08[1]

[1] Value from OFMSW characterization carried out by ARC (2007).

If process indicator values obtained in this plant are compared with those calculated in Chapter 5 or with other results obtained at full-scale facilities, (Blengini, 2008) it can be observed that the tons of compost produced per ton of OFMSW entering the plant tunnels were considerably lower than the values obtained in Chapter 5 and by Blengini (2008) (0.52 and 0.28 t compost/t OFMSW, respectively). This is mainly due to the low quality of the input materials, where the refuse had a high content of compostable materials that reduced plant performance, because the materials are not converted to compost. The quantity of refuse per ton of OFMSW can also be related to the low quality of input materials in this plant. The refuse calculated correspond to 25% of the total input materials, whereas with the plant described in Chapter 5 this number was practically negligible. The quality of the input materials can be explained by the collection system employed at each plant. Tunnel technology plants use street bin collection system whereas confined windrow plant uses a door-to-door system. The complexity of the implemented pre and post-treatment operations has been stated in other studies, including Alvarez et al., 2008. The efficiency of pre and post-treatment operations is another cause that could influence this issue. A ratio of 2.5 t refuse produced/t refuse initially present in OFMSW has been determined in this plant. Values of this ratio higher than 1 mean that some organic matter is separated as refuse (detected in refuse characterization, where organic matter accounted for 34% of the total refuse) with the consequent loss in the capacity of the plant for compost production. On the other hand, complexity of these treatment steps can theoretically affect the energy consumption of the entire process by having a clear influence in the global environmental performance of the plant.

6.4.2.2 Resources consumption indicators

Table 6.3 shows the consumption indicators in the studied plant.

Table 6.3. *Resources consumption indicators in tunnel technology plant.*

Resources consumption indicators	
kWh/t OFMSW	95
L diesel/t OFMSW	3.60
m^3 water/t OFMSW	0.33
Total energy kWh/t OFMSW	133.41

Energy consumption per ton of OFMSW in the plant studied was 95 kWh of electricity and 3.6 L of diesel. Contribution percentages for both electricity and diesel to total energy use were of 71.3 and 28.7% respectively. These percentages are in accordance with those calculated by Blengini (2008) who determined the electricity (60.83 kWh) and diesel (2.06 L) necessary for the composting of 1 ton of OFMSW in a forced aerated windrows composting plant. Values given by Blengini (2008) represent a contribution to the total energy consumption of 73% for electricity and 26.5% for diesel. Diggelman and Ham (2003) reported an energy consumption of 417 kWh/t food waste in a composting plant, including in-vessel decomposition followed by windrow maturation. Fricke et al. (2005) determined the electricity needs for a forced-aeration composting process resulting in 50-60 kWh/t of waste. The electricity used in the study plant is mainly used for aeration of the material, while diesel consumption is related to material handling and pre and post treatment operations. By comparing the consumption of electricity and diesel of this plant with the plant discussed in Chapter 5, it can be noticed that diesel consumption in this plant is 2.5 times lower. This is due to the fact that most of the machines in this plant use electricity to operate (reflected in electric consumption).

Regarding water consumption, 0.33 m^3 water/t OFMSW were consumed in this plant, where 0.19 m^3 water/t OFMSW were consumed in the gas treatment process and 0.14 m^3

water/t OFMSW were used in the composting process. The main reason for this high consumption of water is because of the plants location next to a WWTP. This location allows plant managers to use treated water from the WWTP in the wet scrubber to treat gaseous emissions. Therefore, the scrubber system in this plant works with an open water circuit. Evidently, water consumption is much higher in this tunnel technology plant when compared with the technology used at the confined windrow plant (Chapter 5). This is due to the reason explained above and due to the process requirements in tunnel system.

6.4.2.3 Emission indicators

Table 6.4 lists NH_3, VOCs and CO_2 emitted in the composting process and due to energy consumption (electricity and diesel).

Table 6.4. *Emission indicators in tunnel technology plant.*

Emission indicators (kg/t OFMSW)			
	NH_3	**VOCs**	**CO_2 (energy)**
Process	3.90	0.20	—
Electricity	1.86E-04	0.02	50.16
Diesel	2.79E-07	0.06	10.26
Total	3.90	0.28	60.42

As in the case described in Chapter 5, the emissions of both ammonia and VOCs come mostly from the composting process. Since the emissions from the decomposition stage were mitigated through the gas treatment systems of the plant (scrubber and biofilter), the ammonia in this case was entirety emitted in the maturation phase. Even though VOC emissions were detected in the biofilters, almost 100% of the global VOC emissions were detected at the maturation stage (without gas treatment equipment).

On the other hand, CO_2 emissions were due to energy consumption, as most of the machinery in the plant is operated by electric power. Further, emissions coming from electricity consumption were almost 4 times greater than those produced due to diesel consumption.

To summarize the three types of indicators calculated above, Table 6.5 presents all the input and output flows that are associated with the studied composting plant (materials, energy and water).

Table 6.5. *Inventory analysis for tunnel technology plant. Data is related to the treatment of 1 ton of OFMSW.*

Inputs	Raw materials	t OFMSW/year	6082
		t bulking agent/year	1419
	Resources	kWh electricity/t OFMSW	95
		L diesel/t OFMSW	3.60
		m^3 water/t OFMSW	0.33
		Total energy/t OFMSW	133.41
Outputs	Liquid emission	m^3 leachate/t OFMSW	[a]
	Atmospheric emissions	kg CO_2 energy/t OFMSW	60.42
		kg NH_3 /t OFMSW	3.90
		kg VOCs/t OFMSW	0.28
	Product	t compost/year	567
		t compost/t OFMSW	0.10
	Refuse	t refuse/t OFMSW	0.25

[a] Leachate quantity was not calculated because it is sent directly to the WWTP.

6.4.3 Environmental impact analysis

Environmental impact analysis was calculated using a LCA perspective. Inventory data was collected and analyzed in relation to the functional unit, much like the case discussed in Chapter 5.

Impact categories analyzed in this plant were those described in Chapter 3.

6.4.3.1 Functional unit

The functional unit chosen was the treatment of 1 ton of source-selected OFMSW.

6.4.3.2 System boundaries

LCA methodology was applied to the composting plant, excluding transportation of OFMSW, compost and refuse to its final destinations and wastewater treatment, much like the study developed in Chapter 5. Fuel, electricity and water consumption, and process atmospheric emissions (NH_3 and VOCs) were studied in depth. System boundaries and input and output flows considered are represented in Figure 6.15.

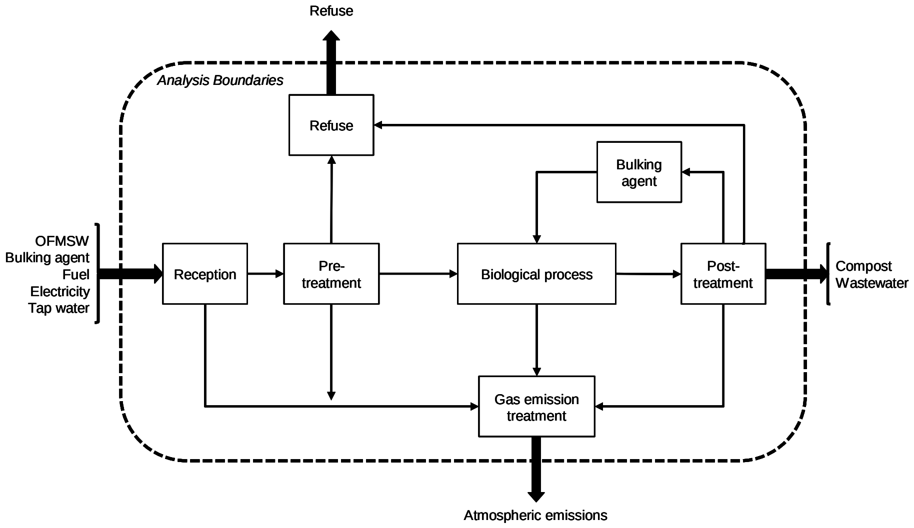

Figure 6.15. *General input and output flowchart analyzed in tunnel composting plant.*

6.4.3.3 Life cycle impact assessment

The contribution of the emissions from organic matter degradation in the composting process (ammonia and VOCs), electricity and fuel consumptions are presented in Figure 6.16.

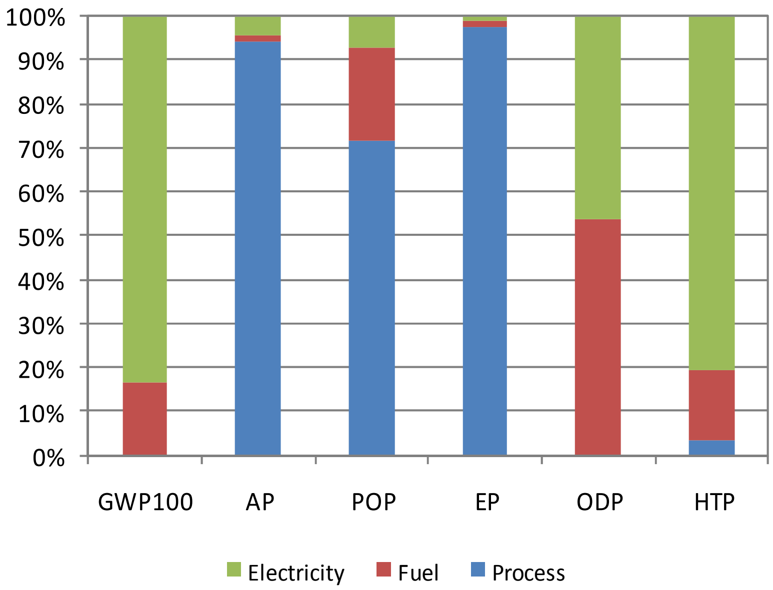

Figure 6.16. *Environmental analysis for tunnel technology plant, including composting process, fuel and electricity consumption.*

To complement the information in Figure 6.16, Table 6.6 summarizes the total value of the impact factors calculated for this plant, as well as the value of the contribution of the biological process, electricity and fuel consumption. Numerical values of the percentages represented in Figure 6.16 are also provided in the table.

Table 6.6. *Impact characterization results for tunnel technology plant, including the contribution of the composting process, fuel and electricity consumption to the total value of the potential (percentage contribution of each item to the total value of the plant in brackets).*

Impact potential	Process	Fuel	Electricity	Total
Global warming (kg CO_2 eq/t OFMSW)	0.00 (0%)	10.55 (16.5%)	53.35 (83.5%)	63.90
Acidification (kg SO_2 eq/t OFMSW)	6.70 (94%)	0.11 (1.5%)	0.32 (4.5%)	7.13
Photochemical oxidation (kg C_2H_4 eq/t OFMSW)	0.09 (69.8%)	0.03 (23.2%)	9.00E-03 (7%)	0.13
Eutrophication (kg PO_4^{3-} eq/t OFMSW)	1.47 (97.3)	2.4E-02 (1.6%)	1.7E-02 (1.1%)	1.51
Ozone layer depletion (kg CFC^{-11} eq/t OFMSW	0.00 (0%)	8.9E-06 (53.8%)	7.7E-06 (46.2%)	1.66E-05
Human toxicity (kg 1,4-DB eq/t OFMSW)	0.42 (3.3%)	2.04 (16%)	10.30 (80.7%)	12.76

- **Global warming potential**

Process contribution to GWP100 (Figure 6.16) was negligible since CO_2 produced during the composting process, as commented in Chapter 5, has not been considered as it comes from a biogenic source (IPCC, 2006). GWP100 obtained was 63.90 kg CO_2 eq/t OFMSW from which 83.5% was due to electricity consumption (Table 6.6).

- **Acidification potential**

The main contribution to acidification potential was produced by process emissions, specifically ammonia emissions, similar to the case evaluated in Chapter 5. Acidification potential in the plant was 7.13 kg SO_2 eq/t OFMSW. Ammonia process emissions represent 94% (6.70 kg SO_2 eq/t OFMSW) of the total AP. Plant emissions from the decomposition phase (tunnel) and the reception zone were treated with a wet scrubber and biofiltration process (with an ammonia removal yield near 100%) while those from the maturation phase are directly released to the atmosphere, and are responsible for the overall ammonia emissions.

- **Photochemical oxidation potential**

Regarding POP, process emissions were 0.09 kg C_2H_4 eq/t OFMSW (Table 6.6) representing the main contribution of its potential (70%). VOCs emitted during composting process in the maturation stage were the principal responsible of this potential.

- **Eutrophication potential**

Ammonia emissions during the composting process in the maturation stage were also the main contributors to the eutrophication potential representing the 97% (1.47 kg PO_4^{3-} eq/t

OFMSW) (Table 6.6). On the other hand, electricity and fuel consumptions had negligible contribution to this potential.

- **Ozone layer depletion potential**

ODP is primarily derived from energy consumption and is also discussed in Chapter 5. In this case the value of the indicator was $1.66 \cdot 10^{-5}$ kg CFC-^{11} eq/t OFMSW (Table 6.6). Fuel consumption represented the major impact with 54% of the contribution, whereas the other 46.2% was due to electricity consumption. The impact contribution coming from the composting process was null.

- **Human toxicity potential**

Energy consumption represents the major contribution to the human toxicity potential (Figure 6.16). Electricity consumption was 80.7% of the total HTP whereas only 16% was derived from fuel consumption (Table 6.6). The composting process emissions had a minimal contribution (3.3%) to this impact potential during the maturation stage.

Electricity consumption during the decomposition and maturation phases (forced aeration) was mainly responsible for two of the six studied potentials (GWP100 and HTP). Diesel consumption was the principal contributor to only one (ODP) of the six potentials although it also contributes in GWP100, POP and HTP in percentages between 16 and 20%.

AP, POP and EP were mainly caused by emissions coming from the composting process, and principally from the maturation stage. NH_3 emissions had a high impact on AP and EP, while VOC emissions were the major contributor to POP.

The implementation of an exhaust gas treatment in the composting processes offers a clear opportunity for reducing the values of the potential environmental burdens related to these impact categories, as can be deduced from the process corresponding to AP, POP and EP, Further, scrubbing and biofiltration efficiently remove ammonia present in emissions from the composting tunnels and other parts of the plant, even if the curing emissions are still significant.

6.4.4 Composting plant improvements

In order to improve the process and reduce its environmental impacts, some changes and observations that have arisen from the study of this plant will be proposed.

- Aeration pattern at the maturation windrows should be optimized. High electricity consumption is due to constant ventilation during the first three days and subsequent intermittent ventilation (15 min with aeration and 15 at rest). This constant ventilation also causes a loss of moisture in many parts of the windrow that ultimately cools the material. As a result, a lack of biological activity during this phase has been evidenced during the study.
- The maturation windrow should be watered more frequently depending on the moisture content present in the materials, in order to obtain optimal composting conditions.

- Maturation and storage areas should be closed, like the reception and tunnels. A scrubber and biofiltration system should be installed due to their high removal efficiency of gaseous emissions. Out of the other improvements proposed, this requires the highest investment.

6.4.5 Environmental analysis of the tunnels composting plant after two years, improvements and changes incorporated

The tunnel system plant was analyzed for the second time two years after the first study, from June to September 2009.

The composting process in the plant operates as specified at the beginning of this chapter (section 6.2). The facility has the same type of tunnels system and operates as discussed before.

The annual input and output materials calculations in this second study was as follows: 7435 t OFMSW/y and 5362 t sewage sludge/y were treated, 1132 t of wood chips as bulking agent were consumed and 1071 t of compost/y were produced. According to this assumption, 1591090 kWh/y during in-vessel composting of OFMSW and windrow maturation provide the desired aeration and 19777 L/y (for the machineries) that were needed. On the other hand, 4163.6 L/y were needed for moisture adjustment, biofilter watering and for the wet scrubbers. As in the first study, the water used by the scrubbers systems came from the WWTP located next to the composting plant. Further, the leachate generated is re-sent to the WWTP for treatment.

Constant complaints from neighbors adjacent to the installation regarding malodorous emissions resulted in gaseous emission mitigation measures for the facility. Maturation and storage areas were closed and a scrubber and biofilter system for each area were installed. As a result, the plant is fully closed and emissions are treated by means of three scrubbers and three biofilters. The characteristics of each treatment system are described in Table 6.7.

Table 6.7. *Biofilters characteristics and fluid used in the previous scrubber at tunnels system plant (2009).*

Characteristics	Biofilter 1	Biofilter 2	Biofilter 3
Dimensions (m) (length x width x height)	30 x 16 x 1.5	26 x 16 x 1.5	18 x 8 x 1.5
Packing material	Compost refuse + wood chip	Compost refuse + wood chip	Compost refuse + wood chip
Origin area of the treated gasses	Composting tunnels, reception and pre-treatment area	Maturation area	Post-treatment and storage area
Scrubber fluid	Tap water (closed loop) + sulfuric acid	Water from WWTP (open loop)	Water from WWTP (open loop)

Because all of the processes occur in closed areas under the new configuration of the composting plant, only the external surfaces of the 3 biofilters described above were considered as emission points in this second study.

As indicated by the methodology described in Chapter 4, the surfaces of biofilters were tested to identify possible preferential pathways. It was observed that in this case both air velocity and VOC emissions showed a difference between sampling points of over 1%, whereas NH3 emissions did not present significant differences along the biofilter's surfaces. Therefore, according to these results and the size of each biofilter, a matrix of 12 sampling points along each biofilter was considered.

To calculate the ammonia and VOC emissions coming from the OFMSW treatment at each biofilter, the following criteria were considered:

- Biofilter 1 treats the emissions from the composting tunnels (either OFMSW and sewage sludge). The criteria described in Section 6.3 was used to provide ammonia emissions and are found to be 5 folds higher in OFMSW than in sewage sludge, and VOC emissions from OFMSW are 7 folds lower than those emitted by the sewage sludge (Pagans et al. 2006a and b).
- In the case of Biofilter 2, all the emissions come from the OFMSW maturation area, and the entirety of the emissions are considered.
- Finally, for Biofilter 3 the OFMSW and sewage sludge compost storage emissions are treated and the same emission criteria established for Biofilter 1 was considered.

Figure 6.17 shows the total ammonia emissions on the surface of each biofilter. For the treating gases coming from composting tunnels of Biofilter 1, (Figure 6.17), the emissions were between 0 and 3.5 kg NH_3/d, with ammonia concentrations ranging from 0 to 6 ppmv. The emissions in Biofilter 1 fluctuated throughout the studied period, showing a low depuration in some sampling days (days 1 and 76). This could be attributed to some adjustments made in the scrubber during those days and could result in a mismatch of the system. Average ammonia emissions found in Biofilter 1 (1.1 kg NH_3/d) were lower than those performed at a OFMSW tunnels composting plant by Colón et al. (2009) (2 kg NH_3/d). For that plant, there was no scrubber was present before the biofilter.

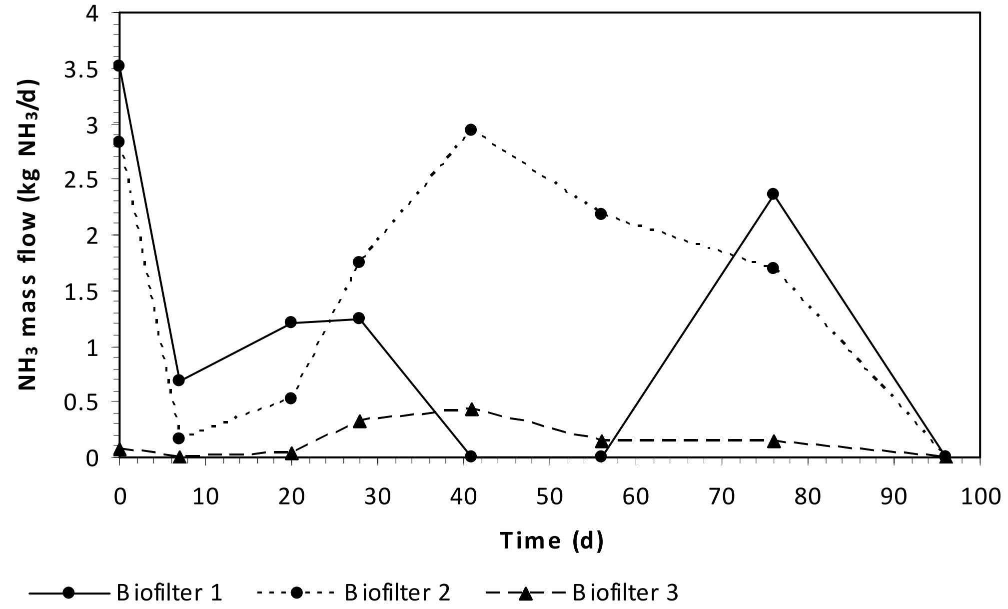

Figure 6.17. *NH_3 mass flow emitted from the three biofilters during the studied period (2009) at the composting tunnels plant. Biofilter 1 (emissions from composting tunnels), Biofilter 2 (emissions from maturation area) and Biofilter 3 (emissions from materials storage).*

With respect to the Biofilter 2 (Figure 6.17), ammonia emissions were between 0 and 2.9 kg NH_3/d, while ammonia concentrations were between 0 to 8 ppmv. As discussed in the initial part of this chapter, the maturation phase takes place in aerated windrows and as observed in the previous results, during the early days of this stage this compound has a maximum emission. As can be seen in Figure 6.17 ammonia emissions in Biofilter 2 were higher than Biofilter 1 and could be related to the different efficiencies of the biofilter and scrubber fluid with NH_3 removed.

Finally, the emissions were between 0 and 0.44 kg NH_3/d. for Biofilter 3 (Figure 6.17). As can be seen in Figure 6.16c ammonia emissions were below those found in the other two biofilters. Further, the treating the emission of Biofilter 3 coming from compost storage area have been seen in earlier results and have been reported by other authors, and NH_3 emissions from stabilized materials are negligible (Pagans et al 2006a).

VOCs emissions obtained in Biofilter 1 (Figure 6.18) were between 3.7 and 13.6 kg C-VOCs/d. In this case total VOC emissions were not affected by the scrubber adjustments. The emissions presented an erratic behavior along the studied period and have been attributed to the biological process complexities that occurred in the different composting tunnels. Comparing the average emission obtained for this compound (10.2 kg C-VOCs/d) with the average emissions found by Colón et al. (2009) (22.4 kg C-VOCs/d), it can be seen that value was two times lower in the plant studied Colón et al (2009), and can be attributed, among other things, to the lack of scrubber.

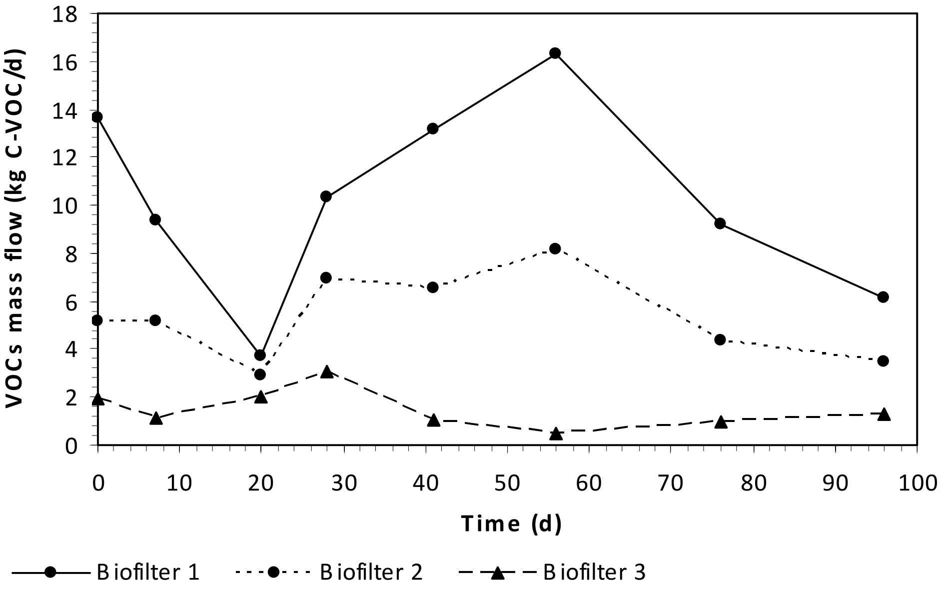

Figure 6.18. *Total VOCs mass flow emitted from the three biofilters during the studied period (2009) at the composting tunnels plant. Biofilter 1 (emissions from composting tunnels), Biofilter 2 (emissions from maturation area) and Biofilter 3 (emissions from materials storage).*

Values for Biofilter 2 were between 2.9 and 6.9 kg C-VOCs/d. Emission behavior in this biofilter was similar to the pattern that was found in the ammonia emissions.

VOC emissions in Biofilter 3 were between 0.5 and 3.1. As in the previous case, VOC emissions have followed a similar pattern and, in the case of NH_3, were lower than in the other two biofilters. The lower emissions in this biofilter are related to the origin area of the gas treated (compost storage), as was commented on previously.

In Figure 6.17 and 6.18 the total emissions for each compound and biofilter were calculated. Table 6.8 shown the total emissions for each biofilter and those that correspond to the OFMSW treatment (as mentioned above this plant treats OFMSW and sewage sludge).

Table 6.8. *Total NH_3 and VOCs emission on each biofilter during the studied period (June to September 2009) and total emissions corresponding to OFMSW treatment in tunnels plant.*

	Emission (kg)			
Biofilter	**Total NH_3**	**NH_3/t OFMSW**	**Total C-VOCs**	**C-VOCs /t OFMSW**
Biofilter 1	92.10	6E-03	1004.53	0.01
Biofilter 2	148.41	0.02	529.71	0.07
Biofilter 3	15.57	1E-03	127.36	0.05
Total	256.08	0.03	1661.16	0.13

As can be seen in Table 6.8, the highest emissions of total ammonia correspond to Biofilter 2, which were three times higher than those in Biofilter 1 and 20 times higher than those from Biofilter 3. The repercussions due to the acid scrubber on the total ammonia emissions can be seen in this case. Regarding to the NH_3 emissions per ton of OFMSW year, Biofilter 3 showed the greatest emissions output.

In the case of total VOCs and ammonia, Biofilter 2 accounted for the greatest emission, representing 7 folds greater than those calculated in Biofilter 1 and 1.4 folds greater than those emitted from Biofilter 3. VOCs emissions per ton of OFMSW year were similar on each biofilter, meaning that Biofilter 2 showed the highest emission calculated (Table 6.9).

6.4.5.1 Overall process evolution

Table 6.9 shows the moisture, organic matter, pH and biological activity (dynamic respiration index, DRI) from four sample areas taken at the plant: reception (OFMSW initial), tunnels (second day), maturation windrow (third week of maturing) and final compost.

Table 6.9. *Main characteristics of solid samples taken at four points in the tunnels system plant during 2009.*

	Moisture (%)	Organic matter (%)	DRI (mg O_2/kg OM h)
OFMSW initial	59.65 ± 3.7	70.55 ± 1.5	6.07 ± 0.3
Tunnel sample	52.76 ± 1.6	69.08 ± 6.9	2.58 ± 0.1
Maturation windrow	43.31 ± 0.4	62.56 ± 6.4	2.22 ± 0.1
Compost	48.46 ± 0.2	56.08 ± 0.1	2.08 ± 0.3

DRI= dynamic respirometric index

The values of both moisture and organic matter (between 40 and 60% and between 50 and 70% respectively) throughout the stages of the plant were within the optimal parameters for the composting process development (Haug, 1993). Organic matter is reduced in accordance with the materials degradation.

Regarding to the DRI calculated in these samples, it can be seen that the DRI dropped drastically within the two first samples (initial OFMSW to tunnels sample) but later in the maturation phase and in the final compost this index was virtually unchanged. This also happened in the previous study, where the SRI remained almost unchanged during the maturation stage, but the value obtained in the final compost (2.08 mg O_2/kg OM h) was above the value considered stable (1.1 mg O_2/kg OM h) (Adani et al., 2004).

6.4.5.2 Plant performance

All the inputs and outputs obtained during this second study (June to September 2009) and those calculated in the fist analysis of this plant (February to March 2007) are summarized in Table 6.10.

Table 6.10. *Inventory analysis of the two studies for tunnel technology plant. Data is related to the treatment of 1 ton of OFMSW.*

			2007	**2009**
Inputs	Raw materials	t OFMSW/year	6082	7435
		t bulking agent/year	1419	1132
	Resources	kWh electricity/t OFMSW	95	214
		L diesel/t OFMSW	3.60	2.66
		m^3 water/t OFMSW	0.33	0.56
		Total energy/t OFMSW	133.41	242.4
Outputs	Liquid emission	m^3 leachate/t OFMSW	[a]	[a]
	Atmospheric emissions	kg $CO_{2\,\text{energy}}$/t OFMSW	60.42	121.02
		kg NH_3 /t OFMSW	3.90	3.04E-02
		kg VOCs/t OFMSW	0.28	0.23
	Product	t compost/year	567	1071
		t compost/t OFMSW	0.09	0.14
	Refuse	t refuse/t OFMSW	0.25	0.13

[a] Leachate quantity was not calculated because it is sent directly to the WWTP.

Values in Table 6.10 allow the comparison between inputs and outputs of the two studied plant configurations.

Firstly, it should be noted that the amount of OFMSW treated in this second study is higher than the amount treated in 2007 (7435 t and 6082 t OFMSW, respectively). The quantity of final product (compost) obtained per t of OFMSW was slight higher in this second study than in the first study (0.14 and 0.09 t compost/t OFMSW, respectively). Finally, with respect to the refuse calculated in the second study (0.13 t refuse/t OFMSW), the amount was almost two times lower than the refuse calculated in the first analysis (0.25 t refuse/t OFMSW), showing slight improvement in the OFMSW pre-treatment system.

Regarding resources consumption, the amount of electrical energy consumed was higher in this second study than in the first (214 in front of 95 kWh/t OFMSW). This was mainly due to the installation of two scrubber systems and two biofilters, which require air collection and transportation, combined these things lead to high electricity consumption as a consequence. The actual electricity requirement for the plant supposes an increase of 125% with respect to the initial consumption. On the other hand, fuel consumption was lower in this second study (2.66 L/t OFMSW) (Table 6.10) than in the first, where diesel consumption was 3.60 L/t OFMSW (26% lower). Finally, the water consumption in the second study was over 1.6 times higher (0.56 m^3 water/t OFMSW) compared with the first, mainly due to the amount of water required for irrigation of the biofilters and scrubber operations.

In the case of atmospheric emissions, Table 6.11 details the emissions of NH_3, VOCs and CO_2 while considering each aspect (process, electricity and fuel consumption). Ammonia and VOC emissions came mostly from the composting process, while CO_2 emissions were due to energy consumption (namely electricity). In this second study, CO_2 emissions were double those emitted in the first study due to the increment in energy consumption commented above.

NH_3 and VOC emissions in this second study were more than 128 folds and 1.2 folds lower respectively than those calculated in the first study. This shows how the implementation of gaseous emission treatment systems has a positive repercussion on the mitigations of environmental impact for this type of biological treatment system. The influence of the treatment equipment in NH_3 emissions is higher than the influence in VOCs emissions. Biofiltering materials emit VOCs as a consequence of the biological activity, and could be a reason for this influence (Pagans et al., 2006a). Identification of the different VOCs present before and after the biofilter will be careful to confirm that fact.

Table 6.11. *Emission indicators in tunnel technology plant during 2009 period.*

Emission indicators (kg/t OFMSW)			
	NH_3	**VOCs**	**CO_2 (energy)**
Process	0.03	0.13	—
Electricity	4.19E-04	0.05	112.94
Diesel	2.20E-07	0.05	8.08
Total	3.04E-02	0.23	121.02

6.4.5.3 Environmental impact analysis

The environmental impact analysis was undertaken using the LCA perspective and based on the inventory data discussed above, as well as for the previous study developed for this plant.

- Functional unit and system boundaries

The functional unit chosen was the treatment of 1 ton of OFMSW. The system boundaries in the second study were limited to the composting plant (Figure 6.15), excluding transportation of OFMSW, compost and refuse to its final destinations and wastewater treatment.

- Life cycle impact assessment

The contribution of the emissions from composting process (ammonia and VOCs), electricity and fuel consumptions was used to calculate the different impact potentials (Figure 6.19).

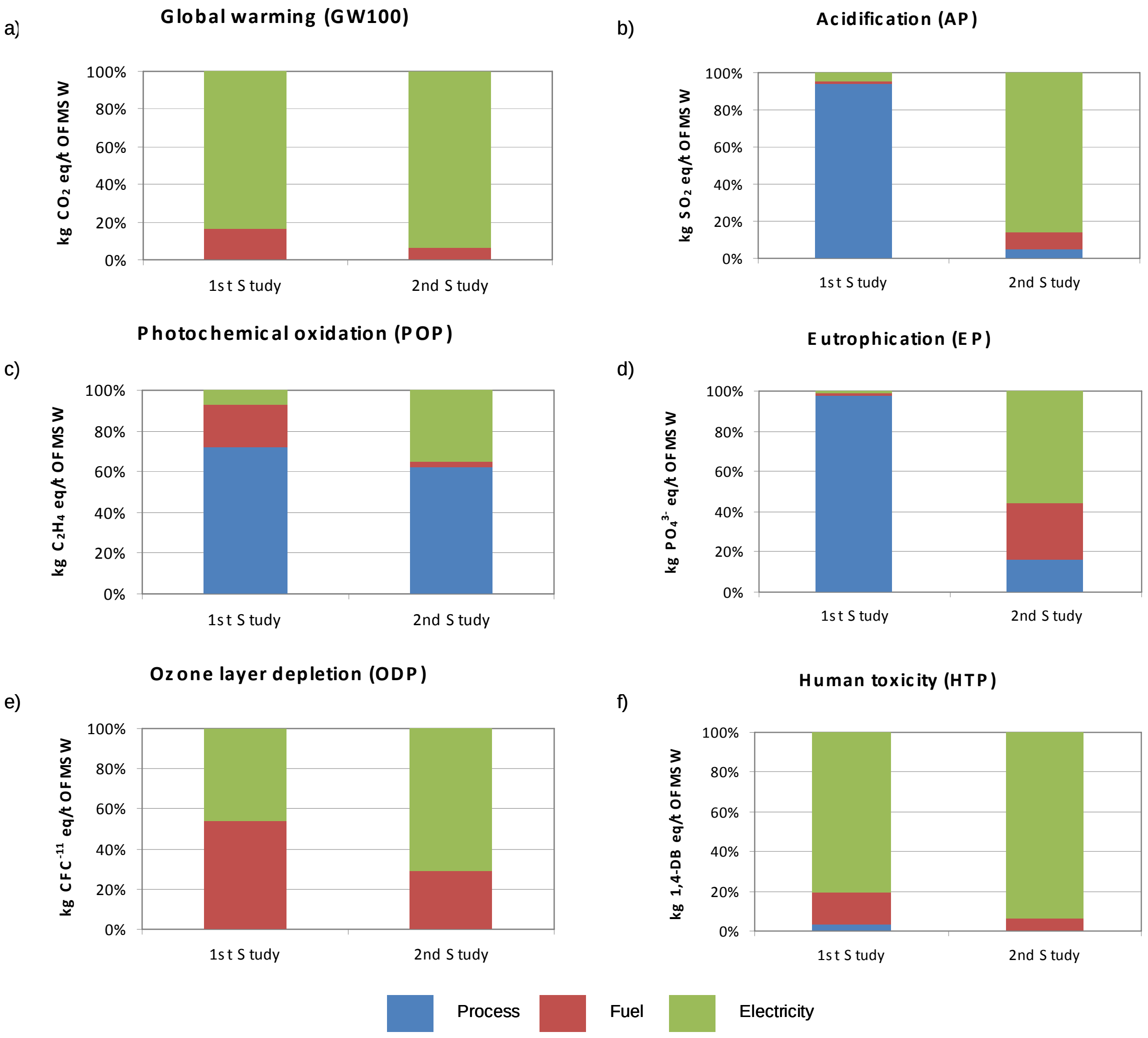

Figure 6.19. *Environmental analysis obtained in the two studies on the tunnel technology plant, including composting process, fuel and electricity consumption. a) global warming potential, b) acidification potential, c) photochemical potential, d) eutrophication, e) ozone layer depletion, and f) human toxicity.*

Table 6.12 summarizes the total value of the impact factors calculated for this plant during the studied period on 2009.

- **Global warming potential**

Global warming potential obtained in this second study was 128.43 kg CO_2 eq/t OFMSW (two times higher than the first study), where 93.4% was due to electricity consumption (Table 6.12) and the rest was due to fuel consumption. Electricity consumption represent the greatest impact on this potential, as it did in the first study, however, in this case was more than two times higher.

- **Acidification potential**

Unlike the previous study, acidification potential was mainly represented by the consumption of electricity (85.7%) (Table 6.12). Composting represented 4.9% (4.8E-02 kg SO_2 eq/t OFMSW) of the total AP. The implementation of gaseous emissions treatment equipment achieved NH_3 emissions reduction that were clearly reflected in its impact potential.

Table 6.12. *Impact characterization results for tunnel technology plant including the contribution of composting process, fuel and electricity consumption to the total value of the potential (percentage contribution of each item to the total value of the plant in brackets).*

Impact potential	Process	Fuel	Electricity	Total
Global warming (kg CO_2 eq/t OFMSW)	0.00 (0%)	8.43 (6.6%)	120 (93.8%)	128.43
Acidification (kg SO_2 eq/t OFMSW)	4.8E-02 (4.9%)	8.7E-02 (8.9%)	0.84 (85.7%)	0.98
Photochemical oxidation (kg C_2H_4 eq/t OFMSW)	5.4E-02 (62.1%)	1.9E-03 (2.2%)	3.1E-02 (35.6%)	8.7E-02
Eutrophication (kg PO_4^{3-} eq/t OFMSW)	1.1E-02 (16.4%)	1.9E-02 (28.4%)	3.7E-02 (55.2%)	6.7E-02
Ozone layer depletion (kg CFC^{-11} eq/t OFMSW	0.00 (0%)	7.1E-06 (29.6%)	1.7E-05 (70.8%)	2.4E-05
Human toxicity (kg 1,4-DB eq/t OFMSW)	3E-03 (0%)	1.6 (6.5%)	23.2 (93.5%)	24.80

- **Photochemical oxidation potential**

Photochemical oxidation potential was 8.7E-02 kg C_2H_4 eq/t OFMSW (Table 6.12) from which 62.1% was due to composting process (Table 6.12). Electric consumption was 35.6% of this potential and the rest was represented by fuel consumption. Total POP in this second study was 1.5 times lower than the first study and the contribution of the composting process to the value of this potential has decreased from a 70% in the first study to a 62% in the second. These changes are mainly due to the gaseous emissions treatment systems that were implemented during 2009 (Figure 6.19) with a focus of VOCs reduction.

- **Eutrophication potential**

Eutrophication potential and AP was mainly represented by energy consumption (electricity and fuel) while the composting process percentages were around 16.4.3% (Table 6.12). The potential impact in this second analysis was 16.8 times lower than the first study. Similar to AP, the reduction in NH_3 emissions was responsible for this difference (Figure 6.19).

- **Ozone layer depletion potential**

ODP was 2.4E-05 kg CFC^{-11} eq/t OFMSW (Table 6.12). Electric consumption represented the major impact with 70.8% of the contribution whereas the other 29.2% was represented by fuel consumption. ODP was 1.4 times higher than the first study due to the incremental electric consumption.

- **Human toxicity potential**

Finally, human toxicity potential was almost entirely represented by the electric consumption (93.5%), the rest was represented by fuel used. The process impact was negligible in this case (Figure 6.19). In this second study HTP was 1.9 times higher than the first analysis.

Out of five of the six studied potentials, electricity consumption in this second analysis was the major contributor (GWP100, AP, EP, ODP and HTP). Electric consumption, as explained before, has risen to its current levels due to the gas treatment mitigation systems installed. Diesel consumption, on the other hand, has decreased slightly when compared to the study developed in 2007.

As has been noted, the implementation of gas treatment systems have had a great repercussions when it came to reducing the contribution of the composting process to the value of the six potential impacts studied. However, due to the incremental electric consumption in this second study GWP100, ODP and HTP were higher than in the first study. Consequently, it should be kept in mind that the reduction of atmospheric emissions from the composting process itself implies the incremental emissions derived from electric energy consumption. This fact should be considered by environmental authorities and plant responsible in the decision making process when balancing the benefits of release reductions of process emissions and the inconvenience of electricity consumption increments. However, as discussed, improvements in the plant were made due to resident complaints. It is expected that the redesigning of the plant will contribute to a reduction in complaints, ultimately justifying in this way the increase in energy consumption.

On the other hand, it is important to highlight that there are different "local social indicators" that are used to assess the social sustainability of waste management systems. These indicators have not been considered in this dissertation because they are not included in the ISO 14040:2006 (ISO, 2006), however the recent years some authors (Meneses et al., 2005; Ebreo et al., 1999; Berger, 1997; Taylor and Todd, 1995) have worked on the indicators with the greatest success with instillation on the social level. These indicators identify three main social aspects: i) social acceptability, ii) social equity, an equitable distribution of benefits and harms of the treatment system, and iii) social function, the social benefits of the treatment system.

The impact of the odor emissions from the treatment plant facility also lies within the social indicators. This indicator describes the odor nuisance potential of a biological treatment plant to a population. The methodology used is based on the estimation of the odor emission from the plant when compared to the estimate of the potential scale of the odor emission in the population. Another indicator is the visual impact of the installation; this indicator assesses the visual impact of each treatment plant.

Evaluation of these indicators to the study developed in this chapter, i.e. impact of malodor emission potential between the neighbors, is a relevant fact that it is worth researching in order to see if the implementation of gas mitigation systems have repercussions on reducing complaints from neighbors and improving plant image.

Chapter 7

Environmental impact analysis of a full-scale anaerobic digestion and composting plant

As discussed in Chapter 1 of this dissertation, anaerobic digestion technologies have received a great deal attention in recent years. Using OFMSW, a renewable feedstock as a clean fuel source can displace fossil fuel use and lessen environmental impacts, but the environmental impact potentials of doing this have not been deeply studies for full-scale treatment plants. This chapter presents the study of the environmental burdens of a full-scale anaerobic digestion plant. The anaerobic digestion process is followed by a composting process to accomplish organic matter stabilization and sanitation.

7.1 Objective

The principal goal of this chapter is to evaluate the environmental impacts of OFMSW treatment in a full-scale plant, which uses anaerobic digestion followed by composting. This is done from a life cycle perspective with a focus on impact indicators, potential impacts and process efficiency.

7.2 Anaerobic digestion plant description

The plant studied is located in Vallès Occidental (Catalonia, Spain). Per year, the anaerobic digestion and composting plant treats roughly 25000 t of OFMSW, which comes from a street bin collection system. Annual biogas production is estimated to be 3500000 Nm^3 given that 140 Nm^3 is produced per ton of OFMSW. This quantity of biogas is sufficient to meet the power demand of the plant, and to generate a surplus, which is sold to an electric company. The plant was studied from March to May 2008.

The general process of this plant is shown in Figure 7.1.

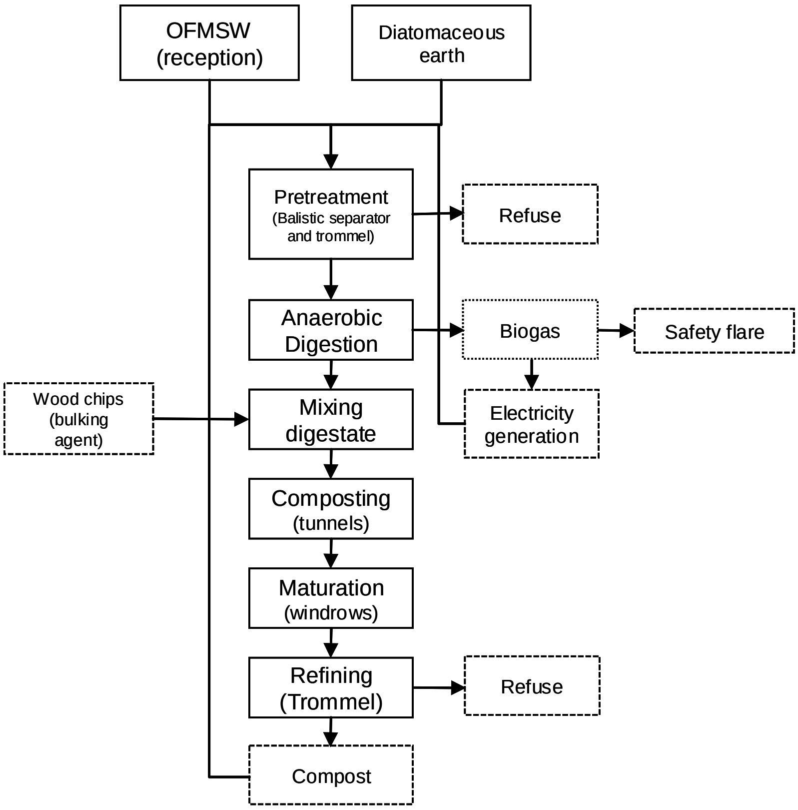

Figure 7.1. *Flowchart of anaerobic digestion and composting in the plant.*

Before the pre-treatment operations, the input materials are mixed with residual diatomaceous earth and compost (which is produced in the plant) at a volumetric ratio of 10:1:1. Diatomaceous earth is a natural material used for industrial-scale filtration processes. In this case the residual diatomaceous earth has a high vegetable fat content since it is waste generated in the biodiesel industry. Pre-treatment stage includes selection of the OFMSW, in order to treat it specifically and to facilitate the separation of reusable by-products followed by the elimination of refuse. The objective of this stage is to obtain an organic fraction that is adequate for submission to the anaerobic digestion process, which requires waste with less than 8% inappropriate components. The OFMSW is sent to shredder and to a ballistic separator, which separates it into three streams: i) light-flat fraction, the majority of which contains items such as paper, cardboard, and plastic bags, ii) rolling elements, which includes heavy items and roundish items such as plastics and iii) fine particles of organic matter and metals. The first two fractions are collected and temporarily stored in a container until they are picked up from the plant for appropriate treatment elsewhere. The fraction composed mainly of organic matter is placed on a conveyor belt mounted with a magnetic separator. Once the metals are removed, the OFMSW is moved to the anaerobic digestion module. The exhaust air of the reception and pre-treatment areas is directed to a scrubber that uses water as a cleaning fluid followed by a biofilter (Biofilter 2) (Figure 7.2).

The treatment capacity of the anaerobic digestion module (a stainless-steel vertical cylindrical digester with 1700 m^3 of capacity) (Figure 7.3) is estimated to be 25000 t OFMSW/year. The expected production of digested matter is 21260 t/year with an average biogas production of 3500000 Nm^3/y.

Figure 7.2. *Detail of the Biofilter 2 (first day of analysis) at the anaerobic digestion and composting plant.*

Figure 7.3. *Anaerobic digester installed in the plant.*

This anaerobic digestion process is based on the Dranco (DRy ANaerobic COmposting) anaerobic digestion technology from Organic Waste System (OWS), a Belgium-based firm (Juniper Consultancy Services, 2005). As its name suggests, this is a dry process, which operates at thermophylic temperature (49°C). The digester is thermally insulated.

Before the organic matter is transferred to the digester, it goes through a series of steps to facilitate the digestion process. First, a shredder reduces the size of the waste to 40 mm particles. Next, the waste is sent to a hopper where it is mixed with a portion of the digested matter, which is taken from the digester outlet. During mixing, a small amount of low-pressure steam is injected into this mixing chamber to increase the temperature of the matter. The steam is supplied by fire-tube boiler, which has maximum production rate of 500 kg/h of low-pressure saturated steam (1.05 bar, 105°C). Once the fresh OFMSW is mixed with the digested matter, the mixture is pumped into the digester by means of a piston pump. The retention time in the digester is 20 days. The digested waste is removed through the bottom of the digester by means of a spiral conveyor and it is sent to the composting line. A small portion is re-circulated into the mixing chamber. The biogas generated is stored in a 270 m^3-volume biogas-store gasometer.

Energy is recovered from the biogas as electricity using two biogas-fired combustion Guascor motor generation sets (630 kWh each). The exhaust gas from the engines feeds the boiler mentioned before. The motors are also able to burn diesel, which ensures they operate continuously if biogas production is insufficient.

As mentioned, some of the electricity produced by the engines is used to power the treatment plant and some is sold to an electricity supply company, which supplies energy to the public power grid.

A high-temperature safety flare is installed to ensure complete combustion of the biogas to prevent its escaping to the atmosphere in the event of engine failure or maintenance shutdowns.

The treatment facility operates Monday through Friday. During the weekend, the plant closes and no operations take place (including the recycling of the digestate, motors, etc.). The biogas produced during the weekend is stored in the gasometer. The two combustion motor generators do not switch on automatically and thus, if the gasometer capacity is exceeded, the biogas is burned by the safety flare.

Once digestion is complete, the digested material is recovered from the anaerobic digestion reactor and is composted. The plant uses a mixed composting system comprised of two stages: controlled fermentation of the organic matter in tunnels and second, the maturing of the compost on a platform where it is periodically turned. Digested material is mixed with wood chips that are added as bulking agent in a 1:4 ratio (v:v) to facilitate the aerobic fermentation process and to ensure the compost quality meets market-standards. The plant has 6 aerated tunnels, which are each 5 m high, 18.3 m deep and 2.5 m wide. The materials remain in the tunnels for 5 days. Aspirated air is used to oxygenize the matter and leaves the tunnels through the holes drilled in the floor. The leachate generated here is collected and stored for further processing outside the plant. Gases emitted in the composting tunnels are directed to a wet scrubber (water-sulfuric acid), and are then sent to a biofilter (Biofilter 1) (Figure 7.4).

Figure 7.4. *Detail of the Biofilter 1 (first day of analysis) at the anaerobic digestion plus composting plant.*

The final maturing stage consists of turned piles. The output tunnel material is turned daily, for a period of 5-7 days. The dimensions of the maturation piles vary depending on the space available in the maturation area. This area is closed and the gaseous emissions are conducted to a wet scrubber (water) and then to a biofilter system (Biofilter 3) (Figure 7.5).

Figure 7.5. *Detail of Biofilter 3 (first day of analysis) at the anaerobic digestion plus composting plant.*

Next, the compost obtained is refined in two steps: screening (using a 12 mm trommel screen) and ballistic separation. The screening process produces bulking material (wood chips) that is re-circulated through the process and, a small particle size fraction, which is sent to the ballistic separator. This separator is equipped with a filter bag for collecting the dust released. The refined compost is sent from the refining area to the storage zone.

Gaseous emissions from the refining area are treated in Biofilter 2. The annual compost production is estimated to be 7500 t.

As discussed, the emissions from the various processes are captured and treated, first using a scrubber system and second, a biofilter. To summarize, the plant has 3 different biofilters, which treat exhaust gases from 3 different emissions areas. The size, areas served and other characteristics of these biofilters are specified in Table 7.1.

Table 7.1. *Biofilters characteristics at the anaerobic digestion and composting plant.*

Characteristics	**Biofilter 1**	**Biofilter 2**	**Biofilter 3**
Dimensions (m) (length x width x height)	32.1 x 10 x 1.5	21 x 21 x 2	21 x 21 x 1.5
Packing material	Pine bark	Wood chips	Pine bark
Area where treated gasses originate	Composting tunnels and anaerobic digester feeding area.	Pre-treatment and refining area	Maturation area
Scrubber fluid	Tap water (closed loop) + sulfuric acid	Tap water (closed loop)	Tap water (closed loop)

7.3 Materials and methods

To observe the behavior of the processes in this plant, six representative sampling points were considered: OFMSW input, OFMSW plus diatomaceous mixture, output from anaerobic digestion, output from composting tunnels, maturation windrow and final compost. Material samples taken at these points were analyzed to determine the routine parameters described in Chapter 3. In case of the biological stability test, the dynamic respiration index (DRI) was used. DRI test is also explained in Chapter 3. Duplicates were performed for each the analysis.

The methodology used to inventory the environmental impacts associated with this plant's operation were described in Chapter 4.

Gaseous samples for analyzing NH_3 and VOC emissions were collected from the three emission points in the plant (three biofilters) as stated in Chapter 4.

7.4 Results and discussions

7.4.1 Inventory analysis

Data on all input and output materials and the biogas energy generated and sold to the electric company during the studied period (March to May 2008) were obtained from reports and data resources provided by the plant managers.

During the study period, the plant treated 4429 t OFMSW, produced 148.5 t compost and 437667.7 Nm^3 of biogas. To treat this amount of OFMSW, it was necessary to add 580.6 t of diatomaceous earth before the pretreatment steps and to mix the digestate with 446.5 t of wood chips (bulking agent) before the composting treatment within the tunnels.

Before the beginning of the study period, tests to observe the preferential gas pathways through biofilters were conducted over three consecutive days. A matrix of sampling points on each biofilter surface was established to determine if there was any uniformity across them. Ammonia and air velocity were measured in each point.

According to the size of the different biofilters and the preferential pathways test results (the differences observed between sampling points in the ammonia and VOC emissions exceeded 1%), five sampling profiles were taken from each biofilter. Each profile included three sampling points, which results in a 15 sampling points matrix for each biofilter-emitting surface.

As explained in Chapter 4, the total emissions estimation for each biofilter was conducted from the values obtained on different sampling days. These data have been obtained for each emission profile, compound (NH_3 and VOCs) and biofilter.

7.4.1.1 Process emissions analysis

Ammonia and VOC emissions from the 3 biofilters determined in the treatment plant are presented in Figures 7.6 and 7.7 respectively.

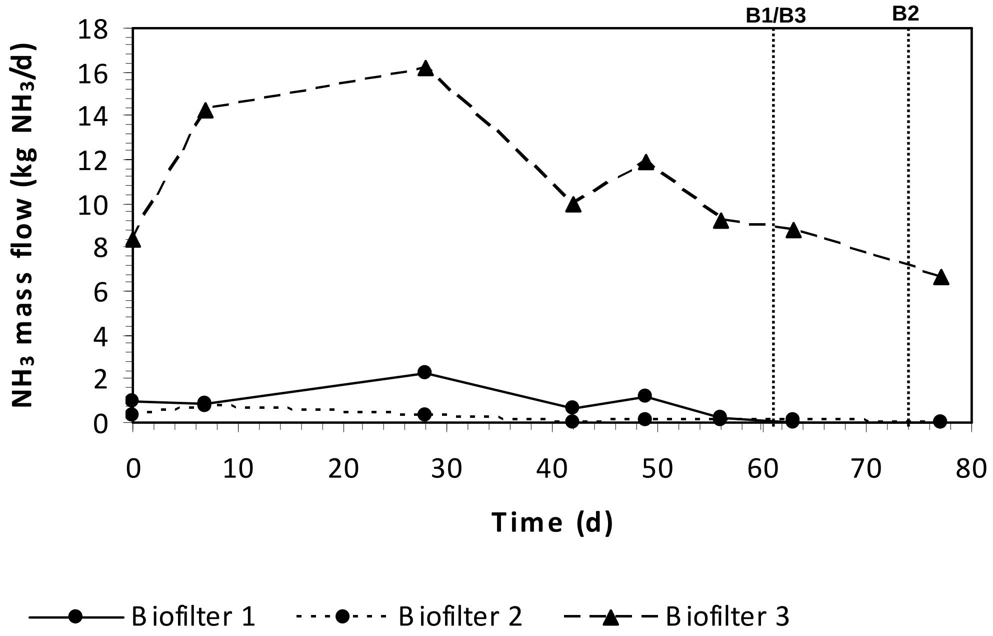

Figure 7.6. NH_3 mass flow emitted from the three biofilters during study period in the anaerobic digestion and composting plant. Biofilter 1 (emissions from decomposition in tunnels), Biofilter 2 (emissions from pre and post-treatment) and Biofilter 3 (emissions from maturation area). The dashed vertical lines corresponds to when biofilter modifications were made (B1= Biofilter 1, B2= Biofilter 2 and B3= Biofilter 3).

For Biofilter 1, the emissions were determined during a period in which, due to operational problems, the wet scrubber was not in service. During the last phase of the study period (day 61), this equipment was once again operational (dashed vertical line in Figure 7.6 (B1) and 7.7 (B1) indicates the beginning of scrubber operation). Figure 7.6 shows the total NH_3 mass flow emitted from Biofilter 1 during the study period. Ammonia emissions obtained after the scrubber became operational, show that the overall ammonia removing capacity was nearly 100%. This shows that optimal gas treatment can be achieved using a combination of scrubbing and biofiltration. During this study, the concentrations of ammonia emissions in Biofilter 1 were between 0 and 55 ppmv (0 and 2.3 kg NH_3/d). The average ammonia emissions for this plant for Biofilter 1 (0.85 kg NH_3/d) is lower that that obtained for Biofilter 1 in the tunnels composting plant (presented in Chapter 6), which treats the tunnels emissions during the second study, from June to September 2009. The value obtained for Biofilter 1 in the tunnels composting plant (1.1 kg NH_3/d) is 1.3 times higher than the value obtained in the present analysis.

The total VOC emissions for Biofilter 1 for this study period are shown in Figure 7.7. While the scrubber system (dashed vertical line (B1)) was found to be effective in treating the emissions of NH_3, it has no clear effect on the treatment of VOCs. VOC emissions in Biofilter 1 were between 40 and 1000 mg C-VOCs/m^3 (23 and 136 kg C-VOCs/d). The average of VOC emissions (74.8 kg C-VOCs/d) at Biofilter 1 for anaerobic digestion and composting is 7.3 times greater than the average obtained for Biofilter 1 (10.2 kg C-VOCs/d) in the tunnels composting plant (second study, Chapter 6). In comparing the two plants, it should be remembered that Biofilter 1 of the anaerobic digestion plant also treats the emissions coming from the anaerobic digester feeding area, where the inputted OFMSW is manipulated and transported.

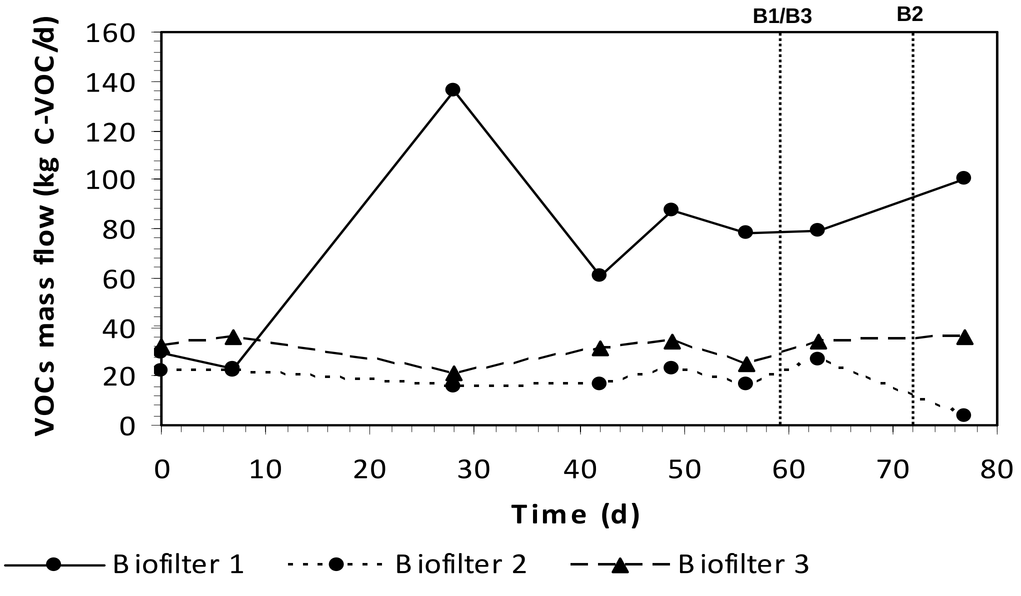

Figure 7.7. *Total mass flow of VOCs in the three biofilters during the anaerobic digestion and composting plant study period. Biofilter 1 (emissions from decomposition in tunnels), Biofilter 2 (emissions from pre and post-treatment) and Biofilter 3 (emissions from maturation area). The dashed vertical lines corresponds to when biofilter modifications were made (B1= Biofilter1 , B2= Biofilter 2 and B3= Biofilter 3).*

During the sample period, changes were made to the operation of Biofilter 2. The packing material within the filter was restructured in specific areas. Samples prior to day 77 were taken before material restructuration, which lead to a considerable reduction in VOC emissions (Figure 7.7) and a slight decrease in NH_3 emissions (Figure 7.6). Ammonia emissions in Biofilter 2 ranged from 0 to 19 ppmv (0 to 0.74 kg NH_3/d) while VOC emissions were between 12 and 163 mg C-VOCs/m^3 (3.8 and 26.9 kg C-VOCs/d).

Changes were also made to Biofilter 3, in this case on the 56th day, when biofilter watering maintenance took place. This maintenance had no effect on VOC (Figure 7.7) emission but resulted in a a slight but detectable decrease in ammonia emissions (Figure 7.6), which is most likely due to increased absorption capacity of the biofilter materials. The concentration of NH_3 and VOCs were consistent along the filter surface. Ammonia emissions in this biofilter were between 24 and 68 ppmv (6.6 and 16.2 kg NH_3/d) and VOC emissions ranged from 75 to 199 mg C-VOCs/m^3 (20.9 to 35.8 kg C-VOCs/d).

Two of the VOC emissions maps obtained (days 63 and 77) for Biofilter 2 are shown in Figure 7.8. This figure shows how the re-structuring of filter materials impacted VOC emissions. On day 63, VOC emissions were 26.9 kg C-VOCs/d and after restructuring (day 77), they decreased 7-fold to 3.82 kg C-VOCs/d (Figure 7.8b). The adjustment also resulted in a change in how the emissions were distributed across the surface of the biofilter. Initially, VOC emissions were evenly distributed over the surface of the biofilter (Figure 7.8a) but after the adjustment (Figure 7.8b) the emissions were localized in a small. Since the gas exhaust velocity over the surface of the filter was uniform on day 77, the localization of emissions in this area cannot have been caused by an unequal distribution of emissions received. Thus, a possible reason why VOC emissions were concentrated in this area could be that the area had low moisture levels.

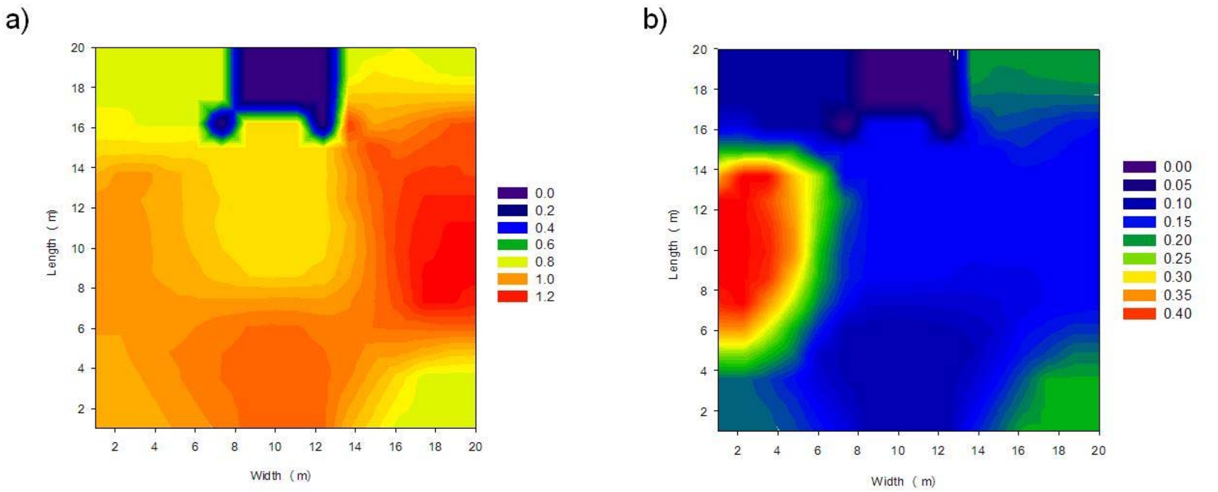

Figure 7.8. *Two dimension representation of the total VOC emission (mg C-VOCs/m^2 s) at Biofilter 2, a) profile at day 63 (before biofilter modifications), and b) profile at day 77 (after biofilter modifications). The air-flow inlet is located on the top of the figure.*

Using data from Figure 7.6 and 7.7, NH_3 and VOC emissions were determined for each biofilter allowing for the total emissions of the plant to be calculated (Table 7.2).

Table 7.2. Total NH_3 and VOC emissions in the anaerobic digestion and composting plant during the study period.

Biofilter	**Emissions (kg)**	
	NH_3	**VOCs**
Biofilter 1	79.88	6150.15
Biofilter 2	21.08	1418.11
Biofilter 3	900.11	2331.73
Total	1001.07	9899.99

As shown in Table 7.2, total VOC emissions (measured as C-VOCs) were higher than NH_3 emissions and each biofilter contributed different amounts to total emissions. NH_3 emissions mostly came from Biofilter 3, which treats the gases in the maturation area. Therefore, the operational problems of the Biofilter 1 scubber had little impact on total NH_3 emissions. NH_3 emissions from Biofilter 1 were expected to be the highest because Biofilter 1 treats the exhaust gases from the composting tunnels. Thus, it is important to mention that the maintenance of moisture levels and biofiltration media in Biofilter 1 is better than it is for Biofilters 2 and 3, which may have resulted in it having a higher ammonia emission mitigation efficiency. In the case of VOCs, the lowest emissions came from Biofilter 2, where pre and post-treatment (refining) area emissions are treated. The highest emissions values were obtained for Biofilter 1 (composting tunnels and anaerobic digestor feeding area).

Based on these values, total annual emissions were found to be 4004.4 kg of NH_3 and 39600 kg of total VOCs.

7.4.1.2 Energy consumption and electricity generation

The plant generated 437667.7 Nm^3 of biogas during the study period (Table 7.3), with an average methane content of 63%. Extrapolating these data, the annual production of biogas was found to be 1750671 Nm^3, from which 1584284 Nm^3 are used to generate electricity and steam to heat the digester, and the remaining 9.5% was burned in the safety flare. The difference between the theoretical biogas production (3500000 Nm^3/y) and the biogas production measured in this plant (175071 Nm^3/y) can partly be attributed to the difference between the theoretical amount of OFMSW (25,000 t/y) that can be treated by the plant and the amount that was treated (17726 t/y) during this study period.

Table 7.3. *Resources consumptions and generation during the study period (three months) in the anaerobic digestion and composting plant.*

Resource	**Value**
Nm^3 biogas produced/studied period	437668
kWh generated/studied period	882040
kWh consumed from its own electric production/studied period	821780
kWh purchased from electric company/studied period	204538
L diesel consumed/studied period	16124

The plant's annual production of electricity is 3528160 kWh. From the electricity generated, 821780 kWh (23.3%) were consumed in the plant and the rest, 2706380 kWh was sold to the electric company (Table 7.3). During the study period, the plant purchased 204538 kWh of energy from the electric company, thus the plant's annual energy consumption is 818152 kWh. During the study period, the plant's total electricity consumption (produced externally or internally) was 409983 kWh (1639932 kWh annually).

Another contribution to the energetic consumption in this plant was the diesel used both to fuel the tractors and the fuel to heat the digester when the cogeneration engines were turned off. Diesel consumption during the study period was 16124 L, and thus annual consumption is equal to 64496 L.

7.4.1.3 Safety flare emissions

When the cogeneration engines are turned off (at least every weekend) or when the digester production exceeds the engine combustion capacity, the excess biogas is burned by the safety flare. The system burned 41596.8 Nm^3 of biogas during the study period. According to Juniper Consultancy (2005), the average methane content of biogas for this type of process is 63%. Thus, according to this value, the combustion of methane by the safety flare resulted in 51508.2 kg of CO_2 emissions. According to Juniper Consultancy (2005), the biogas has a ratio of 1000 ppmv H_2S. Thus, assuming that the combustion process that takes place at the flare converts H_2S to SO_2, 118.9 kg of SO_2 were emitted during the study period. According to these results, the combustion of gases by the safety flare results in annual emissions of 206033 kg of CO_2 and 475.7 kg of SO_2. The CO_2 emitted due to the combustion of methane by the safety flare and the CO_2 originally contained in the biogas treated (120928 kg) were not included in the impacts determination because they result from the biological decomposition of organic material (IPCC, 2006).

7.4.1.4 Water consumption

In the plant, water is consumed to water the biofilters, to operate the scrubber and to generate the steam used to heat the digester. Total water consumption during the study period was 525 m^3. Based on this value an annual consumption of 2100 m^3 was calculated, where 39% (816 m^3/year) was used to heat the digester (steam production).

7.4.1.5 Solid waste refuse and leachate generation

Solid wastes are generated in the plant during pre and post-treatment operations. A total of 1824.11 t solid waste was produced during the study period, 9.72 t of which were metals. Refuse materials represent an average of 7296.44 t each year. All of this waste, excluding metal waste (which is recycled), is sent to a landfill. Refuse represents 41% (by weight) of the total OFMSW which enters the plant. The theoretical amount of refuse which enters the plant can be calculated using the official input material characterizations developed by the *Agència de Residus de Catalunya* (ARC, 2008). During the study period (first quarter 2008), the theoretical amount of refuse that entered the plant was found to be 13% (weight percentage). Thus, the real value obtained for refuse generation is 3.14 times greater than the theoretical value, a difference, which has to do with the pretreatment systems performed in the plan. For example, during pre-treatment, the OFMSW is mixed with diatomaceous material, some of which is lost with the refuse.

Leachate is generated in the reception area and in the composting tunnels where it is collected in a tank and sent away for treatment. During the study period, 150 m^3 of leachate was collected (600 m^3/year). In this plant, the biogas condensates produced in the gasometer during the anaerobic digestion process are collected and are also treated outside the plant. 220 m^3. Of condensate was produced during the three-month study period, which corresponds to an annual condensate production of 880 m^3.

7.4.1.6 Overall process evolution

The routine analytical methods described in section 7.3 of this chapter were applied to the solid samples obtained from different parts of the plant (OFMSW input, OFMSW plus diatomaceous, anaerobic digestion output, tunnel output, maturation windrow and final compost) and used to obtain an indicator of the plant's operation during the study period. These analyses were developed in collaboration with Michele Pognani (a pre-doctoral coworker in this research group).

The change in the bulk density of the treated material during the process is presented in Figure 7.9. For this type of waste, the inputted OFMSW had normal mean density (0.40-0.60 kg/L) (Haug, 1993). Anaerobic digestion resulted in a decrease in the density of the waste. When the material was mixed with the bulking agent and composted, its density was between 0.57 to 0.70 kg/L.

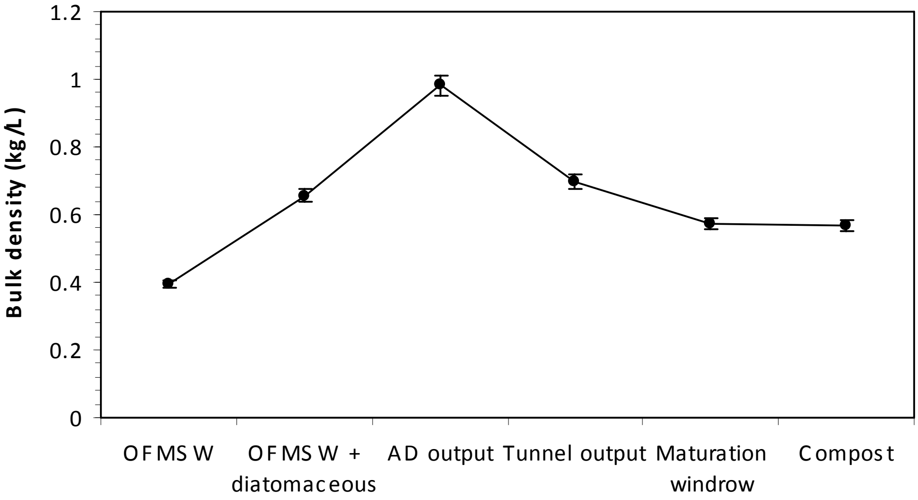

Figure 7.9.*Change in sample material bulk density (kg/L) in the anaerobic digestion and composting plant.*

The change in the dynamic respiration index (DRI) of each sample throughout this study is shown Figure 7.10. The values represented in Figure 7.10 are the averages of two samples, whih were taken on different days. As expected, the DRI values decrease as the process progresses. For Compost 1, the DRI was reduced by 68% during anaerobic digestion and during the composting process they were further reduced to 75%. An overall reduction in DRI of 92% was observed for Compost 1. For Compost 2, a 77% reduction in DRI was observed. These results are similar to the results in other works such as those reported in Ponsá et al. (2008).

In the case of Compost 1, the DRI value obtained (386 mg O_2/g OM h) indicates that the product was very stable (values under 1000 mg O_2/g OM h are considered indicators of stable materials), whereas for compost 2 (1115 mg O_2/g OM h) the value is slightly above this stability limit (Adani et al., 2004).

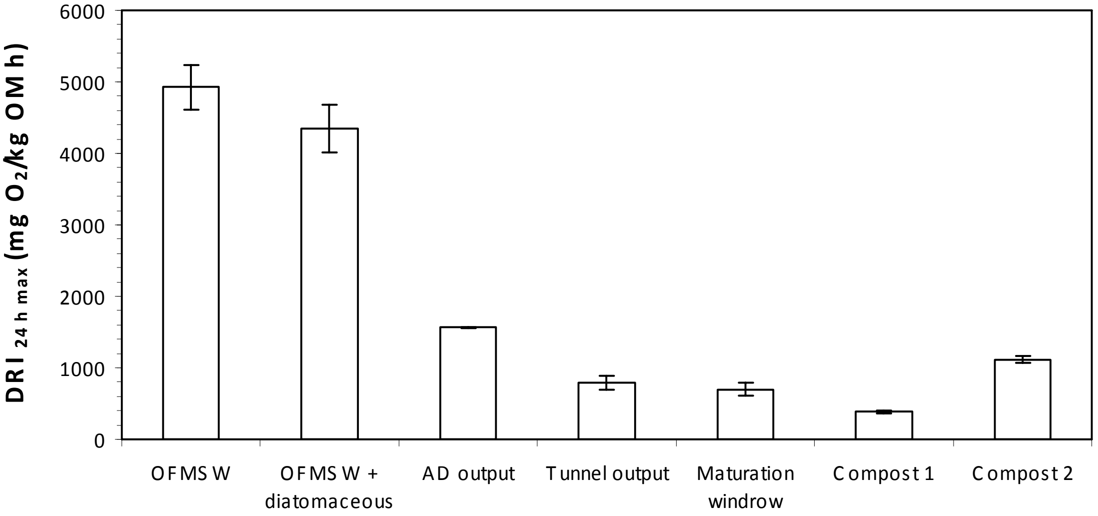

Figure 7.10.*Evolution of the dynamic respiration index (mg O_2/g OM h) in the anaerobic digestion and composting plant.*

The values organic matter and moisture content values of the material for each sample point are shown in Figure 7.11. Organic matter content followed and DRI follow the same pattern. Beginning at 70% (OFMSW input) the organic matter content decreases to 40% in the finished compost. Moisture content followed a different trend because the materials are subjected to two different biological processes during the treatment. The moisture content of the inputted material was 67.9% and the OFMSW mixed with diatomaceous material had a moisture content 48.4%, a reduction of nearly a 20% due to conditioning. Following anaerobic digestion this value increased by more than 24% to reach moisture values greater than 70%. However, during the composting process (decomposition in tunnels and maturation phase throughout turned windrows), this value again decreases. Moisture content during composting was within the recommended optimal values (40-60%) for this process (Haug, 1993), finally reaching 40% moisture content in the final compost.

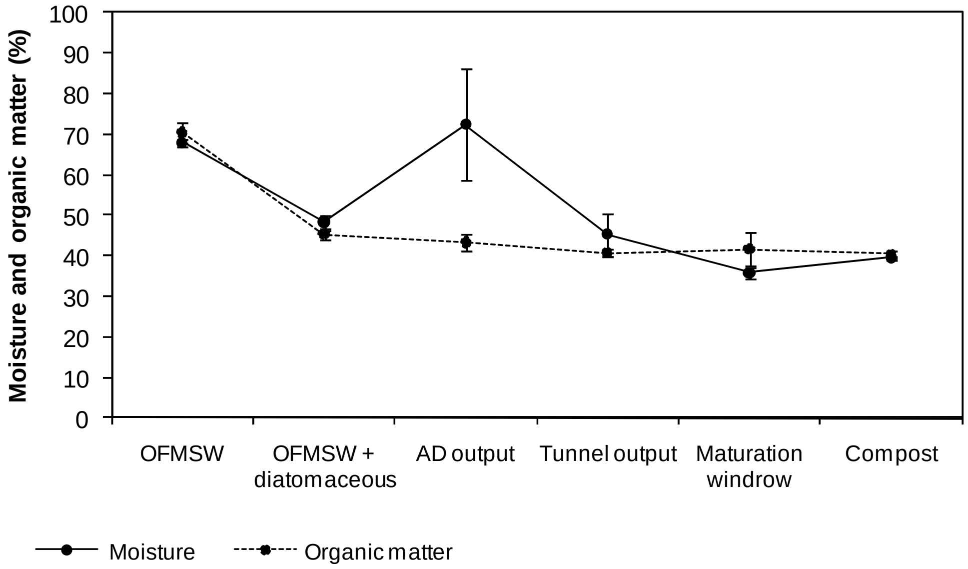

Figure 7.11. *Evolution of moisture content and % organic matter in the anaerobic digestion and composting plant.*

7.4.2 Plant performance indicators

As in previous chapters, plant performance indicators were identified for one ton of OFMSW treated.

7.4.2.1 Process indicators

As shown in Table 7.4, the yield of compost in this plant is very low compared to the yield (0.14 t compost/t OFMSW) obtained for the tunnels system composting plant (Chapter 6). It is important to mention that this plant uses an anaerobic digestion process prior to composting, where there is a considerable degradation of the organic matter which is converted to biogas, and thus the direct composition of compost yield values is not possible.

Table 7.4. *Process indicators of the anaerobic digestion and composting plant.*

Process indicators	
t compost/t OFMSW	0.03
t real refuse/t refuse theoretical	3.14[1]
t refuse/t OFMSW	0.41
Nm^3 biogas/t OFMSW	98.8
kWh generated/t OFMSW	199.2
m^3 leachate/t OFMSW	0.03
m^3 condensate/t OFMSW	0.05

[1] Theoretical refuse data obtained from *Agència de Residus de Catalunya* (ARC, 2008).

Dividing biogas production by the amount of OFMSW entering the digester (OFMSW without inadequate materials, after pre-treatment), gives a biogas production rate of 137.4 Nm^3/t OFMSW. This value is in the range (121-190 Nm^3/t OFMSW) published in the Juniper guide (Juniper Consultancy Services, 2005) for treatment plants using the same technology and is also within the predicted range of the plant design (140 Nm^3/t OFMSW).

The kWh of electricity produced per Nm^3 of biogas, determined from the values in Table 7.4, was found to be 2.23 kWh/Nm^3 biogas. In this case, the value is lower than the range proposed in the Juniper guide (5.5-6.4 kWh/Nm^3 biogas). The biogas burned by the safety flare and motor malfunctions likely explain this difference.

7.4.2.2 Resources consumption indicators

Table 7.5 presents the resource consumption indicators considered. As mentioned, the electricity produced (from biogas) and consumed by the plant can be distinguished from the electricity that is purchased from the electrical company.

Table 7.5. *Resources consumption indicators in anaerobic digestion plus composting plant.*

Resources consumption indicators	
kWh electricity external consumption/t OFMSW	46.2
kWh electricity internal consumption/t OFMSW	46.4
Total kWh used/t OFMSW	92.6
L diesel/t OFMSW	3.64
m^3 total water/t OFMSW	0.12

It is important to highlight that internally produced and externally produced electricity are used to power different areas of the plant. The electricity which is purchased from the electric company is used in the composting tunnel ventilators, the post-processing machinery and to light the plant and offices. While the electricity generated within the plant is consumed by the pre-treatment machines, gasometer, safety flare, scrubber systems, digester and to pump the condensates produced during anaerobic digestion. If pre-treatment and refining are associated with, respectively, digestion and composting, their energy consumptions are quite similar. The total external energy consumed in the plant (85.04 kWh/t OFMSW) includes the energy purchased from the electrical company and the energy obtained from diesel (see Chapter 4).

Water consumption in this plant (120 L/t OFMSW) was much lower than the value obtained for the tunnels system of the composting plant (330 to 560 L/t OFMSW) discussed in the previous chapter. The reason for this is that in this case, the plant's gas treatment equipment uses a closed-loop water system. This consumption was also higher than the value (89 L water/t OFMSW) reported in other full-scale composting treatment plants (Blengini, 2008).

7.4.2.3 Emission indicators

Table 7.6 shows the emission indicators for ammonia, VOCs and CO_2 emitted during the biological process degradation and due to energy consumption (electricity and diesel). As mentioned, CO_2 from biological degradation of organic matter, either aerobic or anaerobic, is not taken into account in the emissions inventory.

Table 7.6. *Emission indicators in anaerobic digestion plus composting plant.*

	Emission indicator (kg/t OFMSW)		
	NH_3	**VOCs**	**CO_2 (energy)**
Process	0.23	2.23	—
Electricity (external)	9.05E-05	1.11E-02	25.93
Diesel	2.82E-07	6.48E-02	10.66
Total	0.23	2.31	36.59

As shown in Table 7.6, VOC and NH_3 emissions were mainly generated in the composting process while CO_2 emissions were due to electricity consumption (25.93 kg CO_2/t OFMSW). Diesel consumption has a lower impact on CO_2 emissions than electricity (10.66 kg CO_2/t OFMSW).

Table 7.7 summarizes input and outputs in this plant.

Table 7.7. *Input and outputs summary of the anaerobic digestion plus composting plant.*

Inputs	Raw materials	t OFMSW/y	17716
		t bulking agent/y	17860
		t diatomaceous/y	2322.40
	Resources	kWh/t OFMSW (external)	46.20
		L diesel/t OFMSW	3.64
		m^3 water/t OFMSW	0.12
		Total energy/t OFMSW	85.04
Outputs	Liquid emissions	m^3 leachate/t OFMSW	0.03
		m^3 condensate/t OFMSW	0.05
	Atmospheric emissions (energy)	kg CO_2 energy/t OFMSW	36.59
		kg NH_3 /t OFMSW	0.23
		kg VOCs/t OFMSW	2.31
	Product	t compost/year	594
		t compost/t OFMSW	0.03
		kWh/t OFMSW (produced)	152.80
		Nm^3 biogas/t OFMSW	98.79
	Refuse	t refuse/t OFMSW	0.41

7.4.3 Environmental impact analysis

Based on inventory and experimental data, the impact potentials of the plant were calculated from a LCA perspective. The impact categories considered are described in Chapter 3. It is important to highlight that this study does not consider CO_2 savings resulting from the hypothetical substitution of fossil fuel with the biogas generated in this plant.

7.4.3.1 Functional Unit

1 ton of OFMSW is the functional unit used in this study.

7.4.3.2 System boundaries

As for the other plants considered in this study, the system boundaries for the LCA of this plant are its process activities (anaerobic digestion and composting). Hence, transporting, manufacturing and products entering and exiting the plant were excluded from this study (refuse and leachate treatment are excluded from the system considered as it is done outside the plant).

7.4.3.3 Life cycle impact assessment

The contribution of the emissions generated during organic matter degradation (NH_3 and VOCs) and electricity and fuel consumption to the total impact potentials are presented in Figure 7.12.

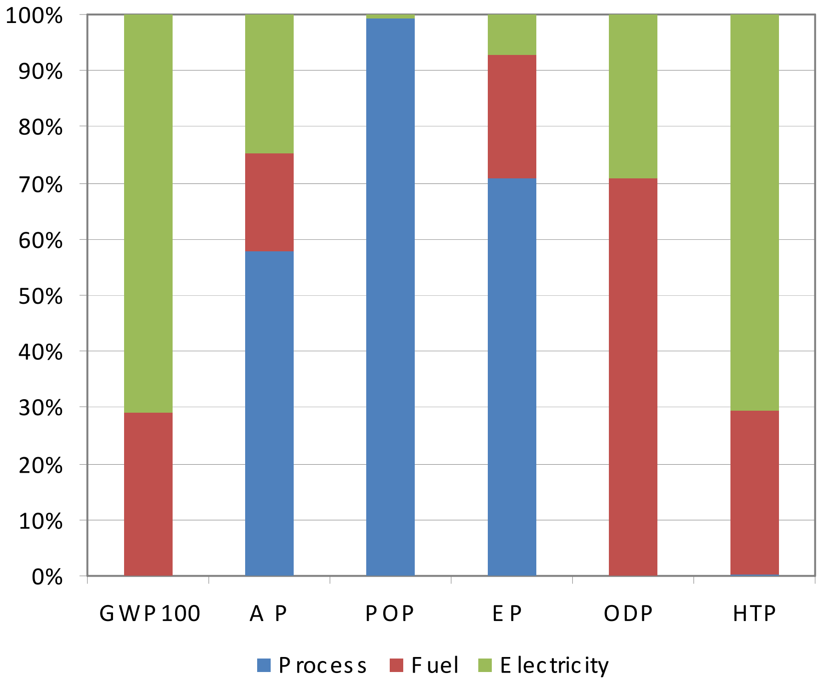

Figure 7.12.*Environmental analysis in the anaerobic digestion and composting plant showing contributions of process, fuel and electricity.*

To supplement Figure 7.12, total impact potentials are presented in Table 7.8 with the percent contributions of the different aspects considered to each potential.

Table 7.8. *Impact characterization results for the anaerobic digestion and composting plant including the contribution of process, fuel and electricity to the total value of the potential (the percent contribution of each item to the total value is shown in brackets).*

Impact potentials	Process	Fuel	Electricity	Total
Global warming (kg CO_2 eq/t OFMSW)	0 (0.0%)	10.67 (29.1%)	25.95 (70.9%)	36.61
Acidification (kg SO_2 eq/t OFMSW)	0.36 (57.8%)	0.11 (17.4%)	0.16 (24.8%)	0.63
Photochemical oxidation (kg C_2H_4 eq/t OFMSW)	0.93 (99.2%)	3.00E-03 (0.3%)	5.00E-03 (0.5%)	0.94
Eutrophication (kg PO_4^{3-} eq/t OFMSW)	0.08 (70.8%)	0.02 (21.9%)	8.05E-03 (7.2%)	0.11
Ozone layer depletion (kg CFC-^{11} eq/t OFMSW	0 (0.0%)	9.04E-06 (70.7%)	3.74E-06 (29.3%)	1.28E-05
Human toxicity (kg 1,4-DB eq/t OFMSW)	0.02 (0.3%)	2.05 (29%)	5 (70.7%)	7.07

- **Global warming potential**

The main contribution to this potential was the CO_2 emitted during electricity consumption, 71% (25.95 kg CO_2 eq/t OFMSW), while diesel consumption was responsible for 29% (10.67 kg CO_2 eq/t OFMSW) of the total potential (Figure 7.12).

- **Acidification potential**

NH_3 generated during the biological process contributed the most to acidification, accounting for 57.8% (0.36 SO_2 eq/t OFMSW) of the total potential. The contributions of electricity and fuel consumption were 24.8% and 17.4%, respectively (Table 7.8).

- **Photochemical oxidation potential**

The emissions from the process contribute the most to the photochemical oxidation, with VOC emissions being responsible for 99.2% (0.93 kg C_2H_4 eq/t OFMSW) of the entire potential. The remainder of the impact was due to energy consumption (5E-03 C_2H_4 eq/t OFMSW for electricity and 3.00E-03 C_2H_4 eq/t OFMSW for fuel).

- **Eutrophication potential**

As for acidification potential, NH_3 emissions generated by the biological process contributed the most to eutrophication (Figure 7.12), being responsible for 70.8% (0.08 kg PO_4^{3-} eq/t OFMSW) of the total impact potential. Electricity consumption has a minimal impact on EP (Table 7.8).

- **Ozone layer depletion potential**

Energy consumption was responsible of the entire ozone layer depletion potential. Diesel is responsible for the majority of this, contributing 70.7% (9.04E-06 kg CFC-11 eq/t

OFMSW), while electricity consumption accounted for the remainder (29.3%). Process emissions have no impact of ODP.

- **Human toxicity potential**

Electricity was responsible for 70.7% (5 kg 1,4-DB eq/t OFMSW) of the human toxicity potential and the remaining 29% was due to diesel consumption. The contribution of the biological process to this potential was minimal (0.3%).

In this plant, process emissions have the greatest impact for three impact categories (AP, POP and EP) due to the emissions of NH_3 (for AP and EP) and VOCs (for POP). This result is consistent with the results obtained in previous chapters (confined windrows composting in Chapter 5 and tunnels composting in Chapter 6. Energy consumption had major impacts on GWP100, ODP and HTP (Table 7.8), where electricity contributes most to GWP100 (79.9%) and HTP (70.7%) and fuel consumption to ODP (70.7%). From these results it can be concluded that the facilities studied in this dissertation follow the same trend in terms of potential impacts. However, the final value of the impact potentials was due to different reasons (plant dimensions, lack of emissions mitigation, resources consumption, etc).

7.4.4 Suggestions for improving the composting plant

Improvements can be made at this plant to reduce the environmental impacts observed in this study. Some of these are discussed below:

- Biogas recovery for energy production should be maximized. For example, gasometer capacity could be increased to accumulate all the biogas generated during periods of maximum production; engine systems should be improved to turn-on automatically during the weekend to reduce the amount of biogas burned by the safety flare.

- As it has been seen it is essential to conduct periodic maintenance of the biofiltration systems for effective emissions mitigation. This maintenance consists of regular irrigation of the systems, materials restructuration and, if necessary, material changes. In addition, the wet scrubber systems should always be in operation and controlled.

- In the case of pre-treatment, the diatomaceous earth should be added after the OFMSW pre-treatment, to prevent material losses.

- They should try to meet all of its energetic needs using internally produced energy; this would lower its impact on the environment. Using biogas as an energy source also results in global benefits because it is a cleaner alternative to fossil fuels.

Chapter 8

Potential odour measurements and environmental impact analysis at a full-scale anaerobic digestion plant

Part of the results presented in this chapter have been submitted for publication in Environmental Science and Technology as: V. Orzi, E. Cadena, G. D'Imporzano, A. Artola, M. Crivelli, E. Davoli and F. Adani, "Potential odour emissions measurement in organic waste during anaerobic digestion: Relationship with process and biological stability parameters".

The experiments presented in this chapter were developed at the *Università degli Studi di Milano, Dipartamento di Produzione Vegetale* (Milan, Italy) as part of a research stage that was undertaken from September 2008 to February 2009 in Dr. Fabrizio Adani research group (RICICLA). Part of the results obtained here are focused on the determination of malodours and malodorous compounds coming from the feedstock mix degradation in biological treatment processes (anaerobic digestion in this case). Different methods to measure odours and related compounds were used: olfactometry, gas chromatography plus mass spectrometry and the electronic nose. Also the possible relationship between odorous emissions and biological stability of organic materials was assessed.

8.1 Introduction

Odorous emissions can be generated in all the areas of an anaerobic digestion plant, but the main sources of odours are the volatile substances produced by the uncontrolled fermentation of organic wastes during the storage and the pre-treatment of the waste. The establishment of the anaerobic metabolism produces a set of odorous compounds including inorganic molecules (ammonia and hydrogen sulphide) and organic molecules such as VOCs, aromatic compounds, mercaptanes and alkyl sulfides (Komilis et al., 2004) that can be released to the atmosphere. Substances generated during the anaerobic digestion process and arising from the post-stabilization and maturation stages of the digested waste, usually performed by aerobic processes are also responsible of malodours (APAT, 2005).

Typically sensory measurements employing the human (olfactometry) or the electronic nose (EN) and chemical analysis are used to quantify the odorants present in sewage treatment works where odour abatement and health control are important objectives (Baby et al., 2005). Olfactometry, using trained human panellists, is the internationally accepted method for the determination of odour concentration. One of the main drawbacks to Olfactometry is that it must be conducted in a controlled laboratory setting and a sufficient number of panellists must be available to conduct the analysis. Gas chromatography (GC) is frequently used to identify and quantify odorous compounds (Hobbs et al., 1995) because it is easier to use and apply when compared to olfactometry. However, gas chromatography is not suitable to detect synergic effects within gaseous compounds that can generate odours not directly related to each odorant concentration. Electronic nose analysis with a sensor array is a potential technology for odour evaluation because it is able to mimic the smell sense in the human nose (Gralapp et. al., 2000).

Several applications of the electronic nose have been reported, for example, odour classification of grains (Börjesson et al., 1996), analysis of urine samples (Di Natale et al., 1999), monitoring sausage fermentation (Eklöv et al., 1998), detection of pollen (Kalman et al., 1997), characterisation of wastewater (Stuetz et al., 1999) or monitoring of microbial and cellular growth in bioreactors (Bachinger and Mandenius, 2001; Mandenius, 1999). A particular interest in bioprocess engineering is that the EN has been shown to respond to characteristic changes in microbial activity. For example, the EN has been successfully applied in strain classification (Gardner et al., 1998; Holmberg et al., 1998; Gibson et al., 1997) and to identify the growth phases of bacteria, yeasts and hamster cells typically used in bio-processing (Bachinger and Mandenius, 2000; Bachinger et al., 1998; Gardner et al., 1998).

EN has also been used in environmental applications. The relationship between the EN responses of quiescent sewage liquors and biological BOD_5 illustrates the ability of a non-specific sensor array to respond to sewage samples with different biodegradable content (Stuez, 1999). Romain et al. (2005) used a lab-made EN to monitor exhaust gases from a composting pile. In a recent study D'Imporzano et al. (2008) evaluated the relationship between the biological stability of a substrate and the odours produced during a lab-scale composting process measured by an EN. A good correlation between odour production and biological stability will result in important practical applications in waste management and related environmental management practices, and focuses the need to improve the research in this field.

8.2 Objectives

The main objective of the research work presented in this chapter is to evaluate the correlation between biological stability and odour coming from a mixture of organic wastes treated in a full-scale anaerobic digestion plant in its three process phases: mixing of organic wastes (incoming materials, non digested), anaerobic digestion and post-digestion. Odorous emissions have been measured using three different methodologies: olfactometry, electronic nose (EN) and gas chromatography plus mass spectrometry (GC-MS) (VOCs determination) while biological stability has been evaluated by means of aerobic (oxygen demand, OD_{20})

and anaerobic (anaerobic biogasification potential, ABP) measurements. The relationship between the results obtained using the different odour measurement techniques has also been investigated as well as different correlations between chemical parameters and biological stability measures. The data of EN and GC-MS have been evaluated using principal component analysis (PCA).

The second objective of this work is to analyze the environmental impacts of the OFMSW treatment through anaerobic digestion from a life cycle perspective focusing on impact indicators and potential impacts, as has been done in the rest of treatment plants studied.

8.3 Anaerobic digestion plant description

The studied plant is located in northern Italy and treats around 30000 t feedstock mix/year coming from a OFMSW source-selection collection system and farm wastes (Figure 8.1). In this plant, 1 MW of electrical power is produced co-digesting, in four parallel continuously stirred thermophilic reactors plus a post-digester, a mixture of energetic crops, pig manure slurry, agro-industrial waste and OFMSW (Pognani et al., 2009). The plant activity is divided into two principal processes: the biological treatment of organic substances itself (anaerobic digestion), with the biogas produced as the main valuable by-product and the transformation of this biogas into electricity. The plant includes a reception area where the tractors unload the material, a biomass grinding system, a device to squeeze the organic wastes and a material mixer, 4 anaerobic digesters, one post-digester and 5 cogeneration units with the respective generator and related electrical controls. This plant, unlike that studied in the previous chapter (anaerobic digestion plus composting plant), does not have a safety flare.

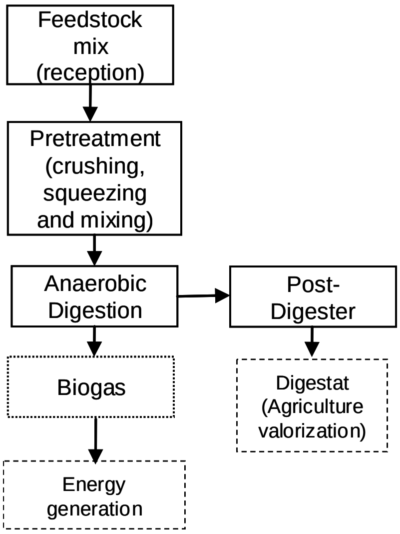

Figure 8.1. *General flowchart of the studied anaerobic digestion plant.*

The first stage of the treatment process consists in crushing, squeezing and mixing the organic wastes (pre-treatment). The resulting mixture is then fed to the anaerobic digesters at a rate of 0.22 t each 15 min. The digesters work under thermophilic conditions (at 55°C) with a waste retention time of 40 days and are followed by post-digestion tanks where the material remains for 10 more days. Finally the digestate goes to an storage tank, open to the atmosphere, where it remains for 10 days (post-digestion) without additional stirring or aeration. The digested material is used in agriculture benefit. All the electricity produced in this plant, as opposed to the facility studied in Chapter 7, is sold to the electric company and the electricity consumed in their operations is entirety purchased to the public electric company.

During the monitoring period (September–December 2008), three sampling campaigns were undertaken, collecting in each of them samples from feed-in mixture (indigestates) (ND), output from thermophilic anaerobic digesters (D) and output from post-digester (PD).

8.4 Materials and methods

8.4.1 Chemical characterization

Representative material samples were used to carry out all the analytical tests. Dry matter (DM), organic matter (OM), total N-Kjeldahl (TKN), NH_3 and pH were analyzed on fresh samples according to methods described in Chapter 3. Volatile fatty acids (VFA) and total organic carbon (TOC) were determined according to standard procedures (US Department of Agriculture & US Composting Council, 2001, and APHA, 1998, respectively).

Biological stability tests, oxygen demand (OD_{20}) and anaerobic biogasification potential (ABP) were performed on all the samples taken. Those tests were also explained in Chapter 3. These analyses were performed in duplicate.

8.4.2 Gaseous samples collection

All the gaseous samples for odour detection were collected using an Italian standard method described in APAT (2003), which consist in a static flux chamber system. In brief, four litres of waste sample were put on a tray container and covered with a Plexiglas® (polymethyl-methacrylate) chamber (38.8 x 50.5 x 40 cm) with 0.196 m^2 surface (Figure 8.2). The chamber was then continuously flushed during 10 minutes with air at 0.35 m^3/h flow rate. Output gas from the chamber was then taken from the outlet port and stored in Nalophane homemade bags (Figure 8.3). Bags of different volume, i.e. 20 L, 2 L and 3 L, were filled and used for olfactometric, electronic nose and gas chromatography and mass spectrometry (GC-MS) analyses, respectively. NH_3 concentration was determined directly at the outlet port of the static chamber by means of the ammonia sensor described in Chapter 3.

Figure 8.2. *Static chamber (Plexiglas® material) used to sample gaseous emissions from the anaerobic digestion plant.*

Figure 8.3. *Gaseous samples collection in the static chamber. A Nalophane bag was connected and filled at the output port at a flow rate of 0.35 m^3/h.*

8.4.3 Gas chromatography and mass spectrometry analysis

Volatile organic compounds (VOCs) from gaseous samples were analyzed by solid phase micro-extraction-gas chromatography mass spectrometry (SPME/GC-MS) (Davoli et al., 2003) at Mario Negri Laboratories (Istituto di Ricerche Farmacologiche, Milan, Italy).

A manual SPME device and divinylbenzene (DVB)/Carboxen/polydimethylsiloxane (PDMS) 50-30 µm fiber (Supelco, Bellefonte, PA, USA) were used. The analytes were adsorbed from the gas samples by exposing the fiber, preconditioned for 3 h at 250°C, as suggested by the supplier, in the Nalophane bags for 30 min at room temperature. A solution of deuterated p-xylene in methanol was used as internal standard for quantitative analysis. VOCs analysis was performed using an Agilent 5975C Series GC/MSD. Volatiles were separated using a capillary column for VOCs (Meta VOC, Teknokroma, Sant Cugat del Vallès, Barcelona, Spain) of 30 m x 0.32 mm, film thickness 3.0 m. Carrier gas was helium at a flow-rate of 1 ml/min. VOCs were desorbed exposing the fiber in the GC injection port for 3 min at 250°C. A 0.75 mm glass liner was used and the injection port was in splitless mode. The temperature program was isothermal for 3 min at 35°C, raised to 200°C at a rate of 8°C/min. The transfer line to the mass spectrometer was maintained at 250°C. Compounds were tentatively identified by comparing their mass spectra with those contained in the NIST (USA) 98 library. A semi-quantitative analysis, for all the identified compounds, was performed by direct comparison with the internal standard. Results were expressed as part per billion on a volume basis (ppbv). Organic compound classes were successively elaborated by principal component analysis (PCA) by using SCAN software (Minitab Inc., State College, PA).

8.4.4 Electronic Nose

Air samples were analyzed by using a PEN3 electronic nose (Airsense Analytics, Schwerin, Germany) (Figure 8.4) equipped with 10 thermo-regulated (150–500°C) sensors made of metal oxide semiconductors (MOS) as reported by D'Imporzano et al. (2008). Each sensor is sensitive to a group class compounds. Sensors in the EN used in this work presented the following sensibility profile (from S1 to S10): S1-aromatic compounds, S2-polar compounds and nitrogen oxides, S3-aromatic compounds, ketones, and aldehydes, S4-H_2, S5-low polarity aromatic and alkane compounds, S6-methane compounds, S7-sulfur compounds and terpenes, S8-alcohols, ketones and partially aromatic compounds, S9-sulfur containing and aromatic compounds and, S10-methane at high concentration. Principal component analysis (PCA) was used to compare odours qualitatively. Multivariate analysis was carried out by means of SCAN software (Minitab Inc., State College, PA).

Figure 8.4. *Electronic nose PEN3 by Airsense Analytics (Schwerin, Germany).*

The following work parameters were used: 300 s for the clean cycle and 100 s for the measure cycle, 3 cycles for each bag were repeated. Only the last 20 s of the measures, when the response of the sensors was stabilized, were chosen to create the sensor patterns.

8.4.5 Olfactometry analysis

Olfactometric analyses were carried out in conformity with EN 13725 (CEN, 2003) by the

Facoltà di Medicina Veterinaria laboratories at the Università degli Studi di Milano (Milano, Italia). An Olfaktomat-n 6 olfactometer (PRA-Odournet B.V., Amsterdam, NL), based on the yes or no method was used as a dilution device.

The measuring range of the olfactometer starts from a maximum dilution factor of 33000 with a dilution step factor of 2. Results of olfactometry were expressed as odour concentration value (OU/m^3). To calculate the odour emission rate in this study (OE, in OU/m^2 h), the odour concentration (OU/m^3) was multiplied by the air filling flow used in the sampling chamber.

8.4.6 Statistical analysis

All statistical analysis were conducted by the Università degli studi di Milano and PCA Technologies group during the research stage at this university.

All statistics developed in this chapter were performed by using SPSS 13 (Chicago, IL). Partial least squares regression (PLS) analysis (Einax et al., 1997) was used for modeling the relationship between odors (OU/m^2 h) and EN data (dimensionless arbitrary unit) and GC-MS data (molecular classes expressed as ppbv), and odors *vs.* respiration data (OD_{20} expressed as mg O_2/g DM) by using the SCAN software (Minitab Inc., State College, PA). Both goodness of fit, i.e regression coefficient (R^2), and goodness of prediction, i.e cross validate regression coefficient (R^2_{cv}), were reported. Cross-validate regression coefficient was calculated as an error of the prediction by cross validation-leave-one out procedure. This procedure allows selecting the more appropriate latent vectors, reducing their total number (i.e. selection of EN sensor and organic compound classes able to predict olfactometric units).

For the GC-MS data, organic compounds classes were used for PLS application taking into consideration those classes that were effectively represented, i.e. esters, aromatic, alcohols, carbonyl and terpenes.

8.5 Results and discussions

8.5.1 Chemical parameters and biological stability

Chemical characteristics and biological stability indices of the analyzed samples are reported in Tables 8.1 and 8.2 respectively. There was a clear difference between values of DM, OM and TOC in Table 8.1 for non digested and digested materials, a fact that was expected, since the anaerobic digestion process provides a high organic matter reduction (Braber, 1995). However, this difference did not exist for samples corresponding to digested and post-digested wastes.

Table 8.1. *Chemical characteristics of the samples analysed (ND: non digested, D: digested and PD: post digested).*

Sample	Collection date	Process time (days)	pH	DM* (%FM)	OM (%DM)	TOC (%DM)	TKN (%DM)	NH_3 (%DM)	VFA (mg/L)
NDa	15/09/08	0	4.41	15.25 ± 0.35	87.49 ± 0.13	45.02 ± 0.84	3.26 ± 0.30	0.68 ± 0.07	21566 ± 708
NDb	07/10/08	0	4.56	22.05 ± 0.02	89.90 ± 0.14	46.56 ± 0.37	2.61 ± 0.16	0.56 ± 0.01	26188 ± 9330
NDc	11/11/08	0	3.84	14.29 ± 0.13	86.16 ± 0.20	49.11 ± 0.89	3.40 ± 0.05	0.58 ± 0.03	27220 ± ,881
Mean ND				**17.15 ± 3.81 b**	**87.86 ± 1.69 b**	**46.90 ± 1.93 b**	**3.09 ± 0.41 a**	**0.61 ± 0.06 a**	**24135 ± 3024 b**
Da	15/09/08	40	8.20	3.83 ± 0.19	67.61 ± 1.00	39.25 ± 1.09	13.35 ± 0.15	9.02 ± 1.23	7539 ± 148
Db	07/10/08	40	8.22	4.42 ± 0.39	67.11 ± 1.00	37.09 ± 0.84	10.90 ± 0.09	6.60 ± 0.15	6579 ± 598
Dc	11/11/08	40	8.13	5.69 ± 0.29	72.02 ± 0.01	42.34 ± 0.71	9.00 ± 0.08	5.51 ± 0.09	10180 ± 573
Mean D				**4.65 ± 0.88 a**	**68.91 ± 4.05 a**	**39.56 ± 2.46 a**	**11.08 ± 1.95 b**	**7.04 ± 1.80 b**	**8100 ± 1709 a**
PDa	15/09/08	50	8.35	3.77 ± 0.42	64.84 ± 0.13	36.23 ± 0.88	13.07 ± 0.32	9.61 ± 0.10	1021 ± 322
PDb	07/10/08	50	8.28	6.49 ± 0.58	58.06 ± 0.49	35.24 ± 0,89	6.45 ± 0.57	4.82 ± 0.01	1839 ± 166
PDc	11/11/08	50	8.20	4.30 ± 0.01	64.45 ± 0.11	38.35 ± 1.20	11.83 ± 0.06	7.60 ± 0.02	9498 ± 779
Mean PD				**4.85 ± 0.60 a**	**62.45 ± 3.41 a**	**36.61 ± 1.61 a**	**10.45 ± 3.16 b**	**7.34 ± 2.41 b**	**4832 ± 4519 a**

* FM: Fresh Matter.

a, b: Values in the same column followed by the same letter are not statistically different (Tukey test, $P<0.05$).

Table 8.2. *Biological stability indices (OD$_{20}$ and ABP) and emission characteristic of the samples analysed (ND: non digested, D: digested and PD: post digested).*

Sample	Date	Process time (days)	OD$_{20}$ (mg O$_2$ /g DM)	ABP (Nl/kg DM)	Odour (OU/m^{2}h)	VOC (ppbv)	NH$_3$ (ppmv)
NDa	15/09/08	0	279.70 ± 21.50	546.89 ± 2.42	119446	2248	0
NDb	07/10/08	0	218.50 ± 18.67	483.12 ± 8.71	76017	1672	15
NDc	11/11/08	0	293.79 ± 0.18	481.60 ± 1.36	36243	6005	5
Mean ND			**264.00 ± 40.03 b**	**503.87 ± 33.60 b**	**77235 ± 41614 a**	**3308 ± 2353 a**	**6.67 ± 7.64 a**
Da	15/09/08	40	116.28 ± 19.55	217.16 ± 0.84	5458	1208	104
Db	07/10/08	40	152.99 ± 11.21	338.00 ± 3.06	17550	1178	54
Dc	11/11/08	40	136.46 ± 0.87	441.05 ± 0.41	29331	1694	45
Mean D			**135.24 ± 18.40 a**	**332.07 ± 100.24 ab**	**17446 ± 11936 a**	**1360 ± 290 a**	**67.67 ± 31.79 b**
PDa	15/09/08	50	88.93 ± 3.31	101.49 ± 10.1	13314	1303	30
PDb	07/10/08	50	83.13 ± 3.84	192.87 ± 2.54	40213	1180	45
PDc	11/11/08	50	112.89 ± 12.36	178.75 ± 0.38	23087	2784	57
Mean PD			**94.98 ± 15.80 a**	**157.70 ± 44.25 a**	**25538 ± 13615 a**	**1756 ± 893 a**	**44 ± 13.53 ab**

a, b: Values in the same column followed by the same letter are not statistically different (test Tukey, $p<0.05$).

Regarding to VFA content (Table 8.1), values were higher in initial samples (ND) due to organic matter hydrolization during waste transport and stock. In the anaerobic digestion and post-digestion processes VFA were transformed into biogas, resulting in a lower VFA concentration for digested (D) and post-digested (PD) samples. TKN and NH_3 concentrations were lower in non digested waste samples than in digested and post-digested materials due to the fact that nitrogen is scarcely consumed during the anaerobic digestion process while organic matter was degraded.

Biological stability of materials (Table 8.2), measured as OD_{20} and ABP, increases through the treatment process as reflected by the decreasing trend of the values of these parameters with process evolution A relationship was found between OD_{20} and some chemical parameters as VFA, %OM and TOC. These chemical parameters were indicative of material stability since lower values normally indicate stable samples (Adani et al., 2004). A linear regression coefficient (r^2) of 0.859 was obtained for OD_{20} vs. VFA content, r^2 of 0.852 for OD_{20} vs. %OM and r^2 of 0.822 in the case of OD_{20} vs. TOC content (Figure 8.5). However, a linear regression coefficient of 0.767 was obtained for OD_{20} vs. ABP indicating a lower correlation between the values of these two biological stability indicators. A low correlation between OD_{20} vs. ABP (r^2 of 0.697) was also found by Schievano et al. (2008) in studies developed at the same organic feedstock full-scale anaerobic digestion plant, this phenomenon may indicate that low values of OD20 present low sensibility with respect to the ABP method. In their study they analyze many samples and their results were similar to those found in this analysis. Ponsá et al. (2008) reported a good correlation (r^2=0.94) between the values obtained for the static respirometric index (expressed as g O_2/kg DM h) and the anaerobic biogasification potential (Nl biogas/kg DM) at 21 days when studying samples from a mechanical-biological treatment plant.

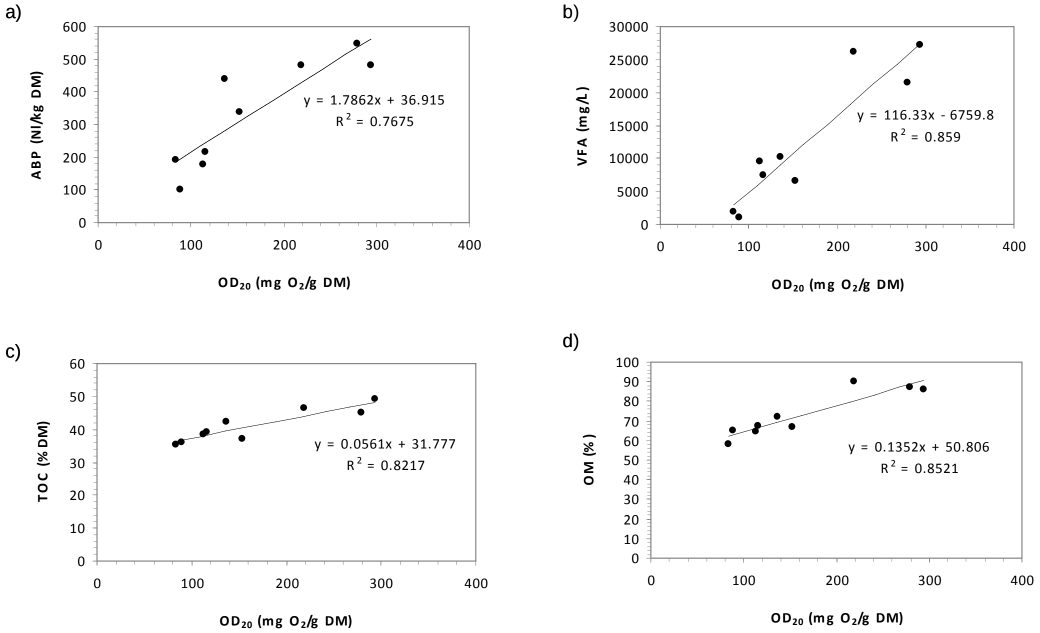

***Figure 8.5**. Linear regression between OD_{20} (mg O_2/g DM) and chemical and biochemical parameters determined on the 9 samples taken during this study. a) OD_{20} vs. ABP (Nl/kg DM), b) OD_{20} vs. VFA (mg/L), c) OD_{20} vs. TOC (%DM) and c) OD_{20} vs. OM (%).*

The increase of the biological stability seems that leaded, as average, to a strong reduction of the odour emitted (OU/m^2 h) by the digested samples with respect to the ingestate (Table 8.2), although no correlation was found between these parameters, and no statistical significance was observed. This was due to the high variability in the measures of the odours emitted by the initial material samples, which reflected the variability of the feeding mixtures that generally occurs at full-scale waste treatment plants (Schievano et al., 2009). However, this variability was also present in this plant within samples of digested materials as well as post digested materials. The odours increment in the final sample (PD) with respect to the digested sample (D) (Table 8.2), could be ascribed to the fact that lower microbiological activity (see ABP data) for PD, leaded to a limitation in the odour stripping from the biomass via biogas production, allowing its concentration in the waste.

8.5.2 Gaseous emissions

In addition to the physico-chemical and biological parameters stated above, gaseous emissions from the different materials were also analyzed to determine their content in total VOCs, ammonia and odour. The identification of the different groups of organic compounds present in gaseous emissions was performed both by gas chromatography-mass spectrometry (GC-MS) (associated to total VOCs analysis) and by using an electronic nose (EN).

Regarding ammonia emissions (ppmv) (Table 8.2) at each sample stage (ND, D and PD) these show that the higher average emission was found in D samples (67.67 ppmv), while the lowest emission was found in the initial undigested samples, ND (6.67 ppmv). The NH_3 emission behaviour was related to the pH of the samples (Table 8.1) and the NH_3 and TKN content, in this case the ammonia emissions increase as the organic nitrogen passes to ammonia when the pH changes from 4 to 8.

Values obtained for total VOCs determined by GC-MS are summarized in Table 8.2. Initial samples (ND) present some variability but values seem higher than those of D and PD samples. Although a numerical correlation can not be established, a tendency indicating that VOC emissions decrease after digestion process can be observed. This result agrees with VFA content (Table 8.1), indicating that highly hydrolyzed samples produce higher VOC emissions. Moreover VOCs contents appeared very similar (not statistically different), as average, for all samples considered, although a reduction during the process seems to occur (Table 8.2).

Dynamic olfactometry and total VOCs measurement, both represented quantitative measurements of molecules present in air samples, but they were not able to identify the kind of molecules present in the air directly responsible of odours. A detailed analysis of the type of VOCs present in the samples of different origin could help to this identification.

Table 8.1. *Volatile organic compounds (VOCs) emissions (ppbv) from the samples studied: non digestate (ND, 0 days of process); digested (D, after 40 days of process) and post-digested (PD, after 50 days of process) wastes. The mean percentage of each type of VOCs over the the total VOCs emissions is also presented.*

Type of VOCs	**ND (ppbv)**				**D (ppbv)**				**PD (ppbv)**			
	a	**b**	**c**	**Mean %**	**a**	**b**	**c**	**Mean %**	**a**	**b**	**c**	**Mean %**
Nitrogen compounds	0.00	0.00	3.42	0.04	0.00	7.11	0.66	0.19	0.00	3.19	0.00	0.06
Esters	224.31	310.29	355.26	11.29	0.00	12.73	1.66	0.35	39.55	6.90	60.12	2.02
Aliphatic hydrocarbons	2.07	30.23	20.43	0.67	0.48	13.65	14.02	0.69	14.42	9.44	22.55	0.88
Aromatic hydrocarbons	19.27	39.07	28.49	1.10	22.28	40.03	55.05	2.88	41.77	29.68	20.93	1.75
Alcohols	762.00	412.93	585.00	22.33	81.89	82.64	38.50	4.98	76.26	35.40	0.00	2.12
Ethers	78.70	0.17	0.00	1.00	4.19	6.36	1.72	0.30	1.30	1.72	4.26	0.14
Carbonyl compounds	558.05	171.38	19.03	9.50	685.38	515.01	423.63	39.80	680.96	463.60	671.64	34.47
Terpenes	603.73	708.29	4749.90	76.90	413.91	500.50	1150.79	50.61	449.33	630.18	1999.43	58.44
Sulphur compounds	0.00	0.00	79.60	1.01	0.00	0.00	8.08	0.20	0.00	0.00	4.80	0.09
Halogenates	0.08	0.00	0.00	0.00	0.00	0.00	0.33	0.01	0.00	0.00	1.05	0.02
Acids	0.00	0.00	164.75	2.09	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
TOTAL VOCs	**2248.21**	**1672.35**	**6005.87**	**100**	**1208.12**	**1178.03**	**1694.44**	**100**	**1303.60**	**1180.12**	**2784.77**	**100**

a (15/09/08); b (07/10/08); c (11/11/08).

Several works reported that N-compounds, S-compounds, volatile fatty acids, ketones, esters, terpenes and hydrocarbons were the most common compounds present in vegetables, fruit and garden (VFG) waste, municipal solid waste and composting plants as odour emission (Tsai et al., 2008; Mao et al., 2006; Defoer et al., 2002; Goldstein, 2002; Smet et al., 1999; Eitzer, 1995; Van Durme et al., 1992). Table 8.3 summarizes the emissions of the different groups of VOCs determined for the studied samples. Table 8.3 shows that aliphatic hydrocarbons, aromatic hydrocarbons, carbonyl compounds and terpenes (i.e. limonene, cymene) were present constantly in the emissions of all the sampled points in the plant (all process steps) while esters (i.e. ethyl acetate), alcohols (i.e. ethanol) and ethers (i.e. furans) decreased from indigestate (ND) to digestate (D) and post-digestate (PD) samples. VOCs present in the air of the ingestate consisted, especially, (average of 3 measures) of terpenes (61%), alcohols (18%) and esters (9%). Changes occurred during anaerobic digestion leaded to digested samples characterized by the still high presence of terpenes (51% and 58% for D and PD samples respectively), by the strongly reduction of both alcohols and esters and by the high presence of carbonyl compounds (40% and 34% for D and PD samples respectively).

Figure 8.6 reports the total VOCs finger-print analyzed by GC-MS spectroscopy and elaborated on a bi-dimensional PCA plot, highlighting that samples having similar biological stability (OD_{20} values in parenthesis in the figure) showed similar VOCs pattern. Again a good regression was found by using PLS methods for OD_{20} vs. GC-MS spectrometry data (compounds classes) (Table 8.3) ($R^2 = 0.95$; $R_{cv}^2 = 0.78$, $P<0.01$, organic compounds selected: ester, aromatic and alcohols). These compounds were selected because they show a high presence and correlation.

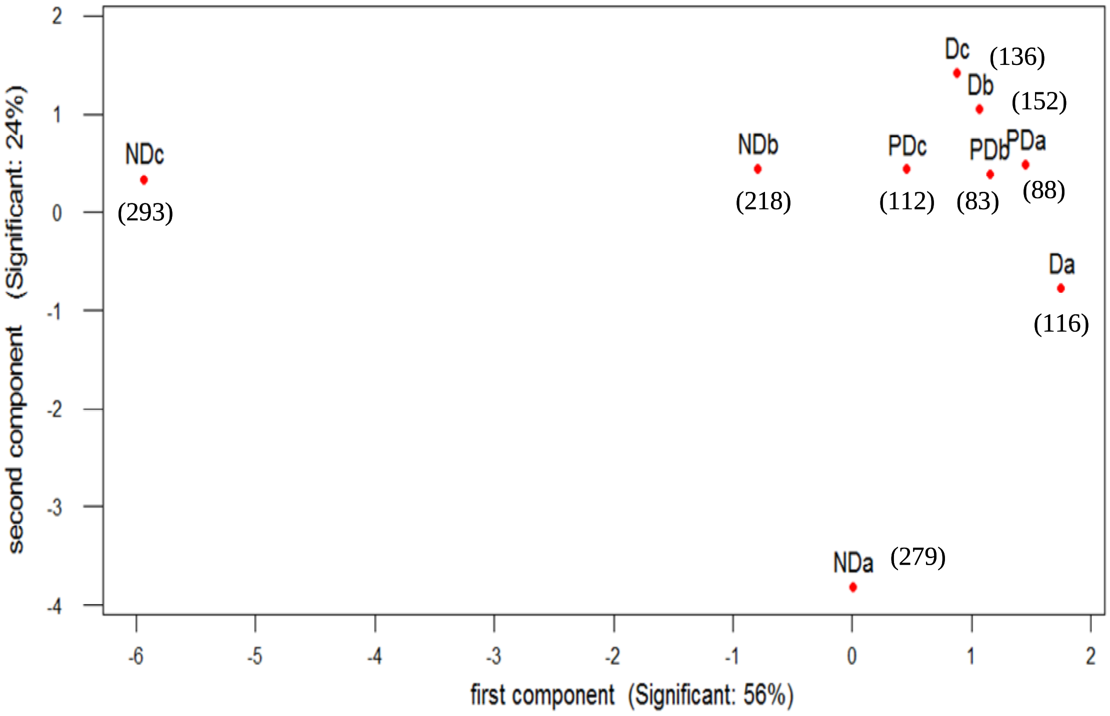

Figure 8.6. *PCA plot of total VOCs measured by GC-MS analysis (organic compounds classes from Table 8.3). PC1 in x axis and PC2 in y axis. Respirometric data (OD_{20} as mg O_2 /g DM) are included in parenthesis.*

In order to get a more in depth knowledge of the organic molecules composing waste air single molecules were elaborate from GC-MS spectra and quantified. Fresh wastes (ND) were particularly characterized by the presence of terpenes (Table 8.4): limonene and beta-pinene, both widespread in fruits, vegetables and pine species (Staley et. al., 2006) and so compose VOCs of OFMSW (Smet et. al., 1999). Terpenes have been reported to be stripped during the first stage of aerobic biological processes, decreasing its concentration (Pierucci et al., 2005). As the anaerobic conditions were far different form aerobic as no forced aeration was performed, terpenes were more or less conserved indicating, also, that no significant degradation of this compounds occurred, except for sample c (Table 8.3). On the other hand, post-digested samples showed the highest presence of p-cymene (Table 8.4), probably coming from the microbial transformation of D-Limonene (Termonia and Termonia, 1999). P-cymene has been reported to be a marker of the presence of old refusals kept under anaerobic condition with production of biogas (Davoli et. al., 2003).

The high presence of alcohols (ethanol, butanol and propanol) for fresh material was the consequence of the microbial-alcohols formation from waste substrate, during the period of the storage under nearly anaerobic conditions at low pH (Staley et. al., 2006). In the successive stable methanogenic phase, the oxidation of alcohols to ketones due to bacteria activity has been reported by means of the reduction of CO_2 to CH_4 (Widdel, 1986). The high presence of ketones, mainly, 2-butanone, (Table 8.4) and the reduction of alcohol concentration in D and PD samples, seems to confirm that fact.

Table 8.4. *Main compounds found in the air samples analyzed (GC-MS analysis).*

Organic Compounds	**ND (ppbv)**			**D (ppbv)**			**PD (ppbv)**		
	a	b	c	a	b	c	a	b	c
Ethanol	281	176	365	-	7	4	-	-	-
2-Butanol	208	105	131	75	63	17	57	24	12
1-Propanol	121	79	36	-	-	-	-	-	-
Ethyl acetate	55	130	158	-	-	-	-	-	-
D-Limonene	450	433	4389	178	323	920	10	26	778
p-Cymene	80	64	-	134	53	59	352	501	1083
Beta-pinene	78	80	26	-	-	-	-	-	-
Camphor	-	-	-	-	28	21	29	37	40
Acetic acid	86	43	-	-	-	-	-	-	-
Propanoic acid	4	85	-	-	-	-	-	-	-
2-Butanone	337	32	12	556	416	311	589	337	558
2 and 3-Pentanone	-	-	-	15	18	12	20	16	12
2-Heptanone	-	-	-	24	24	43	24	32	35

Collection date: a = 15/09/08; b = 07/10/08; c = 11/11/08.

Some authors as Davoli et al. (2003) and Smet et al. (1999) mentioned that, among the different VOCs detected in gaseous emissions from waste materials, limonene is a typical tracer of fresh waste, while p-cymene is characteristic of leachate and biogas samples. Figure 8.7 presents terpenes emissions of the studied samples (based on the values presented in Table 8.4), showing how when the material was fresh (ND) limonene has a correspondent emission ratio of 80% within the other terpenes, while in the final stabilization phase (PD) cymene represents almost all the terpenes emitted.

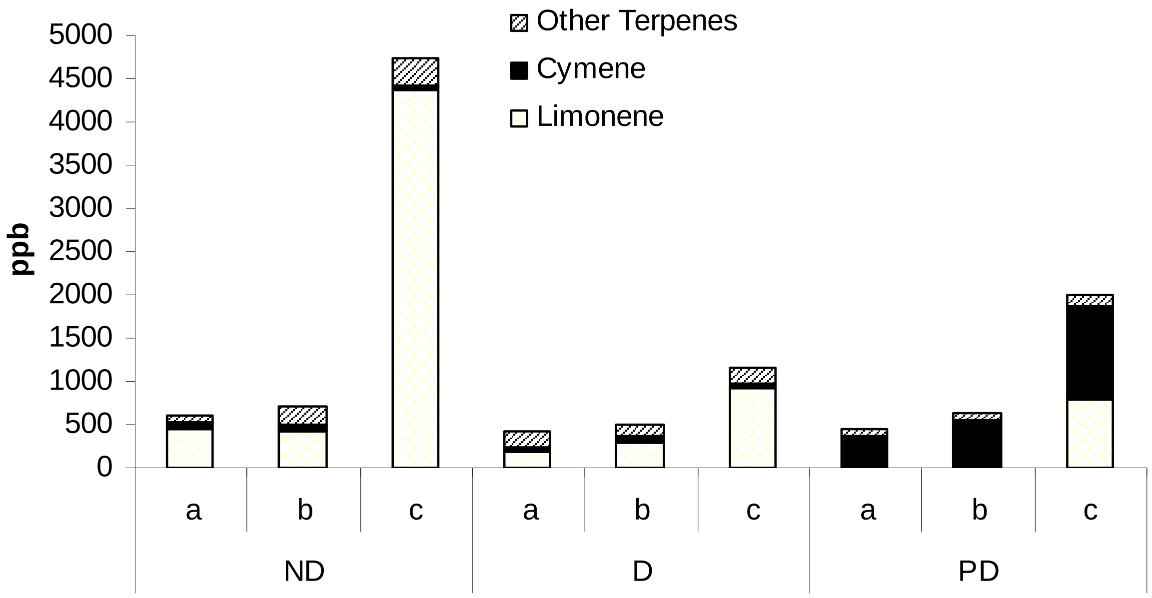

Figure 8.7. *Concentration of limonene, cymene and other terpenes in samples of non-digested (ND), digested (D) and post-digested (PD) waste taken during the three sampling campaigns performed (a: 15/09/08, b: 07/10/08 and c: 11/11/08).*

8.5.3 Electronic nose measurements

Electronic nose response patterns for the samples analyzed are presented in Figure 8.8. Graphs in Figure 8.8 show a decrease in the extension of the response of the different EN sensors with the increment in process time, which means (according to results shown before), an increase in material stability (Table 8.2). These results point to a relationship between EN response and material OD_{20} and ABP. Before the anaerobic digestion process begins (ND samples) aromatic compounds (S1 and S3), polar compounds and nitrogen oxides (S2), methane compounds (S6) and alcohols (S8) have a strong presence in EN response, as well as the presence of H_2 (S4) and CH_4 at high concentrations (S10) indicating the start of anaerobic conditions in the analyzed materials. After 40 days of anaerobic digestion (D samples) and after post-digestion (PD samples) the compounds sensors signals listed above were lower with S2 signal remaining the larger (Figure 8.8).

Figure 8.9 shows the final elaboration of EN data by using a PCA bi-plot graph in which two principal components were reported (PC1 = 66% and PC2 = 21%, in which % represent the total variance explained). PC1 and PC2 that together resumed the 87% of the total variance indicated that the odour finger-prints were similar when the biological stability of the samples was similar (OD_{20} values are included in parenthesis in the graph for each sample). This fact was confirmed by the good regression found by using PLS analysis, for OD_{20} vs. EN ($R^2 = 0.99$; $R_{cv}^2 = 0.98$, $P<0.01$, sensor selected S1 to S7).

From these data it could be concluded that odours reduction due to the acquirement of biological stability commented above, was accompanied by a change in the organic molecules composing the gas of the wastes studied.

EN

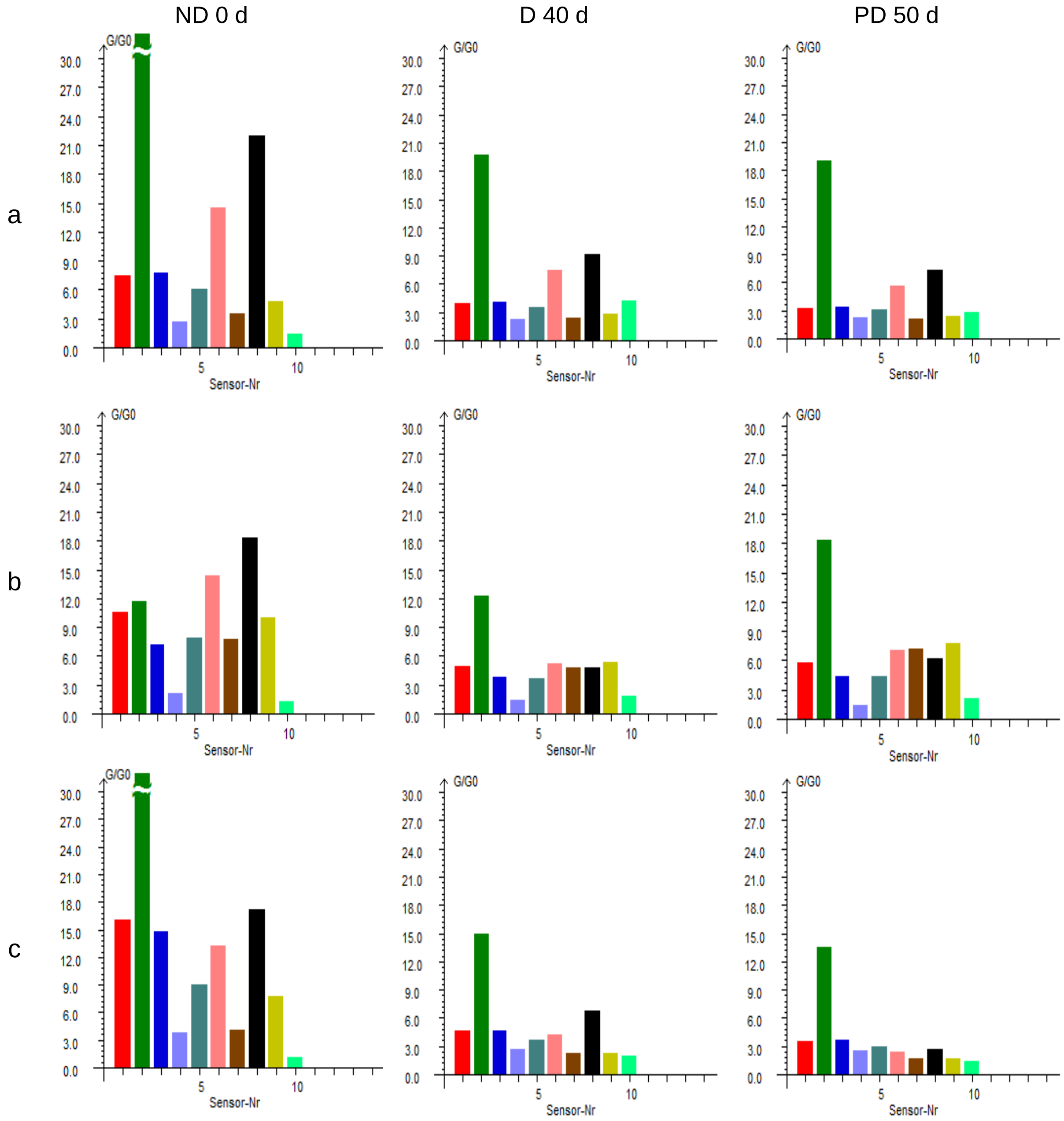

Figure 8.8. *EN signal registered in samples of indigestate (ND), digestate (D), and post-digestate (PD) coming from the three sampling campaigns (a: 15/09/08, b: 07/10/08 and c: 11/11/08). x axis represents EN sensor number and y axes relative sensor signal.*

Comparing the sensors response in EN (Figure 8.8) with VOCs analysis by GC-MS (Table 8.3) it can be observed that alcohols and esters (represented in S8 in EN response) present the same behaviour in both analytical techniques having a strong presence in ND samples and decreasing in D and PD samples.

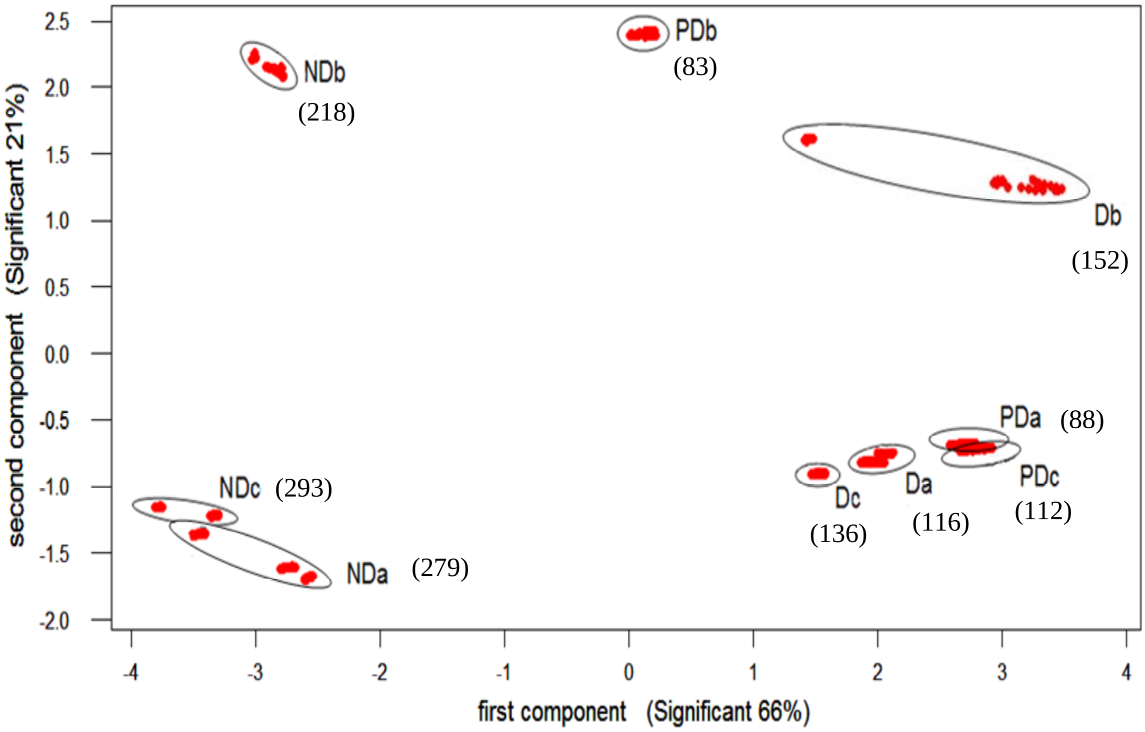

Figure 8.9. *PCA plot of odours measured by the electronic nose (x axis PC1 and y axis PC2). Respirometric data (OD_{20} as mg O_2/g DM) are included in parenthesis.*

8.5.4 Odour measurement by olfactometry

The presence of odorous compounds was also measured by olfactometry (log odour dilution) and the results obtained are summarized in Table 8.2. These results show, as commented early in this chapter, how odour emissions (OU/m^2 h) were high in fresh samples (ND) and decrease after 40 days of anaerobic digestion (D), but began to rise again in post-digested samples (PD). This fact may be due to the important presence of p-cymene (Figure 8.7) and ammonia (Table 8.2) in post-digested samples emissions. While ammonia emissions were similar in D and PD samples, emissions of cymene were clearly higher in PD than in D samples possibly affecting the odour panellists sensory response (Gralapp et al., 2000).

Therefore the synergic effect and the complexity of the organic molecules in the air, more that the single organic or inorganic compounds, was responsible of the odours impact. Both EN and GC-MS spectrometry data when successively elaborated by principal component analysis allowed to describe organic molecules all together. Therefore it can be assumed that PCA and GC-MS elaboration allows studying the synergic effect of the organic molecules. In effect when sensors data from EN analysis, and organic compounds classes from GC-MS analysis were considered together by PCA analyses, and results correlated with odour unit by using PLS analysis, very good regressions were found for EN (EN vs OU: $R^2 = 0.99$, $Rcv^2 = 0.95$; $P<0.01$, sensor selected S1 to S7) but much less for GC-MS data (GC-MS vs OU: $R^2 = 0.80$, $Rcv^2 = 0.40$; $P<0.01$, organic compounds selected: esters).

Therefore, EN and much less GC-MS were able, by multivariate statistical analysis, to describe the odours impact of the waste. Especially EN could replace olfactometry in the odour impact measurement, after a correct calibration (Defoer et al., 2002). The olfactometry approach, although it is the accepted method for the determination the odour concentration (CEN, 2003) in air, must be conducted in a controlled laboratory with a sufficient number of panellists that need be available to conduct the analysis, which is one of its main drawbacks. Moreover, panellist should smell odours including, sometimes, hazardous molecules.

It can be concluded that anaerobic digestion produces odours as a consequence of microbial activity under anaerobic conditions. Nevertheless, the acquirement of biological stability due to OM degradation greatly reduces the odours impact of the residual wastes (digestate) and the biological stability measurement, i.e. OD_{20} represents a first indication of the potential odour impact of the digestate.

The impacts of odours can be measured by the reference method (olfactometry approach) but does not give any indication about the origin of the odours, which depend of the synergic effect of the different organic molecules present in the waste gas. Nevertheless the approach of electronic nose and GC-MS spectroscopy, jointed with multivariate statistical analysis and appropriate calibration on the plant, could replace the odour unit measurement (EN), when given the qualitative reporting information about the nature of organic molecules in composing waste air (GC-MS).

8.5.4 Plant performance indicators

Plant performance indicators have been calculated by using the measured parameters discussed above and operational data provided by the plant manager.

8.5.4.1 Process indicators

Table 8.5 shows the process indicators obtained in this facility.

As it was stated above, all the electricity produced is sold to the public company. Refuse generation in this plant was practically negligible due to input feedstock mix quality.

Table 8.5. *Process indicators in the anaerobic digestion plant.*

Process indicators	
t digestate/t feedstock mix	0.83
t refuse/t feedstock mix	[a]
Nl biogas/t feedstock mix	145.1
kWh generated/t feedstock mix	231.5

[a] Negligible.

The biogas production per ton of feedstock mix treated was between the range (121-190 Nm^3/t OFMSW) stipulate by the Juniper guide in anaerobic digestion processes with the same characteristics (Juniper Consultancy Services, 2005).

8.5.4.2 Resources consumption indicators

All the electricity consumed in the plant was purchased from the electric company (Table 8.6) while diesel and water consumption due to the process activities were practically negligible. Tractors are rarely used in this plant, therefore the consumption of fuel was very low when compared to other studied plants.

Table 8.6. *Resources consumption indicators in anaerobic digestion plant.*

Resources consumption indicators	
kWh external consumption/t feedstock mix	18.5
L diesel/t feedstock mix	1.2E-03
m^3 total water/t feedstock mix	0*

* No water consumption is reported, digestate is recycled if it is required.

8.5.4.3 Emission indicators

The surface of the post-digester (storage tank) was identified as emission point, in order to identify process emissions coming from the feedstock treatment mix of the plant. As discussed previously, this is because it is an open area where digestate is stored and its gaseous emissions are not treated.

Table 8.7 shows the emission factors (NH_3, VOCs and CO_2) from this facility. Electricity impact was represented only by the amount purchased to the electric company (external).

Table 8.7. *Emission indicators in the anaerobic digestion plant.*

	Emission indicators (kg/t feedstock mix)		
	NH_3	**VOCs**	**CO_2**
Process	0.48	0.08	—
Electricity	1.64E-04	1.83E-02	10.96
Diesel	9.29E-11	2.14E-05	3.42E-03
Total	0.48	0.10	10.96

VOCs and ammonia data come from the emission samples obtained in the static chamber described previously. The values obtained (ppmv or ppbv) were converted into mg/m^2 h taking into account the air flow used and the emitting surface of the sample analyzed. These values were then extrapolated to the corresponding emissions on the post-digester tank that has an open (emitting) surface of 3000 m^2. Finally, the emissions were calculated as kg/d for each substance (ammonia and VOCs). The average of the three samples taken divided by the tons of waste treated in this plant corresponds to the value shown in Table 8.7.

As in the previous studies, the inputs and outputs (related to one ton of feedstock mix treated) of the plant were summarized in Table 8.8, where the values represent the performance indicators of the plant.

Table 8.8. *Inputs and outputs summary in the anaerobic digestion plant studied.*

Inputs	Raw materials	t feedstock mix/year	30000
	Resources	kWh/t feedstock mix (external)	18.5
		L diesel/t feedstock mix	1.2E-03
		m^3 water/t feedstock mix	0[a]
		Total energy/t feedstock mix	18.5
Outputs	Atmospheric emissions	kg CO_2 energy/t feedstock mix	10.96
		kg NH_3 /t feedstock mix	0.48
		kg VOC/t feedstock mix	0.10
	Product	t digestate/t feedstock mix	0.83
		kWh/t feedstock mix (produced)	231.5
		Nl biogas/t feedstock mix	145.1
	Refuse	t refuse/t feedstock mix	0[b]

[a] No water consumption, digestate is recycled if required.
[b] Negligible, refuse is recycled into the process.

8.5.5 Environmental impacts analysis

Based on the data presented above (experimentally determined or obtained from plant manager), the impacts of the plant were calculated from an LCA perspective. Impact categories considered were those described in Chapter 3. As in the previous chapter (Chapter 7), it should be highlighted that CO_2 savings, which could result from the hypothetical substitution of fossil fuel by biogas generated in this plant, have not been taken into consideration. Italian power generation model (Italy B250) was used to calculate impact potentials related to electricity consumption.

8.5.5.1 Functional unit

One ton of feedstock mix was used as the functional unit to analyze the environmental impacts in this study.

8.5.5.2 System boundaries

Process activities in the anaerobic digestion plant were taken as limits of the system. Transport, manufacture and use of the products (electricity, digestate) were excluded from this analysis.

8.5.5.3 Life cycle impact assessment

Figure 8.10 represent process, fuel and electricity contribution to each of the impact potentials considered. Table 8.9 reports the impact category results with the respective contribution percentages on each potential.

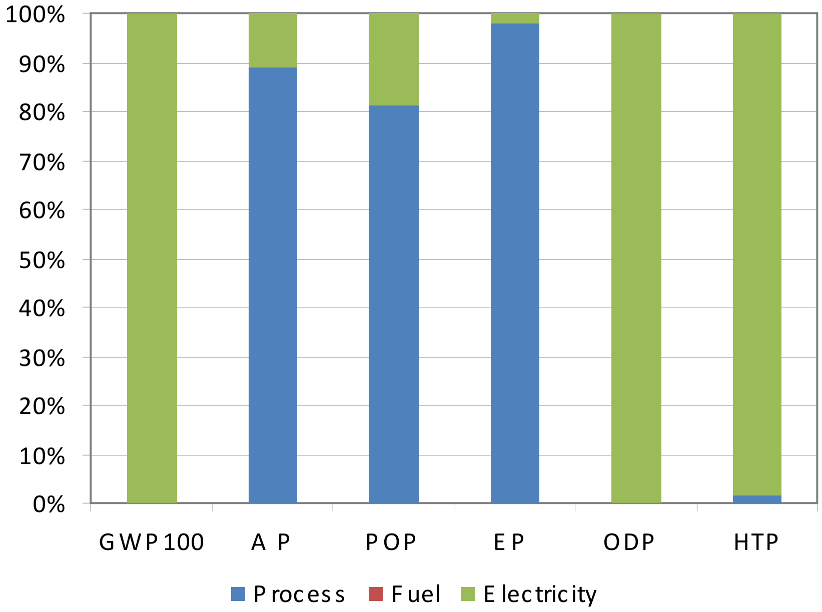

Figure 8.10. *Environmental analysis in the anaerobic digestion plant including contributions to each potential of process, fuel and electricity.*

Table 8.9. *Impact characterization results for the anaerobic digestion plant including the contribution of process, fuel and electricity to the total value of the potential (percentage contribution of each item to the total value of the plant in brackets).*

Impact potentials	**Process**	**Fuel**	**Electricity**	**Total**
Global warming	0	3.52E-03	11.44	11.44
(kg CO_2 eq/t feedstock mix)	(0.0%)	(0.03%)	(99.9%)	
Acidification	0.77	3.59E-05	0.11	0.88
(kg SO_2 eq/t feedstock mix)	(87.5%)	(0.0%)	(12.5%)	
Photochemical oxidation	0.03	8.88E-07	4.21E-03	3.4E-02
(kg C_2H_4 eq/t feedstock mix)	(88.2%)	(0.0%)	(11.8%)	
Eutrophication	0.17	8.09E-06	3.37E-03	0.17
(kg PO_4^{3-} eq/t feedstock mix)	(98%)	(0.0%)	(2%)	
Ozone layer depletion	0	3.23E-09	6.46E-06	6.46E-06
(kg CFC^{-11} eq/t feedstock mix)	(0.0%)	(0.05%)	(99.9%)	
Human toxicity	0.05	7.33E-04	3.07	3.12
(kg 1,4-DB eq/t feedstock mix)	(1.6%)	(0.02.%)	(98.4%)	

- **Global warming potential**

Figure 8.10 shows the contribution of electricity, fuel and process on GWP100. The main impact on this potential was caused by the CO_2 emitted due to electricity consumption representing almost 100% of GWP100 (11.44 kg CO_2 eq/t feedstock mix). Instead, diesel consumption in this plant was practically negligible.

- **Acidification potential**

The maximum impact on this potential comes from NH_3 emissions in the biological process (0.77 SO_2 eq/t feedstock mix). Diesel consumption was almost null while electricity impact was 0.11 SO_2 eq/t feedstock mix (12.5%).

- **Photochemical oxidation potential**

VOC emissions from the biological process represent the largest potential impact with 88.2% (0.03 C_2H_4 eq/t feedstock mix). The remaining potential was represented by electricity consumption 4.21E-03 C_2H_4 eq/t feedstock mix.

- **Eutrophication potential**

NH_3 emissions during the biological process have a major impact on this potential representing a 98% (0.17 kg PO_4^{-3} eq/t feedstock mix). Diesel consumption was minimal in this case (2%).

- **Ozone layer depletion potential**

ODP was almost entirely represented by electricity consumption based in the Italian electric sources production (99.9%) (Table 8.9). On the other hand, fuel consumption, represents only the 0.05% of this potential. Process emissions have had null impact on ODP.

- **Human toxicity potential**

HTP was due to diesel and electricity consumption. In this case, unlike other studies, HTP was almost entirely (3.07 kg 1,4-DB eq/t Feedstock mix) represented by electricity consumption (low fuel consumption) (Table 8.9).

The lack of gaseous emissions in the treatment system of the anaerobic treatment plant studied has several implications on the acidification, eutrophication and photochemical oxidation categories, which would be greatly mitigated or reduced with its implementation (i.e. results obtained in Chapters 6 and 7). To implement a gas treatment system, a cover and gas collection system would have to be placed over the post-digester surface. This action has economical implications in capital investment and higher operational costs due to the incremental energy requirements of the plant (as has been seen in the two scenarios presented in Chapter 6).

It is important to note that the study developed in this chapter was conducted in a different manner from those described in the preceding chapters because of the emplacement of the plant and the main objectives of the study, which compared odour measurement techniques and established a relationship between odour emissions and material stability. Additionally, some of the methods used have also been different, in order to join measurement techniques with the RICICLA research group.

Chapter 9

Integral analysis comparison of the biological treatment plants studied

The results presented for each of the studied plants will be compared in this chapter in order to withdraw conclusions and recommendations on environmental performance of the different treatment plant technologies. A new functional unit will also be introduced in order to take into account the performance of the biological treatment process and the impacts derived from the plants. This unit is based on the use of the respiration index of the different materials as an indicator of the plant efficiency.

However, it should be noted that the results presented in the previous chapters were only the reflection or photograph of the treatment plants operating at a given time and under the specified conditions. To understand the real behavior of the plants and their subsequent environmental impacts, further studies should be carried out in different seasons in to compare results to those calculated in this work.

9.1 Objectives

Previously states, the principal objective of this chapter is to analyze and compare the four OFMSW full-scale facilities studied in Chapter 5 to 7 (in Chapter 6 the two studies carried out will be considered as two different facilities in this chapter) with the addition of one full-scale composting plant with turned windrows technology. The waste stabilization degree achieved will be used as the functional unit from a LCA perspective. The anaerobic digestion facility studied in Chapter 8 has been excluded from this comparison because the methodology for its analysis was different and because of the differing characteristics of the final product obtained (digestate was considered in Chapter 8, where compost was considered in the rest of plants studied).

9.2 Summary of facilities characteristics

The five industrial plants have been numbered from Facility 1 (F1) to Facility 5 (F5):

- F1, composting tunnel technology, which corresponds to the first analysis developed on the plant studied in Chapter 6,
- F2, confined windrow composting technology, which corresponds to the facility studied in Chapter 5,
- F3, anaerobic digestion plus composting technology, which corresponds to the facility analyzed in Chapter 7,
- F4, turned windrows composting technology, which corresponds to a composting plant in the Baix Llobregat area (Catalonia, Spain) and
- F5, composting tunnels technology, which corresponds to the second study developed on the plant discussed in Chapter 6.

First it is important to highlight that the analysis of F4 was performed by Joan Colón (a pre-doctoral co-worker of the research group) from November to December 2008 as part of the same project where environmental impacts are determined for the OFMSW treatment plants developed in collaboration with the *Agència de Residus de Catalunya.*

Facility 4 (Figure 9.1) treats OFMSW coming from a street bin collection system. This facility uses turned piles as its technology treatment. In brief, the process in F4 is developed as follows: the OFMSW is mixed with wood chips in a 2:1 ratio by volume before the decomposition phase, which is conducted using turned windrows. The decomposition stage takes 8 to 9 weeks in which the piles are turned and watered (with leachate and rain-water) two times throughout the phase. Subsequently, the pile material is screened by means of a 80 mm trommel screen to separate inadequate materials that entered with the OFMSW (refuse). The remaining material is used to construct maturation piles that remained stagnant for 10 weeks during the curing stage, meaning that the piles are neither turned nor watered. Finally, at the end of the decomposition stage the materials are screened again, now using a 10 mm trommel screen and refined by means of a ballistic separator. All the refuse materials are compacted and sent to landfill. The compost obtained is mainly used in agriculture and forestry activities.

Since this is an open facility, F4 has no gas emission mitigation equipment. The piles are watered with the same leachate generated during the composting process, therefore, water consumption it is practically zero. Further, as it is stated above, materials are not watered during the maturation stage.

Using the methodology described in Chapter 4, input and output calculations were made by monitoring the different representative decomposition and maturation windrows during the study period (November to December 2008). The input and output data on an annual basis (2008) were: 3000 t/y OFMSW treated, 700 t/y wood chips and 600 t/y of compost produced. Electricity, diesel, water, NH_3 and VOC emissions and another input and output flows in the plant were calculated according to the methodology used in the other study plants and are presented further in this chapter.

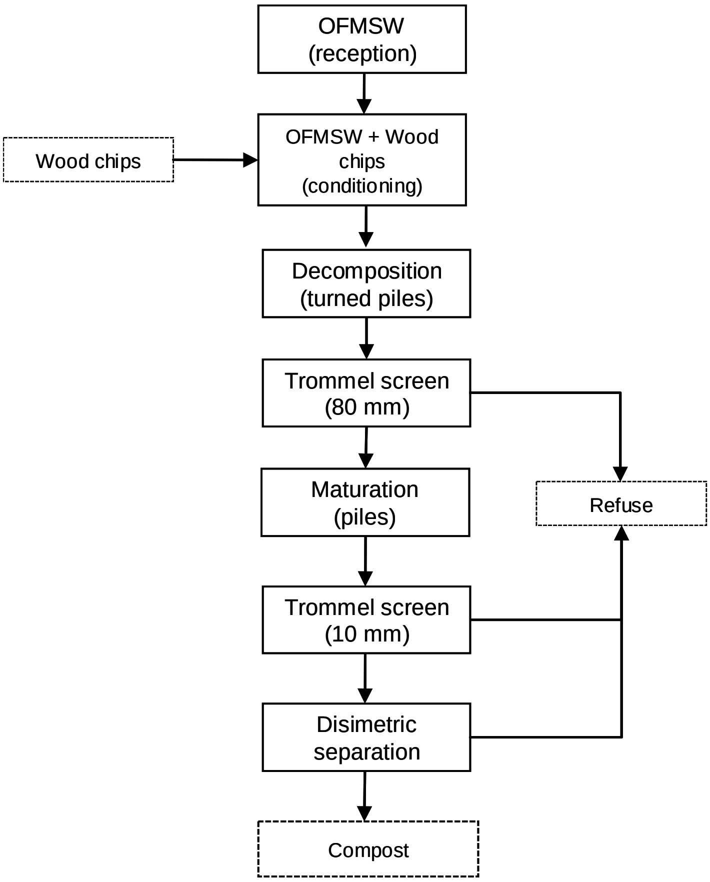

Figure 9.1.*Flowchart operations in F4 (turned windrows).*

Proceeding with the five facilities analysis, Table 9.1 recapitulates the main characteristics of each plant.

These plants (Table 9.1) can be classified in two principal categories. The F2 and F4 facilities are based on low technologies combined with simple operational processes. No pretreatment step is used in F2 and in F4, however, a screen process is present after decomposition. The final product of F2 and F4 facilities is processed using a trommel screen. Materials are aerated by periodic mechanical turnings in F4 and during the maturation phase in F2. Both plants are open to the atmosphere and have no treatments for gaseous emission.

Table 9.1. *Main characteristics of the studied industrial OFMSW treatment facilities.*

Facility	F1	F2	F3	F4	F5
Main biological process	Composting	Composting	Anaerobic digestion + composting	Composting	Composting
Pre-treatment	Trommel* screen (80 mm)	No	Balistic separator + Magnetic separator	Trommel* screen (80 mm)	Trommel* screen (80 mm)
Decomposition phase	In-vessel composting	Aerated confined windrow composting	Anaerobic digestion (solid phase, themophilic conditions) + in-vessel composting	Turned windrow composting	In-vessel composting
Curing phase	Aerated windrow	Turned windrow	Turned windrow	Turned windrow	Aerated windrow
Post-treatment	Trommel screen (10 mm) + ballistic separator	Trommel screen (10 mm)	Trommel screen (12 mm) + ballistic separator	Trommel screen (10 mm) + ballistic separator	Trommel screen (10 mm) + ballistic separator
Type of facility	Closed except maturation and storage zones	Completely open	Completely closed	Completely open	Completely closed
Exhaust gas treatment	Wet Scrubber + biofilter	Not present	Wet Scrubbers + biofilters	Not present	Wet Scrubbers + biofilters
Waste treated (t/year)	6082	91	17715	3000	7435
Improper materials (%)[a]	10	1	13	11	5
Refuse (%) sent to landfilling	25	0	41	26	13
Period studied	February-May 2007	July-November 2007	March-May 2008	November-December 2008	June-September 2009

*The screening process is performed after the decomposition phase.

[a] Values coming from input materials characterization.

F1, F3 and F5 plants based on more complex technologies. The decomposition phase is performed in aerated in-vessel systems using forced aeration in their composting phases. As can be seen in Table 9.1, wet scrubbers and biofilters are used in these facilities to treat their gaseous emissions.

The five plants presented in this chapter represent practically all of the options available in the Catalan market for OFMSW biological treatment.

9.3 Inventory analysis

Main input and output materials and energy flows for each facility are presented in Table 9.2. The functional unit used to calculate these values was 1 t of OFMSW processed in each facility.

Table 9.2. *Input and output flows in the studied OFMSW treatment plants. All parameters are referred to 1 t of OFMSW.*

		F1	**F2**	**F3**	**F4**	**F5**
Inputs	kWh electricity	95	65.5	46.2	9.3	214
	kWh electricity self generation	0	0	46.4	0	0
	L diesel	3.60	9.00	3.64	5.33	2.66
	Total kWh (electricity + diesel)	133.41	161.53	131.4	65.8	242.4
	Total m^3 water	0.33	0.02	0.12	0.00	0.56
Outputs	m^3 leachates	n/e	0.00	0.03	0.00	n/e
	m^3 biogas condensates	n/a	n/a	0.05	n/a	n/a
	kg $CO_{2\ \text{electricity + diesel}}$	60.40	60.23	-55.20*	20.10	121.02
	kg NH_3	3.9	2.00	0.23	8.63	3.04E-02
	kg VOCs	0.29	7.48	2.23	8.98	0.23
	t compost	0.10	0.52	0.03	0.20	0.14
	t refuse sent to landfilling	0.25	0.00	0.41	0.26	0.13
	Nm^3 biogas	n/a	n/a	98.80	n/a	n/a
	Electricity kWh	n/a	n/a	152.80	n/a	n/a

n/a: not applicable, n/e: not evaluated.
*CO_2 net balance (see point 9.3.3.3).

9.3.1 Energy consumption

Electricity consumption is highly dependent on the type of machinery used in each plant and the technologies applied. Electricity was mainly consumed for in-vessel and windrows aeration. In general, low technology plants that base their process on turned windrows (F4) will consume less electricity than more complex plants (F1, F3 and F5). However, this lower electricity consumption coincides with a higher use of diesel. Diesel consumption was mainly used in waste transportation and handling within the treatment plant, which strongly depends on the distance between process steps that the tractors must cover. This was particularly true in the case of F2, where the decomposition zone and the curing zone are separated (provisional configuration) by 200 m, amounting to the highest levels of diesel consumption. In addition, F2 plant uses diesel machinery in the mixing and post-treatment processes.

Total energy consumption (electricity and diesel) ranges between 65 and 242 kWh/t OFMSW, where the lowest value correspond to F4 (low technology process), and the highest value correspond to F5 (high technology process). For F5 the implementation of a gas emission mitigation system, where scrubber and biofilter systems, as well as automation of many of their operations including aeration and watering operation, contribute to high energy consumption. On the contrary, F2 is considered a low technology process, however, it can be seen in this study that the energy consumption in this plant was mainly due to the diesel consumption that was used to turn and transport the material in addition to forced aeration (note: diesel consumption in F2 was two times higher than F5).

Fricke et al. (2005) reported a energy consumption rate (including electricity, heat generation and diesel) between 55.6 and 119.4 kWh/t OFMSW in facilities with anaerobic treatment and aerobic post-treatment. Blengini (2008) reported a total energy consumption of 82.5 kWh/t OFMSW in an aerated windrow composting plant. The minimum value of energy consumption (64 kWh/t OFMSW) found in our study was within the range obtained by the authors mentioned above, but the maximum value (214 kWh/t OFMSW) calculated was clearly above this range.

In the case of the F3 facility, which includes an anaerobic process, energy is recovered from the biogas that was produced. During the studied period the plant produced 98.8 m^3 biogas/t of input OFMSW. This value is in the range reported by Fricke et al. (2005) (60-110 Nm^3 biogas/t OFMSW). Biogas is converted to energy in the same treatment plant yielding 199.2 kWh/t OFMSW. Part of this self-produced electricity is consumed in the plant (46.4 kWh/t OFMSW) and the rest is sold to an external electricity company. This means that 21% of the produced energy is consumed in the plant; this value is in the lower range of values (20-40%) reported by Braber (1995) for this type of facilities. As a result of these values (also reported in Table 9.3), the gross energy yield in F3 facility was 60.2 kWh/t OFMSW (Table 9.3).

9.3.2 Water consumption

Water consumption, like that of energy consumption, also depends on the technological level of the facility. Low technology facilities that use aerated or turned windrow technology with partial or no exhaust gas treatment will normally re-circulate leachates to the decomposition phase, reducing water consumption. In fact, water consumption in F2 and F4 plants was negligible. However, complex facilities, which should include exhaust gas wet treatment processes (F1, F3 and F5), have higher water requirements (F3, 0.12 m^3/t OFMSW). Water consumption in F1 and F5 (0.33 and 0.56 m^3/t OFMSW, respectively) should be considered as extreme values, since in this particular plant the scrubber used in the gas treatment process operates in an open loop mode (one biofilter in open loop in F1 was used, whereas in F5 two biofilters in open loop were used). Water consumption calculated in F5 was relatively lower than that in F1, when considering that three scrubbers and three biofilters are installed in F5, while only one scrubber and one biofilter system were operating in F1. Therefore, water consumption values, when rounded off, will range between 0 and 0.56 m^3/t OFMSW, where the lower values correspond to low technological processes. However, the low or null water consumption is not necessarily a sign that the process works well, may be this means that the materials were not watered during maturation stage as in the case of F2 and F4. As commented on in the previous chapters, Fricke et al. (2005) reported a range between 0.1 and 0.17 m^3/t OFMSW for OFMSW treatment plants with anaerobic treatment and aerobic post-treatment, while Blengini (2008) found a water consumption of 0.09 m^3/t OFMSW in an aerated windrow composting plant with gas treatment system (biofilter).

The implementation of gas treatment systems had an impact on both consumption of electricity and water and can be seen in the case of the tunnel plants (F1 and F5). In the case of electricity consumption, this increased over 50% in the second study after changes in technology were applied to the plant. For example water use increased 1.7-fold in the second study, while diesel consumption decreased 26% when compared to the first study. This increase in resources consumption was necessary to treat the total of the plant emissions, which, as discussed below, decreased significantly when compared to the first study.

9.3.3 Gaseous emissions

9.3.3.1 Ammonia

Ammonia emissions ranged between 3.04E-02 and 8.63 kg NH_3/t OFMSW, where the higher values are those corresponding to facilities that do not include exhaust gas treatment equipment. The case of the F1 plant was particularly interesting since the gas treatment process only treats part of the emissions. Decomposition, pre-treatment and post-treatment steps are performed in closed installations connected to gas treatment equipment (scrubber plus biofilter). Emissions measured at output flow of the biofilter reported negligible NH_3 concentrations during the study period. Thus emissions generated in the curing zone were the unique NH_3 emissions that were directly released to the atmosphere and accounted for 3.9 kg NH_3/t OFMSW. Ammonia emissions in F5, after the gas emissions systems were installed (scrubber and biofilter), were very low (3.04E-02 kg NH_3/t OFMSW) when compared with those calculated in F1. The range obtained for ammonia emissions is consistent with values reported by other authors. To get a better idea of this information, the values that were found in other publications are listed. Blengini (2008) estimated 0.6 kg NH_3/t bio-waste in an aerated windrow composting plant with gas treatment process. Gronauer et al. (1997) reported 0.67 kg NH_3/t OFMSW in aerated pile composting. Clemens and Cuhls (2003) reported ammonia emissions between 0.018 to 1.15 kg NH_3/t waste when analyzing the OFMSW composting process in different mechanical-biological treatment plants. Beck-Friis et al. (2001) found a value of 2.12 kg NH_3/t waste in their study on ammonia emissions during OFMSW composting in 200 L aerated reactors. Finally, Elkind and Kirchmann (2000) reported an ammonia emission of 9.6 kg NH_3/t waste (representing a 70% of the initial nitrogen present in the waste).

9.3.3.2 Volatile organic compounds

Process VOC emissions ranged between 0.23 and 8.98 kg VOCs/t OFMSW, where for ammonia, the higher values are those corresponding to facilities that do not include gas treatment systems. As can been seen in the VOC emissions obtained from the facilities F1, F3 and F5, the results show how the gas treatment systems have low depuration when compared to the ammonia removal, since in some cases the NH_3 depuration was almost total. This is congruent with the findings of Pagans et al. (2006) whose study of the composting process includes the determination of NH_3 and VOC emissions developed at laboratory scale. In this study, it was observed that a certain amount of VOCs (50 mg C-VOCs/m^3) due to system metabolism were emitted by the biofiltration systems. Likewise, the performance of these systems depends of their maintenance and the biofiltrate material used, along with the input load. This can be seen in Chapter 7 where the biofilter performance increased after maintenance adjustments (i.e. VOC emissions in biofilter 2 were reduced after biofilter material restructuration from 27 kg C-VOC/d to 3.8 kg C-VOC/d). On the other hand, F3 emits more VOCs than F1, even though F1 is partially open and the emissions from the maturation phase were emitted directly to the atmosphere without any treatment. In the case of F3, the biofilter performance was inefficient due to poor maintenance (equally, F3 is a larger plant with more inflow material). Moreover, comparing the VOC emissions obtained in F1 and F5, reminding that in F1 the tunnels plant partially closed and in F5 the tunnels plant is completely closed, it can be seen that installation of gas treatment systems had an important impact on the mitigation of VOC emissions, representing a 41% reduction after the implementation.

As commented on in the Introduction (Chapter 1), to the best of our knowledge there few literature reporting data on total VOC emissions at full-scale facilities for the treatment of biowaste. Smet et al. (1999) and Baky and Eriksson (2003) reported VOC emission factors in pilot-scale composting experiments of 0.59 and 1.7 kg VOCs/t OFMSW, respectively. Diggelman (2003) reported 4.3 kg VOCs/t OFMSW from bibliographic data.

9.3.3.3 Carbon dioxide

In this study CO_2 emissions were related to energy consumption (electricity and diesel) because, as stated in the Introduction (Chapter 1) of this dissertation, CO_2 emissions that come from biological treatment processes are not accounted for as an environmental impact because it comes from biogenic source (it belongs to the short CO_2 cycle). The biogenic carbon is sequestered from the atmosphere by biomass and reported as carbon stock change in agriculture, forestry and other land uses sector. However, it does not contribute to additional emissions of CO_2 once it is released back into the atmosphere and should therefore not be considered in the carbon footprint estimations (IPCC, 2006). With consideration to this, 60 kg CO_2/t OFMSW were emitted in F1 and F2 facilities in which aerated composting is performed (in case of F1 decomposition and maturation phases are forced aerated while in F2 just the decomposition phase is aerated). Plant F4 emitted 20 kg CO_2/t OFMSW, mainly due to diesel consumption. F5 emitted more CO_2 (121.02 kg CO_2/t OFMSW) almost entirely due to electricity consumption. It is clear how the facility dimensions, the infrastructure and the systems automation play a key role when considering energy consumption, which is reflected in major or minor CO_2 emissions and has a major or minor impact on the environmental indicator such as the global warming potential.

In the case of the F3 facility, which includes an anaerobic digestion process, the electricity produced from the generated biogas can be considered as a negative contribution to the CO_2 releases, since CO_2 generation from non-biogenic sources emitted during energy production is avoided (IPCC, 2006). According to this, Table 9.3 shows the CO_2 emission balance from electricity consumption and generation. 56.3 kg CO_2/t OFMSW were avoided in F3 facility due to the electricity produced from the biogas.

Table 9.3. *Electricity and CO_2 from electricity balance in the Facility 3 (F3).*

	kWh/t OFMSW	**kg CO_2/t OFMSW**
Electricity consumption	46.2	24.4
Self generated electricity from biogas and consumed in the facility	46.4	0
Self generated electricity from biogas and sold to an electricity distribution company	-152.8	-80.7
Net balance	**-60.2**	**-56.3**

To obtain the total CO_2 emissions coming from the energy consumption of this facility the diesel emissions (1.1 kg CO_2/t OFMSW) must be added to the value calculated above (-56.3 kg CO_2/t OFMSW). In this case, the total CO_2 emission obtained for energy consumption corresponds to 55.20 kg CO_2/t OFMSW (Table 9.2)

On the other hand, some authors such as Rabl et al. (2007) consider that all CO_2 emissions, regardless of their origin, should be considered in the emissions inventory in order to obtain an objective assessment of the impacts coming from OFMSW treatment processes, such as landfilling, composting, anaerobic digestion or incineration. Even if the biogenic CO_2 is not considered in impacts determination, CO_2 emissions from the composting process can be calculated from the data obtained on OFMSW composting assays developed at laboratory scale by the research group. These analyses are based on respiration assays where the biological activity and CO_2 production are determined on OFMSW samples over a long period of time until their biological activity is significantly demised. Mean CO_2 emissions measured during these assays were 344 kg CO_2/t OFMSW (assuming 60% of humidity). Therefore, taking into account the amount of waste treated in the studied facilities CO_2 emissions from the composting process were between 3.1E04 to 6.1E06 kg CO_2/t OFMSW. Then, if CO_2 has to be considered in the environmental impacts, it will have a high repercussion on the emissions inventory. The contribution of the emissions of the biodegradation process to the impact potentials (i.e. global warming) will then be clearly higher than that of the energy consumption in all the studied cases.

9.3.4 Compost production

Composting yield of a treatment plant depends mainly on three parameters: (i) the real content of organic matter in each OFMSW ton (in front of the improper materials content), (ii) the efficiency of pre and post-treatment steps used to separate improper materials and bulking agent from biowaste, and finally (iii) the type of biological process used (aerobic or anaerobic/aerobic steps). Composting yield (t compost/t OFMSW) ranges between 0.03 and 0.52 in the studied plants, being the mean yield 0.21. Blengini (2008) reported a yield of 0.28 t compost/t OFMSW for an aerated windrow composting plant. The lower value corresponds to F3 in which the OFMSW is treated by two consecutively biological processes (anaerobic and aerobic) yielding in a lower quantity of compost due to organic matter transformation into biogas. The higher value corresponds to F2, a facility with a high quality OFMSW and in which pre-treatment processes were not performed.

9.4 Environmental burdens and process performance

The main objective of the OFMSW treatment plants in addition to a proper waste treatment is to stabilize the waste to a level that allows its use as compost or, in general, as organic amendment. Stated in the Introduction of this dissertation, the biological stability is defined as the measure of the degree of decomposition of biodegradable organic matter that is contained in a matrix (Lasaridi and Stentiford, 1998). The biological stability degree of waste materials can be directly measured by means of respiration indices (Adani et al., 2006; Gea et al., 2004). 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 minimized and where the Dynamic Respiration Index (DRI) is below 1.0 mg O_2/g OM h. Consequently the performance of OFMSW treatment plants can calculate direct measure of its efficiency through finding the difference between the stability degrees for input and output materials.

When considering this information, some questions arise, including: Have we compared the studied facilities under the same conditions and restrictions? Did all the studied plants produce a stabilized material? When looking at stability, were the input materials equivalent? Was the emphasis to produce the same type of stabilized material for each plant?

When answering these questions, a fair comparison of the study plants can be drawn from both the efficiency and the environmental point of view. In able to make this comparison, each facility has been analyzed using the stabilization degree achieved in each of the treatment processes and by its environmental impact. Table 9.4 presents DRI values of the input and output materials for each studied plant as well as values of this index within the main process steps. As it can be seen in the previous chapters, it is important to highlight that the stabilization analysis in F1 and F2 were made using the static respiration methods, therefore it was necessary to obtain an equivalence of SRI to DRI. Consequently, it was recurred to the experience of the research group in this field, which was showed that samples with low activity (i.e. final compost) had SRI and DRI values that were practically equal, whereas these same values for samples with high activity differ (i.e. OFMSW input) (Barrena et al., 2009). In reference to DRI input value, it should be supposed that the OFMSW should report similar DRI values in all plants. However, the values obtained for OFMSW input in F3 (4.95), F4 (3.47), and F5 (6.07) showed great differences. These differences are probably due to the heterogeneity of the samples and that a unique sampling was performed in each plant. It has to be thought that if more samples were analyzed the mean values obtained in each plant would be closer to the mean values obtained in the other facilities. According to this, the DRI value used for the input OFMSW in the present analysis was the average of the DRI values obtained in F3, F4 and F5 for the input OFMSW. The DRI value considered is then 4.83 mg O_2/ g OM h.

Table 9.4. *Respiration Index at different process steps for each studied facility (DRI expressed as mg O_2/g OM h).*

	F1	F2	F3	F4	F5
OFMSW	4.83	4.83	4.83	4.83	4.83
Anaerobic digestion output	n/a	n/a	1.59	n/a	n/a
Decomposition output	n/e	1.30	n/e	2.96	n/e
Compost	1.03	0.70	0.75	2.7	2.08
DRI reduction (units)	3.80	4.13	4.08	2.13	2.75
DRI reduction (%)	78.7	85.5	84.5	44.1	56.9

n/a: not applicable. n/e: not evaluated.

According to the legislation draft value stated above (DRI below 1.0 mg O_2/g OM h is considered as stabilized material), compost from F4 and F5 facilities (2.7 and 2.08 mg O_2/g OM h respectively) does not fulfill the stability criteria. In Facility 4 only 2.13 units of RI were reduced during the whole composting process deriving in a 44.1% stabilization yield. The yield of F1, F2 and F3 are over 75%.

NH_3, VOCs or CO_2 emission factors or total energy consumption could be normalized if they are expressed referred to DRI reduction achieved in each plant. This new ratio reports the environmental impact and the energy consumption produced to stabilize the waste in one unit of DRI. This ratio has been named Respiration Index Efficiency (RIE). The RIE could be used for the comparison of different plants including its performance, even when different technologies are used. Table 9.5 shows the Respiration Index Efficiency (RIE) (i.e. amount of contaminant emitted or energy required per ton of OFMSW and RI reduction) for significant factors that characterize a waste treatment plant from an environmental point of view.

Table 9.5. *Respiration Index Efficiency (RIE) for energy consumption, ammonia, VOCs and CO_2 emissions.*

RIE	Units	F1	F2	F3	F4	F5
Total energy consumption	(kWh/t OFMSW)/ (g O_2/kg OM h)	35.11	39.11	32.21	30.89	88.15
Ammonia emission	(kg/t OFMSW)/ (g O_2/kg OM h)	1.03	0.48	0.056	4.05	0.01
VOCs emission		0.076	1.81	0.55	4.22	0.084
CO_2 emission		15.89	14.58	-13.53	9.44	44.01

When comparing the values in Table 9.5 with those presented in Table 9.2 it can be seen that F5 was still the facility with the greatest impact on energy consumption and CO_2 emission due to the high-energy consumption and the low DRI reduction obtained in this plant compared with other facilities. Moreover, F4 had the lowest electricity consumption by reduction of DRI, due to low consumption of energy in the facility. When looking at the F1 to F3 calculated values (NH_3 and VOC emissions) and comparing Table 9.2 the results are very similar, i.e., F4 has the highest emissions of both pollutants due to the lack of a gas mitigation systems and F5 presented low emission due to gas emission treatments. The use of RIE permits the plant comparison despite the use of different technologies, size or even different input wastes and can be proposed as a tool to help policy makers and stakeholders for decision making when selecting OFMSW treatment technologies.

However, the reduction of DRI achieved in each plant does not completely reflect its real performance. A facility can consume low energy in its operation and therefore present low CO_2 emissions for example, but its final product (compost) may have a low quality. In order to get the real performance of the facilities studied, the Quality and Respiration Efficiency Index (QRIE) was calculated. It is obtained by multiplying the RIE (Table 9.5) by the value of DRI found in the final compost (Table 9.4).

Table 9.6. *Quality and Respiration Index Efficiency (QRIE) for energy consumption, ammonia, VOCs and CO_2 emissions.*

QRIE	Units	F1	F2	F3	F4	F5
Total energy consumption	(kWh/t OFMSW)	36.15	27.37	24.15	83.43	183.33
Ammonia emission	(kg/t OFMSW)	1.06	0.34	0.04	10.94	0.02
VOCs emission		0.078	1.27	0.41	11.39	0.17
CO_2 emission		16.37	10.21	-10.15	20.11	91.54

As can be seen in Table 9.6, the environmental performance of F4 and F5 are clearly affected when QRIE is used. With the lowest energy consumption, F4 consumed 65.8 kWh/t OFMSW, which was attributed to the low technology process used in the plant. When the QRIE is applied to energy consumption, however, the F4 facility becomes the plants with the second highest energy consumption per unit of reduced DRI due to the low stabilization efficiency commented above as well as F5.

When applying the QRIE, the effect on CO_2 emissions was the same as those described for energy (given that CO_2 emissions are related to energy consumption) and no variations on the impact order of the different facilities were found, either for NH_3 emissions nor for VOC emissions.

QRIE parameter includes the performance of the plant, or, its the operation of the plant during the studied period. A plant that obtains a quality compost (DRI around 1.0 g O_2/kg OM h) should have similar RIE and QRIE values. Then, using the QRIE it is possible to effectively quantify an abstract concept as the real plant operation.

On the other hand, it is noteworthy that the functional units proposed (RIE and QRIE) can also be useful for analyzing other systems such as wastewater treatment, by using the equivalent unit for these facilities, in this case the biochemical oxygen demand (BOD) reduced (Cadena et al., 2009).

9.5 Life cycle impact assessment and stabilization degree

The use of RIE and QRIE to compare the different plants can be also extended to the LCA study. In this dissertation QRIE has been used. Then the functional unit proposed in this study to improve the representativeness of LCA studies on OFMSW biological treatment plants is the reduction of 1 DRI unit in 1 ton of OFMSW taking into consideration the quality (DRI) of the compost obtained.

In the previous chapters the impact potentials used in LCA studies performed have been recalculated using this new functional unit and are presented in Figure 9.2. Gaseous emissions (CO_2, NH_3 and VOCs) from electricity and fuel consumption and organic matter degradation during composting were used to calculate the different impact potentials. As occurred with process performance indicators, relative values of potentials will also be affected by the lower DRI reductions and compost quality reached in F4 anf F5.

- **Global warming potential**

As stated above, CO_2 from biogenic sources has not been considered to calculate the global warming potential. GWP100 values ranged between 6.73 and 97 kg CO_2 eq/t OFMSW. The contribution to this potential from each plant was directly related to energy consumption. F5 presented a high impact in this category due to its high electricity consumption (Figure 9.2).

CH_4 and N_2O are two gaseous compounds that also have an important contribution to GWP100 and can be emitted during composting, CH_4 is 21 and N_2O is 296 times more effective at trapping heat in the atmosphere when compared to CO_2 over a 100 years time period (IPCC, 2001).

The presence of these two compounds in gaseous emissions from the studied plants have not been determined in this dissertation but a theoretical approach can be made: IPCC (2006) reported an emission of 4 kg CH_4/t waste treated and 0.3 kg N_2O/t waste treated. If F4 (turned windrows, open-to-atmosphere facility) is considered and taking into account its treatment capacity (3000 t OFMSW/y), the contribution of the biological process to the GWP100 will be of 5.18E05 kg CO_2 eq/t OFMSW, making virtually zero (20.83 kg CO_2 eq/t OFMSW) the contribution of energy consumption to the value of this impact potential. These values also highlight that the consideration of these gases is necessary and essential to obtain an objective and representative assessment of these environmental potentials. On the other hand, if this potential is calculated by considering the QRIE in this installation, the result would be even greater (6.56E05 kg CO_2 eq/t OFMSW) because the index reduction was very low.

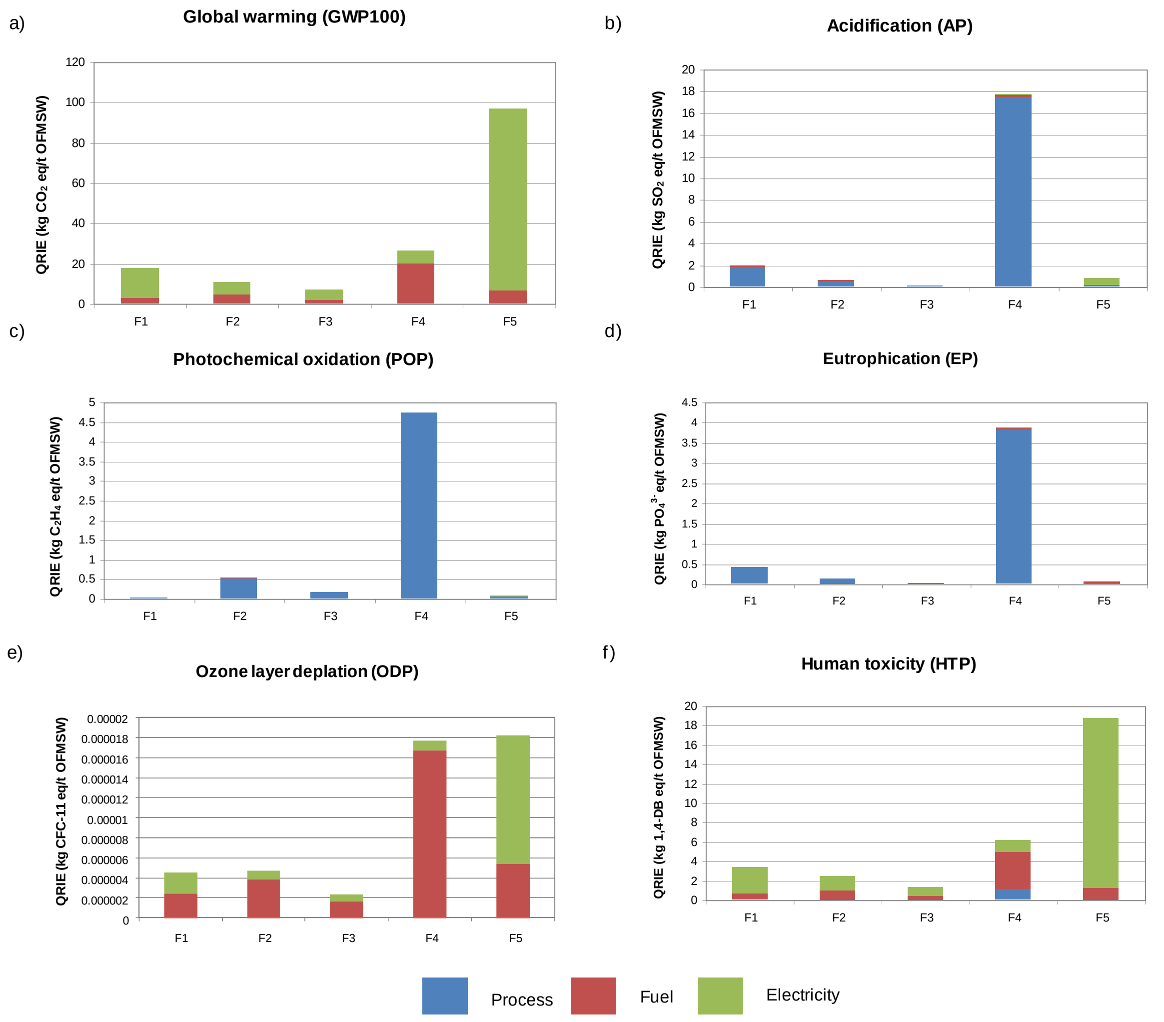

Figure 9.2. *Environmental burdens associated to each studied facility. a) global warming potential, b) acidification potential, c) photochemical oxidation potential, d) eutrophication potential, e) ozone layer depletion potential and f) human toxicity potential are represented.*

- **Acidification potential**

Contribution to AP was mainly attributable to the ammonia emissions that occurred during the process, except in the case of F5. AP ranged between 0.12 and 17.7 kg SO_2 eq/t OFMSW. The lower values were related to facilities with gas treatment processes reporting a low effect on AP; meanwhile AP values were significantly higher when no gas treatment was reported.

In the case of AP the presence of gas treatment equipment is much more relevant than the quality of the final compost, since F5 facility produced a non-estabilized compost and in spite of that its impact on AP is very low.

- **Photochemical oxidation potential**

Process VOC emissions mainly contribute to POP, which ranged between 2.8E-02 and 4.74 kg C_2H_4 eq/t OFMSW. Electricity and fuel consumption suppose a low contribution to this impact category. Lower values were obtained in the facilities with gas treatment equipment.

Again in this case the final quality of the compost is hidden by the presence of a gas treatment system.

- **Eutrophication potential**

In the same way as AP, EP values were mainly due to ammonia emissions. EP ranged between 1.31E-02 and 3.88 kg PO_4^{3-} eq/t OFMSW. F4 facility reported the higher value since it was the facility with the lower DRI reduction and without gas treatment. EP as well as AP and POP were highly dependent of process emissions (VOCs and NH_3) (except in the case of F5 where the value was very low and mainly attributable to the presence of a gas emission mitigation systems).

- **Ozone layer depletion**

Contribution to ODP was related to energy consumption; however, in this impact category contribution of fuel consumption was higher than electricity except in the case of F5. ODP ranged between 2.35E-06 and 1.82E-05 kg CFC^{-11} eq/t OFMSW.

- **Human toxicity potential**

Finally, HTP will be related to ammonia process emissions and to total energy consumption taking into account that fuel contribution to HTP was lower than electricity. HTP ranged between 1.68 and 6.33 kg 1.4-DB eq/DRI reduction t OFMSW.

F4 presents the highest values for three (almost four) out of the six potentials evaluated. However, this is contradictory to what can be deduced from the values obtained when using a conventional functional unit such as the treatment of 1 t of OFMSW. Since the purpose of the organic waste treatment plants is to valorize the waste, it should be thought that the valorization process should be performed producing the less environmental impact and, at the same time, obtaining the better final product (stabilized compost). The use of respiration indices (in the form of RIE or QRIE) appears as a suitable technique to accomplish this purpose. Respiration indices appear to be a suitable tool in order to accomplish this purpose.

9.6 Final remarks

In summary, to determine the environmental performance and compare biological waste treatment plants, it is necessary to use a functional unit that allows this comparison under the same efficiency conditions. Hence, in this chapter it has been proposed and demonstrated that both proposed efficiency index (RIE and QRIE) are effective functional units that accomplish this goal. These functional units have been used to compare OFMSW biological treatment facilities using different technologies, including anaerobic digestion and composting. This approach will be very useful for policy makers and stakeholders when selecting new OFMSW treatment technologies.

As stated in Chapter 6, nowadays there is no impact potential that incorporates the impact of malodors, because odors represent local impacts, while the potential impact considered in the LCA studies correspond to global impacts. In recent years, the community of people working around LCA (Meneses et al., 2005; Ebreo et al., 1999; Berger, 1997; Taylor and Todd, 1995), have begun to discuss how to incorporate the social impacts (social indicators). These indicators identify social aspects such as: social acceptability, social equity (an equitable distribution of benefits and harms of the treatment system), and social function (benefits obtained from the installation). Within these local social indicators the impacts caused by malodors i.e. complaints from neighbors regarding to the installation can be studied. The local community plays an important role in the approval or disapproval of a waste treatment facility and has a great affect on the continuance, prohibition or shut down of plant operations or whether they need to implement gas treatment systems, such as those implemented in F5. At this moment we cannot ascertain or demonstrated if the neighbor's complaints have decreased after the implementation of gas treatment systems in F5 or on the contrary the situation was the same. To know the actual situation about neighbor complains will be very useful since it should justify or not the performed investment. Even more the odors emission reduction from the installation has represented a global impact on GWP100 and HTP for example, mainly due to greater energy consumption.

On the other hand, considering the results obtained on each of the studied plants in this dissertation we may venture to consider, even while knowing that the results were part of a reality for a certain period, which type of technology provides the greatest benefits from the environmental standpoint. However, when considering the greatest benefits the impacts of each system and the plant efficiency as it related to the stability of its final product, needs to be taken into account, among other factors. With each indicator evaluated, it is true that there is no ideal technology that satisfies all of the features, dependant on the characteristics of the material input at each region. The quality of input material plays a key role in these biological processes and depends on the type of collection system. Material was collected door-to-door (F2) showed improved quality (fewer inappropriate materials) in relation to the selective collection, this demonstrates the eminent need for improved collection systems, since this would optimize the biological systems performance and maximize organic matter utilization. An optimized operating protocol is critical in plants in order to maintain efficiency and control of biological processes. Having control of physico-chemical and biological parameters aid the smooth running of the process. Also the gas treatment systems play an important role in reducing the impacts from the process, but also, as it can be seen throughout this thesis, if mitigation systems are not kept under control the impact reduction could be negligible. Moreover, energy consumption on these installations, both in electricity and fuel, have a heavy weight on the environmental impacts, as has been observed when implementing mitigation measures on a large scale where high consumption of energy resources is derived from gas or the automation of the entire system.

Chapter 10

Final conclusions

Several conclusions can be made from the studies developed in this dissertation.

The implementation of the methodology:

- Has aided in determining ammonia and total VOC emissions in OFMSW biological treatment plant.
- Can be successfully applied to full-scale biological treatment plants (composting and anaerobic digestion) in order to identify potential environmental impacts and performance of the studied plants.
- Can be useful to compare the environmental performance indicators of biological treatment technologies as they relate to organic wastes. However, when comparing different facilities it should be taken into account that they must have similar objectives regarding the quality and stability required for the final product (compost).
- Can be an effective tool when combined with the LCA methodology for analyzing environmental burdens of full-scale biological plants.
- May contribute to enhance LCA in its utility in the waste treatment field with the data obtained from these types of studies
- The performance of the biological treatment process should be considered when the environmental impact is determined. Two new functional units have been proposed based on material biological activity determination. The Respiration Index Efficiency (RIE) reflects resources consumption and process emissions needed or generated in the reduction of one unit of the Dynamic Respiration Index (DRI) value. Further on the Quality and Respiration Index Efficiency (QRIE) takes into consideration the quality of the final compost obtained determined by means of its DRI value. These new functional

units can help in presenting the emissions of a biological treatment plant on a more realistic basis since, for instance, low emissions can be incorrectly interpreted as low impacts if they are due to an inefficient biological process. Also emissions associated to energy consumption, energy and water needs and rejected waste production were determined for the studied plant and related to both functional units used.

Regarding the input and output resources in the studied plants:

- The values calculated for the plant performance, especially those related to refuse generation are strongly dependent on the quality of input OFMSW. At the same time these values calculated are directly influenced by the collection system used. The importance of the system boundaries considered in LCA studies for waste management and treatment operations should be emphasized.

- According to these results, total energy (electricity and fuel) needed for OFMSW treatment plants depends on the configuration/technology used. Total energy consumption (electricity and diesel) will range between 65 and 242 kWh/t OFMSW, where the lower value corresponds to low technology processes, and the highest value corresponds to high technology processes where systems of gas emissions are implemented.

- Water consumption also depends on the process activities as well as the plant capacity. Low technology facilities, with partial or no exhaust gas treatment, will normally recirculate leachate to the decomposition stage, whereas complex facilities should include gases in the wet treatment processes and could have higher water requirements. According with this study, water consumption will range between 0 and 0.56 m^3/t OFMSW, being the lower values correspond to low technology processes.

- Composting yield (t compost/t OFMSW) ranges between 0.03 and 0.52, where the highest yield of compost corresponds to the highest quality input materials. Refuse ranges between 0 and 0.41 t/t OFMSW, in this case, the highest refuse was obtained from the plant with the lowest quality of input materials.

Regarding to the analysis of NH_3 and VOC emissions on the full-scale OFMSW treatment plant analyzed, the following conclusions were obtained:

- Ammonia emissions ranged between 3.04E-02 and 8.63 kg NH_3/t OFMSW, where the higher values correspond to facilities that do not include exhaust gas treatment equipment in their treatment process.

- VOC emissions ranged between 0.23 and 8.98 kg VOCs/t OFMSW, as well as for ammonia, the higher values correspond to facilities that do not include gas treatment systems.

- The different ammonia and VOCs emissions found for the different biological treatment plants studied and their relative contribution to eutrophication, acidification, and photochemical oxidation potentials as well as global warming, human toxicity, and ozone layer depletion, depending on the type of energy source used, highlight the need for concrete data. This is strictly necessary in order to accurately perform LCA studies as the use of general data on the biological treatment process may lead to an increase in results uncertainty. These emissions should be carefully determined when using LCA to compare biological treatment technologies.

- Gaseous emissions from the biological process represent the main contribution to eutrophication, acidification, and photochemical oxidation potentials in four of the five plants analyzed, however these results are related to the type of technology used and the gas treatment systems applied.

- Designing and implementing efficient OFMSW treatments of gaseous emissions, technologies such as the scrubber and biofiltration could drastically decrease the values of these potentials (acidification, eutophication and photochemical oxidation).

Regarding to energy consumption and CO_2 emissions:

- Dimensions and distribution of the plant, as well as energy consumption in process represents a large portion of the final impacts of the biological treatment plants.

- CO_2 emissions were calculated only from the energy consumption (electricity and diesel) since CO_2 coming from biological treatment processes is considered as biogenic. CO_2 emissions ranged between 20 and 121.02 kg CO_2/t OFMSW, the highest emission corresponds to the plant with high technology implemented and gas emissions mitigation measures. A different consideration has been taken for the anaerobic digestion plant, since in this case there is a self-producing electricity process. The net CO_2 balance reported CO_2 emission savings of 55.2 kg CO_2/t OFMSW.

- Taking into account that process CH_4 and N_2O emissions were not measured, energy use represents the maximum contribution to global warming, human toxicity and ozone layer depletion potentials in all the studies cases and the relative values of these potentials depend on the different contribution of electricity and diesel consumption to the total energy used.
- When energy produced in plant by biogas cogeneration is taken into account, impact potentials from electricity item (global warming, ozone layer depletion and human toxicity) would be reduced.

Based on the results obtained in this dissertation several future works can be suggested:

- As a consequence of the above stated, if a facility finds the balance between an efficient treatment of OFMSW and moderate environmental impacts, according to the results obtained, one of the best options if this facility will be installed in a urban core, it is the combination of anaerobic digestion and composting including a biofiltration system as gaseous emissions mitigation system. This technology produces materials with high stability, and also produces energy that can be used for all operations within the plant, and can further be sold as a product. On the other hand, if the plant will be installed in a rural area where the input OFMSW are less than in a urban core and has fewer improper materials, the best option is to install a small closed composting plant with a simple biofiltration system, because with this type of systems the OFMSW and their gaseous emissions would treated in a good way without a high investment and getting a quality final product which can be applied in agriculture. However it is clear that to get proper conditions for OFMSW treatment, all the factors described above should be considered: input material quality (including materials collection), operation of plants protocol and monitoring of process and maintenance of gas mitigation systems.

- All the studies developed on the biological treatment plants presented in this dissertation have been a photograph of the operations of these facilities at a particular period. Further, depending on the type of study is would not be unusual to obtain differing results from those found in this study. This has to do with many factors, including input material characteristics and plant operations protocol, thus a future work that might arise after this work is encouraged to carryout further studies in the same facilities but in differing seasons of the year and for a longer period of time in order to observe the repeatability of the results and to see if a study such as this thesis was sufficient enough to correctly evaluate the impacts of the plants.

- Moreover and as noted in Chapter 9, it is essential to determine the greenhouse gases emissions such as CH_4 and N_2O from the biological process, because these values have an important role on the environmental potential impacts calculation such as global warming.

- Likewise, it is essential to study and incorporate in the future analysis the local social indicators such as the impact coming from malodors emissions, because it has severe implications on the facility success at social level. As it was noted in Chapter 8, these studies can be done using several methodologies such as electronic nose or olfactometry.

Chapter 11

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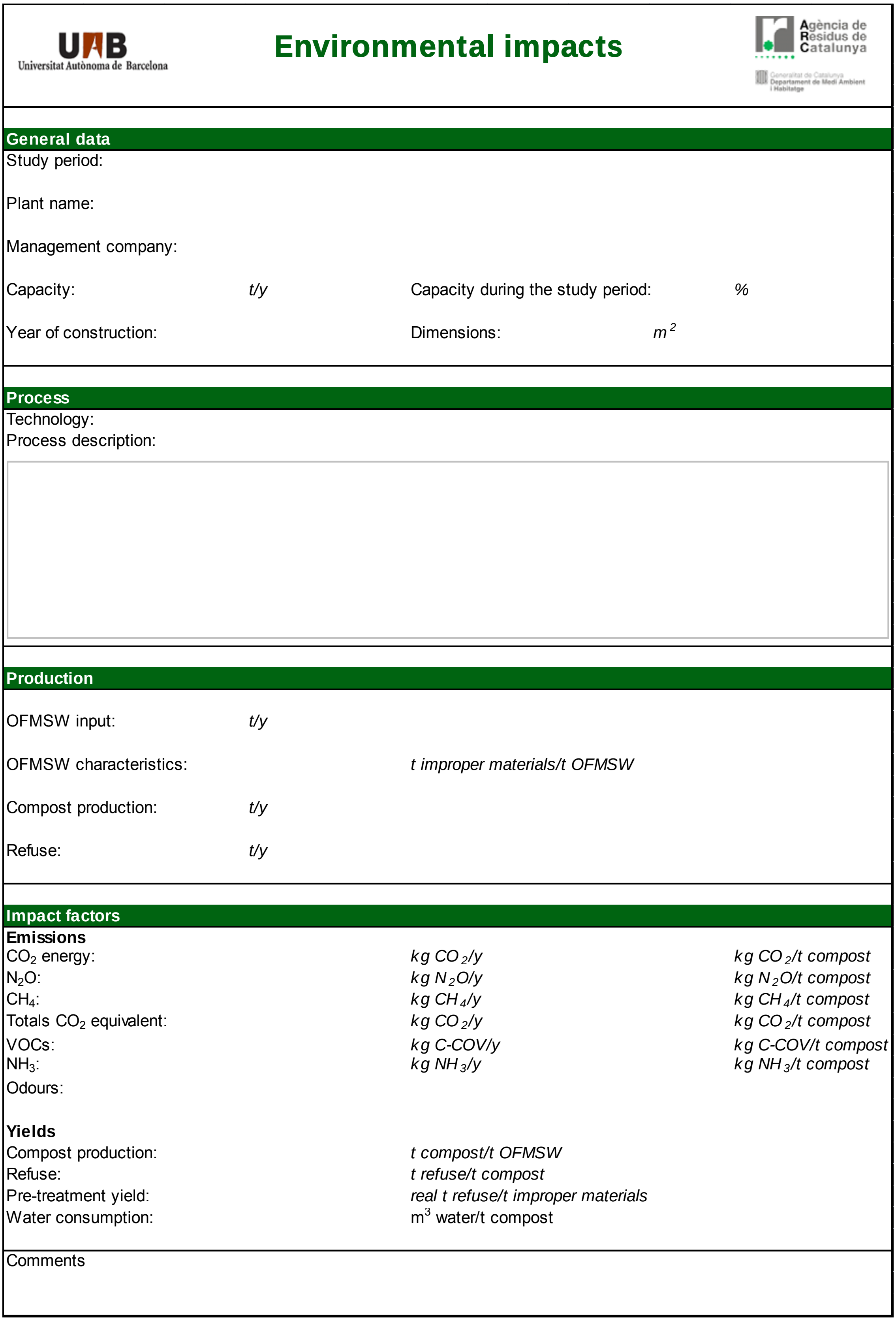
Annex

The main parts of the questionnaire used during this study are presented as follows.



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Environmental impacts

Agència de Residus de Catalunya

Generalitat de Catalunya Departament de Medi Ambient i Habitatge

General data

Study period:

Plant name:

Management company:

Capacity: *t/y* Capacity during the study period: *%*

Year of construction: Dimensions: *m^2*

Process

Technology:

Process description:

Production

OFMSW input: *t/y*

OFMSW characteristics: *t improper materials/t OFMSW*

Compost production: *t/y*

Refuse: *t/y*

Impact factors

Emissions

CO_2 energy:	*kg CO_2/y*	*kg CO_2/t compost*
N_2O:	*kg N_2O/y*	*kg N_2O/t compost*
CH_4:	*kg CH_4/y*	*kg CH_4/t compost*
Totals CO_2 equivalent:	*kg CO_2/y*	*kg CO_2/t compost*
VOCs:	*kg C-COV/y*	*kg C-COV/t compost*
NH_3:	*kg NH_3/y*	*kg NH_3/t compost*
Odours:		

Yields

Compost production:	*t compost/t OFMSW*
Refuse:	*t refuse/t compost*
Pre-treatment yield:	*real t refuse/t improper materials*
Water consumption:	m^3 water/t compost

Comments

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General data

Agència de Residus de Catalunya
Generalitat de Catalunya
Departament de Medi Ambient i Habitatge

Plant name:
Responsible:
Phone number:
E-mail:

Plant information

Year of construction

Modifications date

Modifications

Type Public ☐ Private ☐

Dimensions m^2

Technology process:

Tunnels

Windrows

Tunnels + windrows

Others

Capacity (t/y)

Solid waste

		Destination
OFMSW		
S. Sludge		
Other		

Solid waste treatment

Year

City	OFMSW (%)	% refuse

Comments:

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Reception

Agència de Residus de Catalunya
Generalitat de Catalunya
Departament de Medi Ambient i Habitatge

Equipmetns	Model	Type of fuel	on (kWh)	Time used (h/dia)	Capacity

Reception

Time remained in reception ☐ hr

Is used water? Yes ☐ No ☐ ☐ m³

Are generated leachate? Yes ☐ No ☐ ☐ m³

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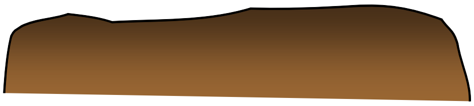
Curing & Maturation phase

Agència de Residus de Catalunya
Generalitat de Catalunya
Departament de Medi Ambient i Habitatge

Windrow dimensions

Width ☐ (m)

Windrow shape



Length ☐ (m)

Height ☐ (m)

Leachate generation? Yes ☐ No ☐

Biological process

Input ☐ (t/d)

Curing phase ☐ (weeks)

Maturation phase ☐ (weeks)

Refuse ☐ kg/d

Refuse characterization

☐ kg characterized

☐ kg glass
☐ kg plastic
☐ kg paper
☐ kg metals
☐ kg organic
☐ kg other

Week 1

Nº of turns ☐ (times) ☐ turns duration (h)

№ of material water ☐ (times) ☐ (m³)
Water source

Leachate ☐ (%)

Rain ☐ (%)

Public ☐ (%)

Leachate generation ☐ (l)

Refuse ☐ (t)

Temperature ☐ °C

Moisture ☐ %

Oxigen ☐ %

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Equipments

Agència de Residus de Catalunya

Equipments	Energy				Water		Refuse		
	Type of fuel	Theoretical consumption (kWh)	Time of use (h/d)	Capacity	Consumption (L/d)	Source	t/d	Treatment	Composition

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Emissions

Agència de Residus de Catalunya

Atmospheric emissions		**Yes**	**No**
Has the plant emissions treatment?		☐	☐
Has the plant emissions data determination?		☐	☐
Are calculate the process emissions?		☐	☐
Comments:			
Leachate generation		**Yes**	**No**
Has leachate tank?		☐	☐
Is the tank open to the atmosphere?		☐	☐
Leachate generatio:		☐ (m³)	
Leachate used in process:		☐ (m³)	
Are treated the leachate?		☐	☐
Type of treatment			
Internal or external?			
Rain water is collected?	N/A	☐	☐
Rain water is used in process?	N/A	☐	☐
Volume of the rain water tank:		☐ (m³)	
Comments:			