

Life Cycle Assessment of organic waste management strategies: an Italian case study



C. Buratti*, M. Barbanera, F. Testarmata, F. Fantozzi

CRB – Biomass Research Centre, Via G.Duranti, 63, 06125 Perugia, Italy

ARTICLE INFO

Article history:

Received 9 May 2014

Received in revised form

27 October 2014

Accepted 3 November 2014

Available online 12 November 2014

Keywords:

Life Cycle Assessment

Organic waste

Landfill

Composting

Mechanical-biological treatment

ABSTRACT

The aim of the paper was to evaluate the environmental burden of the organic waste management systems implemented in Umbria region in Italy, in order to provide useful information for taking strategic decisions aimed at the improvement and optimization. The two most widespread scenarios were analyzed: source-segregated collection followed by organic fertilizer production and not-differentiated collection followed by mechanical and biological treatment and disposal in landfill of the biostabilized material. The environmental performance was assessed through Life Cycle Assessment methodology, assuming one ton of organic waste as functional unit. Most of data for life cycle inventory were provided by actual facilities while background data were obtained from EcoInvent database. The alternative scenarios were compared through the IMPACT 2002+ method and the assessment was carried out on both midpoint and endpoint levels.

Results showed that the landfilling of the undifferentiated organic waste has the least impact on the analyzed impact categories, except on the Global Warming, mostly due to the uncollected methane released by the landfill. As regards the aerobic composting of the source-segregated organic fraction, the efforts to reduce the impact should be mainly focused on the reduction of the air emissions (hydrogen sulfide, particulate, ammonia and NMVOC) from the biostabilization process.

Furthermore the sensitivity analysis indicated that the increase of the biogas collection efficiency could significantly improve the performance of the not-differentiated collection scenario.

© 2014 Elsevier Ltd. All rights reserved.

1. Introduction

Life Cycle Assessment (LCA) methodology (ISO, 2006a,b) can support decision making by public and private actors in order to find the best solution for the municipal solid waste (MSW) management. The opportunity to take into account the characteristics of

the territory, as well as those of the waste disposal and treatment plants is one of the main advantages of this tool. Many applications are focused on the use of the LCA methodology as a decision support tool in the selection of the best MSW management strategy, from an environmental point of view, in a wide range of countries including Italy (Brambilla Pisoni et al., 2009; De Feo and Malvano, 2009), Spain (Bovea et al., 2010), Sweden (Eriksson et al., 2005), Germany (Wittmaier et al., 2009), China (Dong et al., 2013), Belgium (Belboom et al., 2013), and the U.S. (Kong et al., 2012).

This trend was also encouraged by the European Waste Framework Directive (EC, 2008), that emphasized the role of Life Cycle Thinking in the definition and evaluation of more sustainable strategies for MSW management.

A management of the organic fraction (OF) different from landfills and alternative management options such as composting, anaerobic digestion, as well as other conversion technologies are currently popular topics for discussion among policy makers, regulators, and waste management industries. The EU Landfill Directive 99/31/EC established a reduced amount of the biodegradable

Abbreviations: ABF, aerobic biological facility; ATO, optimal territorial area; BT, biological treatment; CP, compost production; DM, dry matter; LF, landfilling; LCA, life cycle assessment; LTP, leachate treatment plant; MBT, mechanical and biological treatment; MSW, municipal solid waste; MT, mechanical treatment; MTF, mechanical treatment facility; NMVOC, non-methane volatile organic compounds; OF, organic fraction; PP, power production; rMSW, residual municipal solid waste; rMSW DF, dry fraction of residual municipal solid waste; SOF, stabilized organic fraction; SS, source segregated; SSOF, source segregated organic fraction; TR, transport; WBTP, wastewater biological treatment plant.

* Corresponding author.

E-mail addresses: cinzia.buratti@unipg.it (C. Buratti), barbanera@crbnet.it (M. Barbanera), testarmata@crbnet.it (F. Testarmata), francesco.fantozzi@unipg.it (F. Fantozzi).

fraction of MSW going directly to landfills. Based on the amount of MSW generated in 1995, the Directive imposed a mandatory stepwise reduction of 25%, 50% and 65%, respectively, by 2006, 2009 and 2016.

The Italian waste management system is ruled by the Italian Legislative Decree no. 152 (ILD, 2006), which specifies as priority measures the prevention and reduction of waste production and its harmfulness followed by measures such as recovery of waste through recycling, reuse, or any other action aimed at extracting secondary raw materials and at using waste as energy source.

The management of MSW in Italy is achieved through an integrated system, divided into Optimal Territorial Areas (ATO). The ATO is a special form of cooperation between local authorities, with legal, regulatory autonomy, within the organizational and budgetary resources allocated to it by the municipalities, the Province, and the Region because of its functions transferred or delegate.

In Italy, about 10.8 million tons of OF, consisting of food and green waste, are currently produced yearly and about 42% is recovered by source segregated (SS) collection (ISPRA, 2013a). The source segregated organic fraction (SSOF) is mainly (about 78%) sent to aerobic biological pretreatment, in order to obtain a high quality compost. The other fraction of the OF is contained in the residual municipal solid waste (rMSW), characterized by a mean biodegradable concentration of 32% w/w (ISPRA, 2013a). Actually, about 40% of the rMSW is subjected to mechanical and biological treatment (MBT), in order to stabilize the biological degradable components. In the widespread scenario, the MBT facility operates the mechanical treatment, such as shredding, screening, and metal sorting of the rMSW in order to separate the rMSW OF from the other recyclable materials. The rMSW OF is then biologically pretreated to reduce its reactivity and mass, obtaining the stabilized OF (SOF). About 59% of the SOF is actually disposed of in landfills (ISPRA, 2013a).

The actual environmental burden of OF diversion from landfills has yet to be thoroughly evaluated and whether such a diversion provides significant environmental benefits must be answered. Laurent et al. (2014), in a Literature review, showed that there is no consensus about which treatment is the best option for the organic fraction both because LCA studies depend on local specificities and furthermore there are a rather limited number of studies about OF management systems. In particular, only few of the studies have attempted to cover multiple environmental impacts, with most tending to focus mainly on climate change (Morris et al., 2013). Therefore, as remarked by Morris et al. (2013), additional studies are necessary to determine the best option for the organic waste management with regard to environmental impacts other than climate change.

The aim of this study is to compare the environmental impacts caused by two most widespread OF management methods in Italy: source segregated collection for the production of high quality compost and undifferentiated collection with landfill disposal after the mechanical and biological stabilization.

As case study area, a district of the Umbria Region was selected, in which a mechanical treatment facility, a biological treatment facility, and a landfill are present.

2. Materials and methods

2.1. Case study area

According to the ruling of the Regional Planning for the Urban Solid Waste Management, the Umbrian territory has been divided into 4 ATOs. The case study area refers to the ATOs no. 1 and 2, with a population of 550,000 inhabitants, producing around 525 kg of

waste per capita per year: 225 kg (43%) is collected separately and 66 kg (12.5%) is the food waste fraction (ARPA Umbria, 2013). The organic waste management in the studied area is shown in Fig. 1.

The rMSW produced in this territory is sent to a mechanical treatment facility (MTF), where the waste stream enters a bag opener and is then conveyed to the first metal separation section, before screened with drum sieves. The waste stream passing through the sieve holes (100 mm diameter) is the rMSW OF which, after metal separation, is sent to the aerobic biological facility (ABF). The dry fraction of the rMSW (rMSW DF), after metal separation, is sent to a mechanical press, obtaining bales which are disposed of in landfill. This fraction still contains a certain percentage of organic matter that has not been possible to separate. In the same site of the MTF, there is also a wastewater biological treatment plant (WBTP) with activated sludge process. The plant treats the wastewater produced in the MTF site as well as sludge from septic tanks and sewage waters of the city nearby. The wastewater first passes through a bar screen and a sediment/oil separator and then is sent to denitrification and oxidation-nitrification sections, followed by final clarification of the effluent. The ABF is located 25 km away from MTF site. The rMSW OF and the SSOF are conveyed at the ABF site, where the waste streams are sent to a continuous flow aerobic basin, with an aerated floor, on which moves a crane bridge with screws used to stir and move the waste from the inlet to the outlet section. After about four weeks, the SSOF is placed in windrow heaps over a concrete platform for further aerobic treatment (62 days), while the rMSW OF, become stabilized (SOF), is disposed of in landfill. The platform is also equipped with ducts and electric fans to supply air to the waste. Before the biological treatment, only the SSOF is mixed with the green waste, while the obtained compost is refined by screening to remove contaminants such as plastic, glass and metals; this flow (Refuse), still containing a fraction of organic waste, is disposed of in landfill. In addition to the ABF, pollution control units are operating for the purification of the various air streams: biofilters, bag filters, and cyclone filters are used.

In the same site there are also a landfill and a leachate treatment plant (LTP). The landfill covers an area of 18 ha and the waste layer is at some points as much as 35 m deep. Biogas extracted from the wells is carried out by conveyance piping system to the combustion engines used to produce electricity. LTP consists of two sections: pretreatment unit, where takes place the equalization and the sedimentation, and the reverse osmosis unit, that produces permeate (discharged into the river) and concentrate (recirculated into the landfill) streams.

2.2. Goal definition

The goal of the study was to compare the environmental burden of the two most common options in Italy for the organic waste disposal, using life cycle approach.

The study is focused on the case of Umbria, a central Region of Italy, with 43% of total separate collection of municipal waste. High-quality data coming from specific mechanical and biological plants are used. The scenarios analyzed are:

1. undifferentiated collection, mechanical and biological treatment, and disposal of in landfill;
2. source-separated collection and production of high quality compost.

The study is performed in compliance with the four steps specified by the International Standardization Organization (ISO) standards (ISO, 2006a,b). The LCA was constructed using SimaPro software version 7.2. Currently, many LCA impact methods have been developed and widely used (Rodríguez et al., 2011), such as

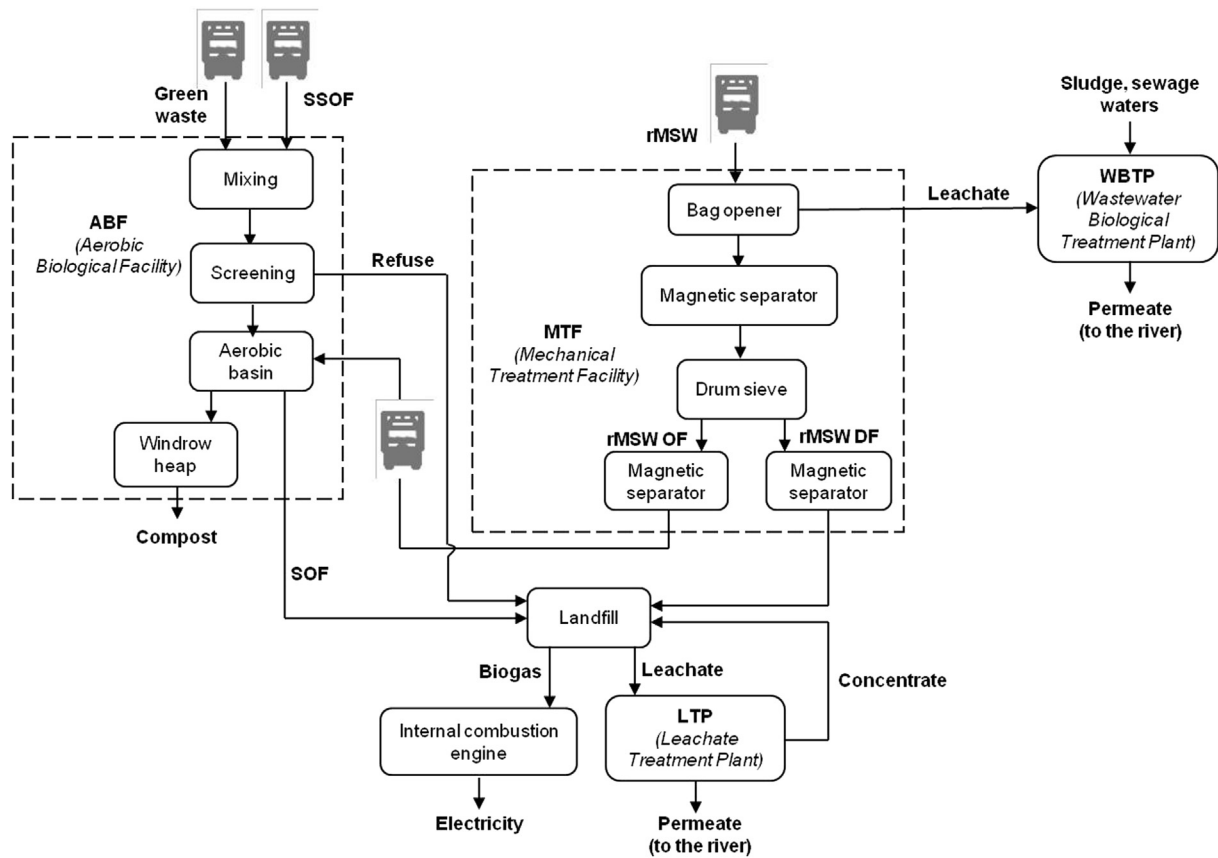


Fig. 1. Organic waste management system in the case study area.

Eco-indicator, EDIP, EPS et al. In this study the IMPACT 2002+ method (Joliet et al., 2003) was used due to the fact that IMPACT 2002+ model is one of the mostly used models in LCA analysis (Hong et al., 2010). IMPACT 2002+ method considers a 500-year time horizon in the global warming.

2.3. Scope definition

The functional unit of the study is one ton of OF. The function is the treatment of this amount of waste, using different technologies. All the input and output flows occurring along the chains were referred to the year 2011.

The system boundaries for the analyzed scenarios are represented in Figs. 2 and 3. The construction, implementation, maintenance, or demolition of the mechanical and biological plants and the equipments are not included, due to a lack of data; furthermore they have been described as negligible in several papers (Finnveden et al., 2005; Gentil et al., 2010).

Since the plants considered in the study treat waste fractions other than the OF, the input and output flows were calculated evaluating the mass percentage of the processed organic waste. Other assumptions considered in the life cycle inventory were:

- the electricity consumed by equipments is drawn from the Italian electricity grid;
- CO₂ from OF by decomposition in landfill site is considered carbon-neutral;
- carbon stock deriving from organic waste landfilling and the use of compost is not considered; in fact, as reported by Bernstad and la Cour Jansen (2012), there is no consensus whether a

carbon sink approach should be used or not. In the sensitivity analysis, the influence of such an assumption on final results was investigated.

MSW management options are multi-output systems. The problem of multi-functionality can be treated through two different approaches: the substitution method and the allocation method. The substitution method requires that a stand-alone process has to be specified to take into account an avoided economic flow. In this case a credit is attributed to the co-product, resulting from its possible use in replacement of an alternative functional product.

In the allocation method, the multi-function process is divided into a number of monofunctional processes. However this operation requires the choice of allocation factors that allow the subdivision of the environmental flows related to the multifunctional process. The choice of these factors is highly subjective and introduces further uncertainty. Both methods were widely used in Literature in various applications (Weiser et al., 2014; Buratti et al., 2013), but in this study substitution approach was applied. For both scenarios, we considered that the produced electricity from biogas combustion and the compost utilization for soil fertilization replace the electricity made by the Italian energetic mix and chemical fertilizers, respectively.

2.4. Data collection

Data were collected during technical visits and meetings with the managers of the MTF and ABF plants (reference year: 2011). Records were used to determine mass flows of the waste within the analyzed

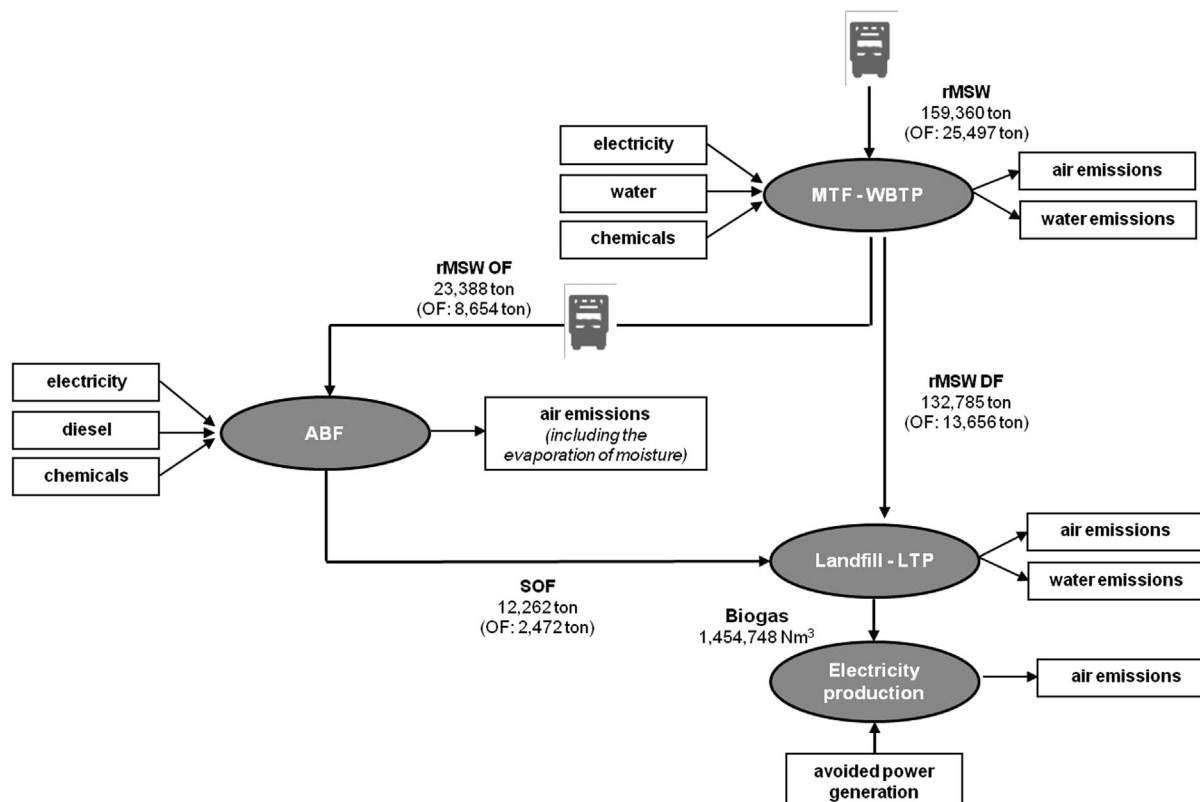


Fig. 2. System boundary for the Scenario 1 (undifferentiated collection).

scenarios, collection quantities and distances, compost recipe, fuel and energy consumption, water emissions from the LTP and WBTP.

Electricity consumption of each plant was determined from the installed power and the yearly operating hours.

Emissions due to the biostabilization process were calculated using emission factors from an extensive Literature review, described in detail below.

Some processes, such as transportation, electricity and fuel consumption emissions were modeled using the Ecoinvent database (Ecoinvent, 2010).

2.5. Life cycle inventory analysis

2.5.1. Feedstock transportation (Scenarios 1 and 2)

In the scenario 1, rMSW is transported at the MTF. Actually, rMSW is mainly collected by proximity separate collection system, consisting in the waste delivery in specific bins located near a group of users. Waste collection and transport is carried out by diesel-fuelled compactor trucks (payload: 11 ton) and the combustion technology is in compliance with Euro 3 standards. The average fuel consumption was estimated at 0.44 l/km, considering the empty return trip. ATO 1 and 2 are overall constituted by 38 municipalities. From the annual rMSW production of each municipality and the average distance from the MTF with 100% load and an empty return trip, a specific fuel consumption of 2.46 l/ton was calculated.

The transport of the rMSW OF from the MTF to ABF is instead carried out by diesel-fuelled articulated lorries (payload: 35 ton) over a distance of 25 km with 100% load and an empty return trip. The average diesel consumption was estimated at 0.43 l/km, considering the empty return trip. From these data, a specific fuel consumption of 0.57 l/ton was determined.

Scenario 2 provides that the SSOF is transported to the ABF. Municipalities neighboring the plant directly confer the waste to

the ABF by diesel-fuelled trucks (payload: 15 ton) with an average fuel consumption of 0.37 l/km. The others convey the SSOF to the MTF and then the waste is transported to the ABF. In both cases the same compactor trucks as for the rMSW collection are employed. Taking into account the average distances between the municipalities and the plants and the annual SSOF production of each municipality, a diesel consumption of 3.58 l/ton was calculated.

For both scenarios, the emission factors for the diesel combustion were drawn from Ecoinvent database (Ecoinvent, 2010).

2.5.2. Mechanical treatment (Scenario 1)

MTF is divided into two lines working in parallel: main line and emergency line. The emergency line is used only to treat the waste during the peak days and the periods of maintenance. Table 1 shows the inventory data collected or calculated for the MTF and WBTP plants.

Electricity consumption of the rMSW OF handling was all assigned to the OF, despite the organic content of the rMSW OF is only about 37%, because the non separated collection of the OF requires the management of a not separable fraction of inorganic waste.

2.5.3. Biological treatment (Scenarios 1 and 2)

The studied ABF has the capacity to treat 105,000 ton of waste (rMSW OF, SSOF and pruning waste) a year. The composting process lasts for about 90 days and is essentially articulated in four steps:

1. mixing of the incoming organic waste (SSOF with pruning waste at a mass ratio of about 3:1);
2. aerobic fermentation of the mixture, using an enclosed agitated bed system (total duration of at least 25 days);
3. maturation of compost in outdoor aerated static piles (total duration of at least 65 days);

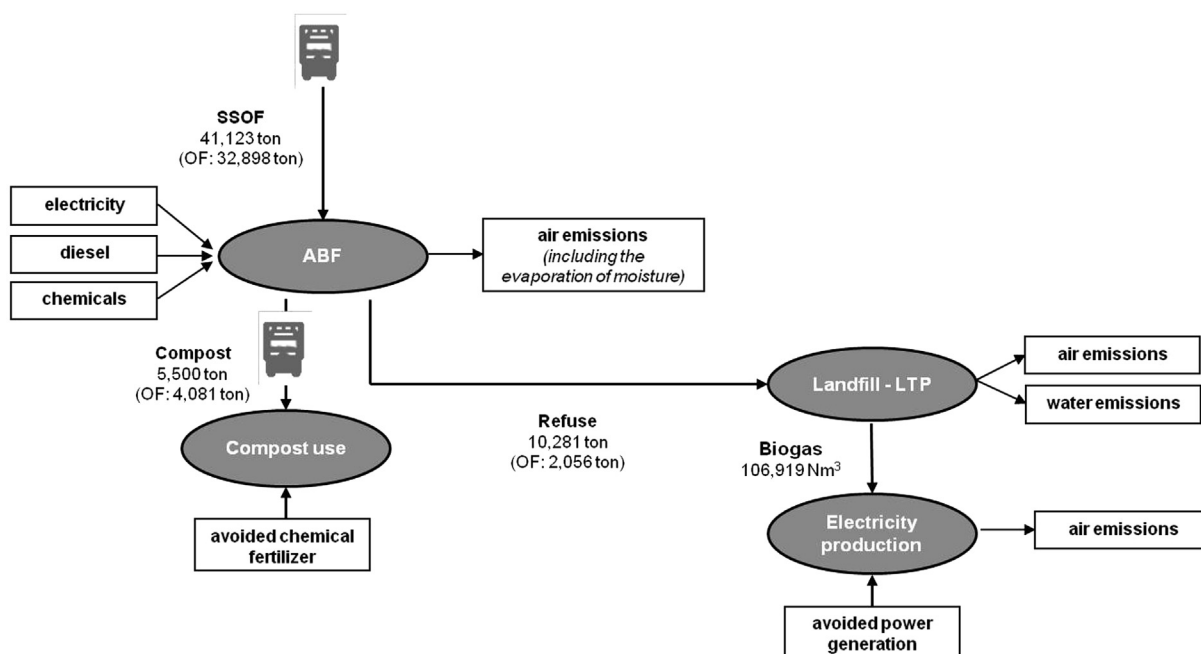


Fig. 3. System boundary for the Scenario 2 (source-separated collection).

4. mechanical refining of compost.

The aerobic fermentation is optimized by blowing air under the organic waste by electric fans and by moistening the bed with a fraction of wastewater from LTP, in order to improve its biological decomposition. This step takes place inside a closed building with air collection; the exhaust air is then conveyed to the curing area where it is blown into the static piles (weekly turned) that have the role of biofilters. At the end of the curing phase, the compost is moved to the refining area, in order to separate the impurities such as plastics, glass and metals; rejected materials are then disposed into the landfill. The stabilization of the rMSW OF takes place in the same plant, where the material is only subjected to the second step in a separate line for 28 days.

Inventory data for the ABF are summarized in Table 2. Electricity is consumed for the aeration of the material and during the operation of the agitated bed system whereas diesel is used in waste handling within the ABF limits.

Table 1

Summary of the inventory data for the MTF and WBTP plants. All the quantities are referred to 1 ton of OF waste (Scenario 1).

Waste treatment	Flow	Amount	Units	Source
MTF				
Energy consumption	Electricity	14.95	kWh/ton	Measured
Water consumption	Freshwater	0.09	m ³ /ton	Measured
Gaseous emissions	Particulate	0.67	g/ton	Measured
WBTP				
Energy consumption	Electricity	0.41	kWh/ton	Measured
Chemicals	Aluminum chloride	12.45	g/ton	Measured
Gaseous emissions	Hydrogen sulfide	1.4E-04	g/ton	Measured
Emissions to water	Suspended solids	0.20	g/ton	Measured
	BOD	0.14	g/ton	Measured
	COD	1.28	g/ton	Measured
	Chromium	5.0E-05	g/ton	Measured
	Lead	2.5E-04	g/ton	Measured
	Copper	2.4E-04	g/ton	Measured
	Sulphate	1.58	g/ton	Measured
	Chloride	2.09	g/ton	Measured
	Total nitrogen	0.47	g/ton	Measured
	Total phosphorus	1.1E-02	g/ton	Measured

The values obtained in our study (electricity: 26.03 kWh/ton of OF, diesel: 1.06 kg/ton of OF) are in the range reported by Bernstad and la Cour Jansen (2012) (electricity: 15.1–55.0 kWh/ton of OF, diesel: 0.01–13.0 kg/ton of OF) in which a literature review was carried out, pointing out that the energy consumption in the composting phase varies largely.

Since the irrigation of the material is carried out with wastewater, the consumption of freshwater was assumed to be zero.

As regards the emissions to air, H₂S (1.56 kg/ton of OF), NH₃ (0.21 kg/ton of OF), and particulate (0.074 kg/ton of OF) were directly evaluated; in particular the first two substances are measured over the static piles, while particulate is measured in the exhaust air from the compost refining area. To our knowledge, few data on H₂S and particulate emissions in full scale composting facilities are reported in the Literature. In particular, Righi et al. (2013) reported the values of 0.11 kg/ton of OF and 0.02 kg/ton of OF for particulate and H₂S, respectively, in a composting facility provided with biofilters for the treatment of the exhaust gases. For the ammonia emission, the obtained value is consistent with those reported in the Literature. Colòn et al. (2012) measured 0.11 kg NH₃/ton of OF for a composting in-vessel plant with biofilter. Blengini (2008) estimated 0.6 kg NH₃/ton of OF in an aerated windrow composting plant with a gas treatment process while Righi et al. (2013) reported a value of 0.04 kg NH₃/ton of OF.

As reported by Saer et al. (2013), the stabilization process of the organic material implies also the emission of four main substances: carbon dioxide (CO₂), methane (CH₄), nitrous oxide (N₂O), and NMVOC. CO₂ emitted from composting is not fossil-derived, and therefore it was not considered as a greenhouse gas emission (Martínez-Blanco et al., 2010), in compliance with the European Commission (Smith et al., 2001).

Few studies reported NMVOC emissions from full-scale composting facilities. Smet et al. (1999) and Baky and Eriksson (2003) considered 0.59 and 1.7 kg/ton of OF, respectively, in pilot-scale composting experiments, while Colòn et al. (2012) measured 0.36 kg/ton of OF in in-vessel composting system.

For methane and nitrous oxide emissions there is no consensus in the Literature on their accounting from composting in LCAs. The

Table 2

Summary of the inventory data for the ABF. All the quantities are referred to 1 ton of OF waste (Scenarios 1 and 2).

Waste treatment	Flow	Amount		Units	Source
		Scenario 1	Scenario 2		
Energy consumption	Electricity	7.08	26.03	kWh/ton	Measured
	Diesel	0.41	1.06	kg/ton	Measured
Gaseous emissions	H ₂ S	0.53	1.56	kg/ton	Measured
	NH ₃	0.07	0.21	kg/ton	Measured
	CH ₄	1.2E-02	3.4E-02	kg/ton	(Martínez-Blanco et al., 2010)
	NM VOC	0.41	1.21	kg/ton	(Martínez-Blanco et al., 2010)
	N ₂ O	3.1E-02	9.2E-02	kg/ton	(Martínez-Blanco et al., 2010)
	Particulate	–	7.4E-02	kg/ton	Measured
Waste production	Refuse	–	312.5	kg/ton	Measured
	SOF	480.9	–	kg/ton	Measured
Valuable materials	Compost	–	167.2	kg/ton	Measured
Avoided products	Calcium ammonium nitrate	–	8.67	kg/ton	Measured
	Single superphosphate	–	5.57	kg/ton	Measured
	Potassium sulphate	–	4.35	kg/ton	Measured

emissions resulting from the biological stabilization of the organic waste can be highly dependent on factors such as climate, local conditions, treatment technology, and exhaust gas cleaning system. Several studies did not consider methane and nitrous oxide emissions because they could not detect them (e.g. Colón et al., 2010; Smith and Jasim, 2009) or because they considered them to be negligible (e.g. US Environmental Protection Agency (EPA), 2006). In both cases the aerobic decomposition of the organic waste is assumed to occur perfectly. In support of this assumption there is also ILCD (2011) stating that toxic emissions other than CO₂ should be excluded from closed composting facilities due to efficient filtering of air emissions.

However, other studies showed that methane and nitrous oxide are emitted from composting systems (Andersen et al., 2010; Amlinger et al., 2008). In particular both Boldrin et al. (2009) and Saer et al. (2013) carried out a Literature review for CH₄ and N₂O emissions from biowaste composting, reporting CH₄ emissions ranging between 0.02 and 1.8 kg/ton of OF and 0.021–11.9 kg/ton of OF and N₂O emissions ranging between 0.0075 and 0.252 kg/ton of OF and 0.0003–0.252 kg/ton of OF, respectively.

The studies taken into account are referred both to the full-scale facilities and to home composting with the lowest values related to large-scale composting. Nevertheless, few studies are available for these two gases in the literature, dealing with fully-enclosed systems with a biofilter to treat exhaust gases. In the Literature the composting facility more similar to that considered in this study was reported by Martínez-Blanco et al. (2010). The study refers to a Spanish composting plant using the in-vessel (tunnel) decomposition technology, with a curing phase in turned windrows in an enclosed building. The plant has also biofilters (1 m layer of vegetal material) for the treatment of the exhaust gases. For this reason the same emission factors for CH₄ (0.034 kg/ton of OF), N₂O (0.092 kg/ton of OF) and NM VOC (1.21 kg/ton of OF) were adopted, which are consistent with the values described previously.

Compost produced in the Scenario 2 was credited by subtracting the life cycle impacts of the mineral fertilizers that compost is replacing, such as calcium ammonium nitrate (N content: 27%), single superphosphate (P₂O₅ content: 21%) and potassium sulphate (K₂O content: 50%). The nutrient composition of the compost produced by the ABF (1.4% of N, 0.7% of P₂O₅, 1.3% of K₂O and 21.9% of organic carbon) was used to calculate the amount of avoided products. The impacts for the production of the mineral fertilizers were drawn from the Ecoinvent database.

The impacts of the compost distribution and use were not considered since it was assumed that they are similar to the scenario of chemical fertilizers employing.

Diesel consumptions in Table 2 include the transportation of the SOF and Refuse streams from the composting facility to the landfill.

2.5.4. Landfill (Scenarios 1 and 2)

Landfill gas is mainly composed of CH₄ and CO₂. The content of other gases such as H₂S, hydrocarbons and aromatics is extremely low. Carbon dioxide emissions associated with the generation of landfill gas are considered biogenic emissions and will not be included in the inventory, in compliance with IPCC methodology (IPCC, 2006). In fact these can be considered equivalent to the carbon dioxide absorbed during plant growth.

The estimation of methane emissions from the landfill was carried out by the software LandGEM (Landfill Gas Emission Model) (US EPA, 2005) and assuming typical gas collection efficiencies. The calculation is based on a first order decay equation that describes the emission rate of methane due to the anaerobic degradation of the organic fraction of the waste. For the application of the model, only two parameters (L_0 and k) required user inputs. L_0 is the methane generation potential (kg/ton) of the waste, while k is defined as the first order decay rate (yr^{-1}). For the OF contained in the rMSW DF, L_0 (72.1 kg/ton) was estimated according to the Eq. (1) (IPCC, 2006):

$$L_0 = F * \text{DOC} * \text{DOC}_f * 16/12 * \text{MCF} \quad (1)$$

where:

- MCF = Methane Correction Factor (the default value is 1.0 for managed-anaerobic landfills);
- DOC = Degradable Organic Carbon (192 kg/ton of OF (ISPRA, 2013b));
- DOC_f = fraction of DOC that can be decomposed (0.50 (ISPRA, 2013b));
- F = fraction of methane in landfill gas (0.56, the average value for 2011);
- 16/12 = stoichiometric factor.

The calculation of L_0 for the SOF and the Refuse was carried out according to Ponsá et al. (2008), assuming that aerobic and anaerobic stability indices are correlated for organic solid wastes in different stages of biodegradation. In particular, L_0 was estimated from the biochemical methane potential when no significant biogas production is detected (BMP_F , L kg DM⁻¹) through the Eq. (2) (Ponsá et al., 2008):

$$\text{BMP}_F = (54 * \text{RI} + 3.8)/0.73 \quad (2)$$

where RI is the Respiration Index ($\text{g O}_2 \text{ kg DM}^{-1} \text{ h}^{-1}$) of the biowaste.

Respiration Index data (yearly average values) were supplied by the plant managers for SOF ($\text{RI} = 412 \text{ mg O}_2 \text{ kg DM}^{-1} \text{ h}^{-1}$; $\text{DM} = 30\%$) and Refuse ($\text{RI} = 163 \text{ mg O}_2 \text{ kg DM}^{-1} \text{ h}^{-1}$; $\text{DM} = 20\%$). Therefore the obtained L_0 values were: 16.5 kg/ton for the SOF and 9.1 kg/ton for the Refuse.

Decay rate (0.144 yr^{-1}) for the organic fraction was taken from Levis and Barlaz (2011). Determination of fugitive emissions is strongly dependent by the gas collection efficiency, varying from zero (during waste placement) to 90–95% (after placement of the final cover) (Levis and Barlaz, 2011; Spokas et al., 2006). However, for the analyzed landfill, data about biogas collection efficiency were not available; the value of 45.3% (ISPRA, 2013b) was assumed to be the average efficiency of the Italian managed landfills in 2011.

As methane gas moves through the soil layer, it is oxidized and converted into carbon dioxide through microbial activities in the soil. Fugitive methane emissions were assumed to be reduced by 10% while passing through a landfill cover system, such as adopted in many protocols including the GHG emission inventory for IPCC, U.S. EPA's mandatory GHG emission reporting program, and the carbon credit calculation guidelines under the Kyoto Protocol, the Chicago Climate Exchange (CCX), Regional Greenhouse Gas Initiatives (RGGI), and the more recent protocol for the Climate Action Reserve (Yoshida et al., 2012).

Table 3 shows the inventory data collected or calculated for the landfill and LTP. Biogas collected from the landfill is used in internal combustion engines with an overall electric power of 2.1 MW. The emission factors (Table 4) for the biogas combustion were calculated from measured data, such as the average composition of the exhaust gases (Particulate: 4.86 mg/Nm^3 , NO_x : 210.46 mg/Nm^3 , SO_2 : 13.81 mg/Nm^3 , CO : 307.60 mg/Nm^3 , NMVOC : 40.62 mg/Nm^3 , HCl : 0.24 mg/Nm^3 , H_2S : 0.18 mg/Nm^3), the average Lower Heating Value (LHV) of biogas (17.35 MJ/m^3), and the specific biogas consumption (11.6 MJ/kWh). Actually, the heat recovered from engine exhaust gases is discarded to atmosphere.

3. Results and discussion

IMPACT 2002+ mid-point life cycle impact assessment results in the two scenarios are compared in Table 5. According to the

Table 3
Summary of the inventory data for the landfill and LTP. All the quantities are referred to 1 ton of OF waste (Scenarios 1 and 2).

Waste treatment	Flow	Amount		Units	Source
		Scenario 1	Scenario 2		
Energy consumption	Diesel	0.87	0.26	kg/ton	Measured
	Electricity	1.55	0.63	kWh/ton	Measured
Chemicals	Sulfuric acid	1.7E-03	5.2E-04	m ³ /ton	Measured
Gaseous emissions	CH ₄	22.91	1.41	kg/ton	Calculated
	Suspended solids	2.3E-02	7.1E-03	g/ton	Measured
Emissions to water	COD	1.6E-01	5.0E-02	g/ton	Measured
	Chromium	5.8E-06	1.8E-06	g/ton	Measured
	Lead	1.7E-05	5.4E-06	g/ton	Measured
	Copper	1.2E-04	3.7E-05	g/ton	Measured
	Sulphate	1.1E-02	3.4E-03	g/ton	Measured
	Chloride	1.1E-02	3.4E-03	g/ton	Measured
	Total nitrogen	6.7E-02	2.1E-02	g/ton	Measured
	Total phosphorus	2.8E-03	8.6E-04	g/ton	Measured
	Biogas	53.59	3.25	Nm ³ /ton	Calculated
Valuable materials					

Table 4

Summary of the inventory data for the power unit. All the quantities are referred to 1 ton of OF waste (Scenarios 1 and 2).

Waste treatment	Flow	Amount		Units	Source
		Scenario 1	Scenario 2		
Gaseous emissions	Particulate	2.03	2.0E-03	g/ton	Measured
	NO _x	87.78	0.09	g/ton	Measured
	SO ₂	5.76	0.01	g/ton	Measured
	CO	128.30	0.13	g/ton	Measured
	NMVOC	16.94	0.02	g/ton	Measured
	HCl	0.10	1.0E-04	g/ton	Measured
	H ₂ S	0.07	7.3E-05	g/ton	Measured
Valuable materials	Electricity	80.15	4.86	kWh/ton	Measured

common practice in LCA studies, positive values mean a burden for the environment (thus negative effects), while negative ones mean savings (thus positive effects for the environment). The processes are named as following: transport (TR), mechanical treatment (MT), biological treatment (BT), landfilling (LF), compost production (CP) and power production (PP).

In order to define the most relevant impact categories, the normalization of results is carried out by applying normalization factors to impact categories (Fig. 4). The IMPACT 2002+ method employs the emission values of Western Europe as reference values. According to the obtained results, it can be deduced that the most affected categories are Non-carcinogens, Respiratory inorganics, Respiratory organics, Terrestrial ecotoxicity, Terrestrial acidification/nutritification, Global warming, Non-renewable energy. These categories are analyzed below in detail.

3.1. Non-carcinogens

Scenario 2 has the most adverse impact on non-carcinogens (Fig. 5). This is due to the emission of hydrogen sulfide from the biostabilization process, that is higher than the credit obtainable from the avoided arsenic emission in atmosphere during fertilizers production. Scenario 1 has less impact thanks to the environmental credits obtained from the electricity production.

3.2. Respiratory inorganics

The highest impact of the Scenario 2 is mainly due to the particulate and ammonia emissions in the biostabilization stage, representing about 70% of the total impact (Fig. 6). Production of compost allows a significant cutback in respiratory inorganics

Table 5

IMPACT 2002+ mid-point results (per ton of OF).

Impact category	Units	Scenario 1	Scenario 2
Carcinogens	kg C ₂ H ₃ Cl eq	−3.17E-02	−5.05E-02
Non-carcinogens	kg C ₂ H ₃ Cl eq	1.43E-01	3.51E-01
Respiratory inorganics	kg PM _{2.5} eq	3.14E-02	5.40E-02
Ionizing radiation	Bq C-14 eq	−5.55E+02	3.91E+01
Ozone layer depletion	kg CFC-11 eq	−9.96E-07	1.47E-06
Respiratory organics	kg C ₂ H ₄ eq	4.12E-01	7.62E-01
Aquatic ecotoxicity	kg TEG water	−6.32E+02	−3.03E+02
Terrestrial ecotoxicity	kg TEG soil	−1.64E+02	5.14E+01
Terrestrial acid/nutri	kg SO ₂ eq	1.81E+00	3.37E+00
Land occupation	m ² org.arable	1.12E-02	−5.77E-02
Aquatic acidification	kg SO ₂ eq	1.19E+00	3.34E+00
Aquatic eutrophication	kg PO ₄ P-lim	−1.88E-03	−5.80E-03
Global warming	kg CO ₂ eq	1.62E+02	3.44E+01
Non-renewable energy	MJ primary	−2.87E+02	2.00E+02
Mineral extraction	MJ surplus	2.74E-02	−5.54E-01

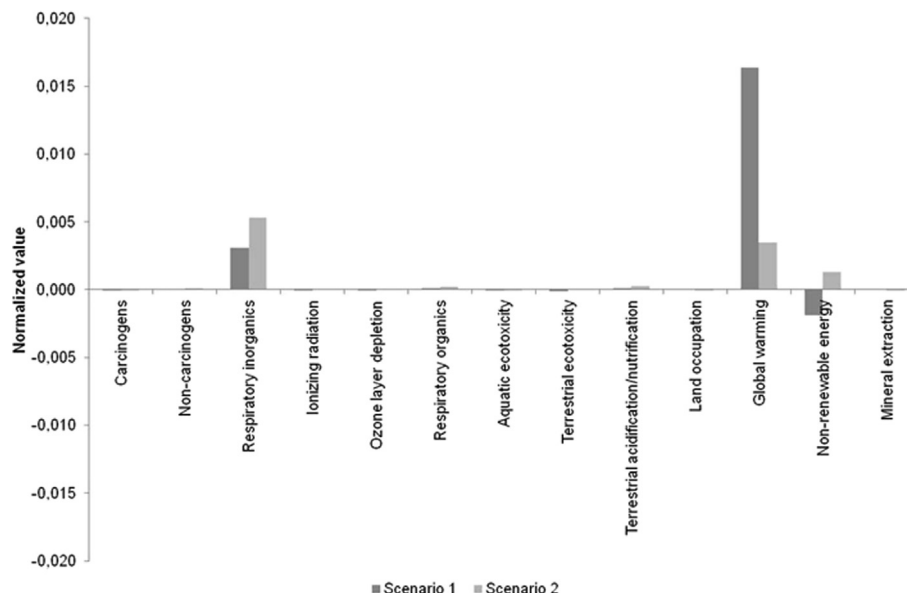


Fig. 4. Normalized impact categories in each scenario.

substances: -0.023 kg PM_{2.5} eq. In the Scenario 1 the highest contributions are caused by diesel combustion in the OF transport (0.012 kg PM_{2.5} eq), in the landfill site (0.085 kg PM_{2.5} eq), and ammonia emissions from biostabilization stage (0.085 kg PM_{2.5} eq).

3.3. Respiratory organics

In both scenarios, the highest contribution to respiratory organics category is caused by NMVOC emissions from the stabilization of the organic fraction, representing 95% and 60% of the total impact respectively for Scenario 2 and 1; in the Scenario 1 methane fugitive emissions from landfill also play an important role (about 34% of the impact) (Fig. 7).

3.4. Terrestrial ecotoxicity

Also in the case of terrestrial ecotoxicity category, the least burden was originated from Scenario 1. In both scenarios the main contributors are the aluminum emissions into the soil and the atmosphere caused by diesel and electricity consumption. Therefore transport (38%) and biostabilization (57%) contributes significantly to this impact category for the Scenario 2 while transport of the organic fraction and the pretreatment of rMSW emit 36% and 29% of terrestrial ecotoxicity pollutants in the Scenario 1. Production of

compost and/or power allows a significant cutback in terrestrial ecotoxicity substances: -483.4 kg TEG soil (Scenario 1), -300.1 kg TEG soil (Scenario 2) (Fig. 8).

3.5. Terrestrial acidification/nutrification

Scenario 2 presents the highest terrestrial acidification/nutrification emissions, with the biostabilization that plays the greatest role due to the ammonia emissions (3.14 kg SO₂ eq). In the Scenario 1 the major contributions are caused by ammonia emissions from biostabilization process (1.05 kg SO₂ eq), nitrogen oxides emissions from diesel use at landfill site (0.21 kg SO₂ eq), and from biogas combustion (0.45 kg SO₂ eq). Production of compost and/or power allows a modest cutback in terrestrial acidification/nutrification substances: -0.77 kg SO₂ eq (Scenario 1) and -0.19 kg SO₂ eq (Scenario 1) (Fig. 9).

3.6. Global warming

Scenario 1 presents the highest GWP, mainly due to the landfilling of the organic waste (178.8 kg CO₂ eq) that generates large emissions of methane. In the Scenario 2 the main contributor is the biostabilization process role (59%) while landfilling is less relevant (21%) since only a small fraction of organic waste is disposed in the landfill. The total amounts of greenhouse gases avoided as a result

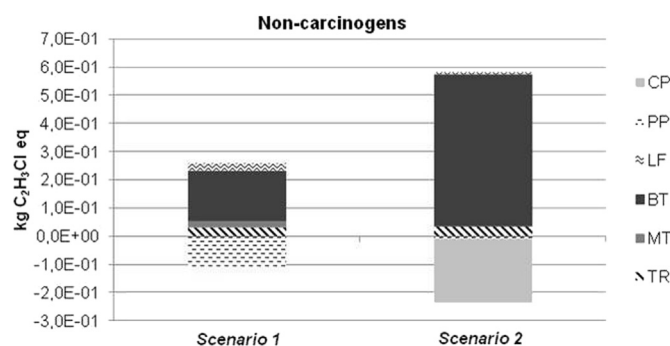


Fig. 5. Process contribution to the non-carcinogens category.

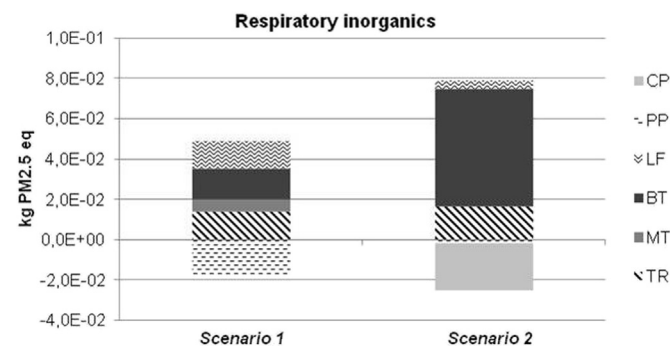


Fig. 6. Process contribution to the respiratory inorganics category.

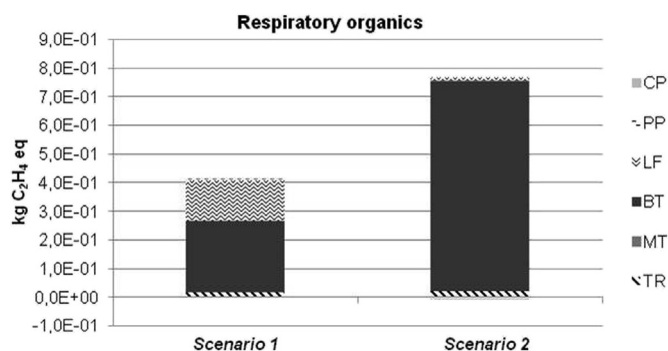


Fig. 7. Process contribution to the respiratory organics category.

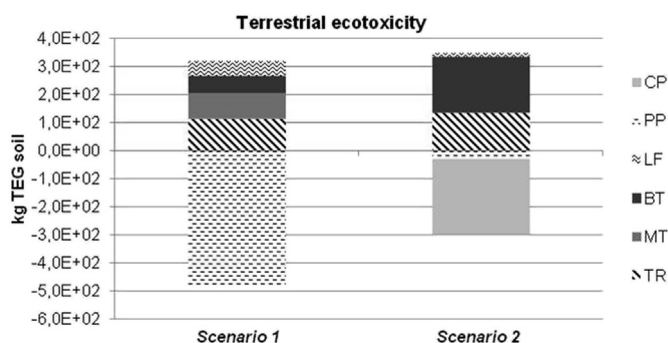


Fig. 8. Process contribution to the terrestrial ecotoxicity category.

of compost or/and power production are: $-45.3 \text{ kg CO}_2 \text{ eq}$ (Scenario 1), $-22.6 \text{ kg CO}_2 \text{ eq}$ (Scenario 2) (Fig. 10).

3.7. Non-renewable energy

Electricity recovery from methane gas reduce significantly the non-renewable energy consumption of Scenario 1. In the Scenario 2 the major contributors are biostabilization (60.8%) and transport (34.6%) while diesel (32%) and electricity (31%) consumption in the waste transport and pretreatment play the most important role in the Scenario 1. In the Scenario 2 chemical fertilizers replacement with compost allows to save 250.9 MJ primary (Fig. 11).

Tables 6 and 7 show the damage endpoint categories in each scenario, combining all the mid-point categories into damages on human health, ecosystems, climate change, and abiotic resources. In the human health and ecosystem quality damage categories, Scenario 2 had the highest potential impacts. In the resources damage category, Scenario 1 had the lowest potential impact due to

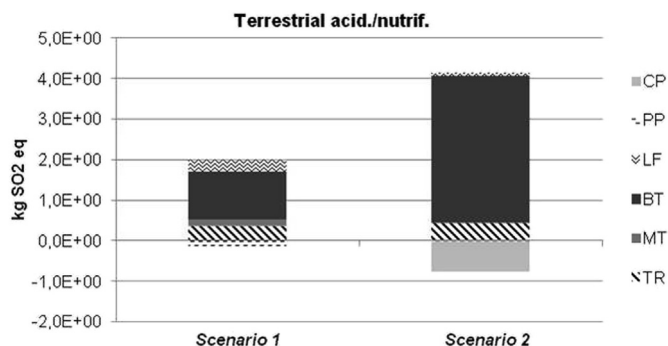


Fig. 9. Process contribution to the terrestrial acidification/nutrient category.

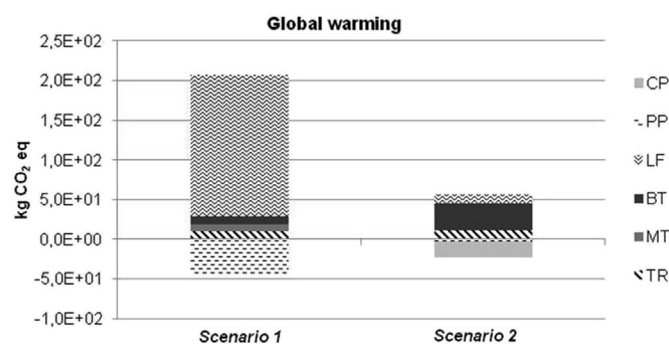


Fig. 10. Process contribution to the global warming category.

the higher electricity recovery from biogas combustion; at the same time, the low collection efficiency of methane generated by the biological degradation caused the highest impact in the climate change category. In particular, Tables 6 and 7 highlight that Respiratory inorganics is the most relevant impact category in the Human Health damage category both in Scenario 1 (94.6%) and 2 (93.8%), mostly due to the biostabilization process (Scenario 1: 31.7%, Scenario 2: 73.6%). As regards the Ecosystem Quality, Terrestrial ecotoxicity would be the main impact category in the Scenario 1 ($2.5 \text{ PDF} \cdot \text{m}^2 \cdot \text{yr}$) but the electricity recovery plays an important role, reducing significantly the score. Therefore, Terrestrial acidification/nutrient is characterized by the highest impact, mostly due to the biostabilization step. The same result was obtained for the Scenario 2, where the avoided production of chemical fertilizers allows to reduce the impact of Terrestrial ecotoxicity category. In the Resources damage category, the use of fossil fuels is the main contributor as previously described for the Non-renewable energy impact category.

Overall, the results obtained in the damage level were consistent with the ones in the mid-point level, showing that Scenario 2 is less impacting than Scenario 1 in terms of GHG emissions but it shows the worst performance across the other damage categories.

The comparison with other studies is difficult mainly because the system boundary of the LCA analysis can be different and also the number of studies that covered any particular impact is very small using different indicators for the same environmental impact (Morris et al., 2013). However, a significant comparison can be carried out on the basis of the climate change impact; GWP results are in the same range for both analyzed scenarios as those from previous Literature studies (aerobic composting: -0.76 to $0.22 \text{ ton CO}_2/\text{ton of OF}$; landfill gas-to-energy; -0.31 to $1.00 \text{ ton CO}_2/\text{ton of OF}$) (Morris et al., 2013).

As final consideration, readers have to be aware of the uncertainty of such a type of exercise (Clavreul et al., 2012). The analysis

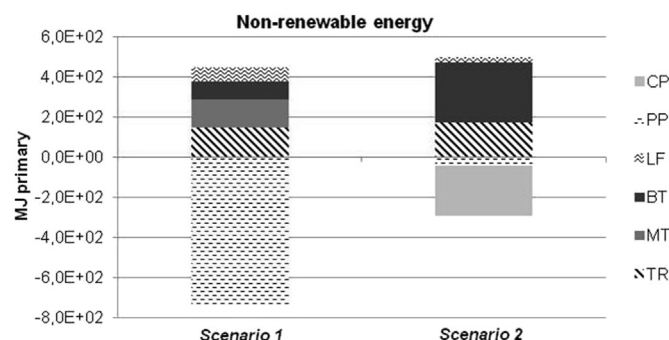


Fig. 11. Process contribution to the non-renewable energy category.

Table 6
IMPACT 2002+ end-point results (per ton of OF) for the Scenario 1.

Damage/Impact category	Units	TR	MT	BT	LF	PP	Total
Human health	DALY	1.01E-05	4.22E-06	1.19E-05	9.84E-06	−1.31E-05	2.30E-05
Carcinogens	DALY	6.39E-08	7.19E-08	6.43E-08	8.40E-08	−3.73E-07	−8.88E-08
Non-carcinogens	DALY	8.75E-08	6.25E-08	5.07E-07	6.39E-08	−3.21E-07	4.00E-07
Respiratory inorganics	DALY	9.92E-06	4.05E-06	1.08E-05	9.39E-06	−1.22E-05	2.20E-05
Ionizing radiation	DALY	8.59E-09	3.59E-08	1.75E-08	7.83E-09	−1.86E-07	−1.17E-07
Ozone layer depletion	DALY	1.27E-09	8.29E-10	5.91E-10	5.71E-10	−4.31E-09	−1.05E-09
Respiratory organics	DALY	3.95E-08	3.63E-09	5.30E-07	3.02E-07	2.76E-09	8.78E-07
Ecosystem quality	PDF ^m m ² yr	1.34E+00	9.01E-01	1.75E+00	7.52E-01	−4.18E+00	5.64E-01
Aquatic ecotoxicity	PDF ^m m ² yr	1.99E-02	1.75E-02	1.14E-02	1.06E-02	−9.12E-02	−3.17E-02
Terrestrial ecotoxicity	PDF ^m m ² yr	9.03E-01	7.35E-01	4.67E-01	4.24E-01	−3.82E+00	−1.29E+00
Terrestrial acid/nutri	PDF ^m m ² yr	3.72E-01	1.35E-01	1.26E+00	3.05E-01	−1.96E-01	1.88E+00
Land occupation	PDF ^m m ² yr	4.54E-02	1.29E-02	8.19E-03	1.25E-02	−6.67E-02	1.22E-02
Climate change	kg CO ₂ eq	9.56E+00	8.73E+00	1.05E+01	1.79E+02	−4.53E+01	1.62E+02
Resources	MJ primary	1.45E+02	1.41E+02	8.91E+01	7.20E+01	−7.34E+02	−2.87E+02
Non-renewable energy	MJ primary	1.45E+02	1.41E+02	8.91E+01	7.19E+01	−7.34E+02	−2.87E+02
Mineral extraction	MJ primary	1.59E-02	1.66E-02	1.77E-02	6.08E-02	−8.37E-02	2.74E-02

Table 7
IMPACT 2002+ end-point results (per ton of OF) for the Scenario 2.

Damage/Impact category	Units	TR	BT	LF	CP	PP	Total
Human health	DALY	1.20E-05	4.39E-05	2.94E-06	−1.72E-05	−1.30E-06	4.03E-05
Carcinogens	DALY	7.54E-08	2.02E-07	2.59E-08	−4.22E-07	−2.26E-08	−1.42E-07
Non-carcinogens	DALY	1.03E-07	1.51E-06	1.99E-08	−6.29E-07	−1.95E-08	9.83E-07
Respiratory inorganics	DALY	1.17E-05	4.06E-05	2.87E-06	−1.61E-05	−1.24E-06	3.78E-05
Ionizing radiation	DALY	1.01E-08	6.32E-08	2.74E-09	−5.66E-08	−1.13E-08	8.21E-09
Ozone layer depletion	DALY	1.50E-09	1.94E-09	1.80E-10	−1.82E-09	−2.61E-10	1.55E-09
Respiratory organics	DALY	4.67E-08	1.56E-06	2.06E-08	−8.55E-09	−1.12E-09	1.62E-06
Ecosystem quality	PDF ^m m ² yr	1.58E+00	5.43E+00	2.36E-01	−3.13E+00	−2.84E-01	3.84E+00
Aquatic ecotoxicity	PDF ^m m ² yr	2.35E-02	3.83E-02	3.36E-03	−7.49E-02	−5.53E-03	−1.52E-02
Terrestrial ecotoxicity	PDF ^m m ² yr	1.07E+00	1.58E+00	1.35E-01	−2.14E+00	−2.32E-01	4.07E-01
Terrestrial acid/nutri	PDF ^m m ² yr	4.40E-01	3.78E+00	9.29E-02	−7.64E-01	−4.21E-02	3.51E+00
Land occupation	PDF ^m m ² yr	5.36E-02	2.76E-02	3.91E-03	−1.44E-01	−4.04E-03	−6.29E-02
Climate change	kg CO ₂ eq	1.13E+01	3.35E+01	1.22E+01	−1.98E+01	−2.76E+00	3.44E+01
Resources	MJ primary	1.71E+02	3.01E+02	2.31E+01	−2.51E+02	−4.45E+01	1.99E+02
Non-renewable energy	MJ primary	1.71E+02	3.01E+02	2.31E+01	−2.51E+02	−4.45E+01	2.00E+02
Mineral extraction	MJ primary	1.88E-02	5.38E-02	1.86E-02	−6.40E-01	−5.07E-03	−5.54E-01

of uncertainty was out of the scope of the paper, but the most relevant parameters and assumptions (i.e. those that most affect waste management LCA) were robust and mostly based on primary data, such as the material flows of the different waste fractions.

3.8. Sensitivity analysis

Sensitivity analyses were performed to assess the robustness of the results by modifying relevant assumptions of the analyzed scenarios. In particular new assumptions were related to the biogas collection efficiency and the carbon sequestration from the use of compost and the waste landfilling. As regards the first parameter, the value assumed in this study (45.3%) reflects the current situation of the Italian landfills and depends on several parameters, such as the type and installation time of the gas collection system and intermediate and final covers (Amini and Reinhart, 2011). It seemed to be interesting to evaluate the environmental impacts under more efficient biogas collection systems. In particular USEPA provides 75% as a default value for the collection efficiency (Niskanen et al., 2013) when the collection system is in use and operates without problems for the new landfills. Obviously, higher biogas collection efficiency increases the electricity production, while direct CH₄ emissions from the landfill to the atmosphere decrease. Biogas collection efficiencies of 55%, 65%, and 75% were assumed and results, reported in Table 8, show that the environmental impacts decrease mostly for the Scenario 1. Impacts in the Scenario 1 were significantly lower for all the impact categories (37%–378%,

for the biogas collection efficiency of 75%), apart for Respiratory organics, Terrestrial acidification/nitrification and Aquatic acidification. For the Scenario 2, the environmental benefit is less important (1–22%), apart for Ionizing radiation (93%) and Terrestrial ecotoxicity (38%). Nevertheless for the impact categories for which Scenario 2 has the best performance in the reference case, only for Carcinogens the increasing of the biogas collection efficiency allows to change the best scenario.

The inclusion of the carbon sequestration in the LCA studies of the organic waste management is one of the main controversial issue. Several studies recognized that landfilling and composting provide for long-term carbon sequestration (Kong et al., 2012) but most of the LCA models available in the Literature (Gentil et al., 2010) do not consider this contribution. Obviously, the carbon binding in the soil only affects the Global Warming impact category because this carbon can be seen as a sink of CO₂ and credited as an avoided downstream emission of CO₂. The permanency of such sinks is difficult to evaluate; however carbon is partially oxidized over a one hundred year time scale, but degradation rates over a 500 year period are not known. Therefore in order to assess the influence of the carbon sequestration into the global warming a 100-year time horizon (CO₂: 1 kg CO₂ eq/kg CO₂, CH₄: 25 kg CO₂ eq/kg CH₄, N₂O: 298 kg CO₂ eq/kg N₂O) (IPCC, 2007) has to be used rather than 500-year time horizon assumed by the IMPACT 2002+ method.

As regards the carbon sequestration factor for the compost, the carbon still bound to soil after 100 years was estimated to be 2–10% of the input (Boldrin et al., 2009); in this study a value of 6% was

Table 8
IMPACT 2002+ mid-point results (per ton of OF) assuming different biogas collection efficiencies.

Impact category	Units	Scenario 1				Scenario 2			
		45.3%	55%	65%	75%	45.3%	55%	65%	75%
Carcinogens	kg C ₂ H ₃ Cl eq	−3.17E-02	−6.08E-02	−9.04E-02	−1.20E-01	−5.05E-02	−5.24E-02	−5.42E-02	−5.60E-02
Non-carcinogens	kg C ₂ H ₃ Cl eq	1.43E-01	1.18E-01	9.22E-02	6.68E-02	3.51E-01	3.50E-01	3.48E-01	3.46E-01
Respiratory inorganics	kg PM _{2.5} eq	3.14E-02	2.76E-02	2.37E-02	1.98E-02	5.40E-02	5.36E-02	5.32E-02	5.28E-02
Ionizing radiation	Bq C-14 eq	−5.55E+02	−7.49E+02	−9.46E+02	−1.14E+03	3.91E+01	2.68E+01	1.47E+01	2.80E+00
Ozone layer depletion	kg CFC-11 eq	−9.96E-07	−1.89E-06	−2.80E-06	−3.71E-06	1.47E-06	1.42E-06	1.36E-06	1.30E-06
Respiratory organics	kg C ₂ H ₄ eq	4.12E-01	3.88E-01	3.63E-01	3.38E-01	7.62E-01	7.60E-01	7.58E-01	7.57E-01
Aquatic ecotoxicity	kg TEG water	−6.32E+02	−1.03E+03	−1.43E+03	−1.83E+03	−3.03E+02	−3.28E+02	−3.53E+02	−3.77E+02
Terrestrial ecotoxicity	kg TEG soil	−1.64E+02	−2.69E+02	−3.76E+02	−4.84E+02	5.14E+01	4.48E+01	3.82E+01	3.17E+01
Terrestrial acid/nutri	kg SO ₂ eq	1.81E+00	1.76E+00	1.72E+00	1.68E+00	3.37E+00	3.36E+00	3.35E+00	3.35E+00
Land occupation	m ² org.arable	1.12E-02	−2.16E-03	−1.57E-02	−2.93E-02	−5.77E-02	−5.86E-02	−5.94E-02	−6.02E-02
Aquatic acidification	kg SO ₂ eq	1.19E+00	1.16E+00	1.12E+00	1.09E+00	3.34E+00	3.34E+00	3.34E+00	3.33E+00
Aquatic eutrophication	kg PO ₄ P-lim	−1.88E-03	−3.04E-03	−4.22E-03	−5.40E-03	−5.80E-03	−5.87E-03	−5.94E-03	−6.01E-03
Global warming	kg CO ₂ eq	1.62E+02	1.22E+02	7.97E+01	3.78E+01	3.44E+01	3.18E+01	2.93E+01	2.67E+01
Non-renewable energy	MJ primary	−2.87E+02	−4.48E+02	−6.11E+02	−7.73E+02	2.00E+02	1.90E+02	1.80E+02	1.70E+02
Mineral extraction	MJ surplus	2.74E-02	9.08E-03	−9.47E-03	−2.80E-02	−5.54E-01	−5.55E-01	−5.56E-01	−5.57E-01

assumed. With respect to landfilling, the carbon binding factors were drawn from AEA (2001). In particular, for the OF contained in the rMSW DF the value of 135 kg CO₂/ton of OF was assumed, for the SOF 10 kg CO₂/ton of OF was considered because this waste fraction is used as a surface dressing for landfill site restoration and decays aerobically, and for the OF in the Refuse 125.5 kg CO₂/ton of OF was taken because this waste fraction is subjected to the biostabilization process.

From the results shown in Table 9, the GWP in Scenarios 1 and 2 decreases by 45.2% and 40.1% of the total CO₂ eq. with the Scenario 2 that has the best performance.

4. Conclusions

LCA was applied in order to evaluate the environmental performance of the organic waste management systems implemented in the Umbria Region. Two waste treatment scenarios were analyzed: undifferentiated collection followed by the biostabilization of the organic fraction and its final disposal in landfill (Scenario 1) and source-separated collection of the organic fraction followed by the production of compost (Scenario 2). The described scenarios are also the typical organic waste management systems actually implemented in Italy.

Taking into account the assumptions and limitations of the study, results show that at midpoint level Scenario 1 has the best performance in ten of the fifteen impact categories considered in the IMPACT 2002+ method, while Scenario 2 is characterized by the lowest impact for Carcinogens, Land occupation, Aquatic eutrophication, Global warming, and Mineral extraction categories. This trend is maintained at endpoint level where Scenario 2 has the best score only for the Climate change damage category. In the Human health category, the most significant contributions are the particulate and ammonia emissions from the biostabilization process for the Scenario 2 and diesel use in the waste transport and landfill site, along with ammonia emissions from the biological treatment for the Scenario 1. The emission of ammonia plays also

an important role in the Ecosystem quality category for both scenarios, along with the aluminum emissions into the soil and atmosphere due to the diesel and electricity consumption in the waste transport and pretreatment for the Scenario 1. As regards the Climate change category, the major impacts are due to the GHG emissions from the landfill in the Scenario 1 and from the biostabilization process for the Scenario 2. At last, the impact of the Resources damage category is mostly due to the fossil fuel consumption in the waste transport and pretreatment for the Scenario 1 and to the electricity consumption in the biological treatment for the Scenario 2.

The sensitivity analysis points out that the inclusion in the LCA analysis of the carbon sequestration in the landfill and from the use of compost significantly decrease GHG emissions of both scenarios (Scenario 1: −45.2%; Scenario 2: −40.1%), while the increase of the biogas collection efficiency from the landfill allows to improve the performance of the Scenario 1, reducing the difference of GHG emissions between the Scenario 2 from 79% to 29% with the efficiency of 75%.

Therefore further research is necessary to better evaluate the biogas collection efficiency and fugitive GHG emissions associated with organic waste landfilling because well-managed landfill could be the best option for all environmental impacts. Considering the actual performance of the biostabilization process, the aerobic composting of the organic fraction could be the most favorable scenario only if a significant reduction of emissions into atmosphere (in particular, hydrogen sulfide, particulate, ammonia and NMVOC) will be achievable.

In conclusion, LCA applied to waste management systems has a great potential, especially in supporting the decisions of planners and companies that manage waste collection, transportation and recovery activities.

As a final remark, the study highlighted the importance to acquire primary data. The applicability of the LCA methodology is in fact strongly subordinated to the availability of data and information on the processes involved and requires that the competent bodies (public and private) provide for a continuously updated information system of collecting and organizing these data.

Table 9
Global warming potential of each scenario considering carbon sequestration.

Impact category	Unit	Reference case		Carbon sequestration	
		Scenario 1	Scenario 2	Scenario 1	Scenario 2
Global warming	kg CO ₂ eq	162.0	34.4	88.7	20.6

Acknowledgments

The authors would like to acknowledge Gesenu S.p.a. for providing part of the primary data used in preparing this study.

References

- AEA Technology, 2001. Waste Management Options and Climate Change: Final Report. AEA Technology Report for European Commission DG Environment. Available at: http://ec.europa.eu/environment/waste/studies/pdf/climate_change.pdf (accessed 04.08.14.).
- Amini, H.R., Reinhart, D.R., 2011. Regional prediction of long-term landfill gas to energy potential. *Waste Manag.* 31, 2020–2026.
- Amlinger, F., Peyr, S., Cuhls, C., 2008. Greenhouse gas emissions from composting and mechanical biological treatment. *Waste Manag. Res.* 26, 47–60.
- Andersen, J.K., Boldrin, A., Christensen, T.H., Scheut, C., 2010. Greenhouse gas emissions from home composting of organic household waste. *Waste Manag.* 30, 2475–2482.
- ARPA Umbria, 2013. Rapporto Rifiuti Urbani Umbria 2012. Available at: www.arpa.umbria.it/resources/documenti/Rapporto%20Rifiuti%202012.pdf (in Italian). (accessed 04.10.13.).
- Baky, A., Eriksson, O., 2003. System Analysis of Organic Waste Management in Denmark. Environmental Project No 822. Danish Environmental Protection Agency, Denmark.
- Belboom, S., Digneffe, J.M., Renzoni, R., Germain, A., Léonard, A., 2013. Comparing technologies for municipal solid waste management using life cycle assessment methodology: a Belgian case study. *Int. J. Life Cycle Assess.* 18, 1513–1523.
- Bernstad, A., la Cour Jansen, J., 2012. Review of comparative LCAs of food waste management systems – current status and potential improvements. *Waste Manag.* 32, 2439–2455.
- Blengini, G.A., 2008. Using LCA to evaluate impacts and resources conservation potential of composting: a case study of the Asti District in Italy. *Resour. Conserv. Recycl.* 52, 1373–1381.
- Boldrin, A., Andersen, J.K., Möller, J., Favoino, E., Christensen, T.H., 2009. Composting and compost utilization: accounting of greenhouse gases and global warming contributions. *Waste Manag. Res.* 27, 800–812.
- Bovea, M.D., Ibáñez-Forés, V., Gallardo, A., Colomer-Mendoza, F.J., 2010. Environmental assessment of alternative municipal solid waste management strategies. A Spanish case study. *Waste Manag.* 30, 2383–2395.
- Brambilla Pisoni, E., Raccanelli, R., Rotelli, G., Botta, D., Melià, P., 2009. Accounting for transportation impacts in the environmental assessment of waste management plans. *Int. J. Life Cycle Assess.* 14, 248–256.
- Buratti, C., Barbanera, M., Fantozzi, F., 2013. Assessment of GHG emissions of biomethane from energy cereal crops in Umbria, Italy. *Appl. Energy* 108, 128–136.
- Clavreul, J., Guyonnet, D., Christensen, T.H., 2012. Quantifying uncertainty in LCA modelling of waste management systems. *Waste Manag.* 32, 2482–2495.
- Colón, J., Martínez-Blanco, J., Gabarrell, X., Artola, A., Sanchez, A., Rieradevall, J., Font, X., 2010. Environmental assessment of home-composting. *Resour. Conserv. Recycl.* 54, 893–904.
- Colón, J., Cadena, E., Pognani, M., Barrera, R., Sánchez, A., Font, X., Artola, A., 2012. Determination of the energy and environmental burdens associated with the biological treatment of source-separated municipal solid wastes. *Energy Environ. Sci.* 5, 5731–5741.
- De Feo, G., Malvano, C., 2009. The use of LCA in selecting the best MSW management system. *Waste Manag.* 29, 1901–1915.
- Dong, J., Ni, M., Chi, Y., Zou, D., Fu, C., 2013. Life cycle and economic assessment of source-separated MSW collection with regard to greenhouse gas emissions: a case study in China. *Env. Sci. Poll. Res.* 20, 5512–5524.
- Ecoinvent, 2010. Ecoinvent Database v2.2. Swiss Centre for Life Cycle Inventories.
- Eriksson, O., Carlsson, M., Frostell, B., Björklund, A., Assefa, G., Sundqvist, J.O., Granath, J., Baky, A., Thyselius, L., 2005. Municipal solid waste management from a systems perspective. *J. Clean. Prod.* 13, 241–252.
- European Commission (EC), 2008. Waste Framework Directive, Directive 2008/98/EC of the European Parliament and of the Council of 19 November 2008 on Waste and Repealing Certain Directives.
- Finnveden, G., Johansson, J., Lind, P., Moberg, Å., 2005. Life cycle assessment of energy from solid waste—part 1: general methodology and results. *J. Clean. Prod.* 13 (3), 213–229.
- Gentil, E.C., Damgaard, A., Hauschild, M., Finnveden, G., Eriksson, O., Thorneloe, S., Kaplan, P.O., Barlaz, M., Müller, O., Matsui, Y., Li, R., Christensen, T.H., 2010. Models for waste life cycle assessment: review of technical assumptions. *Waste Manag.* 30 (12), 2636–2648.
- Hong, J., Li, X., Zhaojie, C., 2010. Life cycle assessment of four municipal solid waste management scenarios in China. *Waste Manag.* 30 (11), 2362–2369.
- Support JRC Scientific and Technical Reports ILCD, 2011. Supporting Environmentally Sound Decisions for Waste Management – a Technical Guide to Life Cycle Thinking (LCT) and Life Cycle Assessment (LCA) for Waste Experts and LCA Practitioners.
- ILD, 2006. Italian legislative decree No. 152, 03/04/2006 and subsequent amendments. In: Environmental Laws. Norme in Materia Ambientale. G.U. (suppl.) n. 88, 14/04/2006 (in Italian).
- IPCC, 2006. 2006 IPCC guidelines for national greenhouse gas inventories – vol. 5: waste. In: Eggleston, H.S., Buendia, L., Miwa, K., Ngara, T., Tanabe, K. (Eds.), National Greenhouse Gas Inventories Programme. IGES, Japan.
- IPCC, 2007. Mitigation of climate change. In: Metz, B., Davidson, O.R., Bosch, P.R., Dave, R., Meyer, L.A. (Eds.), Climate Change 2007: Contribution of the Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, first ed. Cambridge University Press, Cambridge & New York.
- ISO, 2006a. ISO 14044: Environmental Management – Life Cycle Assessment – Requirements and Guidelines. ISO International Organization for Standardization, Geneva, Switzerland.
- ISO, 2006b. ISO 14040: Environmental Management – Life Cycle Assessment – Principles and Framework. International Organization for Standardization, Geneva, Switzerland.
- ISPRA, 2013a. Rapporto Rifiuti Urbani. ISPRA Rapporti-N. 176/2013, ISBN 978-88-448-0596-8 (in Italian).
- ISPRA, 2013b. Italian Greenhouse Gas Inventory 1990–2011. National Inventory Report 2013.
- Jolliet, O., Margni, M., Charles, R., Humbert, S., Payet, J., Rebitze, G., Rosenbaum, R., 2003. IMPACT 2002+: a new life cycle impact assessment methodology. *Int. J. Life Cycle Assess.* 8, 324–330.
- Kong, D., Shan, J., Lacoboni, M., Maguin, S., 2012. Evaluating greenhouse gas impacts of organic waste management options using life cycle assessment. *Waste Manag. Res.* 30, 800–812.
- Laurent, A., Bakas, I., Clavreul, J., Bernstad, A., Niero, M., Gentil, E., Hauschild, M.Z., Christensen, T.H., 2014. Review of LCA studies of solid waste management systems – part I: lessons learned and perspectives. *Waste Manag.* 34, 573–588.
- Levis, J.W., Barlaz, M.A., 2011. Is biodegradability a desirable attribute for discarded solid waste? Perspectives from a national greenhouse gas inventory model. *Environ. Sci. Tech.* 45, 5470–5476.
- Martínez-Blanco, J., Colón, J., Gabarrell, X., Font, X., Sánchez, A., Artola, A., Rieradevall, J., 2010. The use of life cycle assessment for the comparison of biowaste composting at home and full scale. *Waste Manag.* 30, 983–994.
- Morris, J., Matthews, H.S., Morawski, C., 2013. Review and meta-analysis of 82 studies on end-of-life management methods for source separated organics. *Waste Manag.* 33 (3), 545–551.
- Niskanen, A., Värri, H., Havukainen, J., Uusitalo, V., Hörttanainen, M., 2013. Enhancing landfill gas recovery. *J. Clean. Prod.* 55, 67–71.
- Ponsá, S., Gea, T., Alerm, L., Cerezo, J., Sánchez, A., 2008. Comparison of aerobic and anaerobic stability indices through a MSW biological treatment process. *Waste Manag.* 28, 2735–2742.
- Righi, S., Oliviero, L., Pedrini, M., Buscaroli, A., Della Casa, C., 2013. Life cycle assessment of management systems for sewage sludge and food waste: centralized and decentralized approaches. *J. Clean. Prod.* 44, 8–17.
- Rodríguez, R., Ruyck, J.D., Díaz, P.R., Verma, V.K., Bram, S., 2011. An LCA based indicator for evaluation of alternative energy routes. *Appl. Energy* 88 (3), 630–635.
- Saer, A., Lansing, S., Davitt, N.H., Graves, R.E., 2013. Life cycle assessment of a food waste composting system: environmental impact hotspots. *J. Clean. Prod.* 52, 234–244.
- Smet, E., Van Langenhove, H., De Bo, I., 1999. The emission of volatile compounds during the aerobic and the combined anaerobic/aerobic composting of biowaste. *Atmos. Environ.* 33, 1295–1303.
- Smith, S.R., Jasim, S., 2009. Small-scale home composting of biodegradable household waste: overview of key results from a 3-year research programme in West London. *Waste Manag. Res.* 27, 941–950.
- Smith, A., Brown, K., Ogilvie, S., Rushton, K., Bates, J., 2001. Waste Management Options and Climate Change, Final Report. European Commission, DG Environment, Luxembourg.
- Spokas, K., Bogner, J., Chanton, J.P., Morcet, M., Aran, C., Graff, C., Moreau-Le Golvan, Y., Hebe, I., 2006. Methane mass balance at three landfill sites: what is the efficiency of capture by gas collection systems? *Waste Manag.* 26 (5), 516–525.
- US EPA, 2005. Landfill Gas Emissions Model (LandGEM) Version 3.02 User's Guide. Washington, DC.
- US EPA, 2006. Solid Waste Management and Greenhouse Gases: a Life-cycle Assessment of Emissions and Sinks. Available at: <http://www.epa.gov/climatechange/wycd/waste/downloads/fullreport.pdf> (accessed 26.08.14.).
- Weiser, C., Zeller, V., Reinicke, F., Wagner, B., Majer, S., Vetter, A., Thraen, D., 2014. Integrated assessment of sustainable cereal straw potential and different straw-based energy applications in Germany. *Appl. Energy* 114, 749–762.
- Wittmaier, M., Langer, S., Sawilla, B., 2009. Possibilities and limitations of life cycle assessment (LCA) in the development of waste utilization systems. Applied examples for a region in Northern Germany. *Waste Manag.* 29, 1732–1738.
- Yoshida, H., Gable, J.J., Park, J.K., 2012. Evaluation of organic waste diversion alternatives for greenhouse gas reduction. *Res. Cons. Recycl.* 60, 1–9.