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Materials and energy recovery at six European MBT plants

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ABSTRACT

Mechanical Biological Treatment (MBT; called "dirty" Materials Recovery Facilities in the U.S.) is a waste management method, developed mostly in Europe, which combines sorting of recyclable materials (metals, paper, plastics, glass) with composting/digestion of green/ food wastes and, in some cases production of a fuel material. In 2018–19, the authors visited six MBT facilities in Europe that use different approaches for the recovery of materials and energy from mixed MSW. These plants were studied with respect to feedstock composition, operating conditions, capital expenditure, financial viability and environmental impacts. The compost product of most facilities examined did not comply with agricultural standards and, therefore, it was classified as compost-like output (CLO) and used as daily cover in landfills. The best composting practice used source separated organic materials (yard and other green wastes) and yielded a marketable compost. MBT plants that did not include the recovery of fuel materials had lower landfill diversion rates and, also, lower capital and operating costs. It was concluded that an MBT plant must include a very efficient sorting and recyclables recovery line and charge a sufficient gate fee. Also, in addition to the recycled products, there should be a stream to recover fuel materials sent to a power plant or cement plant, thus increasing revenue, and landfill diversion, and maximizing greenhouse gas (GHG) savings.

1. Introduction

Mechanical Biological Treatment (MBT, called "dirty" Materials Recovery Facilities in the U.S.) is a process widely used in the EU for processing mixed municipal solid wastes (MSW). It combines sorting of recyclable materials (metals, paper, plastics, glass) with biochemical treatment of green and food wastes by composting or anaerobic digestion (Vrancken et al., 2017; Cook et al., 2015). In some cases, MBT plants recover fuel materials used in cement manufacturing or in energy recovery (Gerassimidou et al., 2021; Nasrullah et al., 2017). Thus, MBT is addressed to the upper tiers of the hierarchy of sustainable waste management, i.e., the recovery of resources either as recyclables/compostables, or as fuels in the energy/cement sectors, thus diverting wastes from landfills (EU Directive, 2018).

The configuration of MBT facilities depends on the input streams to the facility and the desirable products. The input composition of the feedstock depends on the collection systems, i.e., mixed wastes, single-stream, dual stream, while the MBT products depend on the priorities of the market. Several studies have been conducted on the efficiencies, and the environmental and economic impacts of MBT plants. Pressley et al. (2015) developed a comprehensive hypothetical framework for different recycling facilities but did not include utilization of the organic fraction. Ardolino et al. (2017) conducted a holistic assessment of different configurations of a material recovery facility, by considering several possibilities for the outputs, but the study used data from one facility. Similarly, Beylot et al. (2015), evaluated the performance of one facility in France, and considered the environmental impacts. Other studies have focused on single or dual stream recycling systems (Fitzgerald et al., 2012), or on pre-sorted materials, e.g., packaging materials (Karine et al., 2018; Mastellone et al., 2017; Cimpan et al., 2016). Also, researchers emphasized on the energy implications of recycling facilities (Cimpan et al., 2013), and on policy implications with regard to costs and environmental impacts of waste management systems (Jaunich et al., 2019). Other studies have focused on the quality of the fuel produced from MBT plants (Di Lonardo et al., 2016; Edo-Alcón et al., 2016; Samolada and Zabaniotou, 2014; Velis et al. 2012). Merrild et al. (2012) assessed the environmental impacts of recycling specific materials, as compared to energy recovery, and concluded that it is unclear whether materials with high energy content should be used to produce energy or recycled. Furthermore, Life Cycle studies have been conducted to identify the best available waste management systems of cities (Hadzic et al., 2018; Jia et al., 2018; Madav and Samadder, 2018; Ripa et al.,

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2017; Fernandez-Nava et al., 2014; Laurent et al., 2014; Kaplan et al., 2009; Chester et al. 2008). However, most studies focused on the environmental and economic aspects, not on the technical performance of processes.

This study used primary industrial data to compare the environmental and economic aspects of advanced MBT facilities in Europe, some of which included the use of residual fuel materials in the energy/cement sectors.

2. Materials and methods

2.1. Scenarios examined

Material balances of the MBT systems examined were developed using the STAN software, which performs material flow analysis (MFA) (https://www.stan2web.net/). Input/output values were obtained from reports provided by the operators of the plants, communication of the authors with senior engineers of the facilities, and visits of the plants. These MBT plants use state-of-the-art technology and are located in Spain, Greece and Cyprus.

The MBT facilities assessed use a combination of both biological and physical processes. Initial waste preparation takes place with removal of oversized items, such as mattresses, or other bulky wastes, which could cause problems with processing equipment downstream. Refuse bags are then transferred with the aid of cranes, and conveyor belts to bag openers, which liberate the materials inside the bags. Hand sorting of paper, plastics, and glass materials takes place, which is followed by trommels for the separation of the organic fraction from 'dry' recyclables. Ferrous and non-ferrous metals are separated with magnetic and eddy current separators. Ballistic, disc screens, and near infrared spectra optical separators are used to recover other 'dry' recyclables, such as paper, plastics, and glass. The organic materials are processed aerobically for several weeks, except for Plant S1. The stabilized organic material is refined with the use of screens and shaking tables for the removal of fine 'dry' recyclables such as glass and plastics. Plant S5 uses Anaerobic Digestion for part of the organic fraction and sends the residues of the recycling and composting processes to a moving grate waste to energy (WTE) power plant. In all, six systems were studied:

S1. The total capacity of the plant is 16,500 metric tons of mixed MSW per year. The facility produces recyclables, and a refuse derived Fuel (RDF), consisting of non-recycled paper and plastics (NRPP), used in cement production. The residues are landfilled.

S2. The plant receives 76,000 metric tons of mixed MSW per year and 11,375 metric tons of source separated green wastes per year. The facility produces recyclables and composts aerobically in-vessel green wastes for 6–7 weeks, followed by 2–3 weeks of maturation, and further refinement. The process residues are landfilled.

S3. The facility processes 150,000 metric tons of mixed MSW per year. The plant produces recyclables, compost like output (CLO) used as daily cover in landfills, and refuse-derived fuel (RDF), derived from non-recyclable paper and plastics (NRPP) and used in cement production. The aerobic composting is in-vessel for 12 days followed by further processing in open windrow for one week. The process residues are landfilled.

S4. The plant processes 185,000 metric tons of mixed wastes per year. The facility produces recyclables, CLO used as daily cover in a landfill, and RDF used in cement production. The aerobic composting is in-vessel for 6–7 weeks. The process residues are landfilled.

S5. This facility processes 250,000 metric tons of mixed MSW per annum and consists of an MBT plant, an anaerobic digestion facility, and a moving grate waste-to-energy (WTE) plant. It produces recyclables, CLO used as daily cover in landfills, and electrical energy from the combustion of RDF in the WTE plant (60,000 MWh/year) and the anaerobic decomposition of of 30,400 tons of organics (4,800 MWh/year). The system also processes aerobically 75,500 metric tons of organics per year; the organic fractions are produced from the mechanical separation of organics from mixed MSW. The aerobic composting occurs in aerated static piles for 3–4 weeks, along with an aerobic stabilization turning system. The anaerobic digestion is a "wet" process and involves hydromechanical pre-treatment that includes waste pulping and grit removal. The anaerobic digestion of the cleaned organic suspension is carried out under mesophilic conditions (35–38 °C). The process residues are combusted in the WTE plant.

S6. The MBT plant receives 350,000 metric tons of mixed MSW, and 35,000 metric tons of source separated yard wastes. The facility produces recyclables, and a fertilizer that is used for edible plants and is mainly produced from the source separated yard wastes. Aerobic composting takes place in aerated static piles of 25 composting tunnels for 3–4 weeks, followed by maturation of the produced materials with turning system and further refinement. The residues of the process are landfilled.

2.2. Assessment of contribution of materials to GHG and revenues of systems

The material-specific factors of greenhouse gas (GHG) emissions (Fig. 1) were obtained from the Waste Reduction Model (WARM) of US EPA (EPA Waste Reduction Model, 2016). Raw data are provided in supplementary materials A1. WARM was used to provide the GHG savings of recycling specific materials. Thus, the emissions due to use of electricity were assumed to be the same as the U.S. national average. The revenues from the sale of products recovered from the mixed MSW stream were estimated using commodity prices obtained from Eurostat and WRAP of UK (WRAP, 2018; Eurostat, 2021). The market price of recyclables depends on the quality of the products; the average values of low- and high-quality recyclables were used as confirmed by the operators of the plants. The price of electricity from anaerobic digestion was set at $150/MWh, and from WTE at $60/MWh. The landfilling fee of MBT residues, shown as 'other non-recyclable materials to LF' in the MFA graphs, was assumed to be $50/metric ton. The landfilling fee of the stabilized residue of the aerobic process of the systems examined and the anaerobic process of S5, shown as 'residues to LF' in the MFA graphs, is only $3/metric ton, according to the operators of the MBT plants. However, the operators of the MBT plants receive $5/metric ton for the use of CLO as daily cover in landfills. The market prices of fertilizer used in the production of non-edible and edible plants were set at $70 and $320/metric ton, respectively, according to the operators of the plants and also as reported by Eurostat and WRAP of UK (Eurostat, 2021; WRAP, 2018). Both the S2 and S6 plants recover good quality fertilizers by using mainly source separated green/yard wastes.

In order to understand the effect of utilizing the organic fraction of mixed MSW on the environmental and economic efficacy of the plants, a sensitivity analysis was conducted, where the produced compost of S2 and S6 was not used as fertilizer, but as daily cover in landfills where the plants receive only $5/metric ton of CLO. Also, an extreme case was considered, where the operators pay a gate fee of $3/metric ton to dispose of all the organics in landfills. The WARM factors for the use of mixed organics as fertilizer is (0.09) metric tons $CO_{2\text{-eq}}$/metric ton of material, and for the use of mixed organics in landfills is 0.18 metric tons $CO_{2\text{-eq}}$/metric ton of material. It should be noted that processing of mixed organics in WTE plants results in a WARM factor of (0.15) metric tons $CO_{2\text{-eq}}$/metric ton of material, while processing anaerobically is associated with (0.06) metric tons $CO_{2\text{-eq}}$/metric ton of material.

Furthermore, it was assumed that cement operators pay $20/metric ton of non-recycled fuel materials (Bourtsalas et al., 2018). Revenues from the gate fees received by the MBT plants in this study range from $30 to $80 per ton of mixed wastes, as reported by the operators. Revenues from the recovery of metals and minerals from WTE bottom ash and costs of disposing the ash were not included in the financial calculations. A detailed economic and environmental comparison of the use of alternative fuels in cement manufacturing or as fuel in a Waste to Energy (WTE) plant was conducted by Bourtsalas et al. (2018), thus the

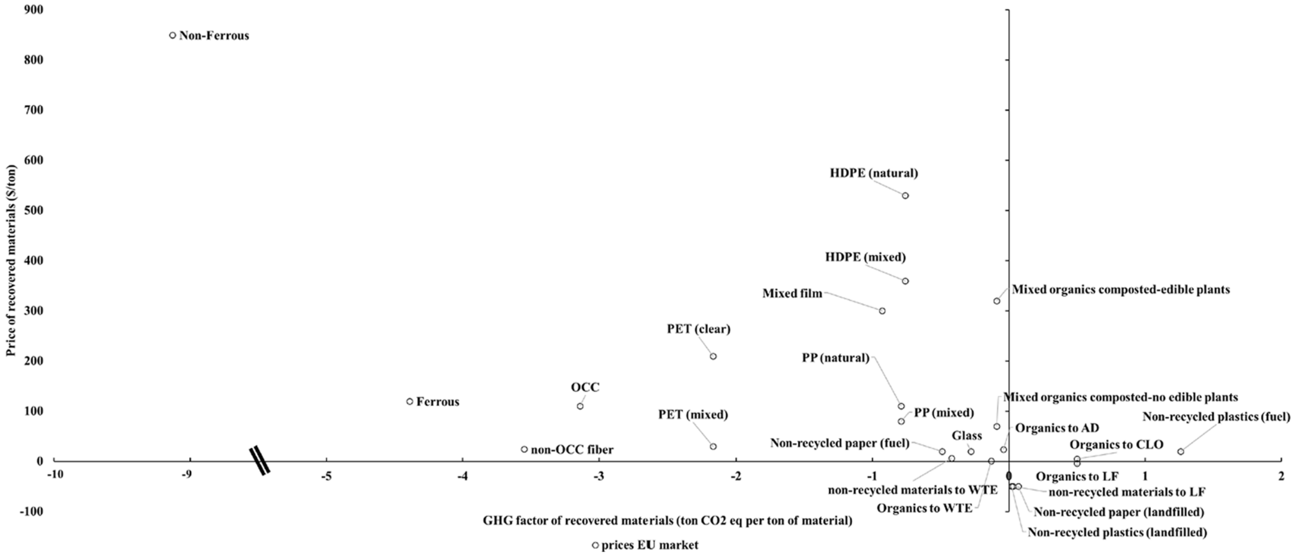


Fig. 1. GHG factors (metric tons CO2-eq/metric ton of material) and prices ($/metric ton of material) of recovered materials produced at MBT plants.

options for utilizing alternative fuels were not included in the analysis.

The capital and operating estimates (CAPEX and OPEX) were based on information provided by the MBT plant operators and the literature. Two scenarios were developed by selecting low and high values for capital and operational charges to capture differences in the permit processes, and other factors that contribute to possible changes in the CAPEX and OPEX of the facilities, such as inflation, living expenses. The CAPEX, OPEX assumptions are provided in the supplementary materials, A2. The mechanical treatment part was assumed to be in the range of $220 and $300/metric ton of feed (Jaunich et al., 2019; Pressley et al., 2015). The CAPEX of the aerobic composting unit ranged between $60 and $120/metric ton of the anaerobic digestion unit between $400 and $600/metric ton, and of the WTE unit between $550 and $700/metric ton (Jaunich et al., 2019; Pressley et al., 2015). The landfill CAPEX was assumed to be $30 – 50 per metric ton (Jaunich et al., 2019; Levis et al., 2014; Levis et al., 2013). The OPEX required for the systems was in the range of $25 and $45/metric ton (Jaunich et al., 2019; Levis et al., 2014; Levis et al., 2013). The ratio of bank loan to equity was assumed to be 70:30, the repayment period of the investment was assumed to be 20 years, and the interest rate was set at 4%. Total $ and $/ton values were calculated. The annual revenues minus expenditures were calculated and the results were compared with landfill diversion rate (%), gate fee ($/metric ton of feed), and GHG savings (10 kg $CO_{2\text{-eq}}$/metric ton of feed) of the systems.

3. Results and discussion

3.1. Material flow analysis (MFA) of MBT plants

The MFAs of the six MBT systems examined in this study are presented in Fig. 2.

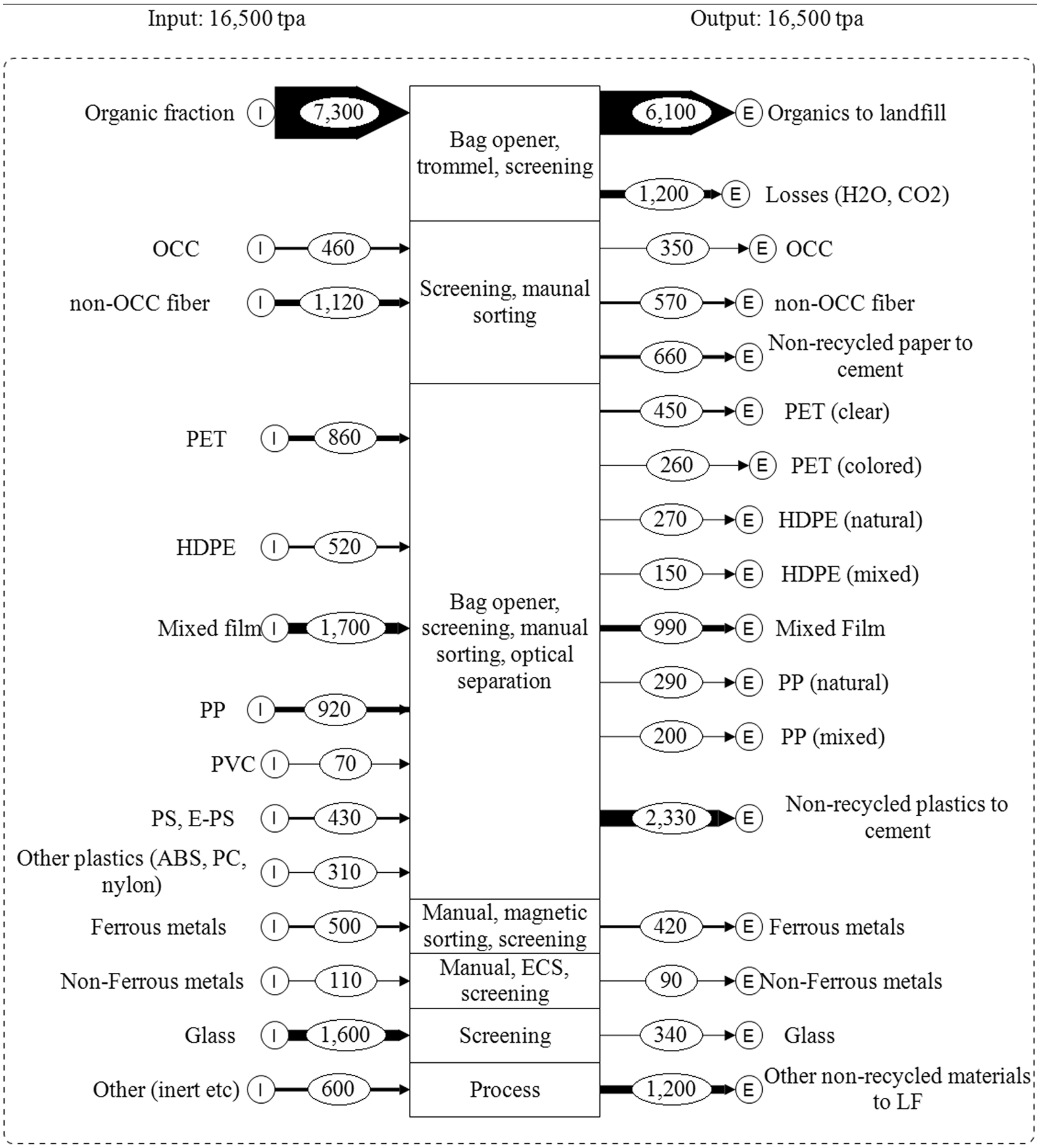
Table 1 is a summary of the recovery of materials and fuel from the mixed waste feedstock to the six MBT plants in this study.

Table 1 shows that for the four MBT plants that recover mostly materials the average landfill diversion was only 52%. Plants S4 and S5 which recover materials and fuel, have a much higher landfill diversion; however, these plants recovered significantly lower amounts of paper fiber and plastics. In terms of efficiencies, metal recovery was very high in all cases examined, with ferrous metals ranging from 83.3% (S3) to 89.9% (S2), and non-ferrous from 78.1% (S3) to 82.8% (S5). For glass, in all cases, the recovery efficiency was below 20%. PET and HDPE materials indicate fairly high recovery efficiencies, over 80% for four systems and 40–50% for S4 and S5; for mixed film (mainly LDPE) and PP the efficiencies are 50–65% for four systems and 20–30%, for systems S4 and S5. The efficiency of recovery of corrugated cardboard (OCC) was at the 70% level. However, the rest of paper fiber had efficiencies in the range of 50 and 70% for S1-S3 and S6, and only 25% for systems S4 and S5. Recovery of fuel materials was between 15 and 20%, except for S4, which reported 38.5% recovery. This is mainly associated with the required quality of the cement manufacturers in the region of the plant, but also with the amount of plastics received at S4. A higher amount of plastics may result in higher amount of recyclables, but a higher concentration of chlorinated plastics, such as PVC, will result in higher Cl content. The utilization of waste fuels in cement manufacturing depends on the willingness to pay for this fuel by the cement producers; this price depends on the quality of the fuel, e.g., chlorine content, and the market price of fossil fuels (Gerassimidou et al., 2021; Waltisberg and Weber, 2020; Bourtsalas et al., 2018; IFC, 2017).

The system S5 includes a WTE plant so all the non-recycled materials are combusted for the production of electricity; it had 53.4% total fuel recovery and the highest landfill diversion of 86.4%. However, the selected boundaries of the S5 system do not include the waste to energy (WTE) power plant; the inclusion of WTE residues would decrease landfill diversion to 73.1%, assuming 25% WTE ash, without metal recovery and beneficial use of the WTE bottom ash in the civil engineering sector. S4 indicated the second highest landfill diversion (69.6%), followed by S2 (57.8%), S1 (51.3%), S3 (49.8%), and S6 (48.9%).

Systems S2 and S6 mix the organic fraction with source separated organics and recover compostable products in the range of 9 and 16%, which are used as soil conditioner for the production of non-edible and edible plants, respectively. As shown in Table 2, in the case of system S2, organics are mixed with 35% of branches and leaves whereas, in system S6, organics are mixed with about 20% yard wastes. In system S2, where a lower quality of compost is acceptable, the total compostables produced are 36.9% of the organic feedstock and the residue of the aerobic composting process is 25.9%. In contrast, for system S6, the total compostables produced are 18.8%, and the residue to landfill is 48.8%. Systems S2, and S6 have process losses of between 32 and 37%.

System S1 sends the organic fraction directly to landfill, and systems S3-S5 produce a CLO which is used as daily cover in landfills and therefore is not included in the landfill diversion rate. System S4 involves a more sophisticated aerobic composting system, which includes refinement of the produced organic fraction, and thus the process losses are higher than for S3. For the same reason, the total compostables produced in S3 are higher compared to S4, however, the residue to landfill is nearly the same, 19.1% and 16.2%. System S5 includes both



Material Flow Analysis (MFA) for S1

Fig. 2. Material Flow Analysis of six systems examined.

an aerobic composting unit and an anaerobic digestion facility; 28.8% of the organic fraction is processed anaerobically to produce 0.16 MWh of energy/ton of organics. The process loss is 38.8%, 40.8% of the residues are landfilled and 20.4% are combusted in the WTE plant. The aerobic process produces 28.5% CLO, has 47% process loss, and the 24.5% residue is combusted at the WTE plant.

At the time of the study, the quality standards for the use of compost products obtained from organic wastes were established at the national level. However, in 2019, the European Commission published guidelines and directives on the subject (EU, 2019a) that place a lot of emphasis on impurities contained in plastic materials (EU, 2019b). National regulations typically depend on the specific conditions of the country/region (soil composition, agricultural practices, market needs) that can vary significantly (EEA, 2020). The quality of compost derived from organic wastes mainly depends on the levels of heavy metals and impurities present, such as plastics, glass, and metals, and the maturity, and stability of the end product (EEA, 2020; Cerda et al., 2018). Systems S2 and S6, which recover good quality compost to be used as fertilizer for non-edible and edible plants, accordingly, use state-of-the-art aerobic processing of the organic fraction, with several refinement and maturation stages. Also, both systems mix the organic fraction produced with source separated organics to improve the properties of the final product, such as by reducing the high salt concentrations often found in the organic fraction of mixed household wastes (Cerda et al., 2018). Such practices reduce the levels of impurities and enhance the quality of the produced compost (EEA, 2020; Cerda et al., 2018; Puyuelo et al., 2019; Slorach et al., 2019).

The latest EU guidelines exclude the use of 'the organic fraction of mixed municipal household waste separated through mechanical, physicochemical, biological and/or manual treatment', for use as

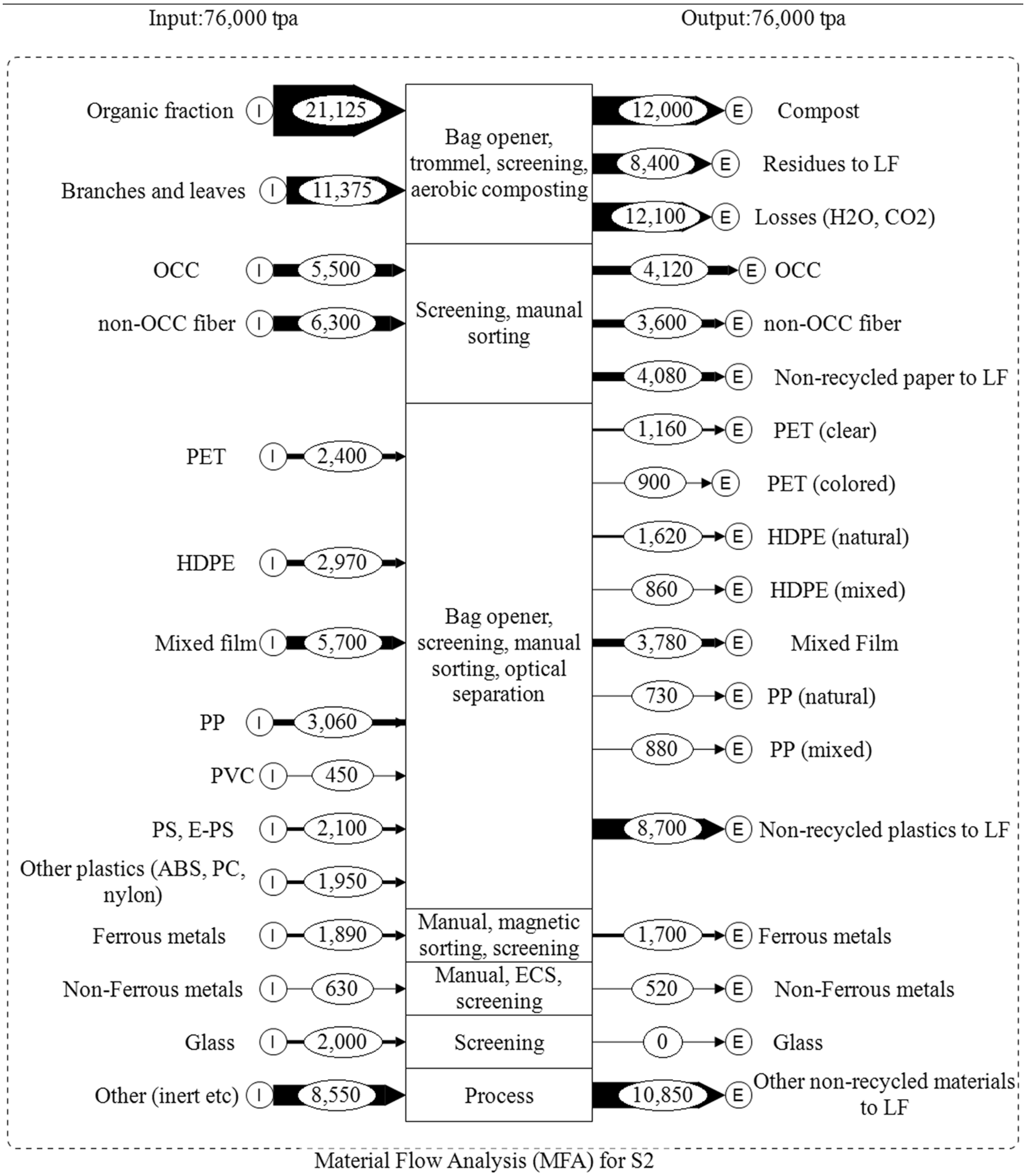


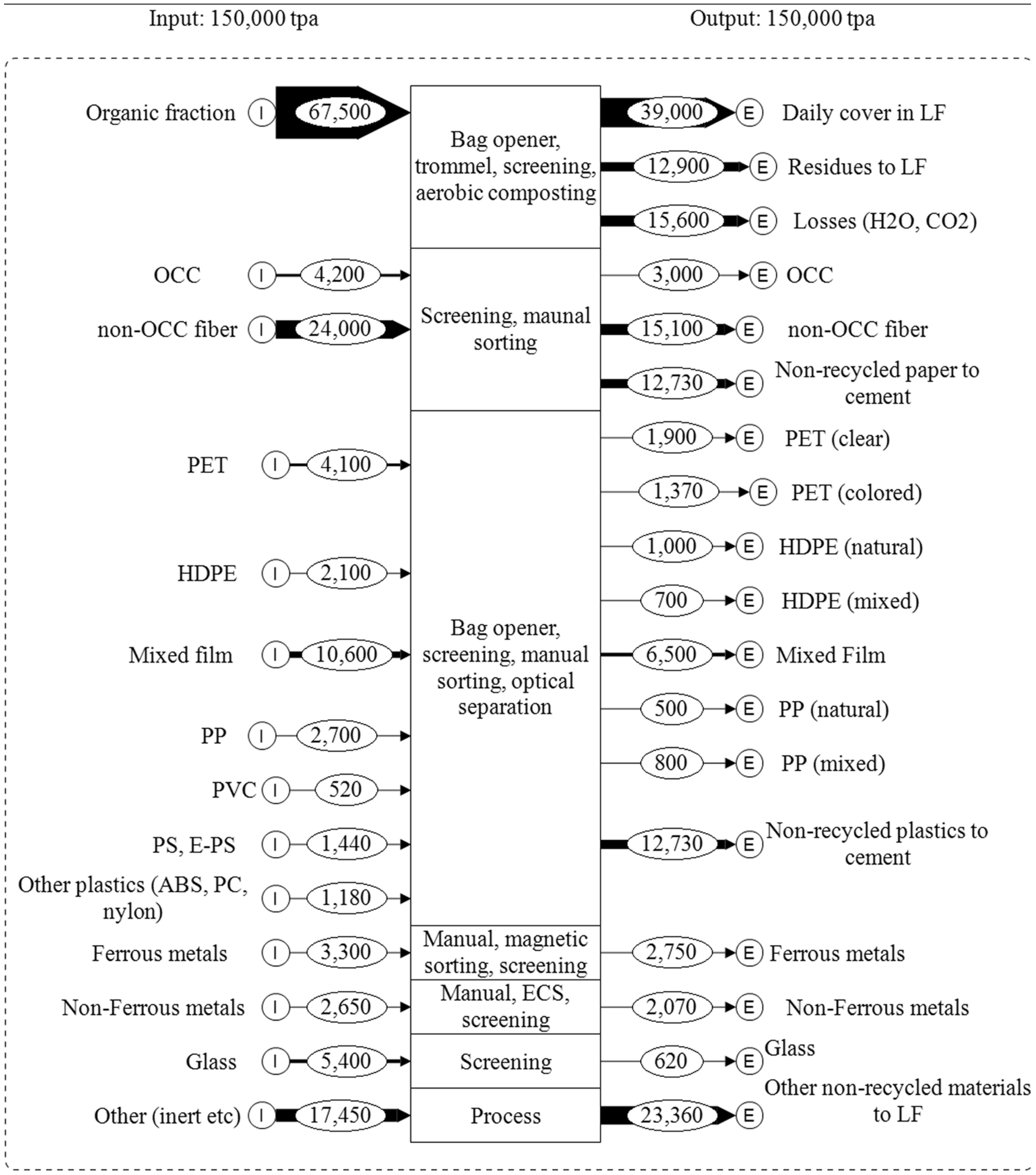
Fig. 2. (*continued*).

fertilizer, and several countries impose bans on its utilization as soil improver/fertilizer (EU, 2019a). As noted earlier, the data used in this study were collected during the 2018–19 period, prior the implementation of the new EU Directives; this may change the national/regional practice for utilization of organic wastes as compost for the production of edible/non-edible plants. Ideally, municipal collection systems should include source separation of organic wastes to avoid contamination with other waste streams and improve the quality of both recyclables and compostables. All MBT plants exhibited significantly lower recovery efficiencies, as compared to Material Recovery Facilities which use single or dual stream recycling and have reported recoveries of over 90% for most recyclables (Pressley et al., 2015; Fitzgerald et al., 2012). However, it is very challenging to ensure stakeholder engagement and citizens participation, both of which are essential for effective collection and recycling/composting systems (Rodrigues et al., 2020; Schanes et al., 2018; Scherhaufer et al., 2018).

Table 2 is a summary of the processes used for managing the organic fraction of the feed to the MBT plants and the results obtained.

3.2. Contribution of materials to GHG savings and to revenues of MBT systems

Fig. 3 shows the contribution of various materials recovered in the MBT process to the GHG emissions and the revenues of each of the six MBT configurations. A breakdown of each category, i.e., recyclables, organic fraction, non-recycled materials and NRPP, is provided in the supplementary materials A3. In all cases examined, the main GHG savings are associated with the recycling of paper fiber, ferrous and non-ferrous metals. Plastics recycling contribute between 10 and 20% to the total GHG savings, with PET (clear and mixed), and mixed film having the highest contribution. A breakdown of the contribution of plastics is provided in supplementary materials (A4). In terms of plastics



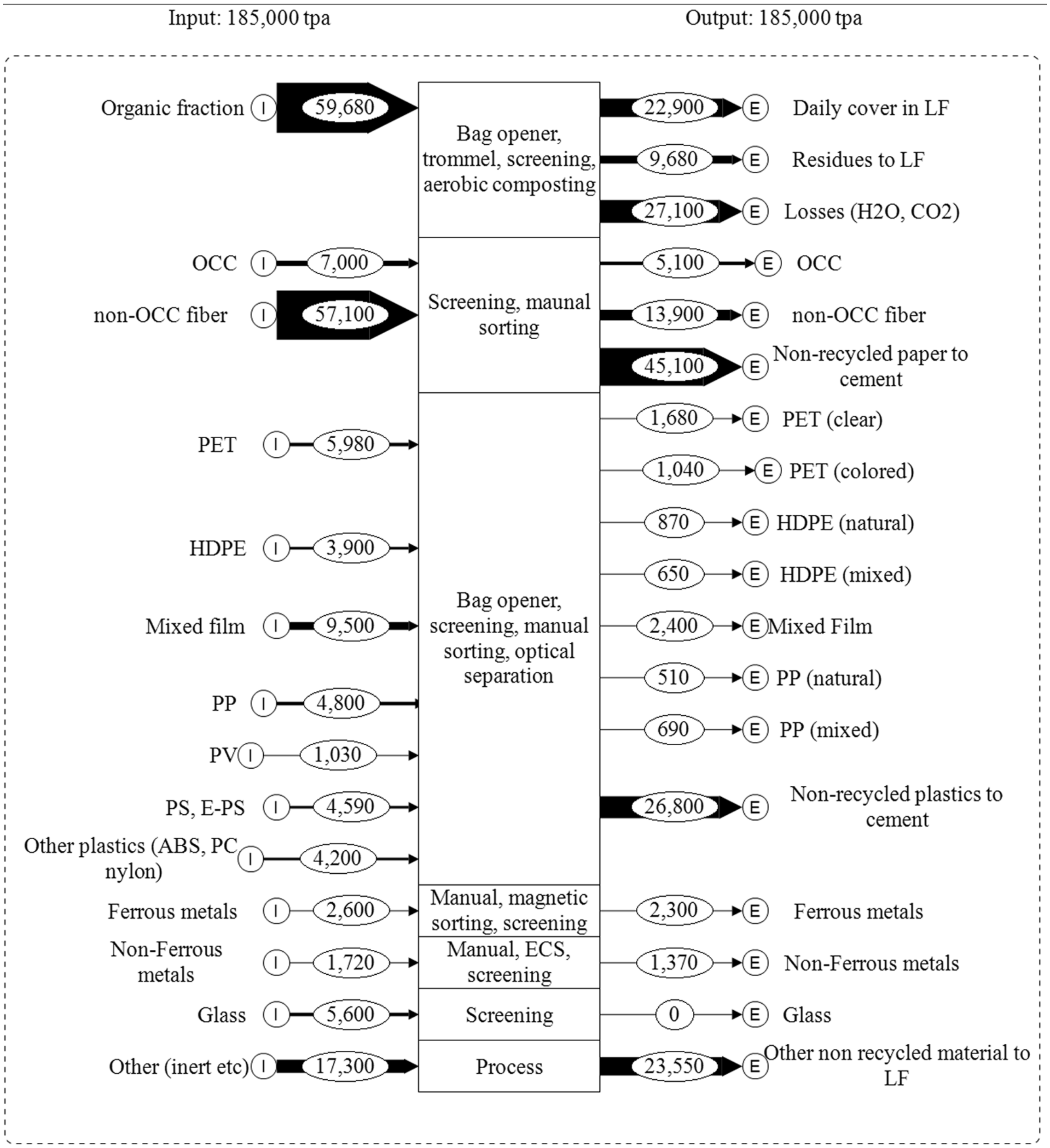
Material Flow Analysis (MFA) for S3

Fig. 2. (*continued*).

recycling, the main contribution is due to PET (clear and mixed) recycling, with both categories contributing between 40 and 60% to the total GHG savings of plastic materials. Mixed film contributes between 20 and 30% to the GHG savings of plastics, except for S5, where a significant fraction of mixed film is combusted with energy recovery. HDPE (natural and mixed), and PP (clear and mixed) contribute between 10 and 20% of the GHG savings, respectively. System S5 that sends the post-recycling residue to a WTE plant, has the lowest GHG emissions of all systems for non-recycled residues.

In terms of total GHG emissions, S2 indicates the lowest (-0.59 metric tons $CO_{2\text{-eq}}$/metric ton of feed), which can be attributed to the high recovery efficiency of recyclables, as compared to the other scenarios, and the use of part of the organics for the production of non-edible plants. All systems indicate GHG savings in the range of −0.38 (S4) to −0.59 metric tons GHG/metric ton of feed (S2), except for S1 (−0.21 metric tons $CO_{2\text{-eq}}$/metric ton of material), which is mainly a result of landfill emissions. The sensitivity analysis showed that the GHG savings of S2, and S6 drop to −0.50 metric tons $CO_{2\text{-eq}}$/metric ton of material, and −0.44 metric tons/ton (from −0.51 metric tons/ton), when the organic fraction is used as daily cover in landfills or is landfilled, instead of using as fertilizer.

The main GHG benefits of S1 are associated with the recycling of non-OCC fiber, followed by ferrous metals, OCC fiber, PET (clear), and non-ferrous metals. The main GHG contribution of S2 comes from the recycling of non-OCC, and OCC fibers, followed by non-ferrous and ferrous metals, mixed film and PET (clear and mixed). S2 and S5 indicate the lowest GHG contribution from the management of the organic fraction, because a fraction of organics is used for the production of non-



Materials Flow Analysis (MFA) for S4

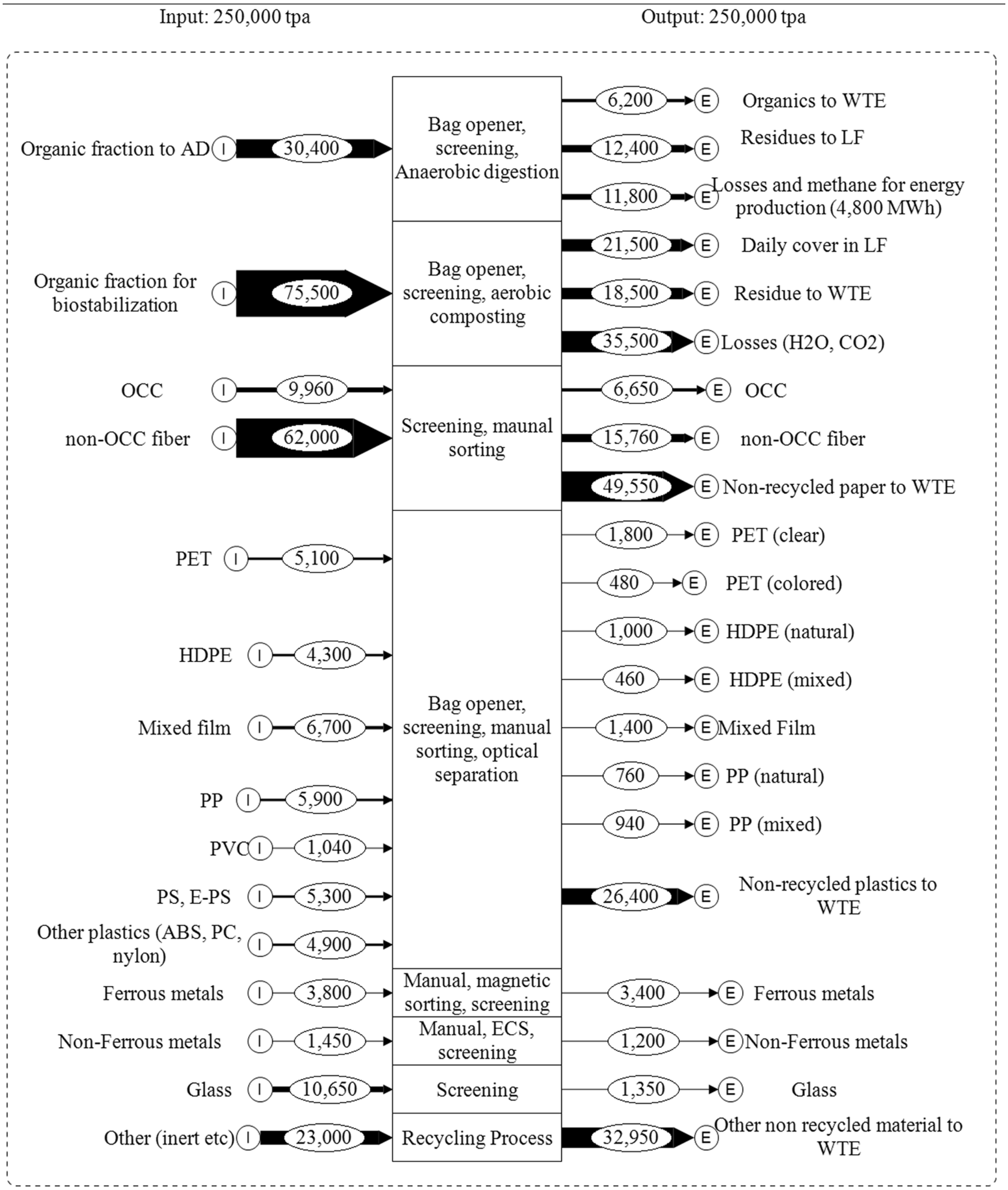
Fig. 2. (*continued*).

edible plants in S2, or is combusted in the WTE plant of S5. However, when the organic fraction is used as CLO or disposed of in landfills, S2 has higher GHG contribution as a result of the organic fraction management. S3 and S4 non-OCC and OCC fibers, ferrous and non-ferrous metals, mixed film, and PET (clear and mixed), are the materials with the major contribution. For S5 and S6, GHG savings are mainly associated with the recycling of non-OCC fiber, and OCC fiber, followed by ferrous and non-ferrous metals, and PET (clear).

The main revenues of the plants are contributed by mixed film, HDPE (clear), PET (clear), non-ferrous and ferrous metals, polypropylene, and OCC fibers. MBT plants that provide RDF to cement plants have an additional source of revenue. Similarly, the utilization of non-recycled materials in a WTE plant (S5) provides a positive revenue, in contrast to the other systems which landfill their residues.

In terms of total revenues, S6 and S5 have the highest, which partially relates to the higher gate fee these plants receive. S5 receives a gate fee of $80/metric ton, S6 $60/ton, S4 $50/ton, S2 $45/ton, S3 $40/ton, and S1 $30/ton, as shown in Fig. 3. Also, for the case of S6, a major revenue is due to the high price of high-quality compost in the EU market, whereas for S5 additional revenues are obtained from the recovery of electricity in the WTE plant; as is the cases of S1, S3, and S4, where the NRPP fraction is sold to the cement manufacturers. S6 ($53.1/metric ton) and S2 ($47.2/metric ton) indicate the highest revenue from products utilization, i.e., excluding gate fee, on a per metric ton basis, which can be attributed to the high recovery efficiency of the plants, along with the advanced utilization of the organic fraction. However, there are also negative revenues, due to the landfilling of the MBT residues.

The sensitivity analysis showed that the utilization of organic wastes contributes significantly to the economic efficacy of the plants. In the case of the S2 and S6 plants where organic wastes are used ass daily cover in landfills, the revenues from products utilization, drop to $36.8/



Materials Flow Analysis (MFA) for S5

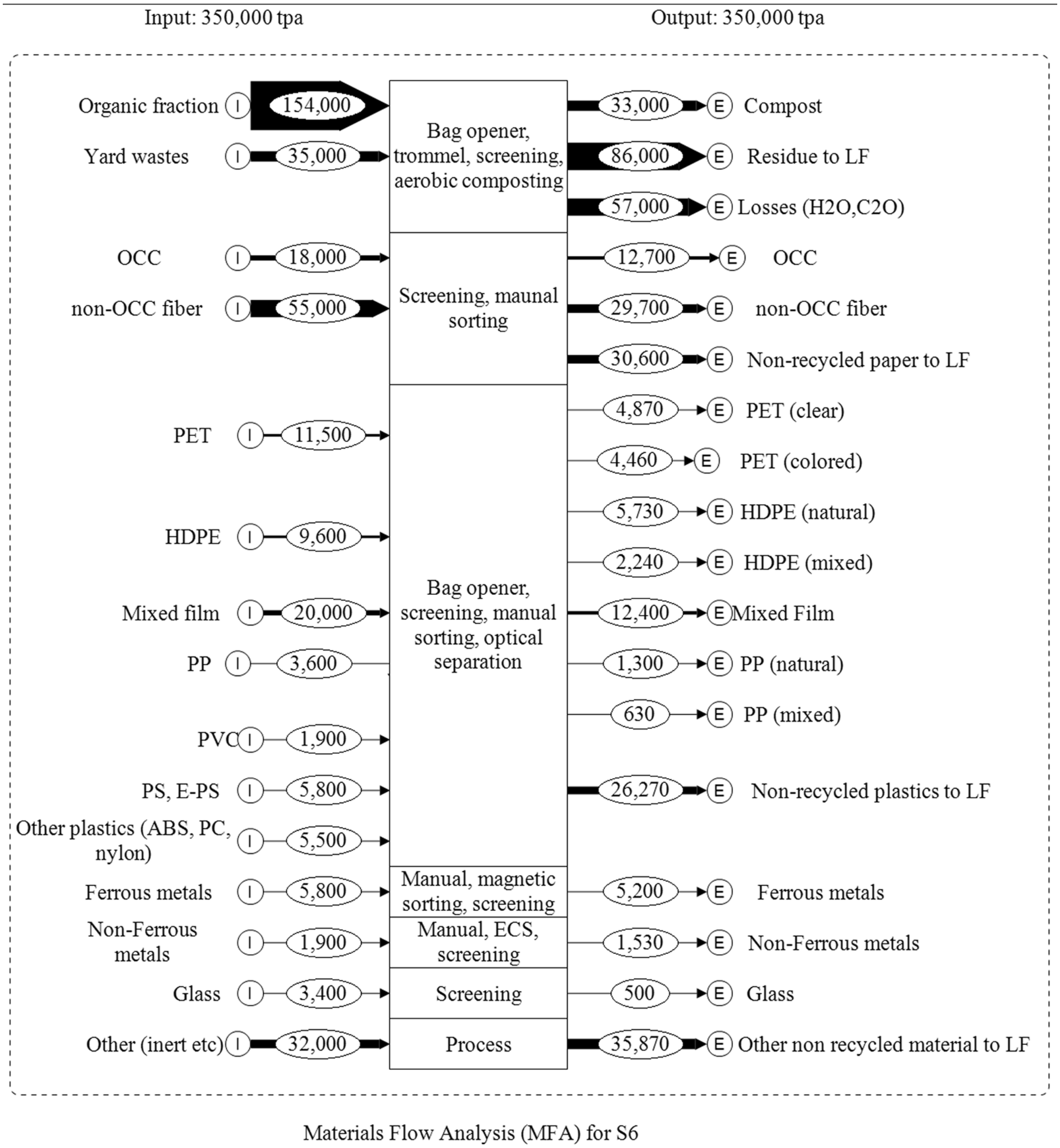
Fig. 2. (*continued*).

metric ton for S2, and to \$23.4/metric ton for S6. Furthermore, in the case where all the plants dispose of the organic wastes in landfills, S6 indicates the lowest revenues from products utilization, \$22.5/metric ton), followed by S4 (\$23.4/metric ton). S1 indicated the highest revenues from products utilization (\$46.8/metric ton); systems S2 (\$34.9/metric ton), S3 (\$34.1/metric ton), and S5 (\$30.4/metric ton) indicated similar values.

3.3. Evaluation of systems

The estimated CAPEX and OPEX values used for the financial assessment, are presented in Table 3. Details on the economic calculations are presented in Supplementary materials, A5.

The system S5, which includes anaerobic digestion of the organic fraction and combustion of residues in a WTE plant, had the highest CAPEX (\$588–733/metric ton), and OPEX (\$49–58/metric ton) ranges, in both cases of low and high costs, but indicated the highest landfill diversion rate (86.4%). S6, which includes a more sophisticated system for the utilization of the organic fraction, indicated the second highest CAPEX, between \$299 and \$360/metric ton, and the lowest landfill diversion rate (48.9%) of all systems examined. S1 (\$263-\$322/metric ton), S2 (\$287–342/metric ton), and S3 (\$264–317/metric ton) had CAPEX close to S6, and indicated landfill diversion rates of 51.3%, 57.8%, and 49.8%, accordingly. S4 indicated the lowest CAPEX on a per ton basis (\$248–294/metric ton), and the second highest landfill diversion rate (69.6%). The OPEX of S1-S4 and S6, ranged between \$26



Materials Flow Analysis (MFA) for S6

Fig. 2. (*continued*).

(S1) and $38/ton (S6), whereas the OPEX of S5 was higher and between $49 and $58/ton of feed. The CAPEX and OPEX ratios of scenario A/B had a range of (1.19, 1.25), and (1.11, 1.23). S2, and S4 indicated the lowest CAPEX ratios, and S5 the highest, whereas S4 indicated the lowest OPEX ratio, with S1 indicating the highest.

In comparing revenues minus expenditures, S6 was the highest, both when high (rev-exp: $56.8/metric ton) and low (rev-exp: $65.6/metric ton of feed) CAPEX and OPEX values were used; also, when the produced compost is used as fertilizer for edible plants. However, this fertilizer is mainly produced from source separated organic materials, and not from the organic fraction of mixed MSW, which, as mentioned earlier and with regard to the recent EU regulations, cannot be used as soil conditioner. Therefore, and as attenuated, for S6, the sensitivity analysis, where the organic fraction is landfilled, resulted in a significant decrease in the revenues-expenditures of the system, which indicated $35.4/metric ton of feed when low CAPEX, OPEX values were used, and $26.7/metric ton when high CAPEX, OPEX values were used.

When the organic fraction is used as daily cover in landfills, the revenues minus expenditures of S6 is the highest ($51.2/metric ton for low CAPEX, OPEX values, and $38.7/metric ton of feed for high CAPEX, OPEX values). This is mainly associated with the composition of MSW processed in S6, the high recoveries of materials with high market prices, such as HDPE, PP, and metals, and partially with the economies of scale. When the organic fraction is landfilled, S4 was the most profitable system ($29.3/metric ton for high CAPEX, OPEX values were used), which is mainly related to the use of a significant fraction of NRPP as alternative fuel in cement production, the low CAPEX, OPEX values the system indicates, and the relatively high gate fee ($50/metric ton) the plant receives. However, when low CAPEX, OPEX values were used, S1 was the most profitable ($36.8/metric ton vs. $31.7/metric ton for S5), which can be explained by the high recovery efficiencies of the plant, and the simplicity of the system. S5 showed the lowest revenues minus expenditures values of all systems examined, which is due to the significantly higher CAPEX and OPEX values of S5, which uses both AD and WTE.

When the organic fraction was used as daily cover in landfills, revenues - expenditures ranged from $15.2/metric ton to $32.1/metric ton, for high and low CAPEX, OPEX values. When the organic fraction was

Table 1
% Recovery of materials and fuel by MBT plants from mixed MSW (output/input).

	Mostly on materials recovery					On materials and fuel recovery	
Flows	S1	S2	S3	S6	Average of four plants	S4	S5
OCC	76.1%	74.9%	71.4%	70.6%	73.3%	72.9%	66.8%
Non-OCC fiber	50.9%	66.7%	62.9%	54.0%	58.6%	24.3%	25.4%
Total paper recycled[a]	**58.2%**	**69.5%**	**64.2%**	**61.6%**	**63.4%**	**29.6%**	**31.1%**
PET (clear)	52.3%	48.3%	46.3%	42.3%	47.3%	28.1%	35.3%
PET (colored)	30.2%	37.5%	33.4%	38.8%	35.0%	17.4%	9.4%
Total PET[a]	82.5%	85.8%	79.7%	81.1%	82.3%	45.5%	48.6%
HDPE (natural)	51.9%	54.5%	47.6%	59.7%	53.4%	22.3%	23.3%
HDPE (mixed)	28.8%	29.0%	33.3%	23.3%	28.6%	16.7%	10.7%
Total HDPE[a]	80.7%	83.5%	80.9%	83.0%	82.0%	39.0%	38.5%
Mixed film, mainly LDPE	61.9%	66.3%	61.3%	62.0%	62.9%	25.3%	20.9%
PP (natural)	31.5%	23.9%	18.5%	36.1%	27.5%	10.6%	12.9%
PP (mixed)	21.7%	28.8%	29.6%	17.5%	24.4%	14.4%	15.9%
Total PP[a]	53.2%	52.7%	48.1%	53.6%	51.9%	25.0%	29.3%
Total plastics recycled	**52.7%**	**48.6%**	**44.3%**	**46.3%**	**48.0%**	**23.1%**	**20.6%**
Ferrous metals	**88.0%**	**89.9%**	**83.3%**	**89.7%**	**87.7%**	**88.5%**	**89.5%**
Non-ferrous metals	**81.8%**	**82.5%**	**78.1%**	**80.5%**	**80.7%**	**79.7%**	**82.8%**
Glass	**18.9%**	**0.0%**	**11.5%**	**14.7%**	**11.3%**	**0.0%**	**12.7%**
Total materials recovery	**25.9%**	**26.1%**	**24.2%**	**23.2%**	**24.9%**	**16.5%**	**14.1%**
Total compostables recovered[b]	**0.0%**	**15.8%**	**0.0%**	**9.4%**	**12.6%**	**0.0%**	**0.0%**
Total fuel recovery	**18.1%**	**0.0%**	**15.2%**	**0.0%**	**8.3%**	**38.5%**	**53.4%**
Process loss (H_2O, CO_2)	**7.3%**	**15.9%**	**10.4%**	**16.3%**	**12.5%**	**14.6%**	**18.9%**
Total diversion from LF[c]	**51.3%**	**57.8%**	**49.8%**	**48.9%**	**52.0%**	**69.6%**	**86.4%**
Total residues landfilled	**48.7%**	**42.2%**	**50.2%**	**51.1%**	**48.0%**	**30.4%**	**13.6%**

* Zero values were not included in the average calculations.

[a] Input: total paper, PET, HDPE, and PP received at plants.

[b] S2 and S6 mix food wastes with other organics to produce soil conditioner, which cannot be used for edible plants. S3, S4, and S5 produce a CLO that is used as daily cover in landfills.

[c] The material flow analysis of S5 does not include the WTE ash. Assuming 25.% WTE ash and zero beneficial use, would decrease the landfill diversion of S5 to 73.1%.

Table 2
Management of organic fraction in six MBT plants.

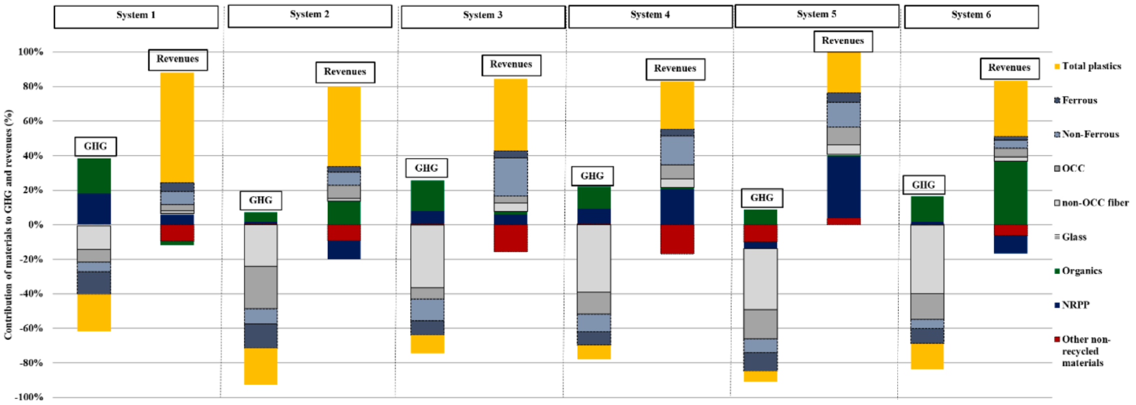
	S1	S2	S3	S4	S6	S5 (incl. AD)
Destination of composting product	Directly to LF	Soil conditioning	Daily cover in LF	Daily cover in LF	Soil conditioning	Feedstock to WTE
Description of composting process	N/A	In-vessel (6–7 weeks); maturation (2–3 weeks); refinement	In-vessel (12 days); then in open windrow, mixed, and wetted (1 week)	In-vessel (6–7 weeks); refinement	Aerated Static Pile (ASP) in 25 composting tunnels (3–4 weeks); maturation with turning system; refinement	Aerated Static Pile (ASP) (3–4 weeks); aerobic stabilization turning system
Material flows in and out:						
Organics, mainly food wastes, contaminated with fine particles[a]	**100.00%**	**65.00%**	**100.00%**	**100.00%**	**80.10%**	**71.20%**
Other organics added	**0.00%**	Branches, and leaves **35.00%**	**0.00%**	**0.00%**	Yard wastes **19.90%**	**0.00%**
Total compostables produced	**0.00%**	**36.90%**	**57.80%**	**38.40%**	**18.80%**	**28.50%**
Compost process losses (H_2O, CO_2)	0.00%	37.20%	23.10%	45.40%	32.40%	47.00%
Residue to WTE	**0.00%**	**0.00%**	**0.00%**	**0.00%**	**0.00%**	**24.50%**
Residue to LF	**100.00%**	**25.90%**	**19.10%**	**16.20%**	**48.80%**	**0.00%**
Organics to AD[b]						**28.80%**
MWh produced per ton organics processed anaerobically						0.16 MWh/ton
AD process losses, incl. biomethanisation						38.80%
AD residues to LF						40.80%
AD residues to WTE						20.40%

[a] the fines include paper, plastics, metals, glass and other non-recyclable materials. The fines were not included in the analysis for lack of data.

[b] Wet process. hydromechanical pre-treatment, including waste pulper and grit removal system. The subsequent anaerobic digestion of the cleaned organic suspension is carried out under mesophilic conditions (35–38 °C).

landfilled, both values decreased slightly to $14.8/metric ton, and $31.7/metric ton, respectively.

The study showed that currently most of the organic fraction is disposed in landfills, at best as daily cover in landfills, due to contamination from ‘dry’ recyclable materials e.g., plastics, glass, and metals. Also, the anaerobic digestion operations reported low recoveries, due to the inconsistent nature of the organic fraction. Similar observations have been reported by other scientists (Al-Wahaibi et al., 2020; Montejo et al., 2013; Salati et al., 2013; Fantozzi and Buratti, 2011; Donovan et al., 2010). Recent developments also use the organic fraction as a feedstock for bioethanol and biodiesel production, but the problematic nature of the organic fraction of mixed MSW hinders commercialization (Barampouti et al., 2019). Within the waste sector, the primary focus should be on the diversion of biodegradable organics from landfills



	S1	S2	S3	S4	S5	S6
tons GHG per ton of feed	(0.21)	(0.59)	(0.48)	(0.38)	(0.40)	(0.51)
total GHG (CO2-eq emissions)	**(3,418.1)**	**(45,164.0)**	**(71,842.9)**	**(70,342.6)**	**(100,387.6)**	**(177,836.4)**
Revenues from products ($)	**$ 772,403**	**$ 3,536,507**	**$ 5,452,050**	**$ 4,578,385**	**$ 7,105,963**	**$ 18,568,471**
Gate fee ($/ton)	$ 30	$ 45	$ 40	$ 50	$ 80	$ 60
Revenues from gate fee ($)	$ 495,000	$ 3,420,000	$ 6,000,000	$ 9,250,000	$ 20,000,000	$ 21,000,000
total revenues ($)	$ 1,267,403	$ 6,956,507	$11,452,050	$ 13,828,385	$ 27,778,617	$ 39,568,471
total revenues ($/ton of feed)	$ 76.8	$ 91.5	$ 76.3	$ 74.7	$ 111.1	$ 113.1

Fig. 3. Contribution of materials to GHG (metric tons $CO_{2\text{-eq}}$) and revenues of systems ($). Items in bold were used in the graph. A breakdown of each category, i.e., recyclables, organics, non-recycled materials and NRPP, is provided in the supplementary materials A2.

Table 3
CAPEX and OPEX values used for financial assessment.

CAPEX ($/ton)	S1	S2	S3	S4	S5	S6
Scenario A	322	342	317	294	737	360
Scenario B	263	287	264	248	588	299
OPEX ($/ton)						
Scenario A	32	36	30	30	58	38
Scenario B	26	31	26	27	49	32
CAPEX Ratio, A/B	1.22	1.19	1.20	1.19	1.25	1.20
OPEX Ratio, A/B	1.23	1.16	1.15	1.11	1.18	1.19

(UNEP, 2021) because it will reduce landfill methane emissions (US EPA, 2021; Duren et al., 2019; Jeong et al., 2017). Technologies to divert biodegradable wastes from landfills are commercially available and in widespread use today. The extent of their existing use is, in large part, directly a result of public policy. While the relative merits of each of these technologies are beyond the scope of this paper, the severity and magnitude of the climate challenge will require a suite of solutions, each of which is capable of being developed and applied in an environmentally protective manner.

4. Conclusions

It can be concluded that the optimal solution for the municipalities should be built upon a thorough sorting and recyclables recovery line, to obtain both the GHG emission savings and the positive revenues that such systems have achieved, from recyclables. The main revenues of the plants when recycling takes place, are contributed by the OCC, the HDPE, and ferrous and non-ferrous metals. For the high calorific value materials that cannot be recovered or do not have a value in the market, there should be a product stream of alternative fuel for the cement/energy sectors, which also reduces GHG emissions, by reducing residues to landfilling and also adds to the revenue stream. Systems that produce a fuel material for cement or energy production, indicated the highest landfill diversion rates. However, the operators of the MBT plants should secure long-term contracts with the cement operators, in order to minimize the risk of the investment. This is mainly associated with the willingness to pay of the cement manufacturers for alternative fuels, which greatly depends on the quality of the fuel, such as chlorine content, as well as the market price of fossil fuels (Gerassimidou et al., 2021; Waltisberg and Weber, 2020; Bourtsalas et al., 2018; IFC, 2017). In order to decide between alternative fuel production and recyclables recovery, the GHG emission savings and the prevailing gate fees are the important factors.

In the case of S5, the mixed alternative fuel and the residues of the MBT process are combusted to produce energy. This was found to be an efficient way that results in the minimum use of land for disposition of materials, without considering the indirect impact of producing energy from the residues of the MBT process in WTE plants, and the use of WTE bottom ash in the civil engineering sector, as practiced in the UK, France, Denmark, and elsewhere (Allegrini et al., 2015). The low quality of compost that is produced in most cases does not allow its use in agriculture and can only be used as daily cover in landfills. However, plants that utilize source separated organic materials, such as leaves and branches, achieve better quality compost that can be used as soil conditioner. It is now recognized that biodegradable organics should be diverted from landfills, in order to reduce methane emissions. However, considering the low capital and operational costs when all the residues are landfilled (S1-S3, and S6), cities that cannot support financially advanced integrated systems, such as anaerobic digesters and WTE plants should mandate the implementation of sanitary landfills equipped with landfill gas capture systems (Noya et al., 2018; Wilson et al., 2013).

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary material

Supplementary data to this article can be found online at https://doi.org/10.1016/j.wasman.2022.01.024.

References

Allegrini, E., Vadenbo, C., Boldrin, A., Astrup, T.F., 2015. Life cycle assessment of resource recovery from municipal solid waste incineration bottom ash. J. Environ. Manage. 151, 132–143. https://doi.org/10.1016/j.jenvman.2014.11.032.

Al-Wahaibi, A., Osman, A.I., Al-Muhtaseb, A.H., Alqaisi, O., Baawain, M., Fawzy, S., Rooney, D.W., 2020. Techno-economic evaluation of biogas production from food waste via anaerobic digestion. Sci. Rep. 10 (1) https://doi.org/10.1038/s41598-020-72897-5.

Ardolino, F., Berto, C., Arena, U., 2017. Environmental performances of different configurations of a material recovery facility in a life cycle perspective. Waste Manage. 68, 662–676.

Barampouti, E.M., Mai, S., Malamis, D., Moustakas, K., Loizidou, M., 2019. Liquid biofuels from the organic fraction of municipal solid waste: A review. Renew. Sustain. Energy Rev. 110, 298–314.

Beylot, A., Vaxelaire, S., Zdanevitch, I., Auvinet, N., Villeneuve, J., 2015 May. Life Cycle Assessment of mechanical biological pre-treatment of Municipal Solid Waste: a case study. Waste Manag. 39, 287–294. https://doi.org/10.1016/j.wasman.2015.01.033. Epub 2015 Feb 21 PMID: 25708404.

(Thanos) Bourtsalas, A.C., Zhang, J., Castaldi, M.J., Themelis, N.J., Karaiskakis, A.N., 2018. Use of non-recycled plastics and paper as alternative fuel in cement production. J. Cleaner Prod. 181, 8–16.

Cerda, A., Artola, A., Font, X., Barrena, R., Gea, T., Sánchez, A., 2018. Composting of food wastes: Status and challenges. Bioresour. Technol. 248, 57–67.

Chester, M., Martin, E., Sathaye, N., 2008. Energy, greenhouse gas, and cost reductions for municipal recycling systems. Environ. Sci. Technol. 42 (6), 2142–2149.

Cimpan, C., Maul, A., Wenzel, H., Pretz, T., 2016. Techno-economic assessment of central sorting at material recovery facilities – the case of lightweight packaging waste. J. Cleaner Prod. 112, 4387–4397.

Cimpan, C., Wenzel, Henrik, 2013. Energy implications of mechanical and mechanical–biological treatment compared to direct waste-to-energy. Waste Management 33 (7), 1648–1658. https://doi.org/10.1016/j.wasman.2013.03.026.

Cook, E., Wagland, S., Coulon, F., 2015. Investigation into the non-biological outputs of mechanical–biological treatment facilities. Waste Manage. 46, 212–226.

Directive (EU) 2018/851 of the European parliament and of the council of 30 May 2018 amending Directive 2008/98/EC on waste: https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:32018L0851&from=EN.

Di Lonardo, M.C., Franzese, M., Costa, G., Gavasci, R., Lombardi, F., 2016. The application of SRF vs. RDF classification and specifications to the material flows of two mechanical-biological treatment plants of Rome: Comparison and implications. Waste Manage. 47, 195–205.

Donovan, S., Bateson, T., Gronow, J., Voulvoulis, N., 2010. Characterization of Compost-Like Outputs from Mechanical Biological Treatment of Municipal Solid Waste. J. Air Waste Manage. Assoc. 60 (6), 694–701. https://doi.org/10.3155/1047-3289.60.6.694.

Duren, R.M., Thorpe, A.K., Foster, K.T., Rafiq, T., Hopkins, F.M., Yadav, V., Bue, B.D., Thompson, D.R., Conley, S., Colombi, N.K., Frankenberg, C., McCubbin, I.B., Eastwood, M.L., Falk, M., Herner, J.D., Croes, B.E., Green, R.O., Miller, C.E., 2019. California's methane super-emitters. Nature 575 (7781), 180–184. https://doi.org/10.1038/s41586-019-1720-3.

Edo-Alcón, N., Gallardo, A., Colomer-Mendoza, F.J., 2016. Characterization of SRF from MBT plants: Influence of the input waste and of the processing technologies. Fuel Process. Technol. 153, 19–27.

EPA Waste Reduction Model (WARM). Program documentation. Environmental Protection Agency. Vers. 14. N.p., Mar. 2016. Web. <https://www.epa.gov/warm/versions-waste-reduction-model-warm>.

EU, 2019a, Regulation (EU) 2019/1009 of the European Parliament and of the Council of 5 June 2019 laying down rules on the making available on the market of EU fertilising products and amending Regulations (EC) No 1069/2009 and (EC) No 1107/2009 and repealing Regulation (EC) No 2003/2003 (OJ L 170, 25.6.2019, pp. 1–114).

EU, 2019b, Directive (EU) 2019/904 of the European Parliament and of the Council of 5 June 2019 on the reduction of the impact of certain plastic products on the environment (OJ L 155, 12.6.2019, pp. 1-19).

European Environment Agency (EEA), Bio-waste in Europe — turning challenges into opportunities 2020, ISBN 978-92-9480-223-1, ISSN 1977-8449, doi:10.2800/630938.

Eurostat secondary material prices, accessed: July 2021, Link: https://ec.europa.eu/eurostat/statistics-explained/index.php?title=Recycling_%E2%80%93_secondary_material_price_indicator#Price_and_trade_volumes.

Fantozzi, F., Buratti, C., 2011. Anaerobic digestion of mechanically treated OFMSW: Experimental data on biogas/methane production and residues characterization. Bioresour. Technol. 102 (19), 8885–8892.

Fernandez-Nava, Y., del Rio, J., Rodriguez-Iglesias, J., Castrillon, L., Maranon, E., 2014. Life cycle assessment of different municipal solid waste management options: a case study of Asturias (Spain). J. Cleaner Prod. 81, 178–189. https://doi.org/10.1016/j.jclepro.2014.06.008.

Fitzgerald, G.C., Krones, J.S., Themelis, N.J., 2012. J, Greenhouse Gas Impact of Dual Stream and Single Stream Collection and Separation of Recyclables. Resour. Conserv. Recycl. 69, 50–56.

Gerassimidou, S., Velis, C.A., Williams, P.T., Castaldi, M.J., Black, L., Komilis, D., 2021. Dimitrios Komilis. Chlorine in waste-derived solid recovered fuel (SRF), co-combusted in cement kilns: A systematic review of sources, reactions, fate and implications. Crit. Rev. Environ. Sci. Technol. 51 (2), 140–186.

Hadzic, A., Voca, N., Golubic, S., 2018. Life-cycle assessment of solid-waste management in city of , Croatia. J. Mater. Cycles Waste Manage. 20 (2), 1286–1298. https://doi.org/10.1007/s10163-017-0693-2.

International Finance Corporation (IFC), Increasing the use of alternative fuels at cement plants: International best practice, 2017. Report available here: Link: https://www.ifc.org/wps/wcm/connect/33180042-b8c1-4797-ac82-cd5167689d39/Alternative_Fuels_08+04.pdf?MOD=AJPERES&CVID=lT3Bm3Z.

Jaunich, M., Levis, J.W., DeCarolis, J.F., Barlaz, M.A., Ranji Ranjithan, S., 2019. Environ. Sci. Technol. 53 (4), 1766–1775. https://doi.org/10.1021/acs.est.8b04589.

Jeong, S., Cui, X., Blake, D.R., Miller, B., Montzka, S.A., Andrews, A., Guha, A., Martien, P., Bambha, R.P., LaFranchi, B., Michelsen, H.A., Clements, C.B., Glaize, P., Fischer, M.L., 2017. Estimating methane emissions from biological and fossil-fuel sources in the San Francisco Bay Area. Geophys. Res. Lett. 44 (1), 486–495. https://doi.org/10.1002/2016GL071794.

Jia, X., Wang, S., Li, Z., Wang, F., Tan, R.R., Qian, Y., 2018. Pinch analysis of GHG mitigation strategies for municipal solid waste management: A case study on Qingdao City. J. Cleaner Prod. 174, 933–944. https://doi.org/10.1016/j.jclepro.2017.10.274.

Kaplan, P.O., Ranjithan, S.R., Barlaz, M.A., 2009. Use of life-cycle analysis to support solid waste management planning for Delaware. Environ. Sci. Technol. 43 (5), 1264–1270. https://doi.org/10.1021/es8018447.

Karine, I., Testa, M., Raymond, A., Graves, S., Gutowski, T., 2018. Performance evaluation of material separation in a material recovery facility using a network flow model. Resour. Conserv. Recycl. 131, 192–205.

Laurent, A., Bakas, I., Clavreul, J., Bernstad, A., Niero, M., Gentil, E., Hauschild, M.Z., Christensen, T., 2014. Review of LCA studies of solid waste management systems - Part I: Lessons learned and perspectives. Waste Manage. 34 (3), 573–588. https://doi.org/10.1016/j.wasman.2013.10.045.

Levis, J.W., Barlaz, M.A., DeCarolis, J.F., Ranjithan, S.R., 2014. Systematic exploration of efficient strategies to manage solid waste in US municipalities: Perspectives from the solid waste optimization life-cycle framework (SWOLF). Environ. Sci. Technol. 48 (7), 3625–3631. https://doi.org/10.1021/es500052h.

Levis, J.W., Barlaz, M.A., DeCarolis, J.F., Ranjithan, S.R., 2013. A generalized multistage optimization modeling framework for life cycle assessment-based integrated solid waste management. Environ. Modell & Softw. 50, 51–65. https://doi.org/10.1016/j.envsoft.2013.08.007.

Mastellone, M.L., Cremiato, R., Zaccariello, L., Lotito, R., 2017. Evaluation of performance indicators applied to a material recovery facility fed by mixed packaging waste. Waste Manage. 64, 3–11.

Madav, P., Samadder, S.R., 2018. Environmental impact assessment of municipal solid waste management options using life cycle assessment: a case study. Environ. Sci. Pollut. Res. 25 (1), 838–854. https://doi.org/10.1007/s11356-017-0439-7.

Merrild, H., Larsen, A.W., Christensen, T.H., 2012. Assessing recycling versus incineration of key materials in municipal waste: The importance of efficient energy recovery and transport distances. Waste Manag. 32 (5), 1009–1018. https://doi.org/10.1016/j.wasman.2011.12.025. Epub 2012 Jan 20 PMID: 22265239.

Montejo, C., Tonini, D., Márquez, M.D.C., Fruergaard Astrup, T., 2013. Mechanical–biological treatment: Performance and potentials. An LCA of 8 MBT plants including waste characterization. J. Environ. Manage. 128, 661–673.

Nasrullah, M., Hurme, M., Oinas, P., Hannula, J., Vainikka, P., 2017. Pasi Vainikka. Influence of input waste feedstock on solid recovered fuel production in a mechanical treatment plant. Fuel Process. Technol. 163, 35–44.

Noya, I., Inglezakis, V., Gonzalez-Garcia, S., Katsou, E., Feijoo, G., Moreira, M.T., 2018. Comparative environmental assessment of alternative waste management strategies in developing regions: A case study in Kazakhstan. Waste Manage. Res. 36 (8), 689–697. https://doi.org/10.1177/0734242X18786388.

Pressley, P.N., Levis, J.W., Damgaard, A., Barlaz, M.A., DeCarolis, J.F., 2015. Analysis of material recovery facilities for use in life-cycle assessment. Waste Manage. 35, 307–317.

Puyuelo, B., et al., 2019. Quality assessment of composts officially registered as organic fertilisers in Spain. Spanish J. Agric. Res. 17 (1), e1101. https://doi.org/10.5424/sjar/2019171-13853.

Ripa, M., Fiorentino, G., Vacca, V., Ulgiati, S., 2017. The relevance of site-specific data in Life Cycle Assessment (LCA). The case of the municipal solid waste management in the metropolitan city of Naples (Italy). J. Cleaner Prod. 142, 445–460. https://doi.org/10.1016/j.jclepro.2016.09.149.

Rodrigues, L.C., et al., 2020. The impact of improper materials in biowaste on the quality of compost. J. Cleaner Prod. 251, 119601. https://doi.org/10.1016/j.jclepro.2019.119601.

Salati, S., Scaglia, B., di Gregorio, A., Carrera, A., Adani, F., 2013. Mechanical biological treatment of organic fraction of MSW affected dissolved organic matter evolution in simulated landfill. Bioresour. Technol. 142, 115–120.

Samolada, M.C., Zabaniotou, A.A., 2014. Energetic valorization of SRF in dedicated plants and cement kilns and guidelines for application in Greece and Cyprus. Resour. Conserv. Recycl. 83, 34–43.

Schanes, K., et al., 2018. Food waste matters — a systematic review of household food waste practices and their policy implications. J. Cleaner Prod. 182, 978–991. https://doi.org/10.1016/j. jclepro.2018.02.030.

Scherhaufer, S., et al., 2018. Environmental impacts of food waste in Europe. Waste Manage. 77, pp. 98 113. https://doi.org/10.1016/j.wasman.2018.04.038.

Slorach, P.C., et al., 2019. Environmental and economic implications of recovering resources from food waste in a circular economy. Sci. Total Environ. 693 https://doi.org/10.1016/j.scitotenv.2019.07.322.

United Nations Environmental Program (UNEP), 2021. Global Methane Assessment: Benefits and Costs of Mitigating Methane Emissions, Link available here: https://www.unep.org/resources/report/global-methane-assessment-benefits-and-costs-mitigating-methane-emissions.

United States Environmental Protection Agency (USEPA), 2021 U.S. Inventory of Greenhouse Gas Emissions and Sinks: 1990 – 2019. https://www.epa.gov/ghgemissions/inventory-us-greenhouse-gas-emissions-and-sinks-1990-2019.

Velis, C., Wagland, S., Phil, L.P., Bryce, R.B., Keith, S.K., Wise, S., Pollard, S., 2012. Solid Recovered Fuel: Influence of Waste Stream Composition and Processing on Chlorine Content and Fuel Quality. Environ. Sci. Technol. 46 (3), 1923–1931. https://doi.org/10.1021/es2035653.

Vrancken, C., Longhurst, P.J., Wagland, S.T., 2017. Critical review of real-time methods for solid waste characterisation: Informing material recovery and fuel production. Waste Manage. 61, 40–57.

Waltisberg, J., Weber, R., 2020. Disposal of waste-based fuels and raw materials in cement plants in Germany and Switzerland – What can be learned for global co-incineration practice and policy? Emerg. Contamin. 6, 93–102.

Wilson, D.C., Velis, C.A., Rodic, L., 2013. Integrated sustainable waste management in developing countries. Waste Resour. Manage. 166 (2), 52–68.

WRAP, UK, secondary materials prices, 2018. Link: https://archive.wrap.org.uk/content/glass-4 (accessed: July 2021).