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About the environmental sustainability of the European management of WEEE plastics

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ABSTRACT

A huge increase of waste of electrical and electronic equipment (WEEE) is observing everywhere in the world. Plastic component in this waste is more than 20% of the total and allows important environmental advantages if well treated and recycled. The resource recovery from WEEE plastics is characterised by technical difficulties and environmental concerns, mainly related to the waste composition (several engineering polymers, most of which containing heavy metals, additives and brominated flame retardants) and the common utilisation of sub-standard treatments for exported waste.

An attributional Life Cycle Assessment quantifies the environmental performances of available management processes for WEEE plastics, those in compliance with the European Directives and the so-called substandard treatments. The results highlight the awful negative contributions of waste exportation and associated improper treatments, and the poor sustainability of the current management scheme. The ideal scenario of complete compliance with European Directives is the only one with an almost negligible effect on the environment, but it is far away from the reality. The analysed real scenarios have strongly negative effects, which become dramatic when exportation outside Europe is included in the waste management scheme. The largely adopted options of uncontrolled open burning and illegal open dumping produce huge impacts in terms of carcinogens ($3.5\cdot10^{+7}$ and $3.6\cdot10^{+4}$ person·year, respectively) and non-carcinogens ($1.7\cdot10^{+8}$ and $2.0\cdot10^{+6}$ person·year) potentials, which overwhelm all the other potential impacts. The study quantifies the necessity of strong reductions of WEEE plastics exportation and accurate monitoring of the quality of extra-Europe infrastructures that receive the waste.

1. Introduction

1.1. The global framework of electrical and electronic waste management

Electrical and electronic equipment (EEE) includes a wide spectrum of goods “containing circuitry or electrical components with either power or battery supply” (STEP, 2014). They greatly contribute to achieving a higher standard of living, and continue to gain importance in several sectors such as health, energy, transport, security and, more recently, education. At the same time, their production, consumption and disposal are no longer sustainable. In particular, the end-of-life stage generates a waste stream, called WEEE (waste of electrical and electronic equipment) or simply e-waste, which is not easy to manage since each item has a specific composition, characterised by hazardous and valuable chemical elements (up to 69, as reported by Deubzer et al., 2019), linked together in a complex way, and then requiring peculiar processes/technologies for recycling or disposal. A recent tech-

Abbreviations: ABS, Acrylonitrile–Butadiene–Styrene; APC, Air Pollution Control; BAT, Best Available Technologies; BAT-AEPLs, BAT Associated Environmental Performance Levels; BFR, Brominated Flame Retardant; CHP, Combined Heat and Power; CP, Carcinogens Potential; CRT, Cathode Ray Tube; DecaBDE, Decabromodiphenyl ether; EEE, Electrical and Electronic Equipment; E-LCA, Environmental Life Cycle Assessment; EU, European Union; FR, Flame Retardant; G&SD, Goal and Scope Definition; GHG, GreenHouse Gases; GWP, Global Warming Potential; HIPS, High Impact Polystyrene; LAM, Lamp; LCA, Life Cycle Assessment; LCI, Life Cycle Inventory; LCIA, Life Cycle Impact Assessment; LCT, Life Cycle Thinking; LE, Large equipment; MFA, Material Flow Analysis; MG, Moving Grate; NCP, Non-Carcinogens Potential; NREP, Non-Renewable Energy Potential; PA, Polyamide; PBDD, Polybrominated Dibenzo-p-Dioxins; PBDE, Polybrominated Diphenyl Ether; PC, Polycarbonate; PC+ABS, Blends of Polycarbonate and Acrylonitrile–Butadiene–Styrene; PE, Polyethylene; POP, Persistent Organic Pollutant; PP, Polypropylene; PS, Polystyrene; PUR, Polyurethane; PVC, Polyvinyl Chloride; SAN, Styrene-Acrylonitrile Resin; PBT, Polybutylene Terephthalate; PMMA, Poly Methyl Methacrylate; RINP, Respiratory INorganics Potential; RK, Rotary Kiln; RR, Recovery Rate; SE+ICT, Small Equipment+Small IT and Telecommunication Equipment; S&M, Screens and Monitors; SFA, Substance Flow Analysis; TEE, Temperature Exchange Equipment; TEP, Terrestrial Ecotoxicity Potential; VF, Variation Factor; WEEE, Waste of Electrical and Electronic Equipment; WEEP, WEEE mixed plastics; WM, Waste Management; WTE, Waste-to-Energy; WWTP, Waste Water Treatment Plant.

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nical report (Forti et al., 2020) quantified the crucial numbers of the WEEE in the world, highlighting how much they are affected by the increasing consumption rates of EEE, their too short life cycles, and the too few repair options that are really available. WEEE generation increases everywhere in the world, reaching in 2019 a global amount of about 53.6 Mt, that means each inhabitant of our planet produces 7.3 kg each year. The generation is concentrated in Asia (24.9 Mt), Americas (13.1 Mt) and Europe (12 Mt), with lower amounts in Africa (2.9 Mt) and Oceania (0.7 Mt). Europe has the highest WEEE generation per capita (16.2 kg), similar to those of Oceania and Americas (Forti et al., 2020). A national policy for a WEEE proper collection and management has been now adopted by 78 countries but their application is still inefficient for a series of reasons (limited investments, lack of stimulating actions, rules not easy to understand and comply with, and absence of harmonisation across countries). This creates a not acceptable gap between the 9.3 Mt of documented WEEE, which are formally collected and properly recycled (17.4% of global e-waste generation), and the 44.3 Mt of "not documented" WEEE (82.6% of the total). The possible, and too often unknown, fates of this not officially quantified e-waste are different: managed by scavenging or informal sector, that is collected and processed by poor or not well developed management infrastructures, mainly in middle- and low-income countries (where the hazardous substances are generally not depolluted, with high potential of severe health effects on informal workers); lost or discarded into waste bins, mainly for small-size items and particularly in high-income countries; illegally traded, and generally shipped from Northern hemisphere to developing countries, especially Asia and Africa (Hosoda, 2007; BioIS, 2013), and/or illegally dumped, as it has been documented by the cited report "the Global E-waste Monitor 2020" (Forti et al., 2020) and the EU-FP7 project "Countering WEEE Illegal Trade-CWIT" (Huisman et al., 2015). The partitioning between documented and not documented WEEE is not uniform around the world, with collection/recycling rates that appear affected mainly but not only by income level, being 0.9% in Africa, 8.8% and 9.4% in Oceania and Americas, 11.7% in Asia and 42.5% in Europe (Forti et al., 2020).

There is a great concern for the potential environmental impacts and human health risks related to the not documented WEEE flows and their related improper treatments (Jonkers et al., 2016). E-waste has a not negligible content of hazardous additives, such as brominated flame retardants (BFRs) or mercury, but it can be also considered a urban mine, due to the content of precious materials, such as critical and not-critical metals, and plastics (D'Adamo et al., 2016; Arduin et al., 2020). Europe ranks first for the amount of documented and properly recycled WEEE, but anyway only about 3.5 Mt/y of the total 12 Mt/y of WEEE are officially collected, and then dismantled, sorted and processed in re-manufacturing units and reported to authorities across Europe. This creates doubts about the overall sustainability of the whole European WEEE recycling chain, which is still too much affected by illegal trading/exportation or illegal collecting/processing (BioIS, 2013; Huisman et al., 2015).

1.2. Main concerns of resource recovery from WEEE plastics

A crucial aspect of WEEE management is related to the fraction of plastics, whatever the system (responsible or not, legal or not) for its collection and processing. WEEE plastics (WEEP) have a key role for their mass percentage, more than 20% of the total waste (Taurino et al., 2010; Beigbeder et al., 2013), and the important advantages that can be obtained from their recycling and return on the market (EERA, 2020) but also for the content of hazardous additives that makes complicate the recycling process and requires appropriate, safe and efficient, processes and technologies. The amount of collected WEEP in Europe is limited, and equal to about 0.7 Mt/y (EERA, 2020). Only a small part of these plastics can be sent to recycling facilities (the maximum treatment capacity in Europe is about 0.2 Mt/y, based on EERA, 2018), even though environmental and economic advantages of their recycling have been demonstrated (EERA, 2020; Forti et al., 2020). Important reasons explain a such limited capacity of proper handling, also in the European Union that has the highest documented rates of collection and recycling (EERA, 2017). The first, and probably principal, reason is that the recycling process of WEEP is technologically difficult, since it has to process a mix of different polymers. A large part of them (about 50%) is made of engineering polymers (ABS, acrylonitrile–butadienestyrene, HIPS, high impact polystyrene, PC +ABS, polycarbonate/ABS blends) that cannot be recycled by conventional mechanical techniques without a high material degradation, which makes weak the access to the market (Beigbeder et al., 2013; Gu et al., 2017). For example, ABS releases volatile compounds (mainly styrene derivates), leading to a significant loss of key physical properties (Ragaert et al., 2017). Moreover, a not negligible amount of BFRs is present in WEEP - and particularly in styrenic polymers (HIPS and ABS) that have a lower limiting oxygen index (Jonkers et al., 2016) - to comply with safety standards (Wagner and Schlummer, 2020). This amount, which can be estimated in the range 10–30% (Vehlow et al., 2003; Haarman et al., 2020), cannot be eliminated by conventional mechanical recycling (Lucas et al., 2018a), and is a risk for the environment and health of waste operators as well as that of users of recycled products (Ionas et al., 2014). Then, there is a huge necessity of specific and efficient management options able to minimise health and ecological risks. New recycling technologies, able to remove hazardous compounds (such as BFRs) from WEEP, are recently approaching the market, even though most of them are still at a pilot- or demonstrative-scale level (González et al., 2016; Schlummer et al., 2016; NONTOX, 2020; Wagner and Schlummer, 2020).

The second important reason is the lack of reliable data about the quantity and composition of plastics separated from WEEE (in terms of the type of polymer and content of BFRs) and those sent to the different recycling or disposal treatments. This concern is further complicated by the large recourse to illegal or not responsible management procedures, without any specialised treatment facility: the wide utilisation of backyard recycling system poses severe risks for human health and the environment (Osako et al., 2004; EERA, 2017; Forti et al., 2020). This dramatically dangerous scenario is also related to the much higher costs for recycling in Europe: the demands for compliance required to European recyclers do not apply to recyclers outside Europe, so that the cost saving determined by noncompliant treatment exceeds the normal economic margins of responsible recyclers (Magalini and Huisman, 2018), making unfair competition. This implies that a too large fraction of WEEP is currently not (or not properly) recycled, and the fraction containing BFRs is sent to incineration for energy recovery (Lucas et al, 2018b; Forti et al., 2020).

Last but not least, there are the difficulties related to the continuous changes in the EU legislation. The WEEE Directive was introduced in 2003 (EC, 2003) and then updated in 2012 (EC, 2012): it requires to regulate collection, recycling and recovery and reduce WEEE disposal to landfill by increasing the collection rate and sustainable management through high reuse and recycling rates (Arduin et al., 2019; EERA, 2018). In the meantime, the so-called RoHS Directive (EC, 2017) establishes a number of restrictions in the use of certain hazardous substances (heavy metals such as lead, mercury, cadmium, and hexavalent chromium, and flame retardants such as polybrominated diphenyl ethers (PBDE)) in electrical and electronic equipment, in order to provide their substitution with safer alternatives. Further accurate rules have been established for all the waste management options by the new Best Reference Documents for waste treatment (EC-JRC, 2018) and waste incineration (EC-JRC, 2019) and by the European regulation on persistent organic pollutants (EU, 2019). It could appear that the legislation evolves faster than technological performances of the recycling industry, which for a (too) long period simply maintained the old conventional processes of mechanical recycling. This implies that recyclers are not always able to match EU targets for resource recovery, so making most of WEEPs unrecyclable, and discarded with a loss of resources (EERA, 2018).

1.3. Previous LCA studies

There are several Life Cycle Assessment (LCA) papers related to WEEE management but most of them focus on the recovery of common or precious metals (Fiore et al., 2019) or other valuables materials, such as liquid crystal displays (Amato et al., 2016) or cathode ray tube (CRT) glass (Song et al., 2018), with a limited attention to WEEP management. Most of the papers refer to one or few types of WEEE (such as televisions and computers (Hong et al., 2015), refrigerators (Xiao et al., 2016) or CRT televisions (Song et al., 2018)), or to only some categories established by the current legislation (Fiore et al., 2019). On the contrary, Gu et al. (2017) provide an LCA of specific treatment for waste plastics, but without any focus on the WEEE sector. Furthermore, the studies or sections dedicated to WEEP management often refer to a generic "mixed plastics" stream, without reliable information about their composition and the specific treatment for each polymer. This is probably due to the difficulties to acquire reliable data about WEEP composition and physico-chemical characteristics (Taurino et al., 2010; Kohl and Gomes, 2018), together with those of recycled polymers, at least in terms of substitution factor of virgin material (Beigbeder et al., 2013). This factor is often assumed as 1:1 (Wager and Hischier, 2015; Fiore et al., 2019) without any specific scientific or technical support. Moreover, almost all LCA papers do not consider the role of flame retardants or that of improper waste treatments and disposal. Both are excluded from the analysis due to the difficulty of a reliable quantification (Biganzoli et al., 2015), even though there are some important exceptions, such as the analysis carried out by Jonkers et al. (2016). Finally, none of the LCA study on WEEP developed an analysis extended to the whole management in the European context, but focused on a specific region: Xiao et al. (2016) and Gu et al. (2017) published papers on Chinese situation, while some papers on specific European regions have been published by Biganzoli et al. (2015), with reference to Lombardia region in Italy, Unger et al. (2017), to Austrian situation, and De Meester et al. (2019), to Flanders in Belgium. Other papers focused on a specific treatment facility, such as that by Hong et al. (2015) in China, Fiore et al. (2019) in Italy or Wager and Hischier (2015) in central Europe. The latter is one of the few LCA papers focused on plastics from WEEE. All the mentioned studies agree on the necessity to establish environmentally sound practices to treat WEEE, due to their complexity and increasing quantities. The reported LCA results highlight the importance of proper collection and treatments, especially recycling, which seems from any point of view better than the alternative treatments: energy demand (Song et al., 2018), greenhouse gases (GHG) emissions (Xiao et al., 2016; Song et al., 2018) or impacts on human health, ecosystems quality and resources availability (Wager and Hischier, 2015; Xiao et al., 2016; Fiore et al., 2019),

1.4. Objectives of the study and its innovative aspects

The study has been carried out in the framework of the European Horizon2020 project "Nontox" (NONTOX, 2020), with the aim of describing in a proper detail the technologies today adopted for the "responsible" management of WEEE plastics, i.e. those in compliance with the European WEEE Directive (EC, 2012), but also the substandard or improper treatments in developing countries, such as illegal dumping and uncontrolled burning. The related limitations and concerns have been detected and quantified, in terms of technical feasibility and, above all, environmental sustainability. As mentioned above, these aspects are generally ignored in LCA studies, even though their importance is largely recognised, as in the two mentioned international reports (Huisman et al., 2015; Forti et al., 2020). The estimated environmental burdens, characterising each specific management option and related to the average composition of WEEE plastics in Europe, have been used to quantify the environmental performances of possible management schemes in the European context. The potential maximum amounts of materials obtainable from officially collected WEEP have been also quantified, by developing a Material Flow Analysis (MFA) referred to an ideal scenario where all WEEP are treated by mechanical recycling and the related residues are sent to thermal treatments. The study first describes the methodological approach, which substantially coincides with the Goal&Scope Definition (G&SD) of the implemented LCA. Then, it analyses the technological options available for WEEP treatment, and reports the necessary Life Cycle Inventory (LCI) tables of direct, indirect and avoided burdens. Finally, the results of Life Cycle Impact Assessment (LCIA) are presented and discussed, with the support of a sensitivity analysis.

2. Methodological approach

2.1. The system under analysis and the functional unit

The study has been developed in compliance with the international standard ISO 14044 (ISO, 2006), by utilising an attributional, process-based approach, then accounting for impacts directly related to the system of interest and attributing them to the activities within the system in a current perspective (Royal Academy of Engineering, 2017). The G&SD identified the "system to be studied" as the European management chain(s) of the mixed plastic wastes coming from WEEE, starting from their sorting and pre-treatment, through different processes to remove hazardous fractions, until the recovery of secondary plastics and the disposal of residues. The analysed scheme of current WEEP management takes into account only the plastics obtained by the dismantling of official collected WEEE in Europe, which are about 0.7 Mt/y (EERA, 2020). The study does not consider plastics contained in WEEE not officially collected and does not investigate illegal trading of these wastes. The flow diagram in Fig. 1 summarises the current management scheme for WEEP, with the indication of involved process units and the identification of their main environmental burdens. The "system boundaries" are those generally indicated as a "gate-to-gate", where the input gate is that of mixed plastic obtained downstream of the WEEE separate collection and dismantling and the final gate is that of recycled plastics or recovered energy. The involved stages are those of mechanical recycling (plastic sorting and re-manufacturing), thermal treatment, landfilling, taking also into account the trading that can interest a large fraction of the sorted stream of total WEEP.

The "function of the systems under analysis" is the management of plastics coming from WEEE, classified in six categories in the Annex III of the European Directive 2012/19/EU (EC, 2012): Temperature exchange equipment; Screens, monitors, and equipment containing screens having a surface greater than 100 cm^2; Lamps; Large equipment (any external dimension higher than 50 cm); Small equipment (no external dimension higher than

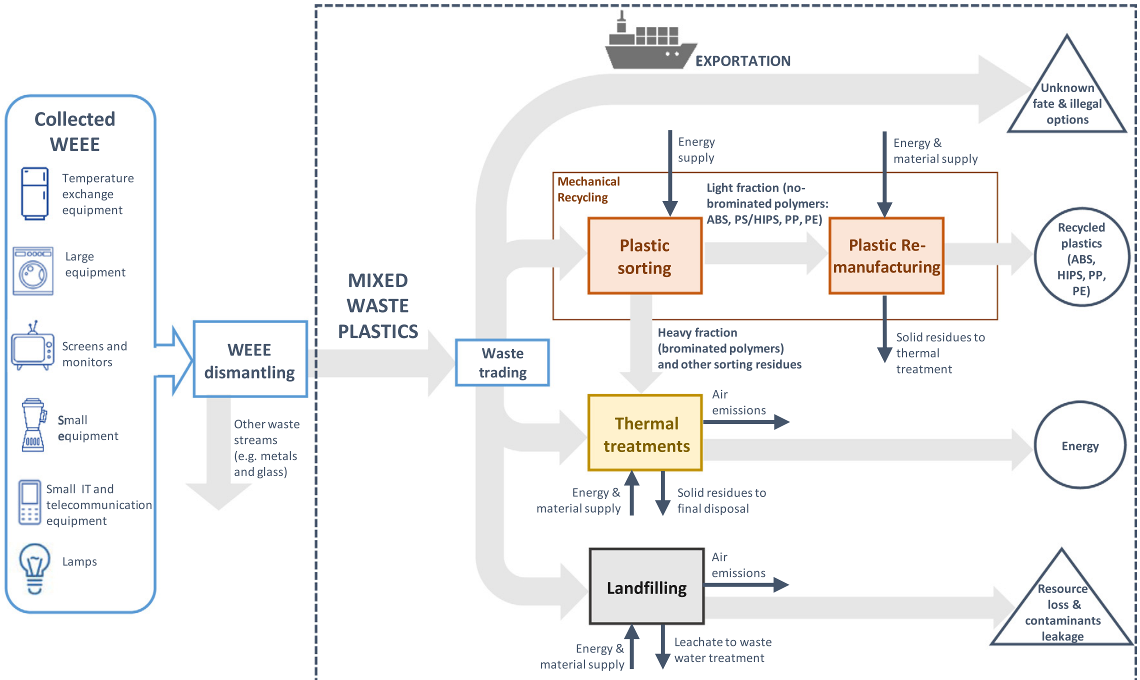

Fig. 1. Qualitative management scheme for WEEE plastics, with the indication of involved process units and main related environmental burdens. Dashed lines identify the system boundaries, i.e. the activities taken into account in the LCA study.

50 cm); Small IT and telecommunication equipment (no external dimension higher than 50 cm); taking into account that the category 6 is sometimes grouped with small equipment.

Table 1 shows the amount of mixed plastics obtained from collected WEEE for each category, in Europe in 2017, and their composition. It has been updated based on a cross-check with samples collected and analysed in the framework of NONTOX project (NONTOX, 2020), adopting a top-to-bottom approach. The "functional unit" has been accordingly defined as the management of 732,000 t/y of WEEP, including the contribution of each category in terms of amount and types of polymers reported in Table 1.

It is not easy to define a reliable composition of WEEE plastics, mainly due to the complexity of new electronic apparatus continuously entering the market and the cited high fraction of not officially documented WEEE. The composition reported in Table 1 has been obtained from specific databases provided by an association of WEEE producer responsibility organisations (WF-RepTool, 2020; WEEE forum, 2020; Prosum Database, 2020). The obtained composition is in substantial agreement with those of three reference studies in the field (Stenvall et al.,2013a; Buekens and Yang, 2014; Haarman et al., 2020), and has been further detailed, quantifying BFR content of WEEP by means of a specific substance flow analysis (SFA). The "allocation problem" has been avoided by utilising the system expansion methodology ("avoided burdens method"), by identifying which products are replaced on the markets by the obtained co-products and including their replacement in the model. The adopted procedure is that proposed by Vadenbo et al. (2017), which quantifies the substitution potential γ of an available market product (virgin plastics) with a secondary resource (recycled plastics), as the product of four parameters:

Table 1
Amount and composition of WEEE plastics for each category, in Europe in 2017. Sources: WF-RepTool, 2020; WEEE forum, 2020; Prosum Database, 2020.

	TEE Temperature exchange equipment	LE Large equipment	S&M Screens and monitors	SE + ICT Small equipment + small IT and telecommunication equipment	LAM Lamps	TOTAL
Collected WEEE (t/y)	693,000	1,227,000	452,000	1,082,000	35,000	3,488,000
Mixed plastics (t/y)	100,000	172,000	80,000	379,000	2,000	732,000
Other waste (t/y)	593,000	1,055,000	372,000	703,000	33,000	2,756,000
Mixed plastics composition (%)						
ABS	2.71	14.67	21.89	35.62	7.55	24.65
PA	0.23	0.21	–	0.48	12.13	0.36
PC	0.12	0.95	2.66	4.17	13.23	2.72
PBT	–	0.98	2.71	2.35	23.47	1.8
PC + ABS	–	1.56	8.18	7.49	–	5.14
PE	0.41	0.92	0.02	1	0.72	0.79
PMMA	–	0.39	1.57	1.13	6.38	0.86
PP	11.63	56.4	12.19	14.55	3.43	23.68
PS (including **HIPS**)	42.27	2.81	40.62	15.91	11.15	19.13
PUR	37.2	0.36	–	0.04	–	5.17
PVC	3.88	3.18	0.01	0.35	7.13	1.47
SAN	0.12	0.04	0.01	0.19	–	0.13
Other plastics	1.43	17.53	10.14	16.72	14.80	14.10

$$\gamma = U \cdot \eta \cdot \alpha \cdot \pi \quad (1)$$

These are: the potential physical amount of the secondary resource (U); the related recovery efficiency of this resource (η), which depends on the waste plastic management option; the substitutability (α), that represents the functionality provided by the recovered resource compared to that of the conventional resource; the market response (π), which is the share of secondary resource that can effectively displace the available product on the market. The "quality of data" is rather high, considering that all amounts and composition data have been acquired by the consortium of thirteen important institutions and private companies active in the NONTOX project, and that an extended analysis of scientific and technical studies has been carried out to estimate and compare all the direct and avoided environmental burdens. Indirect burdens have been obtained from Ecoinvent databank v.3.3 (Ecoinvent, 2016). The burdens related to the infrastructures have not been included. The European energy mix has been utilised to evaluate the avoided burdens related to the exported electricity (IEA, 2019). The "selected LCIA methodology" is Impact 2002+ (Jolliet et al., 2003), which has been used with the support of the software package SimaPro© 8.4 (SimaPro, 2020). The study has to be considered valid within the set of assumed specific conditions and hypotheses.

2.2. The current scenarios of WEEE plastics management

Three European scenarios have been defined and quantified based on different sources (EERA, 2018; 2020; Haarman et al., 2020; Forti et al., 2020). The reference scenario is an "ideal current management scheme", where the collected WEEP are all treated by means of the best options currently available, i.e. the mechanical recycling for material recovery, when this is possible, and the combustion for energy recovery, without any material sent to landfilling (Table 2). The disposal to (legal or illegal) landfills as well as the exportation outside Europe are excluded by this reference scenario but taken into account in two alternative scenarios that quantify the "real current management scheme". The ideal current scenario allows a quantification of maximum achievable recovery rates for WEEP, and the identification of WEEE categories with lowest amount of recycled plastics. The alternative management scenarios (Real_1 and Real_2) have been defined based on two international reports (EERA, 2018; Haarman et al. 2020). The first takes into account the evidence of the large amount of WEEP that is exported to Asia or Africa (Forti et al., 2020), which generally means to undefined treatments due to the lack of reliable information about the real fate of these wastes. These treatments are mainly those of open dumping and open burning, as they will be specified in the next section. The second real scenario has been defined based on data recently published by the Sofies group (Haarman et al., 2020), which indicates large fractions of WEEP sent to material recovery (44%) and direct energy recovery (45%), and only a limited part (11%) disposed in sanitary landfills. The Sofies report does not take into account the exportation after the WEEP official collection.

As already made in similar studies (Ardolino et al., 2017; 2020), a material flow analysis (MFA) has been carried out to quantify the mass flow rates of all the streams in input and output in the three scenarios. Figs. 2 and 3 show the quantitative flow sheets related to the total plastics mass flow rates, whereas the flow sheets related to the mass flow rates of brominated compounds can be found in Annex A (Figures A.1-A.3). Data processing has been carried out based on the information and assumptions reported in the next section. The MFA of the Ideal scenario permits to quantify the highest achievable annual mass flow rate of recycled plastics (about 370 kt), and then the maximum recovery rates for WEEP (50%), expressed as the ratio between the mass flow rate of recycled plastics and that of total collected WEEP. This value has been quantified by taking into account material losses during sorting and re-manufacturing stages for each WEEE category, as detailed in the paragraph 3.1.1 and reported in Figures A.6-A.10 of Annex A. The obtained material recovery rates appear in good agreement with a recent technical report (Haarman et al., 2020). They are high (51% and 68%) for TEE and LE, and lower (44%, 44% and 20%) for S&M, SE + ICT, LAM, indicating the categories that could receive major advantages by the development of innovative recycling solutions. The SFA related to the brominated compounds is based on the bromine content measured in about one hundred samples (including WEEE mixed plastics, single polymers and sorting residues), analysed in the framework of NONTOX project. These analyses permit to quantify a total BFR content in the assumed WEEP composition equal to 8900 ppm (i.e. 8900 g/t_{WEEP}) and to identify its partitioning in the output streams directed to different process units. The BFR content has been also quantified with reference to WEEPs belonging to different WEEE categories, as reported in detail in Figure A.11 of Annex A. The quantification of the related environmental burdens has been developed by considering that the total BFRs are composed by Tetrabromobisphenol A, TBBPA (75%), Decabromodiphenyl ether, DecaBDE (13%) and 1,2-bistribromophenoxyethane, TBPE (12%), as obtained by samples analysed in the framework of NONTOX project.

3. Technological options for WEEE plastics management

The technical options that characterise the so-called responsible management/recycling of WEEP containing BFRs (EERA, 2018) and the substandard management (Forti et al., 2020) are described in the following, while the specific inventory tables of direct environmental burdens, obtained from recognised technical reports and scientific studies, are reported in Annex B.

3.1. Responsible management options

3.1.1 Mechanical recycling: plastic sorting and re-manufacturing

Mixed waste plastics obtained by WEEE collection, shredding and dismantling need to be sorted before re-manufacturing process (Beigbeder et al., 2013; Stenvall et al., 2013b). The WEEP sorting is generally carried out by a flotation process (also called "sink and float"), which allows a density-based separation of polymers. There are new sorting techniques that will progressively acquire a major role (Maisel et al., 2020) but it is reasonable to assume that the sink and float process is currently predominant, even though it has a limited selectivity (Haarman et al., 2020). The sorting process requires preliminary stages of washing and granulation to remove

Table 2
Main features of analysed scenarios of WEEE plastics management in Europe.

ID name	Material recovery	Direct energy recovery[1]	Landfilling	Exportation	Data source
Ideal	100 %	–	–	–	NONTOX, 2020
Real_1	25 %	–	–	75 %	EERA, 2018
Real_2	44 %	45 %	11 %	–	Haarman et al., 2020

[1] It does not include the streams of solid residues from plastics sorting units.

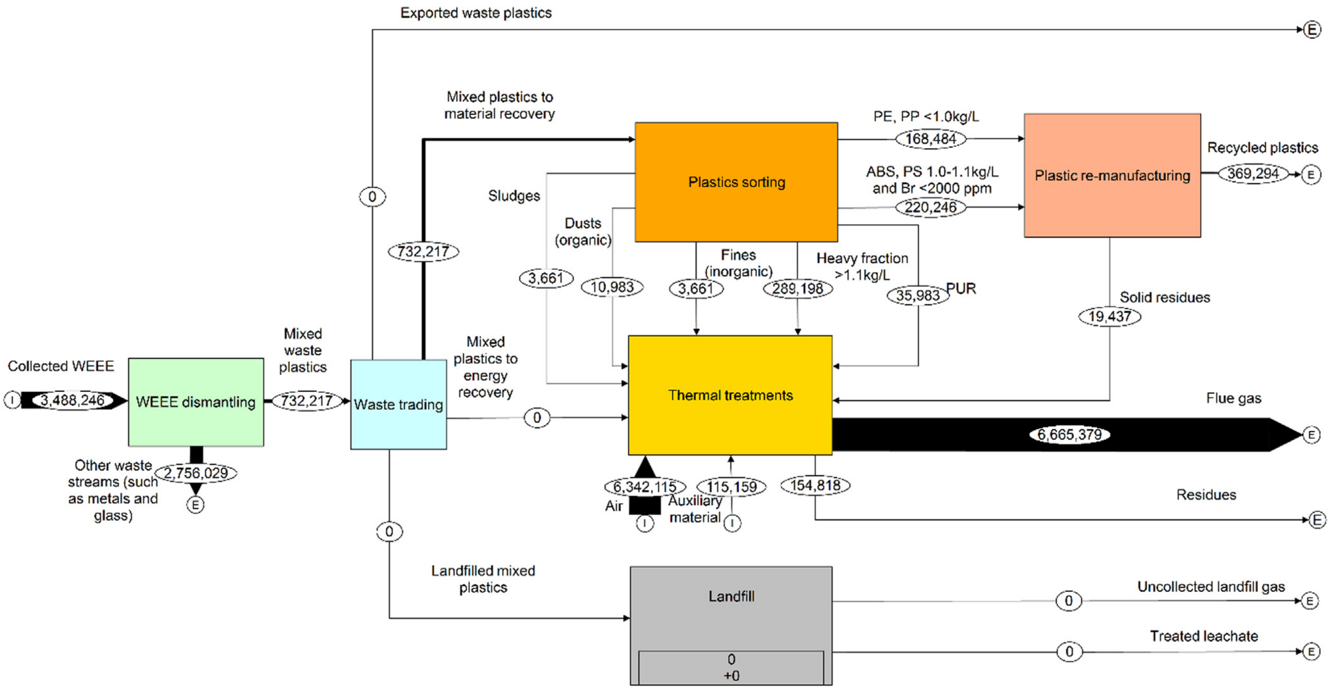

Fig. 2. WEEE plastics management in the ideal current scenario. Data refer to the functional unit and are expressed as t/y. I = import stream; E = export stream.

residual impurities from polymers and further reduce their size: this implies different density baths in big tanks, with different flotation media (such as pure water or water added with salts or other substances) to adjust the threshold density at the desired value (Makenji and Savage, 2012). The light polymers, mainly PP (which is about 24% of total WEEP) but also the limited amount of PE (0.79% of the total), are easily separated, having a density < 1.0 kg/L, and sent to plastic re-manufacturing, together with ABS and PS (including HIPS), having a bromine content < 2000 ppm and a density in the range 1.0–1.1 kg/L (Coolrec B.V., 2020; ERION, 2020; Relight, 2020; Haarman et al., 2020). The heavier fraction, whose composition is shown in Table B.5 of Annex B, includes mainly ABS and PS/HIPS with bromine content >2000 ppm and a density > 1.1 kg/L but also other heavy polymers (such as PC, PC + ABS, PVC, whatever their Br content is), some of the particle-filled and fibre-reinforced versions of styrenics and polyolefins. This heavy fraction is disposed in landfills or thermally treated (Forti et al., 2020), together with other sorting residues, such as dusts, fines, sludge and unrecyclable light polymers (i.e. PUR). This implies that two important WEEP fractions, those of PC (2.72% of the total) and PC + ABS (5.14%) cannot be sent to re-manufacturing (Coolrec B.V., 2020; ERION, 2020; Relight, 2020). It has been assumed, as also made by De Meester et al. (2019), that these sorting residues are sent to a moving grate combustor with energy recovery. The electricity consumption of the sorting process for 1 tonne of WEEP ranges between 10 and 40 kWh, depending on types and number of operating flotation units (Coolrec B.V., 2020). An average value of 25 kWh/t_{WEEP} has been assumed. The values of average material recovery rate (*RR*) for ABS, PS/HIPS, PE and PP, defined by the Eq. (2), have been utilised to quantify the amount and composition of recovered plastics and generated sorting residues.

$$RR\ \ of\ \ polymers\ \ P_i = \frac{Amount\ of\ P_i\ sent\ to\ plastic\ re-manufacturing}{Amount\ of\ P_i\ (included\ brominated\ P_i)\ in\ input\ to\ the\ sorting\ process} \quad (2)$$

Table 3 shows the values of *RR*s for each WEEE category, considering the density value and bromine content of each target polymers, as obtained by the high number of samples analysed in the framework of NONTOX project, together with the sorting efficiencies provided by operating facilities managed by some NONTOX partners (Coolrec B.V., 2020; ERION, 2020; Relight, 2020). A loss of 2.5% in sorting residues (dust, sludge and fines) has been evaluated for all WEEP. PE and PP show good levels of *RR*s (always higher than 85%) for each WEEE category while ABS and PS/HIPS have low levels of *RR* for LE, S&M and SE + ICT categories, due to the high-Br content, which is a strong limitation for conventional mechanical recycling.

The streams of recovered waste plastics (ABS, PS/HIPS, PE and PP) are sent to re-manufacturing, which requires some processes (such as shredding, extrusion and injection moulding) to obtain secondary polymers with mechanical characteristics similar to those of virgin materials (Beigbeder et al., 2013). It is important to note that due to the loss of transparency, the second life of PS is most likely as an opaque material performing in traditional applications of HIPS upon upgrading (Norner, 2020). The re-manufacturing stage generates a further loss of plastics and requires a certain amount of process energy and some material, such as additives and virgin polymers (Norner, 2020). A process efficiency of 95% has been assumed for all polymers, as an average value between 90% reported by De Meester et al. (2019) and 100% suggested by Norner (2020). According to Gu et al. (2017) and Aimplas (2020), an average electricity consumption of 360 kWh/t of plastic sent to the re-manufacturing stage (i.e. 189 kWh/t_{WEEP}) has been used. The addition of additives has been not considered, similarly to the studies by Beigbeder et al., (2013) and Stenvall et al. (2013b), which have been used as sources for technical performances of virgin and secondary polymers. Analogously, the blending of recovered polymers with virgin materials has been not taken into account, since the quantification of the related avoided burdens already considers their net contribution (Table 4). In particular, the overall recovery efficiency of secondary plastics utilises the values of *RR*s specifically quantified for each polymer

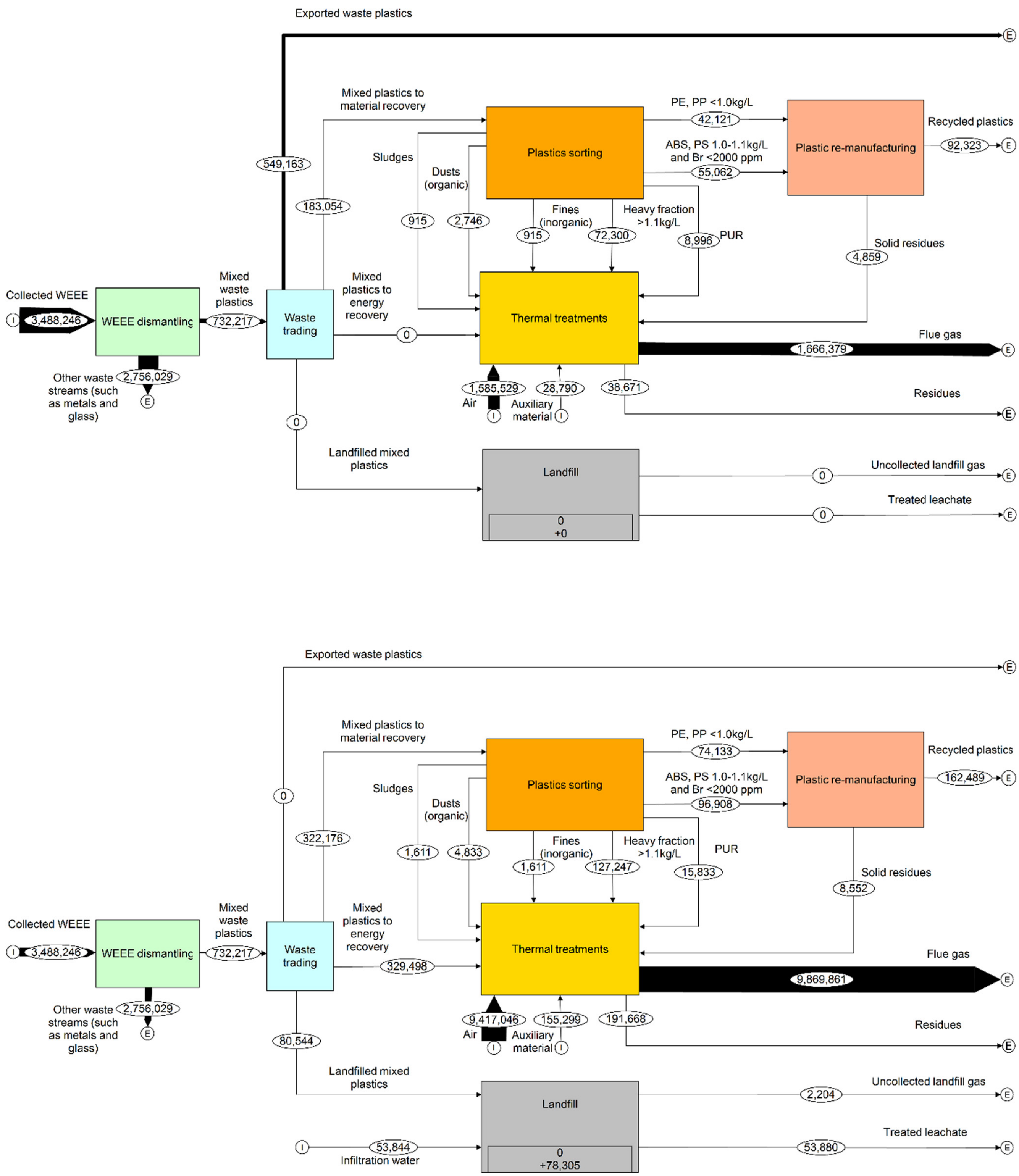

Fig. 3. WEEE plastics management in the real current scenarios (from top to bottom: Real_1 and Real_2), based on hypotheses of EERA (2018) and Haarman (2020). Data refer to the functional unit and are expressed as t/y. I = import stream; E = export stream.

(see Table 3), together with the re-manufacturing losses. For each polymer, the substitutability has been evaluated following the procedure proposed by Rigamonti et al. (2020), then selecting the key technical parameters and taking into account the related values for secondary and virgin polymers: for ABS and HIPS, the impact strength has been used, with the values measured by Beigbeder et al. (2013); for PP and PE, the elongation at yield has been considered, with the values reported by Stenvall et al. (2013b).

The LCI table with the main direct and avoided burdens related to the mechanical recycling of 1 tonne of WEEE mixed plastics is reported in Table B.1 of Annex B.

3.1.2 Thermal treatment

The combustion process is largely utilised for WEEPs that do not have the possibility of an efficient mechanical recycling leading to good-quality recycled polymers. The alternatives to the combus-

Table 3
Quantified material recovery rates for ABS, PS/HIPS, PE and PP, for each WEEE category

	Recovered polymers, %					
	TEE	LE	S&M	SE + ICT	LAM	TOTAL
ABS	93.4%	95.5%	54.4%	64.1%	90.9%	68.0%
PS/HIPS	94.8%	39.5%	56.3%	61.5%	93.0%	69.6%
PE	90.8%	97.5%	90.8%	95.7%	97.5%	95.8%
PP	90.8%	97.5%	97.0%	87.8%	97.5%	94.0%

Table 4
Assumed values of parameters for the quantification of the avoided burdens related to recycled plastics, by following the procedure proposed by Vadenbo et al. (2017) and Rigamonti et al. (2020). Burdens are shown with reference to 1 tonne of polymer sent to mechanical recycling and to the polymers contained in 1 tonne of mixed WEEE plastics sent to mechanical recycling.

	ABS	HIPS	PE	PP
Avoided burdens with reference to 1 tonne of polymer sent to mechanical recycling				
Potential physical amount of the secondary resource, $U_{recycled\ plastics}$ (t)	1	1	1	1
Recovery efficiency of the secondary resource, $\eta_{recycled\ plastics}$ (-)	0.65	0.66	0.91	0.89
Sorting efficiency	0.68	0.70	0.96	0.94
Re-manufacturing efficiency	0.95	0.95	0.95	0.95
The substitutability, $\alpha_{recycled\ plastics}$: $\alpha_{virgin\ plastics}$ (-)	0.76	0.99	0.75	0.75
Market response, $\pi_{plastics}$ (-)	1.00	1.00	1.00	1.00
Avoided production of virgin polymers, (t)	**0.49**	**0.65**	**0.68**	**0.67**
Avoided burdens with reference to polymer's content in 1 tonne of mixed WEEE plastics				
Potential physical amount of the secondary resource, $U_{recycled\ plastics}$ (kg)	247	191	7.9	237
Avoided production of virgin polymers, (kg)	**120**	**125**	**5.4**	**159**

tion process are those of gasification and pyrolysis, which are not yet largely utilised (Ragaert et al., 2017) and consequently have not been taken into account. The study considered the options of moving grate (MG) and rotary kiln (RK), with reference to the specific composition of waste plastics and according with the recent best available technologies (BAT) and BAT Associated Environmental Performance Levels (BAT-AEPLs), recently defined by the European Commission (EC-JRC, 2019). In particular, the moving grate option with energy recovery has been utilised for all the residues from plastics sorting/re-manufacturing (Figs. 2 and 3) and for the WEEP directly coming from waste trading (Fig. 3-bottom), by utilising the average values of BAT-AEPLs (for instance, 0.045 ng/m^3_N for PCDD/F + dioxin-like PCBs, so including PBDD). The rotary kiln option (without energy recovery) has been instead assumed only for the burned fraction of the exported WEEP (Fig. 3-top). In this case, the utilised combustion units have been assumed to be in weak compliance with the BREF document of European Commission, then the worst values of the BAT-AEPLs range have been utilised (for instance, 0.08 ng/m^3_N for PCDD/F + dioxin-like PCBs). The LCI table with the main direct and avoided burdens related to the possible thermal treatments of 1 tonne of WEEE mixed plastics is reported in Table B.2 of Annex B. In particular, it has been estimated an avoided burden of 2350 kWhe for each tonne of WEEP directly burned in a moving grate operated with municipal residual waste, which corresponds to an average efficiency of conversion in electric energy of 24%, based on BREF document (EC-JRC, 2019).

3.1.3 Landfilling

It is the most economical and simple WEEP responsible management option (Lucas et al., 2018a), and it is largely used in developed and developing countries (Forti et al., 2020), with different levels of engineering, e.g. with/without biogas capture systems and with/without liner to prevent soil and water contamination. Landfilling WEEP generates resource losses and can lead to health and environmental damages, since hazardous substances (such as BFRs) can be emitted into air, soil and groundwater by different ways and transferred into the human food chain (Lucas et al., 2018a; WHO, 2018). Simonson et al. (2000) assumed that the release of BFRs from WEEE landfilling is very low and limited to leachate. A different picture is reported by other studies, which detected high concentrations of these hazardous substances (in particular, PBDEs and TBBPA) on the landfill surface (Morin et al., 2017), in the leachate (Kim et al., 2006; Osako et al., 2004), and in different media of areas close to WEEP landfills (Tang et al., 2015). Danon-Schaffer and Mahecha-Botero (2010) developed a kinetic model to predict the debromination of PBDEs in a landfill system, reporting that 100% of them were completely released in a century (as DecaBDE in the first years, and then as congeners with lower molecular weight). The temporal evolutions of production, consumption and atmospheric emission of PBDEs have been investigated on national (Morf et al., 2008; Sakai et al., 2006), European (Earnshaw et al., 2013) and global scale (Abbasi et al., 2019), founding out that landfill disposal can be the main source of PBDEs atmospheric release. The spread of BFRs in the environment is a concern even for engineered landfills because of different causes: diffuse air emissions cannot be fully collected; liners on the landfill surface can be damaged; collected leachate can be sent to waste water treatment plants (WWTP) not efficiently equipped for BFR removal (Lucas et al., 2018a). Furthermore, fugitive emissions can occur in the preliminary stage of WEEP unloading (Earnshaw et al. 2013). The LCA reported here considers a sanitary landfill with a lifetime of 100 years as an option for WEEP responsible management in Europe, and assumes for DecaBDE and TBBPA an emission factor of $1.0{\cdot}10^{-5}$ for the WEEP unloading, and an annual emission factor of 0.016 during landfill lifetime (Earnshaw et al., 2013; Abbasi et al., 2019). The degradation of DecaBDE in its congeners with low bromine content has been taken into account by assuming a half-life of 1 year for each PBDE, according to Danon-Schaffer and Mahecha-Botero (2010). This allowed to quantify the amount of PBDEs annually stocked in the landfill and those released into the atmosphere, as reported in detail in Annex A (Figures A.4-A.5). BFRs air emissions are collected by a biogas capture system, having an efficiency of 55% (Ardolino et al., 2017): this leads to estimate 75 g of higher brominated PBDEs, 337 g of lower brominated PBDEs and 2374 g of TBBPA emitted into the atmosphere for each tonne of WEEP disposed in a sanitary landfill

(Table B.3 of Annex B). DecaBDE and TBBPA released in water have been quantified by assuming an emission factor in leachate equal to 0.0004 (Jonkers et al., 2016; Morin et al., 2017) and considering a waste water treatment plant (WWTP) with a removal efficiency of 71% (Osako et al., 2004), leading to 0.13 g of DecaBDE and 0.73 of TBBPA emitted in water for each tonne of WEEP disposed in a sanitary landfill. Similarly, antimony emissions in water have been quantified assuming an emission factor of 0.96 (Jonkers et al., 2016) and an abatement efficiency of 72% (Osako et al., 2004). Other emissions in water have been estimated, based on average concentration values reported by Osako et al. (2004) for treated leachate. Other air emissions include fossil CO_2 and CH_4, which have been quantified assuming a carbon degradation rate of 3% (Simonson et al., 2000), an average biogas composition (55% of CH_4 and 45% of CO_2), and a biogas collection efficiency of 55%. Biogas is assumed to be sent to a combined heat and power (CHP) system, for energy recovery and minimisation of greenhouse gases: the related emissions have been calculated based on the study by Ardolino et al. (2017). The latter has been also used to quantify process-specific burdens, including soil occupation, landfill infrastructures and maintenance activities. The LCI table with the detail of direct burdens related to the sanitary landfilling of 1 tonne of WEEE mixed plastics is reported in Table B.3 of Annex B.

3.2. Substandard management options

A large fraction of WEEP is exported outside Europe, where it is generally managed by means of substandard options, without any measure to prevent health damages and/or environmental pollution (Forti et al., 2020). The main substandard treatments have been investigated in this study, in order to quantify the negative contribution deriving from WEEP exportation. According to Jonkers et al. (2016), the exported WEEP is assumed to be transported to Asian or African countries by means of a transoceanic ship for an average distance of 19,000 km, and for large part disposed by means of open dumping (33%) and uncontrolled burning (50%), and for the remaining part (17%) combusted in a furnace. Details about substandard treatments are reported in the following, while those related to the combustion have been already described in the thermal treatment Section 3.1.2.

3.2.1 Open dumping

It is the worst condition for landfilling: the waste is disposed on the soil without any protection; no compaction activities are used and no engineered systems to biogas and leachate capture are applied (Manfredi et al., 2009). Uncontrolled dumping is a not responsible option that unfortunately is still largely adopted in developing countries for municipal solid waste (Kaza et al., 2018) as well as for WEEP (Forti et al., 2020), causing large releases of hazardous substances into the environment (Lucas et al., 2018a). The open dumping for the exported WEEP has been modelled by starting from the same assumptions applied for sanitary landfilling, but taking into account the absence of biogas and leachate collection/treatment systems: the pollution control devices have been then assumed absent. The same emission factors and carbon degradation rate (3%) have been assumed. This leads to estimate 167 g of higher brominated PBDEs, 750 g of lower brominated PBDEs and 5300 g of TBBPA emitted into the air for each tonne of WEEP, together with 0.44 g of DecaBDE and 2.53 g of TBBPA emitted in the water, for each tonne of WEEP disposed in an open dumping site. The LCI table with the detail direct burdens related to the open dumping of 1 tonne of WEEE mixed plastics is reported in Table B.3 of Annex B.

3.2.2 Open burning

Uncontrolled burning of solid waste is a widespread phenomenon in dumpsites, treatment facilities and residential areas of developing countries (Kaza et al., 2018), where it has been recognised as the main source of outdoor particulate pollution, with related high rates of respiratory and neurological diseases, and premature deaths (Levis et al., 2017). Open burning of WEEP can lead to strongest consequences, since it generates not only emissions of dusts, greenhouses gases, metals and hydrocarbons as in the case of municipal solid waste, but also remarkable amounts of polibrominated/polichlorinated dioxins/furans (PBDD/F and PCDD/F) as well as DecaBDE (Gullett et al., 2007). Simonson et al. (2000) and Gullett et al. (2007) measured air emissions (such as of DecaBDE, TBBPA, dioxins and metals) deriving from WEEP burning, by carrying out specific firing tests, and providing useful datasets that are in good agreement with those reported by Abbasi et al. (2019). These data have been utilised in this study together with values related to dusts, NOx and SOx, reported by Jonkers et al. (2016) and Levis et al. (2017). Furthermore, greenhouse gases released into the atmosphere have been quantified considering that 69% of carbon content of WEEP is combusted (Simonson et al., 2000) and emitted as fossil CO_2, CO, and CH_4 (with percentage of 90%, 9% and 1%, respectively) according to Levis et al. (2017). The main air emissions quantified for the open burning of 1 tonne of WEEP are: PBDD/F (0.17 g/t_{WEEP}), PCDD/F (0.004 g/t_{WEEP}), particulates (22 kg/t_{WEEP}), PAH (5.7 kg/t_{WEEP}), fossil CO_2 ($1.7{\cdot}10^3$ kg/t_{WEEP}), and metals (408 g/t_{WEEP} and 1072 g/t_{WEEP} of Cu and Pb, respectively). The LCI table with the full list of air and water emissions related to the open burning of 1 tonne of WEEE mixed plastics is reported in Table B.4 of Annex B.

4. Environmental assessment of WEEE plastics management scenarios

4.1. Life Cycle impact Assessment

Fig. 4 reports the normalised results of LCIA of the Ideal scenario of WEEP management in Europe, with reference to the functional unit and in terms of the midpoint impact categories that have a key role in the environmental performances. Values related to the other impact categories have been neglected, based on the normalised results reported in Fig. C.1 of Annex C.

The total values for each of the main impact categories indicate that the ideal scheme of WEEP management would lead to important advantages for the environment. For the impact category of Non-Renewable Energy potential (NREP, which quantifies the consumptions of primary not renewable energy sources, such as oil, natural gas and coal) the value is equal to $-2.0{\cdot}10^{+5}$ person · year. The categories strictly related to human toxicity are Carcinogens potential (CP, that estimates the cumulative toxicological risk and potential impacts associated with a specified mass of a carcinogen, such as dioxins and PBDEs, emitted into the environment) and Respiratory INorganics potential (RINP, which quantifies the effects of the release into the atmosphere of inorganic compounds, such as dust, SOx, NOx, on the human respiratory system) (Jolliet et al., 2003; Ardolino et al., 2020). The values related to CP and RINP are $-7.6{\cdot}10^{+4}$ and $-5.4{\cdot}10^{+4}$ person · year, respectively. Fig. 4 indicates that the avoided productions of ABS, HIPS and PP have a major role in these good environmental performances. The combustion of heavy fraction and that of plastics residues also contribute positively to RINP and NREP thanks to recovered electricity, even though they affect negatively the Global Warming potential (GWP, which quantifies the effects of the emissions of greenhouse gases, such as CO_2, CH_4, N_2O, on the climate change), due to the high content of fossil carbon. Furthermore, the combus-

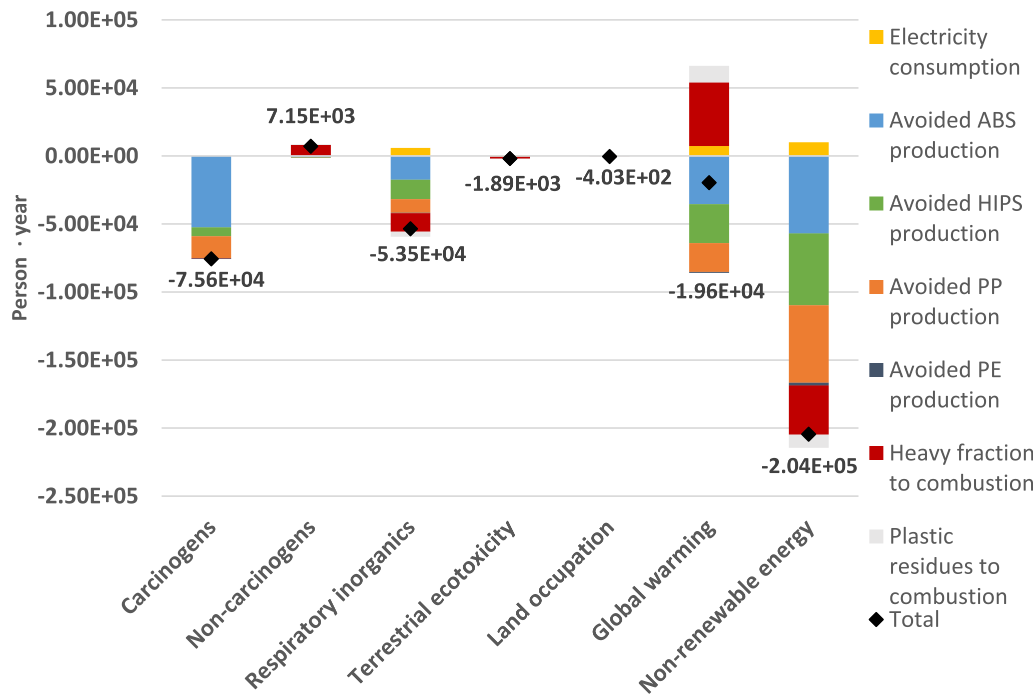

Fig. 4. Normalised results of impact assessment for the scenario Ideal with reference to the functional unit and with the detail of the contribution of each single stage of the life cycle. The shaded rhombus indicates the total value for each impact category. Results are normalised in "Person · year", i.e. the average impact in a specific category caused by a person during one year in Europe.

tion of heavy fraction shows a limited contribution in terms of Non-Carcinogens potential (NCP, which quantifies the cumulative toxicological risk associated with emissions into the environment of substances with non-carcinogen effects, such as dioxins and metals), deriving from the disposal of combustion residues in landfill. Figure C.2 in Annex C shows the same results of Fig. 4 but related to the scenarios Real_1 and Real_2.

An analytical comparison of the potential environmental impacts of all the considered scenarios can be made by using data in Table 5 and Figure C.3 of Annex C. It is evident the worst performance of scenario Real_1, characterised by the huge contribution of the "exportation" option. Most of related potential impacts are dramatic: those of Human Toxicity potential (carcinogens, non-carcinogens and respiratory inorganics), Terrestrial Ecotoxicity potential (TEP, which quantifies the effects of the release into the environment of substances having toxic effects on the ecosystems, such as heavy metals) and Global Warming Potential. The numerical data of these potential impacts ($3.5{\cdot}10^{+7}$ person · year for CP, $1.7{\cdot}10^{+8}$ person · year for NCP, $4.9{\cdot}10^{+5}$ person · year for RINP, $1.2{\cdot}10^{+5}$ person · year for TEP, and $9.3{\cdot}10^{+4}$ person · year for GWP) are so high that overwhelm all the others. The uncontrolled burning produces the main negative contribution to scenario Real_1, accounting for: 100% of CP and 99% of NCP, which are both related to the high release of PBDD in air; 94% of RINP, related to the air emissions of particulate; and 100% of TEP, related to the air emissions of copper. The contribution of open dumping is also dramatic ($3.6{\cdot}10^{+4}$ person · year for CP and $2.0{\cdot}10^{+6}$ person · year for NCP). The role of transportation of WEEP from Europe to Asian/African countries appears rather limited, with low contributions in terms of RINP ($3.8{\cdot}10^{+4}$ person · year) and GWP ($1.2{\cdot}10^{+4}$ person · year) and a high contribution in terms of NREP ($1.2{\cdot}10^{+4}$ person · year). These data depict a not-responsible WEEP management, when these plastics are exported without any reliable information about their fate. The quantified dramatic environmental impacts could be strongly reduced by making possible the WEEP exportation only to facilities able to operate a sufficiently sustainable treatment. Accordingly, the sensitivity analysis considered possible variations of the percentages of uncontrolled burning and combustion in furnace, highlighting how much it is important monitoring and conditioning the fate of the exported WEEP.

Data in Table 5 confirm the importance of mechanical recycling (when it can be applied) and the detrimental contribution related to sanitary landfilling. This latter option has been reported by Haarman et al. (2020) as still adopted in a not negligible percentage (11%), clearly contributing to the poor performances of the scenario Real_2, in particular for CP ($7.2{\cdot}10^{+3}$ person · year) and NCP ($3.9{\cdot}10^{+5}$ person · year).

A further support to the considerations reported above is given by Fig. C.4 of Annex C, which quantifies the LCIA of the treatment of 1 tonne of the considered WEEP by means of each of the responsible or substandard options described above. It is shown the huge contribution of open burning and, to a lesser extent, those of open dumping and sanitary landfilling. The dramatic contributions of open burning are mainly related to the emissions of PBDD, while those of open dumping and sanitary landfilling are mainly related to the emissions of PBDEs into the atmosphere. A comparison of the obtained LCIA results with those of similar analyses can be done with the only LCA study published on the same topic (Butturi et al., 2020). It utilised mainly (secondary) data from Ecoinvent and did not detail the BFR amount and composition taken as reference. Although these assumptions may underestimate the strong contribution of the BFR in terms of environmental impacts, obtained results confirm that substandard management options have significantly worsen performances for impact categories related to human toxicity.

4.2. Sensitivity analysis

The sensitivity analysis of an LCA can utilise two criteria: the definition of scenarios alternative to the base case reference scenario (as it has been made above), and the variation of some selected parameters in a reasonable range (Clavreul et al., 2012;

Table 5
Normalised results of impact assessment of the analysed scenarios of WEEP management, with reference to the functional unit and with reference to the main six impact categories. Results are normalised in "Person · year", i.e. the average impact in a specific category caused by a person during one year in Europe. For each specific impact category, **bold** data indicate the worst performance while underlined data indicate the best performance.

Midpoint impact category	Carcinogens Potential	Non-carcinogens Potential	Respiratory Inorganics Pot.	Terrestrial Ecotoxicity Pot.	Global Warming Potential	Non-Renewable Energy Pot.
IDEAL scenario						
Mech. Recycling	−7.53E+04	−1.22E+03	−3.62E + 04	**−1.02E+02**	−7.86E+04	−1.59E+05
Energy Recovery[1]	**−3.13E+02**	**8.37E+03**	**−1.74E+04**	−1.79E+03	**5.90E+04**	**−4.58E+04**
TOTAL	−7.56E+04	7.15E+03	−5.35E+04	−1.89E+03	−1.96E+04	−2.04E+05
REAL_1 scenario						
Mech. Recycling	−1.88E+04	−3.05E+02	−9.04E+03	−2.56E+01	−1.96E+04	−3.97E+04
Energy Recovery[1]	−7.82E+01	2.09E+03	−4.34E+03	−4.48E+02	1.47E+04	−1.15E+04
Incineration	3.11E+02	1.58E+03	4.95E+03	1.83E+02	2.75E+04	9.31E+02
Open Burning	**3.52E+07**	**1.67E+08**	**4.56E+05**	**1.17E+05**	**5.46E+04**	0
Open Dumping	3.58E+04	1.96E+06	2.06E+01	1.90E-10	3.82E+03	0
Transportation	3.80E+02	3.42E+02	3.80E+04	8.03E+02	1.18E+04	**1.19E+04**
TOTAL	3.53E+07	1.69E+08	4.86E+05	1.18E+05	9.28E+04	−3.83E+04
REAL_2 scenario						
Mech. Recycling	−3.31E+04	−5.37E+02	−1.59E+04	−4.51E+01	−3.46E+04	−6.98E+04
Energy Recovery[2]	−6.63E+02	8.11E+03	−2.58E+04	−2.68E+03	**8.53E+04**	−6.78E+04
Sanitary Landfilling	**7.22E+03**	**3.88E+05**	**−1.20E+01**	**−9.34E-01**	9.97E+02	**−1.41E+02**
TOTAL	−2.66E+04	3.95E+05	−4.17E+04	−2.72E+03	5.17E+04	−1.38E+05

[1]This value takes into account the residues of WEEP sorting and re-manufacturing sent to energy recovery in combustion furnaces.
[2]This value takes into account the WEEP directly sent to energy recovery together with the residues of WEEP sorting and re-manufacturing.

Astrup et al., 2015). The latter criterion is adopted in the following, considering the parameters and related variation ranges shown in Table 6: 1) the percentage of open burning in the disposal options of exported WEEP; 2) the material recovery rates obtained by mechanical recycling of ABS and PS/HIPS obtained by WEEP sorting; 3) the PBDEs and TBBPA emissions from open dumping or sanitary landfilling; 4) the PBDD/F emission from open burning; 5) the DecaBDE content in the BFR compounds of WEEP.

The results are reported in Fig. 5 in terms of variation factor, that is the ratio between the result for the changed parameter in the sensitivity analysis and that estimated for the base case (Ardolino et al. 2018). The same results are also shown in Figure D.1 of Annex D as normalised values. The environmental performances of scenario Real_1, in terms of CP, NCP, RINP and TEP, can be improved up to 50%, by decreasing the amount of WEEP burned under uncontrolled conditions. Moreover, different assumptions in the values of PBDD/F emissions from open burning of WEEP can lead to variations of both CP and NCP of about +/- 50%. This confirms that the sustainability of the European management of WEEP is greatly affected by the quantity of exported waste, and further highlights the necessity of reducing the percentage of WEEP treated by this option. Alternatively, the quality of infrastructures that receive WEEP has to be monitored (and improved) to avoid the application of substandard treatments, which cause large environmental impacts. Environmental performances of scenario Real_2 are mainly affected by the BFRs composition and their annual emission factors, since both imply a decrease/increase of PBDEs and TBBPA emitted into the atmosphere, and then a variation in terms of CP and NCP. The different values assumed for DecaBDE content in BFRs lead to a variation of +/- 54% in terms NCP. The strongest variation relates to a higher BFRs emission factor into atmosphere from landfill or dumping, which implies a huge worsening in terms of NCP up to +372%. This supports the opportunity of more studies on this aspect but, above all, the strong necessity to avoid as much as possible these not sustainable management options. The performances of the scenario Ideal are affected by different amounts of ABS-HIPS sent to plastic re-manufacturing, which cause a variation in terms of GWP (+/-50%) as a consequence of the changed avoided burdens and amounts of heavy fraction sent to energy recovery. The Ideal management scenario is the only scheme that shows environmental impacts relatively limited or close to zero.

Table 6
Parameters considered in the sensitivity analysis, with the indication of the assumed ranges of variation.

Changed parameter	Involved scenario	Base case value	Sensitivity values	Sensitivity scenario ID
% of uncontrolled burning	Real_1	50%	25% (with a corresponding increase of incineration in a RK unit)	**Real_1_25%burn**
% of material recovery rates of ABS and HIPS	All	68% (ABS)	10% lower	**Ideal/Real_LowABS-HIPS**
		70% (HIPS)	10% higher	**Ideal/Real_HighABS-HIPS**
Annual emission factors to air from WEEP disposed by landfilling/dumping	Real scenarios	0.016	0.001	**Real_LowEmissions**
			0.1	**Real_HighEmissions**
PBDD/F emissions from uncontrolled burning	Real_1	0.17 g/t_{WEEP}	50% lower	**Real_1_LowPBDD**
			50% higher	**Real_1_HighPBDD**
DecaBDE content in BFRs	Real scenarios	13%	50% lower (with a corresponding increase of TBBPA)	**Real_LowDecaBDE**
			50% higher (with a corresponding decrease of TBBPA)	**Real_HighDecaBDE**

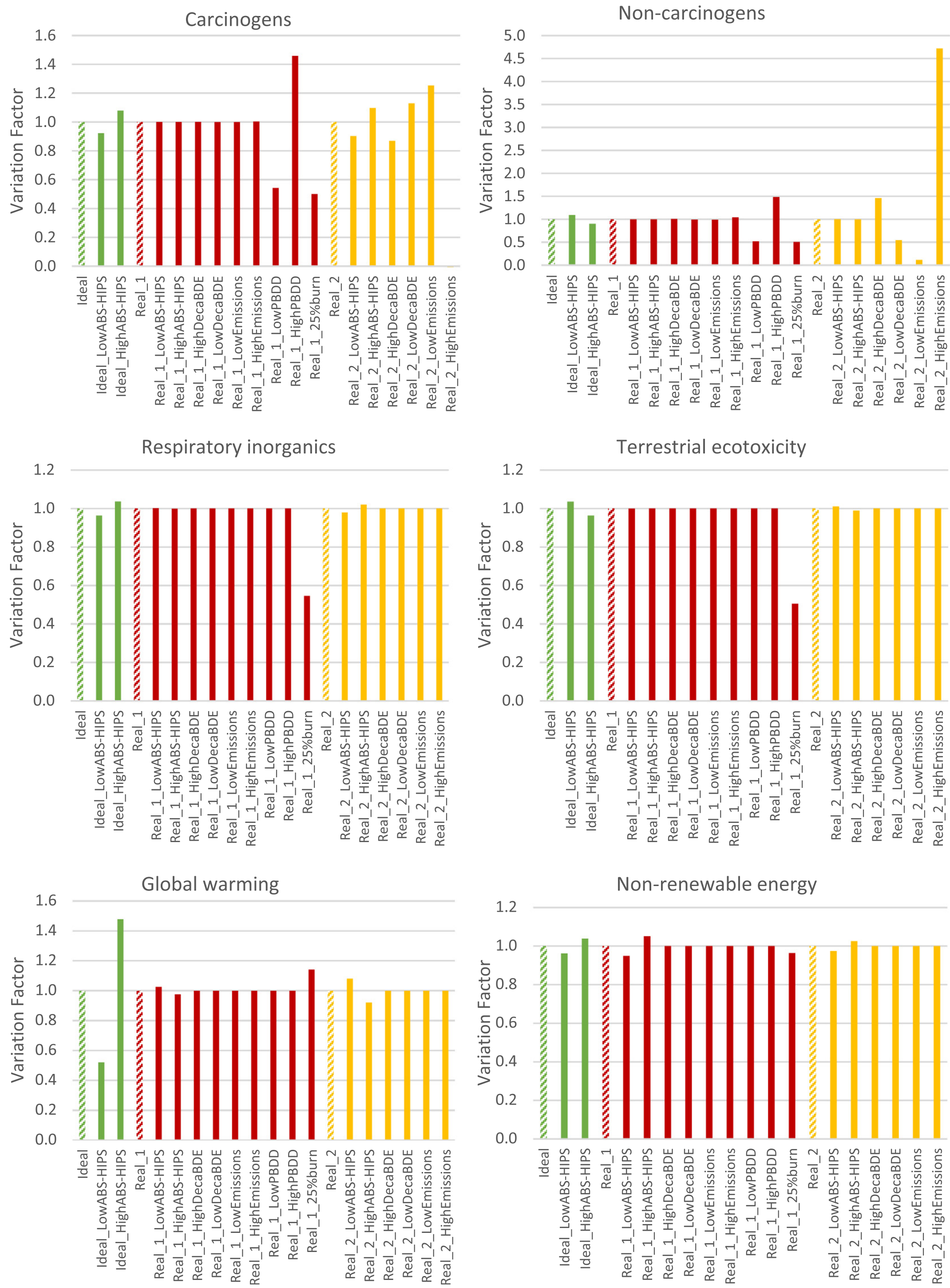

Fig. 5. Results of the sensitivity analysis in terms of variation factor, VF. VF = 1 indicates no variation; some variations occur when VF is < 1 or VF > 1; and a negative value of VF changes the potential impact from positive to negative or vice-versa.

5. Conclusions

An attributional Life Cycle Assessment quantifies the environmental performances of the main management processes for WEEE plastics, i.e. those in compliance with the European Directives ("responsible management options") and the so-called substandard treatments (which generally characterise the fate of WEEP exported outside Europe).

The study estimates the resource recovery rates for each WEEE category, indicating those with lowest values: Screen&Monitors (44%), Small Equipment + Small IT and Telecommunication Equipment (44%), and Lamps (20%). Data have been correlated with the specific content of the main target polymers. Very high levels of recovery rates (generally close to 95% and always higher than 88%) have been estimated for PE and PP in each WEEE category. On the contrary, lower levels (54% and 56% for Screens&Monitors, and 64% and 61% for Small Equipment + Small IT and Telecommunication Equipment) have been quantified for ABS and PS/HIPS, due to the higher content of brominated compounds, which strongly limit the conventional mechanical recycling.

The life cycle impact assessment highlights the huge negative contributions of the waste exportation and its associated not responsible treatments. The Ideal scenario of complete compliance with European Directives is the only one with an almost negligible effect on the environment, even though it is far away from the reality of the current European management of WEEP.

The analysed real scenarios have strongly negative effects on the environment. These effects become dramatic for the scenario that includes the exportation of a fraction of WEEP outside Europe, due to the very poor performances of uncontrolled burning and open dumping. These substandard options produce huge impacts, mainly in terms of carcinogens ($3.5{\cdot}10^{+7}$ and $3.6{\cdot}10^{+4}$ person·year, respectively) and non-carcinogens ($1.7{\cdot}10^{+8}$ and $2.0{\cdot}10^{+6}$ person·year) potentials.

The overall set of LCA results, including those of a related sensitivity analysis, clearly indicates the necessity of strong reductions of WEEP exportation. This could occur by providing economic incentives for European recyclers and supporting an increase of the capacity of sustainable WEEP management in Europe. In the meantime, it is crucial that the European Union operates to improving the quality of infrastructures that receive exported WEEP and making possible only the exportation to facilities able to perform sufficiently sustainable treatments.

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.2021.02.040.

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