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ORIGINAL PAPER

Quality evaluation and improvement of mechanically–biologically treated municipal solid waste in view of a possible recovery

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Abstract This study aimed to identify possibilities to improve the quality of the stabilized biowaste coming from the mechanical–biological treatment of municipal solid waste produced in Rome for possible recovery rather than landfilling. The waste sampled before and after the first and the fourth week of aerobic biodegradation as well as different particle size classes composing the biostabilized materials were characterized in order to investigate the content and distribution of contaminants. Results showed, firstly, that the stabilized biowaste was not biologically stable after 4 weeks of biostabilization process, presenting dynamic respiration index >1,000 mg O_2 kg VS^{-1} h^{-1}. Heavy metal (Cd, Cr, Cu, Ni, Pb, Zn) content fulfilled the requirements for utilization, but by contrast, their release in water phase was quite high and not complying with Italian regulatory limits for waste recovery. In order to raise the quality, the biological process management should be improved by increasing the water amount to add to the rotting waste since the output material had moisture (19.7 % WM) lower than the recommended content for an optimized aerobic biodegradation (>40 %). Furthermore, fractions having particle size higher than 10 mm, composed of high content of impurities and heavy metals, should be separated by obtaining a reduction in chemical–physical contamination. Finer waste flow (<10 mm) can be recovered in environmental remediation activities, whereas coarser fractions (>10 mm) can be used in waste-to-energy plants, given the great content of high calorific value materials and the compliance with characteristics defined for solid recovered fuels.

Keywords Biological stability · Heavy metals · Mechanical–biological treatment · Municipal solid waste · Particle size distribution · Stabilized biowaste

Introduction

Mechanical–biological treatment (MBT) of residual municipal solid waste (i.e., the remaining MSW after source separate collection) was firstly developed to ensure a reduction in biodegradable waste to be landfilled in order to meet the targets of the European Landfill Directive 1999/31/EC (European Commission 1999). Nowadays, a significant objective for MBT, alongside the environmentally sustainable landfill management, is to utilize significant percentages of the outputs for material and/or energy recovery (Farrell and Jones 2009; Di Lonardo et al. 2012a; Zaman 2013). One of the output of MBT is the stabilized biowaste (SBW) produced from the biological treatment, generally consisting of aerobic biostabilization, of the organic fraction mechanically separated from the input MSW. Nowadays, unlike other MBT outputs (such as refuse derived fuels, RDFs), official European standards regulating quality requirements of the stabilized biowaste and its possible use options do not exist despite hundreds of large scale MBT facilities of residual MSW were developed in Europe in the last two decades (Lornage et al. 2007; Barrena et al. 2009; Bayard et al. 2010; Ponsá et al. 2010; Tintner et al. 2010). There is only a draft of the European Commission, i.e., “Biological treatment of biowaste—2nd draft” (European Commission 2001), which states that the mechanical–biological treatment shall have the purpose of stabilizing and reducing the volume of the biodegradable waste in order to ensure that the resulting stabilized biowaste can either be used for ecological improvement or has reduced negative environmental impacts when landfilled. Moreover, the EC

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document specifies that Member states may authorize the use of SBW, fulfilling the requirements indicated in the document, as a component in artificial soils or in those land applications that are not destined to food and fodder crop production (such as final landfill cover, landscape restoration in old and disused quarries and mines, anti-noise barriers and road construction). Such statement came from the evidence that the content of non-compostable materials and heavy metals in stabilized biowaste, which lead to physical and chemical contamination, is higher than that in compost produced from source segregated waste, as observed by many authors (Leikam and Stegmann 1999; Whittle and Dyson 2002; Zennaro et al. 2005; Dimambro et al. 2007; Di Lonardo et al. 2012a; Karak et al. 2013). Modern mechanical separation technologies and process improvements significantly reduced the “impurities” and the metal content in stabilized biowaste compared with 30 years ago (Zennaro et al. 2005; Dimambro et al. 2007). Nevertheless, from a legal viewpoint, in many countries, stabilized biowaste remains waste (Baird et al. 2005; Farrell and Jones 2009), rather than a recoverable material, and its subsequent use needs to be subjected to appropriate regulatory measures (Baird et al. 2005; DEFRA 2013). As a result, there are only few studies assessing the environmental quality (e.g., the leaching behavior) and the reuse options of the MBT waste. The SBW utilizations proposed and investigated are mainly focused on the application in environmental remediation activities, e.g., as landfill final cover (Einola et al. 2008; Izzo et al. 2009; Huber-Humer et al. 2009; Angermeier et al. 2011) and in the restoration of degraded areas (contaminated sites, closed landfills, brownfields, etc.) (Baird et al. 2005; UK Environment Agency 2009). Specifically, it was observed that the main benefits of the SBW use as landfill cover material were the reduction in landfilled waste volume and the use of treated waste rather than natural soils that often are not available or too expensive, as well as the mitigation of greenhouse gas emissions from landfills when SBW is used as a support medium in methane oxidative biocovers. When applied to degraded soils, SBW was found to improve soil properties, such as water retention, porosity, nutrients and organic matter supplying. Nevertheless, all studies recommended to use a well-biostabilized material which otherwise could continue to lose organic mass, resulting, e.g., in increased concentrations of heavy metals. Furthermore, it is pointed out to take into account the risks related to the potential contaminants that can accumulate in the soil where biostabilized waste is applied and, through release and transfer mechanisms, move toward other receptors (groundwater and surface water, plants, animals and, last but not least, humans).

Italy has a long tradition in MBT, but emphasis on recovery of the outputs is mainly focused on the production of refuse derived fuel (RDF), nowadays defined as solid recovered fuel (SRF), with just some plants producing stabilized biowaste suitable for restricted applications (Partl and Cornander 2006). As a result, nowadays, SBW produced from Italian MBT plants is mainly landfilled. Specifically, data provided by the Italian Institute for the Environmental Protection and Research in “Municipal Solid Waste Report—Edition 2013” (ISPRA 2013) show that 84 % of the total quantity of stabilized biowaste produced in 2011 (i.e., 10 % of the total MSW amount treated by MBT plants equal to 7,852,057 Mg MSW/year) was landfilled. The Italian Decree 205/2010, which regulates waste field, delegates the definitions of quality requirements of stabilized biowaste and its uses complying with health and environmental protection to appropriate technical standards that still do not have been issued. Hence, a detailed investigation is essential for the development of technical guidelines, which could provide limit values for characterizing the quality of stabilized biowaste in relation to the specific intended use or disposal (ISPRA 2007).

Furthermore, in order to comply with waste hierarchy (European Commission 2008), the individuation and evaluation of management techniques aimed at ensuring SBW recovery represent one of the priorities of waste management strategies in Italy, as well as in the areas where MBT of MSW is in widespread use (such as Spain, United Kingdom and Austria; Partl and Cornander 2006). In view of such considerations, the present study aimed to assess the quality of the stabilized biowaste coming from the mechanical–biological treatment of residual MSW produced in Rome, in order to evaluate the potential sustainable recovery in alternative to landfilling. Previous characterizations performed during the years 2010–2011 on the stabilized biowaste coming from two MBT plants of Rome showed that such materials did not have a suitable quality for waste recovery both in terms of biological stability degree and in terms of physico–chemical properties (Di Lonardo 2013). Therefore, a further study on SBW quality was carried out by sampling the waste before and after the first and the fourth week of aerobic biodegradation in order to assess the evolution/changing of the investigated characteristics during the process. Moreover, different particle size classes composing the biostabilized materials were characterized in order to investigate the distribution of physical and chemical properties and the contribution to the contamination of the entire stabilized biowaste. The purpose was to identify possibilities to reduce pollutants and to improve the quality in view of a possible recovery, by acting on the treatment process. Such research study was conducted between the late 2011 and 2012 at the Sanitary-Environmental Engineering Laboratory, Department of Civil Engineering and Computer Science Engineering of the University of Rome “Tor Vergata.”

Materials and methods

MBT plant of Rome

In Lazio Region, part of waste management system, besides the source separate collection, was arranged for mechanical–biological treatment with seven MBT plants now in operation. In the year 2011, a percentage by weight equal to 32 % of MSW produced in this Italian region was treated by MBT facilities (ISPRA 2013). Furthermore, it has to be pointed out that, in 2012, the percentage of separate collection in Rome was still quite low and equal to 22.4 % of the total amount of MSW produced in this city (ISPRA 2013). As a result, residual MSWs feeding MBT plants were found to be composed by high percentage of biodegradable organic waste (food and garden waste, paper and cardboard, fine organic waste lower than 20 mm) equal to 60 % (Di Lonardo et al. 2012b).

In this study, waste materials were sampled in a mechanical–biological treatment plant whose maximum treatment capacity is 750 Mg MSW/day and which treats residual MSW coming from south areas of the city of Rome. The received MSW is discharged by the collection vehicles into a receiving area for temporary storage where a manual/mechanized pre-sorting of bulky materials (such as appliances, tires, furniture and mattresses) is carried out in order to avoid clogging phenomena in the downstream equipment. The receiving/storage area is covered and maintained under negative pressure conditions in order to minimize dust and odor emissions to the outside. The first treatment unit consists in bag braking and size reduction by means of hammer-mill shredders, after which the waste, placed on belt conveyors, is fed into trommel screens for size separation (grate spacing of 80 mm). From this latter process, the following two outputs are obtained:

- *biodegradable fraction* (undersize), presenting a high percentage of organic compounds and moisture content
- *dry fraction* (oversize), presenting a lower moisture content and typically enriched in materials characterized by a significant heating value (mainly plastics).

The dry fraction is processed by an air classification unit so to separate the light fraction (refuse derived fuel, RDF) from the heavy fraction (heavy scraps). Ferrous and non-ferrous metals are then separated from these two flows. After this step, densification or shredding of RDF and heavy scraps is carried out in order to facilitate their handling, storage and transport to incineration plant and landfill, respectively. At the same time, the biodegradable fraction, after metal removal by belt-type electromagnetic separators, is sent to a biostabilization basin where aerobic biodegradation occurs for 4 weeks at forced aeration conditions. Three augers moved by a crane have a dual function: turning over the material in order to keep proper free air space (pores) for aeration (avoiding the formation of anaerobic conditions, especially at the bottom of the basin) and moving the material along the basin. During the turning/moving, water is added to the material by nozzles mounted on the crane, in order to keep the water content favorable for the microbial activity. The stabilized output then is sieved in a trommel screen with a mesh opening of 20 mm (refining unit) in order to separate an oversized fraction mainly composed of plastics and inert materials (stabilization scraps which are landfilled) from the undersized fraction consisting of the final stabilized biowaste (SBW) that is not subjected to a further ripening phase. The average mass balance of the MBT plant calculated for the year 2012 showed that the biodegradable fraction feeding the biostabilization process was equal to 56.5 %, whereas the percentage of stabilized biowaste was equal to 19.9 % of the input MSW (100 %). Such quantity is daily disposed of in landfill.

Sampling procedure and analytical methods

During two characterization campaigns performed between the late 2011 and 2012, three macro-samples, consisting of the input material feeding the aerobic stabilization (untreated biowaste, UBW < 80 mm), the stabilized biowaste after 1 week (SBW I) and after 4 weeks (SBW IV) of treatment, prior to the refining unit, were taken. The UBW sample was taken from the conveyor belt which moves the biodegradable fraction to the stabilization basin. Partial quantity (increments) of the material was sampled at different time interval while the conveyor belt was moving by feeding the basin. For the sampling of SBW I and SBW IV (ISPRA 2007), the basin was divided into four parts transversely to the movement direction of biowaste, each area corresponding to a progressive week of process. Again, partial quantities of the materials were collected at four different points along the first area (first week of treatment, SBW I) and along the fourth area (fourth week of treatment, SBW IV) prior to the secondary sieving unit.

Afterward, the different collected increments were mixed in order to obtain the three macro-samples of UBW, SBW I and SBW IV, which then were homogenized and quartered.

Biological and physical–chemical characterization was performed to determine the following parameters: biological stability (determination of dynamic respiration index, DRI), particle size distribution, non-compostable material content, organic matter content (determination of volatile solids, VS and total organic carbon, TOC), heavy metal total content and release of contaminants in water phase (leaching test). A final quantity of roughly 20 kg for each sample was air-dried prior to performing all laboratory analysis, with the exception of the biological stability analysis where an additional amount of about 15 kg of as-received samples was used.

Biological stability, which measures the degradability of the readily available organic matter contained in the waste in a short time period (4 days) under standardized aerobic conditions (Adani et al. 2004), was analyzed by determining the dynamic respiration index (DRI), namely the absolute maximum rate of oxygen consumption due to microbial activity. DRI was measured and calculated according to the procedure reported in the Italian Standard UNI/TS 11184 (2006) by using a 30 l adiabatic respirometric reactor (Costech International Respirometer 3024). Prior to the beginning of the respiration test, moisture content (EN 2006) was determined on the as-received material.

The particle size analysis (ASTM 2007) was performed for SBW I and SBW IV samples. After quartering, an amount of roughly 5 kg for each air-dried sample (SBW I and SBW IV) was sieved by means of the following sieve sizes: 38.1, 25.4, 19.1, 10, 4, 2, 0.84 mm. Materials retained by each sieve were progressively weighed and the cumulative passing was calculated through Eqs. (1) and (2):

$$P_i = \sum_{n=1}^{i} R_i - R_i \tag{1}$$

$$\%P_i = 100 \frac{P_i}{\sum_{n=1}^{i} P_i} \tag{2}$$

where P_i is the weight, expressed in grams, of the material passed through the ith sieve and R_i is the weight, in grams, of the material retained by the ith sieve. The particle size distributions of SBW I and SBW IV were then determined by correlating the percentage by weight of cumulative passing with each sieve size. On the basis of the found particle size distributions, four particle size classes, namely A = [<2 mm], B = [2–10 mm], C = [10–38.1 mm] and D = [38.1–80 mm], were identified and prepared for the subsequent physical–chemical characterization.

The investigation on non-compostable material content was carried out only for coarser classes C and D, since, given the small size of classes A and B (<10 mm), it was difficult to perform the manual sorting of the different materials. In accordance with the ISPRA method 36 (2000), the different material categories (organics, paper/cardboard, textiles, plastics, composite packaging, wood, glass, metals, inert materials, hazardous) were manually sorted and weighed. The percentage material composition was then determined through Eq. (3):

$$\%F_i = 100 \frac{f_i}{\sum_{n=1}^{i} f_i} \tag{3}$$

where f_i is the weight, expressed in grams, of the ith material category.

The volatile solids (VS) content was determined by loss-on-ignition at 550 °C for 8 h (UNI 2006) on 10 g of sample grinded to 0.5 mm and pre-dried at 105 °C for 4 h. Total organic carbon (TOC) content was analyzed by means of Shimadzu SSM-5000A instrument on approximately 0.2 g of dried sample grinded to size lower than 200 µm (UNI 2002). VS and TOC were analyzed in triplicates. Heavy metal content in solid materials (total content) was determined in duplicates by acid digestion with HNO_3, H_2O_2 and HCl, according to EPA method 3050B (1996), for 1 g of dried sample grinded to 0.5 mm. The obtained solution, after filtration at 0.45 µm, was analyzed by inductively coupled plasma atomic emission spectrometry (Varian ICP-AES). For the leaching test, dried samples of UBW, SBW I and SBW IV (in duplicates) were shredded to a particle size lower than 4 mm, as required by the standard procedure EN 12457-2 (2002). A volume equal to 80 ml of deionized water was added to 8 g of each air-dried sample in order to obtain a liquid-to-solid ratio equal to 10, and bottles containing the mixture were stirred for 24 h. The obtained eluates were analyzed by determining the pH (Hanna Instrument pH meter) and, after filtration at 0.45 µm, the heavy metal concentrations (Varian ICP-AES analyzer) and the dissolved organic carbon (DOC) (Shimadzu TOC-V CPH/CPN analyzer).

Results and discussion

Biological stability

Table 1 shows the results of DRI, moisture content and VS (mean ± standard deviation) measured for UBW, SBW I and SBW IV. UBW was characterized by a significant biological reactivity, much higher than the limit value equal to 1,000 mg O_2 kg VS^{-1} h^{-1}, below which the material is considered biologically stable (European Commission 2001; Adani et al. 2004). During the biodegradation process, a reduction in reactivity was observed, even if DRI of SBW IV was found slightly higher than the limit. It has to be noticed that DRI of SBW I was slightly higher than DRI of UBW. A possible reason is that, after 1 week of biodegradation, the microbial activity and the related oxygen consumption were still quite high and comparable to the beginning of the process, whereas organic matter (volatile solids) slightly reduced (Table 1). As a result, the area under the DRI curve of UBW was greater than that of SBW I even if the latter had a higher max DRI (curve peak), as can be observed in Fig. 1 that shows the trends of DRIs hourly registered during the respiration test. Moisture content was quite high for the untreated waste because it was mainly composed of wet biodegradable organic waste (fines <20 mm plus food and garden waste) with a percentage by weight (wet matter, WM) roughly equal to 50 % of the total UBW (Di Lonardo et al. 2012b). Then, moisture tended to decrease in the 4 weeks of biodegradation likely

Table 1 Results of DRI, moisture content, VS and RA_4 measured and calculated for UBW, SBW I and SBW IV

	DRI ($mg\ O_2\ kg\ VS^{-1}\ h^{-1}$)	Moisture content (% WM)[a]	VS (% DM)[a]
UBW	2,546.1	50.1 ± 1.5	65.9 ± 0.8
SBW I	2,397.6	32.0 ± 2.2	59.2 ± 5.1
SBW IV	1,176.8	19.7 ± 0.5	55.3 ± 1.3
	$DRI_{cum\ 96\ h}$ ($mg\ O_2\ g\ DM^{-1}$)	DRI ($mg\ O_2\ kg\ DM^{-1}\ h^{-1}$)	RA_4^b ($mg\ O_2\ g\ DM^{-1}$)
UBW	113.8	1,635.7	103.3 ± 1.5
SBW I	78.4	1,522.4	96.1 ± 1.4
SBW IV	51.7	681.7	43.1 ± 0.2

[a] Values expressed as mean ± standard deviation

[b] Values calculated by means of the regression equation provided by Scaglia et al. (2010)

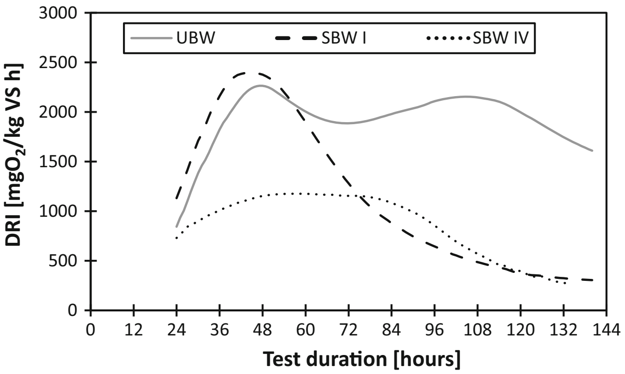

Fig. 1 Trends of DRIs hourly registered during the respiration test and averaged over 24 h

because of the evaporation occurred due to the high temperature reached (50–70 °C). Furthermore, it was presumed that a small amount of water was added during the process since the moisture of SBW IV was quite low (19.7 % WM), and the same was observed in previous characterization campaigns performed during the years 2010–2011 (Di Lonardo 2013). Such fact might have slowed down the biodegradation kinetic by causing a low biostabilization of SBW at the end of the 4 weeks, as confirmed by the DRI value, since the optimum moisture content for an aerobic biological process has to be higher than 40 % WM (Sirini et al. 2009).

In other European countries, such as Austria, biological reactivity of waste is analyzed by means of a different type of respiration test that measures the cumulative oxygen uptake within a period of 4 days (RA_4, Binner et al. 2012). On the basis of the DRI values hourly registered during the respiration test, the cumulative oxygen consumption after 96 h was calculated (DRI_{cum96h}) for the three samples and results are shown in Table 1. Moreover, Scaglia et al. (2010) found a correlation between the two respiration indices (DRI and RA_4) and derived the following regression equation:

$$\mathrm{DRI} = (15.85 \pm 0.34) \cdot \mathrm{RA}_4 - (1.14 \pm 10.75) \tag{4}$$

where DRI and RA_4 are expressed as $mg\ O_2\ kg\ DM^{-1}\ h^{-1}$ and $mg\ O_2\ g\ DM^{-1}$, respectively.

Equation (4) was used to calculate RA_4 for UBW, SBW I and SBW IV as reported in Table 1.

DRI_{cum96h} and RA_4 showed to be quite similar, and both indices were found to be much higher than the limit value equal to 7 $mg\ O_2\ g\ DM^{-1}$ set by the Austrian Landfill Ordinance (Binner et al. 2012), indicating a high reactivity of the material even after 4 weeks of biodegradation. It has to be observed that the Austrian limit is based on a longer duration of biological treatment as carried out in Austrian MBT plants, namely 2–6 weeks of intensive decomposition treatment plus a ripening phase ranging from 6 to 12 weeks (Tintner et al. 2010), unlike MBT plants of Rome. Therefore, the stabilized biowaste should be subjected to a downstream ripening phase, otherwise strong pollutant loading may occur when utilized (Di Lonardo et al. 2014).

Particle size distribution and material composition

Particle size analysis was carried out in order to investigate the distribution of physical and chemical properties among different particle size classes of the stabilized biowaste (Zhao et al. 2012). In Fig. 2, the particle size distributions of SBW I and SBW IV are shown and compared. These materials were found to be quite coarse given the low percentages of sizes below 2 mm (approximately 9–14 % DM). Differences between SBW I and SBW IV are clearly visible. More specifically, SBW IV was characterized by higher percentages of material at sizes in the range 2–38.1 mm than SBW I. This indicated the progressive degradation of the organic matter during the stabilization process, which caused a size reduction in the material (Lornage et al. 2007).

On the basis of the found particle size distributions, four particle size classes, namely A = [<2 mm], B = [2–10 mm],

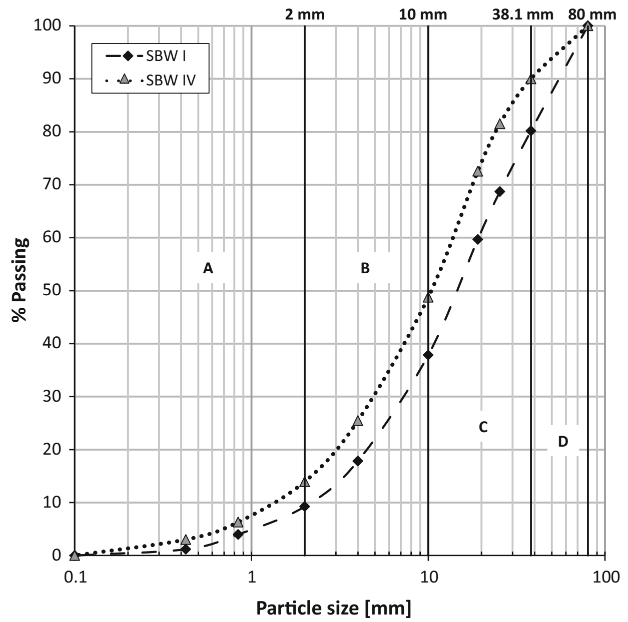

Fig. 2 Particle size distribution of SBW I and SBW IV

C = [10–38.1 mm] and D = [38.1–80 mm], were identified (see Fig. 2). The percentages by weight, on a dry matter (DM) basis, of the four classes were: A = 9.2 %, B = 28.6 %. C = 42.3 %, D = 19.8 % for SBW I; A = 13.9 %, B = 34.8 %. C = 41.2 % and D = 10.0 % for SBW IV. It is more evident that the latter was composed by higher percentages of particle size fractions below 10 mm than SBW I.

The determination of the material composition was carried out in order to investigate the content of non-biodegradable materials, i.e., plastics, glass, metals, inert and hazardous materials, which may cause a physical contamination (Baird et al. 2005; Dimambro et al. 2007; Farrell and Jones 2009; Montejo et al. 2010), as well as a chemical contamination due to high heavy metal content (as thereafter discussed) by compromising the final quality of the stabilized biowaste. As previously mentioned, the analysis regarded only classes C and D of SBW I and SBW IV. However, by a visual inspection, classes A and B showed to be mainly composed of fine organic materials (food and garden waste and small pieces of paper).

Results are reported in Table 2, and it can be observed that for both SBW I and SBW IV, class D showed a very low content of organics (<5 %) compared with class C (25 % for SBW I and 15 % for SBW IV). Furthermore, class D was composed of high quantities of slow and non-biodegradable materials, i.e., paper/cardboard, plastics, glass and inert materials, by total percentages roughly equal to 80 % for SBW I and 90 % for SBW IV. It has to be pointed out that, although paper is a biodegradable material, it is known to have relatively slow kinetics of biodegradation under aerobic conditions (Komilis 2006), especially if compared to the duration of the biostabilization treatment (4 weeks). Hence, this might have influenced the performance of the biodegradation by reducing the rate of the whole process (Lornage et al. 2007; Bayard et al. 2010; Montejo et al. 2010). However, as previously described, "impurities" are reduced in the output of the stabilization process by means of the secondary mechanical sieving at 20 mm. Concern should be focused on the presence of relative high quantities of glass fragments and inert materials (brick and ceramic pieces) in class C of SBW IV (roughly 25 % glass and 11 % inerts) that in part (size < 20 mm) are not diverted from the undersize of the secondary sieving unit.

Table 2 Material composition of PS classes C and D for SBW I and SBW IV (values expressed in % DM)

Categories	SBW I		SBW IV	
	C	D	C	D
Organic	25.1	4.3	14.9	2.7
Paper/cardboard	26.6	35.7	18.9	32.6
Textiles	1.0	2.5	5.1	3.3
Plastics	10.7	21.1	15.6	17.6
Comp. packaging	0.0	2.0	0.0	0.0
Wood	4.8	14.5	2.8	4.8
Glass	19.4	2.7	24.7	23.3
Metals	1.4	2.7	6.9	0.9
Inert materials	10.5	14.5	11.1	14.8
Hazardous	0.5	0.0	0.0	0.0
Total	100	100	100	100

Organic matter content

The organic matter content of total UBW, SBW I and SBW IV, as well as of the particle size classes, was determined by measuring the volatile solids (VS) and the total organic carbon (TOC), and results are shown in Table 3 (mean ± standard deviation). Comparing UBW, SBW I and SBW IV, low reduction in VS and TOC content during biodegradation process can be observed (Leikam and Stegmann 1999; Lornage et al. 2007; Barrena et al. 2009; Bayard et al. 2010). This is because VS and TOC take into account the overall organic matter, namely the non-degradable, slowly and readily degradable. In fact, residual MSW of Rome aimed to MBT, showed to have relative high content of non- and not readily degradable organic materials (plastics and paper), as reported by Di Lonardo et al. (2012b), and a significant percentage quantity of such materials passed through the primary mechanical sieve. For the same reason, class D (enriched in plastics and paper/cardboard, as previously discussed), for both SBW I and SBW IV, showed a very high content of VS (Table 3), as also found by Zennaro et al. (2005) and Bayard et al. (2010) for coarse fractions of MSW undergoing MBT. All other values of VS and TOC showed to be quite comparable among the different particle size classes. In order to better understand how organic matter was distributed among the different particle size classes, mass balances in terms of VS and TOC were calculated taking into account the percentages by dry weight found for each class and the mass loss compared to the untreated sample (UBW), as shown in Fig. 3. VS and TOC showed to have roughly the same distribution in SBW I and in SBW IV (Fig. 3), proving the correlation between these two parameters (Bayard et al. 2010) as they both are directly related to the organic carbon content (Barrena et al. 2009). Differences between SBW I and SBW IV can be noticed since VS and TOC increased in classes A and B and decreased or kept roughly constant in classes C and D for SBW IV compared to SBW I. This confirmed the reduction in the particle size due to degradation of organic matter. Regarding SBW IV, the highest content of organic matter was found in classes B and C. Nevertheless, it has to be pointed out that class C was composed of paper/cardboard and plastics by approximately 35 % (see Table 2) that highly contribute to increase the VS and TOC values, whereas organic matter in class B was mainly related to fine readily degradable organics.

Table 3 VS and TOC (mean ± standard deviation) content measured in UBW, SBW I and SBW IV and in PS classes of SBW I and SBW IV

	VS (% DM)	TOC (% DM)
UBW	65.9 ± 0.8	27.8 ± 0.3
SBW I	59.2 ± 5.1	23.7 ± 0.1
SBW IV	55.3 ± 1.3	23.6 ± 0.9
SBW I classes		
A	45.6 ± 0.6	21.3 ± 0.2
B	49.9 ± 1.3	24.9 ± 0.3
C	59.7 ± 1.0	23.7 ± 2.8
D	77.5 ± 0.0	23.1 ± 0.5
SBW IV classes		
A	46.5 ± 0.1	22.0 ± 0.1
B	47.3 ± 1.5	22.7 ± 1.7
C	49.3 ± 2.1	25.3 ± 0.6
D	74.7 ± 2.2	21.8 ± 1.0

Heavy metal total content

Heavy metals (HM) investigated in this work were cadmium (Cd), chromium (Cr), copper (Cu), lead (Pb), nickel (Ni) and zinc (Zn), i.e., those considered of greatest concern for a possible downstream environmental application (they tend to bioaccumulate, causing short or long-term toxic effects to

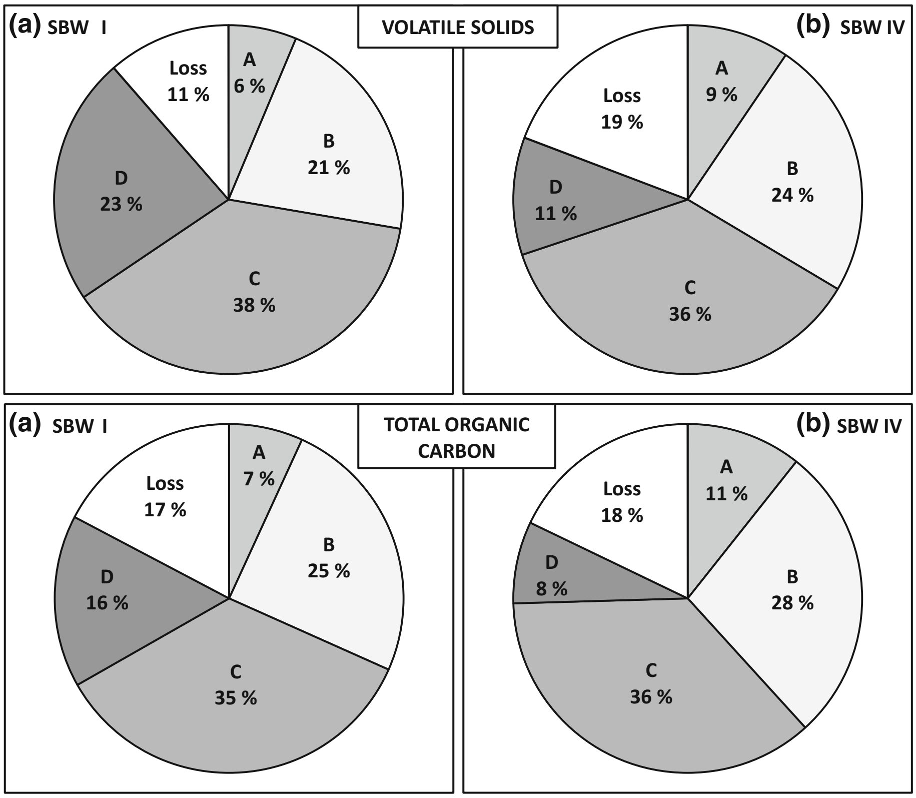

Fig. 3 Mass balance on the basis of VS and TOC content in the particle size classes composing **a** SBW I, **b** SBW IV

Table 4 Total heavy metal content in UBW, SBW I, SBW IV and in SBW IV PS classes and limit values (values expressed as mean ± standard deviation in mg/kg DM)

Samples	Cd	Cr	Cu	Ni	Pb	Zn
UBW	0.3 ± 0.03	15.7 ± 4.4	41.2 ± 0.5	12.5 ± 3.3	53 ± 2.1	226.9 ± 1.6
SBW I	0.3 ± 0.03	21.2 ± 0.8	97.5 ± 32	25.4 ± 7.7	163.8 ± 26	241.9 ± 30.5
SBW IV	0.5 ± 0.02	15.1 ± 1.3	81.8 ± 1.2	27.8 ± 9.9	175.8 ± 4.3	305.7 ± 10.3
Classes						
A	0.5 ± 0.009	79.9 ± 6.4	154.5 ± 17.1	49.3 ± 2.2	499.8 ± 41.9	388.8 ± 7.9
B	0.3 ± 0.2	39.9 ± 24.1	91.3 ± 11.4	36.6 ± 19.3	188.7 ± 26.9	296.5 ± 63.4
C	0.3 ± 0.06	15 ± 2.1	79.9 ± 2.1	18.2 ± 1.5	188.1 ± 0.7	275.1 ± 26.5
Limits						
EC 2001	5	600	600	150	500	1,500
ACO (2001)	3	250	500	100	200	1,800
SEPA 2005	3	400	200	100	200	1,000

organisms in the environment), and indeed, they are the most commonly regulated in the field of organic waste (Whittle and Dyson 2002). Table 4 shows the total heavy metal content (mean ± standard deviation) found in UBW, SBW I and SBW IV, as well as in the particle size classes of the latter. Since class D was found to be mainly composed of non-biodegradable coarse materials (>38.1 mm), which are separated from the output of the stabilization process by means of the secondary mechanical screening unit, heavy metals content in this class was not determined. The results are compared with limits reported by the European Commission document on biowaste (European Commission 2001), previously mentioned, by the Austrian Compost Ordinance (ACO 2001) which identify the class B of compost as suitable for landfill cover, as well as by the Scottish Environmental Protection Agency (SEPA) standard (Baird et al. 2005) which states that “If mixed waste compost (i.e., stabilized biowaste) is able to achieve the indicative standard, then it may be possible to consider applying the material to land subject to risk assessment procedures on a site by site basis.”

Heavy metals do not degrade and volatilized during the aerobic biological process (van Praagh et al. 2009); therefore, constant concentrations might be expected. Results showed some differences between the three analyzed samples, and this was mainly due to the heterogeneity of UBW, SBW I and SBW IV. Such variations might be reduced by sampling the same input material during the treatment rather than sampling at the same time at different stages (van Praagh et al. 2009). To a lesser extent, besides heterogeneity, another factor might have an influence on the HM changes in concentration during the biostabilization. Specifically, the loss of mass due to degradation of organic matter causes an increase in the heavy metal concentration in the biostabilized waste (Zennaro et al. 2005; Dimambro et al. 2007; van Praagh et al. 2009). As a result, metals showed to be more concentrated in the biostabilized materials (SBW IV) comparing with the untreated biowaste (UBW), with the exception of Cr which had roughly the same concentration in UBW and SBW IV. In order to overcome the influence of carbon loss on metal content, HM concentrations were recalculated according to Eq. 5, proposed by Amir et al. (2005):

$$C_{\mathrm{corr}} = C_i \cdot \left(\frac{100 - \mathrm{VS}_{\mathrm{ini}}}{100 - \mathrm{VS}_i} \right) \tag{5}$$

where C_i is the measured heavy metal concentration (mg/kg DM) at ith week of the process, and $\mathrm{VS}_{\mathrm{ini}}$ and VS_i (% DM) are the volatile solids of the untreated material and of the biostabilised waste at ith week, respectively. The HM corrected concentrations are shown in Fig. 4a, and it can be observed that metal content was lower in SBW I and SBW IV compared with UBW. This was probably due to metal loss through percolation (Whittle and Dyson 2002; Amir et al. 2005; Castaldi et al. 2006).

Comparing HM total content of SBW IV with limit values, it can be seen that requirements of EC document, Austrian Compost Ordinance and SEPA standard were all fulfilled. Therefore, the stabilized biowaste from the MBT plant of Rome can be potentially utilized in environmental remediation applications (for non-food production).

Regarding PS classes of SBW IV, results showed that the highest concentration of HM was found in the finest class A, in agreement with other studies (Zennaro et al. 2005; Zhao et al. 2012). Furthermore, it can be observed that Pb concentration in class A approached the EC document limit value and exceeded the Austrian Ordinance and SEPA limit values. Some authors (Zennaro et al. 2005; Zhao et al. 2012) argued that by eliminating the finest fraction, a reduction in heavy metal total content in the stabilized biowaste could be obtained without great losses in terms of mass and organic matter. This is true also in this case since SBW IV was composed of class A by roughly 14 % by dry weight and this class had organic content slightly lower than that in class B (see Table 3). Nevertheless, the issue is on how separating such fraction. For instance, a downstream mechanical sieving would not be very effective since, for wet materials, the fine particles (<2 mm) would tend to be "glued" to the coarser materials.

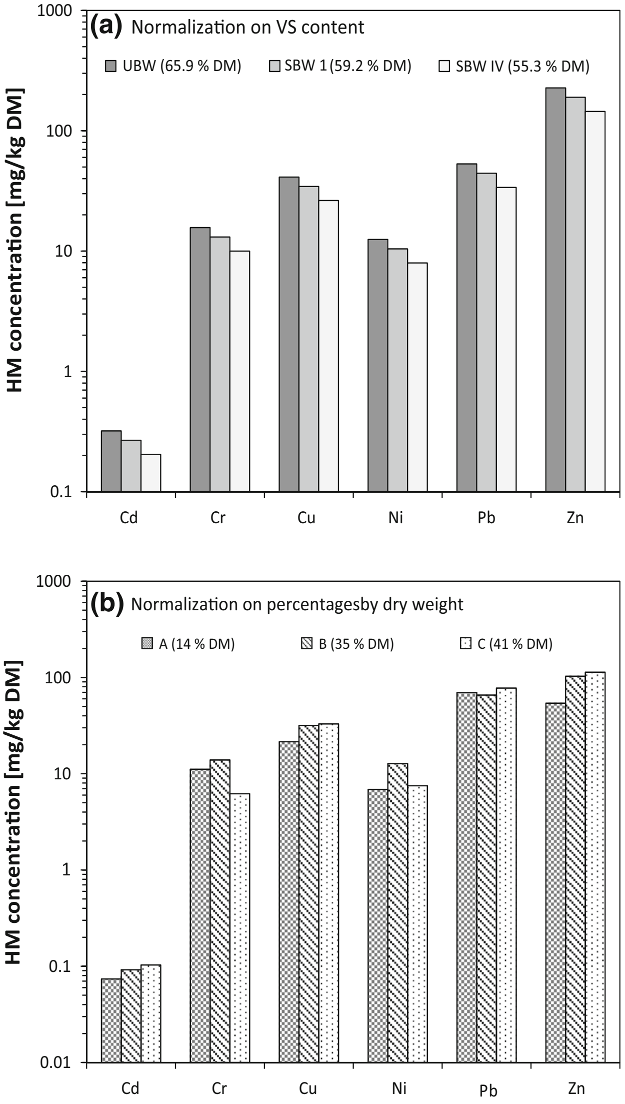

Fig. 4 Metal normalized concentrations on the basis of **a** VS content in SBW I and SBW IV and **b** the percentage by dry weight of SBW IV PS classes

In order to better evaluate the distribution of heavy metal content among the PS classes of SBW IV, the HM concentrations were recalculated and normalized on the basis of the percentage by dry weight found for each class, as shown in Fig. 4b. In this case, class C showed to have the highest metal content, with the exception of Cr and Ni. Therefore, the separation of such class from SBW IV could be taken into consideration also because, as shown by the material composition, class C was composed by 60 % of non-compostable organic materials (see Table 2) which might cause a physical contamination in the case of SBW recovery in environmental remediation. Then, materials having particle size higher than 10 mm (classes C and D), once mechanically separated, can be used in waste-to-energy plants rather than landfilling. In fact, in two studies (Di Lonardo et al. 2012c; Franzese et al. 2013), it was observed that the characteristics of the stabilization scraps coming from the same MBT plant were consistent with requirements set for the different classes defined for the solid recovered fuel (SRF) (EN 2011).

Leaching behavior

Generally, regulations on composted/biostabilized waste set limits on total heavy metal content and restrictions on leaching behavior are not reported. The determination of the total composition does not provide useful information about the potential release of contaminants in the environment (van der Sloot et al. 2004), which could occur if, for example, waste is applied to land. In Italy, limits on waste leaching are set by the Ministerial Decree (MD) 186/2006, established for the regulation of non-hazardous waste recovery. Table 5 shows the results of the leaching tests carried out for UBW, SBW I and SBW IV, and limit values of MD 186/2006 are reported as well. It has to be observed that, regarding SBW IV, only Cd and Zn fulfilled the limit values set by MD 186/2006; hence, from this point of view, the stabilized biowaste could not be recovered, and this is in contrast with what was found for total heavy metal content.

Table 5 Leachate characteristics (mean ± standard deviation) of UBW, SBW I and SBW IV and limit values (metal concentrations expressed in µg/l)

Parameters	UBW	SBW I	SBW IV	MD 186/2006
Cd	1.3 ± 0.2	1.1 ± 0.04	2.3 ± 0.1	5
Cr	37 ± 1.3	60 ± 1.3	100 ± 4.8	50
Cu	404 ± 4.1	560 ± 42.1	702 ± 21.9	50
Ni	115 ± 1.5	269 ± 12.9	316 ± 11	10
Pb	26 ± 3.6	97 ± 4.7	202 ± 10.7	50
Zn	1,433 ± 168.9	1,406 ± 20	1,516 ± 21.4	3,000
DOC (mg/l)	4,912.7 ± 673.8	2,317.3 ± 303.6	1,561 ± 191.2	–
Cl^- (mg/l)	748.6 ± 32.6	895.0 ± 48.8	610.2 ± 40.7	100
pH	6.1 ± 0.05	6.6 ± 0.03	6.8 ± 0.01	5.5–12

The release of contaminants in water phase is generally governed by the variation in pH (Whittle and Dyson 2002; van der Sloot et al. 2004). Furthermore, in the organic matter rich matrices, as in the case of biowaste, the role of solid organic matter and dissolved organic matter (DOC) is a key factor in the transfer of inorganic contaminants to the water phase through the formation of soluble and insoluble complexes (van der Sloot et al. 2004; van Praagh et al. 2009; Karak et al. 2013). Since the pH of UBW, SBW I and SBW IV eluates was found to be around neutral values (Table 5) and did not significantly vary (as also observed by Makan et al. 2014), the release of metals was minimally influenced by this parameter (Whittle and Dyson 2002); therefore, DOC was likely the most dominant factor controlling it (van der Sloot et al. 2004). In order to evaluate how release of contaminants changed during the aerobic biodegradation process, the heavy metals and organic carbon (OC) percentage release was calculated by means of Eq. 6, as follows:

$$\%\mathrm{release} = 100 \cdot C_{\mathrm{leach}} \cdot 10/C_{\mathrm{sol}} \qquad (6)$$

where C_{leach} is the concentration (mg/l) measured in the eluates of the three samples, 10 is the liquid-to-solid ratio (l/kg DM) and C_{sol} is the total content in solid materials (mg/kg DM). Results are shown in Fig. 5. The percentage release of the organic carbon reduced during the process due to the degradation of the organic matter. Similarly, Zn release was found to decrease since it tended to be bound to solid organic matter and to insolubilize (Karak et al. 2013). Differently, the release of Cu, Cr, Ni and Pb increased at the end of the process probably because of the formation of soluble compounds with DOC (Amir et al. 2005; Castaldi et al. 2006; van Praagh et al. 2009; Karak et al. 2013). It has to be observed that the Pb percentage release was the lowest one (≤1 %) comparing with the other metals since lead is known to be almost water-insoluble (Petruzzelli et al. 1994). Finally, the Cd release was found roughly constant during the 4 weeks of biostabilization. The influence of DOC on heavy metals release will need to be deeper investigated through pH dependence test and percolation test as a basis for the development of a risk assessment procedure to apply for any use of the stabilized biowaste in environmental remediation applications.

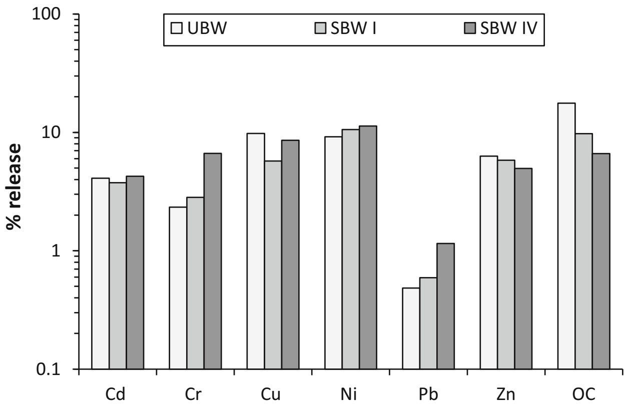

Fig. 5 Comparison of HM and organic carbon (OC) percentage release between UBW, SBW I and SBW IV

Conclusion

Heterogeneous biowaste streams, i.e., the biodegradable fraction of municipal solid waste, pose environmental risks while being a potential feedstock resource for producing valuable materials. Nevertheless, only few studies assessing the environmental quality (e.g., the leaching behavior) and the reuse options of the MBT waste were found. Therefore, this work aimed to identify possibilities to reduce pollutants and to improve the quality of the stabilized biowaste coming from the mechanical–biological treatment of MSW produced in Rome in view of a possible recovery, by acting on the treatment process. It was observed that the stabilized biowaste was still biologically reactive after 4 weeks of biodegradation process, confirming what was found in previous investigations. Therefore, the biological process management should be improved specifically by increasing the water amount to add to the rotting waste since it was found that the output material had moisture lower than the recommended content for an optimized aerobic biodegradation process (>40 %). Furthermore, comparing with a different biological stability limit and the type of biological process carried out in Austrian MBT facilities, it was deduced that an additional and downstream ripening phase should be carried out in order to obtain a highly biostabilized waste leading to a reduction in the pollutant loading when utilized.

The metals total content showed to slightly increase at the end of the biostabilization process due to the degradation of the organic matter but fulfilled the requirements set by some European standards for a possible recovery. However, in order to further improve the SBW quality, fractions having particle size higher than 10 mm, which were found composed of high content of impurities and heavy metals, should be separated (by means of mechanical sieving) by obtaining a whole reduction in chemical–physical contamination without great losses in terms of mass and organic matter. The finer fractions of SBW (<10 mm) can be recovered in environmental remediation activities, whereas the coarser materials (>10 mm) can be used in waste-to-energy plants rather than landfilling, given the great content of high calorific value materials and the compliance with characteristics defined for the solid recovered fuels. Contaminants release in water phase showed to increase for some metals (i.e., Cu, Cr, Ni and

Pb) during the biostabilization comparing with the untreated waste, due to the significant affinity with the dissolved organic carbon. Furthermore, heavy metal concentrations in the eluate of the stabilized waste showed to not fulfill the Italian limits set for non-hazardous waste recovery, and this was in contrast with what was found for the heavy metal total content. Hence, from this point of view, the only feasible utilization for the stabilized biowaste showed to be in monitored environments, such as in landfill sites as cover material, where the release of contaminants is controlled through the collection of the leachate and its subsequent treatment. Other uses, such as in soil applications for landscape restoration, may be considered under the constraint to apply a risk assessment procedure on a site-specific level in order to evaluate the potential risk for receptors (groundwater and surface water, plants, animals and humans) and to establish at which acceptable levels of pollutants, taking also into account the migration routes, such land application is not harmful to the environment and the human health. Therefore, the observations and evaluations highlighted in the present work could be useful in the development of such risk assessment.

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Abbreviations

DRI	Dynamic respiration index
DM	Dry matter
HM	Heavy metals
MBT	Mechanical biological treatment
MSW	Municipal solid waste
PS	Particle size
RA	Respiration activity
RDF	Refuse derived fuel
SBW	Stabilized biowaste
SRF	Solid recovered fuel
UBW	Untreated biowaste
WM	Wet matter

References

ACO (2001) Verordnung des Bundesministers für Land- und Forstwirtschaft, Umwelt und Wasserwirtschaft über Qualitätsanforderungen an Komposte aus Abfällen (Kompostverordnung). Verordnung nr 292, emitted 14 August 2001

Adani F, Confalonieri R, Tambone F (2004) Dynamic respiration index as a descriptor of the biological stability of organic wastes. J Environ Qual 33:1866–1876

Amir S, Hafidi M, Merlina G, Revel JC (2005) Sequential extraction of heavy metals during composting of sewage sludge. Chemosphere 59:801–810

Angermeier R, Tintner J, Smidt E, Ottner R, Matiasch L, Binner E, Böhm K (2011) Development of mechanically biologically treated municipal solid waste under different vegetation types. J Environ Eng 137(5):340–346

ASTM (2007) Standard test method for particle-size analysis of soils. ASTM Standard D422-63, ASTM International, West Conshohocken, PA. doi:10.1520/D0422-63R07

Baird J, Hutton M, Savage A, Hipkin A, Cruz P, MacLeod I (2005) The use of stabilised biowaste in the restoration of former landfill sites. Environment Group Research Report 2005/03, Scottish Executive, Edinburgh, UK

Barrena R, D'Imporzano G, Ponsá S, Gea T, Artola A, Vázquez F, Sánchez A, Adani F (2009) In search of a reliable technique for the determination of the biological stability of the organic matter in the mechanical–biological treated waste. J Hazard Mater 162(2–3):1065–1072

Bayard R, de Araújo Morais J, Ducom G, Achour F, Rouez M, Gourdon R (2010) Assessment of the effectiveness of an industrial unit of mechanical–biological treatment of municipal solid waste. J Hazard Mater 175(1–3):23–32

Binner E, Böhm K, Lechner P (2012) Large scale study on measurement of respiration activity (AT_4) by Sapromat and OxiTop. Waste Manag 32(10):1752–1759

Castaldi P, Santona L, Melis P (2006) Evolution of heavy metals mobility during municipal solid waste composting. Fresenius Environ Bull 15(9b):1133–1140

DEFRA (Department for Environment Food and Rural Affairs) (2013) Mechanical biological treatment of municipal solid waste. http://www.defra.gov.uk/publications/. Accessed 21 July 2014

Di Lonardo MC (2013) Evaluations for a possible recovery/utilisation of mechanically–biologically treated municipal solid waste by means of different operating conditions and retaining times. Dissertation, University of Rome "Tor Vergata"

Di Lonardo MC, Lombardi F, Gavasci R (2012a) Characterization of MBT plants input and outputs: a review. Rev Environ Sci Biotechnol 11:353–363. doi:10.1007/s11157-012-9299-2

Di Lonardo MC, Lombardi F, Gavasci R, Rocca S (2012b) Characterisation of MSW organic fraction flow separated and treated in a mechanical–biological treatment plant. In: Proceedings of SIDISA 2012 (book of abstracts), Sustainable Technology for Environmental Protection, Milano, Italy, 26–29 June 2012

Di Lonardo MC, Lombardi F, Gavasci R (2012c) From RDF to SRF. Assessments and perspectives for RDF producer and/or user plants. In: Proceedings of ISWA World Solid Waste Conference 2012, Firenze, Italy, 17–19 September 2012

Di Lonardo MC, Binner E, Lombardi F (2014) Investigations on biological stability degree of mechanically–biologically treated MSW produced in Rome. In: Proceedings of EURASIA Waste Management Symposium, Istanbul, Turkey, 28–30 April 2014, pp 39–46

Dimambro ME, Lillywhite RD, Rahn CR (2007) The physical, chemical and microbial characteristics of biodegradable municipal waste derived compost. Compost Sci Util 15(4):243–252

Einola J-KM, Karhu AE, Rintala JA (2008) Mechanically–biologically treated municipal solid waste as a support medium for microbial methane oxidation to mitigate landfill greenhouse emissions. Waste Manag 28(1):97–111

EN (2002) Characterization of waste-leaching: compliance test for leaching of granular waste materials and sludges—Part 2. One-stage batch test at a liquid to solids ratio of 10 l/kg for materials with particle size below 10 mm (with or without size reduction). CEN 12457-2, European Committee for Standardization (CEN), Brussels, Belgium

EN (2006) Characterization of waste: calculation of dry matter by determination of dry residue or water content. CEN 14346, European Committee for Standardization (CEN), Brussels, Belgium

EN (2011) Solid recovered fuels: specifications and classes. CEN 15359, European Committee for Standardization (CEN), Brussels, Belgium

EPA (US Enviromental Protection Agency) (1996) Method 3050B: acid digestion of sediments, sludges, and soils. http://www.epa.gov/osw/hazard/testmethods/sw846/pdfs/3050b.pdf. Accessed 17 January 2012

European Commission (1999) Council Directive 1999/31/EC of 26 April 1999 on the landfill of waste. Official Journal of the European Union L 182/1

European Commission (2001) Second working document of the EU commission on biological treatment of biowaste. DG ENV.A.2/LM, 12 February 2001, Bruxelles, Belgium

European Commission (2008) Directive 2008/98/EC of the European Parliament and of the Council of 19 November 2008 on waste and repealing certain Directives. Official Journal of the European Union L 312/3

Farrell M, Jones DL (2009) Critical evaluation of municipal solid waste composting and potential compost markets. Bioresour Technol 100(19):4301–4310

Franzese M, Lombardi F, Gavasci R, Di Lonardo MC (2013) From RDF to SRF. Characterization and first application of national guidelines in a MBT plant of Rome. In: Proceedings of ISWA World Congress 2013, Vienna, Austria, 7–11 October 2013. ISBN: 978-3-200-03229-3

Huber-Humer M, Röder S, Lechner P (2009) Approaches to assess biocover performance on landfills. Waste Manag 29:2092–2104

ISPRA (Institute for the Environmental Protection and Research) (2013) Rapporto Rifiuti Urbani 2013: Edizione 2013 (Municipal Solid Waste Report: Edition 2013). Report n. 176/2013. ISBN: 978-88-448-0596-8

ISPRA (Institute for the Protection and Environmental Research, ex ANPA) (2000) Analisi merceologica dei rifiuti urbani (Material composition analysis of municipal solid waste). Report RTI CTN_RIF 1/2000, Dipartimento Stato dell'Ambiente, Controlli e Sistemi Informativi, Rome, Italy

ISPRA (Institute for the Protection and Environmental Research, ex APAT) (2007) Caratterizzazione chimico-fisica del biostabilizzato proveniente da impianti di trattamento meccanico biologico dei rifiuti (Chemical-physical characterization of the stabilized biowaste coming from mechanical biological treatment plants). Studio APAT-ARPA-CIC, Dipartimento stato dell'ambiente e Metrologia ambientale, Servizio Rifiuti, Rome, Italy

Italian Decree (2010) Disposizioni di attuazione della direttiva 2008/98/CE del Parlamento europeo e del Consiglio del 19 novembre 2008 relativa ai rifiuti e che abroga alcune direttive. Decreto Legislativo del 3 dicembre 2010, n. 205 (in Italian), published on 10 December 2010 in Gazzetta Ufficiale nr 288

Izzo RLS, Mahler CF, Bauer J, Münnich K (2009) Test results of a capillary barrier made of MBT waste. In: CISA (Environmental Sanitary Engineering Centre) (ed), Proceedings of Sardinia 2009, Twelfth International Waste Management and Landfill Symposium, S. Margherita di Pula, Cagliari, Italy, 5–9 October 2009

Karak T, Bhattacharyya P, Das T, Paul RK, Bezbaruah R (2013) Non-segregated municipal solid waste in an open dumping: a potential contaminant in relation to environmental health. Int J Environ Sci Technol 10:503–518. doi:10.1007/s13762-013-0184-5

Komilis DP (2006) A kinetic analysis of solid waste composting at optimal conditions. Waste Manag 26(1):82–91

Leikam K, Stegmann R (1999) Influence of mechanical–biological pretreatment of municipal solid waste on landfill behaviour. Waste Manag Res 17(6):424–429

Lornage R, Redon E, Lagier T, Hébé I, Carré J (2007) Performance of a low cost MBT prior to landfilling: study of the biological treatment of size reduced MSW without mechanical sorting. Waste Manag 27(12):1755–1764

Makan A, Assobhei O, Mountadar M (2014) Initial air pressure influence on in-vessel composting for the biodegradable fraction of municipal solid waste in Morocco. Int J Environ Sci Technol 11:53–58. doi:10.1007/s13762-013-0434-6

Ministerial Decree (MD) (2006) Decreto del 5 Aprile 2006, n. 186 - Regolamento recante modifiche al decreto ministeriale 5 febbraio 1998 « Individuazione dei rifiuti non pericolosi sottoposti alle procedure semplificate di recupero, ai sensi degli articoli 31 e 33 del decreto legislativo 5 febbraio 1997, n. 22 (in Italian), published on 19 May 2006, in Gazzetta Ufficiale nr 115

Montejo C, Ramos P, Costa C, Márquez MC (2010) Analysis of the presence of improper materials in the composting process performed in ten MBT plants. Bioresour Technol 101(21):8267–8272

Partl H, Cornander I (2006) Organics from mechanical-biological treatment (MBT) facilities: International standards, applications and controls. Report NS03039, The Waste Management Association of Australia, Sydney, Australia

Petruzzelli G, Ottaviani Lubrano L, Veschetti E (1994) Characterization of heavy metal mobile species in sewage sludge for agricultural utilisation. Agrochimica 38:277–284

Ponsá S, Gea T, Sanchez A (2010) The effect of storage and mechanical pretreatment on the biological stability of municipal solid wastes. Waste Manag 30:441–445

Scaglia B, Confalonieri R, D'Imporzano G, Adani F (2010) Estimating biogas production of biologically treated municipal solid waste. Bioresour Technol 101:945–952

Sirini P, Tchobanoglous G, Noto La Diega RC (2009) Ingegneria dei rifiuti solidi (Solid waste engineering). Mc Graw Hill, Milan

Tintner J, Smidt E, Böhm K, Binner E (2010) Investigations of biological processes in Austrian MBT plants. Waste Manag 30:1903–1907

UK Environment Agency (2009) A review of human health and environmental risks associated with the land application of mechanical–biological treatment outputs. Report SC030144/SR2, Environment Agency for England and Wales, Bristol, United Kingdom

UNI (Italian National Agency for Standardization) (2002) Characterization of waste: determination of total organic carbon (TOC) in waste, sludges and sediments. UNI EN 13137, Milan, Italy

UNI (Italian National Agency for Standardization) (2006) Waste and refuse derived fuel: determination of biological stability by dynamic respirometric index. UNI/TS 11184, Italian National Agency for Standardization (UNI), Milan, Italy

Van der Sloot HA, Comans RNJ, Meeussen JCL, Dijkstra JJ (2004) Leaching methods for soil, sludge and treated biowaste. ECN—Environmental Risk Assessment, Final Report HORIZONTAL-23

van Praagh M, Heerenklage J, Smidt E, Modin H, Stegmann R, Persson KM (2009) Potential emissions from two mechanically–biologically pretreated (MBT) wastes. Waste Manag 29(2):859–868

Whittle AJ, Dyson AJ (2002) The fate of heavy metals in green waste composting. Environmentalist 22:13–21

Zaman AU (2013) Identification of waste management development drivers and potential emerging waste treatment technologies. Int J Environ Sci Technol 10:455–464. doi:10.1007/s13762-013-0187-2

Zennaro M, Cristofori F, Formigoni D, Frignani F, Pavoni B (2005) Heavy metal contamination in compost: a possible solution. Ann Chim Rome 95:247–256. doi:10.1002/adic.200590027

Zhao S, Liu X, Duo L (2012) Physical and chemical characterization of municipal solid waste compost in different particle size fraction. Pol J Environ Stud 21(2):509–515