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Extended producer responsibility: How to unlock the environmental and economic potential of plastic packaging waste?

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ABSTRACT

Several are the challenges related to plastic waste, spanning from littering, high collection costs, and low recycling rates. Extended producer responsibility (EPR) is considered a key initiative to tackle some of these issues. To evaluate EPR role and effectiveness, 40 management scenarios focused on plastic packaging waste generated by Italian households were investigated, and their environmental performance (via a consequential life cycle assessment) and the economic sustainability of their waste value chain (via a cost-benefit analysis for each stakeholder) were compared to the recycling targets. Overall, packaging waste management represented an environmental burden. Yet, environmental benefits can be achieved by maximizing the collection rate, while minimizing the impurities collected with the source-segregated plastic and the processing losses in the recycling chain. Furthermore, the cost-benefit analysis showed that the recyclers are the weakest link in the value chain, and recycling of soft plastic and mixed polyolefin is generally not profitable. This increases the risk of exporting low-quality materials outside Europe, where their fate is uncertain. Finally, the results demonstrate that improving plastic packaging recyclability and strengthening the market for secondary plastic is critical for reaching the European recycling targets of 55% in 2030.

1. Introduction

While representing a major technological breakthrough in itself, plastic is associated with a wide range of challenges in its waste phase, from marine pollution to limited recycling (Bio Intelligence Service, 2011; EC, 2018a). Following the substantial attention plastic waste has received in recent years, and promises of considerable economical gains voiced by a wide variety of stakeholders (Ellen MacArthur Foundation, 2017, 2016), the EU recently defined a new circular economy strategy (EC, 2018a) by providing guidelines for its management, with reuse, repair, and recycling as the preferred options. Despite this heightened attention in recent years, actual recycling and off-setting of virgin plastic production is extremely limited; for instance, in the EU, less than 30% of plastic was collected for recycling in 2017 (EC, 2018a). With a new EU recycling target for plastic packaging of 50% by 2025 and 55% by 2030 (EC, 2018b), significant improvements are needed in the coming years.

Extended producer responsibility (EPR) and take-back/deposit systems are expected to play a major role in this regard by making producers responsible for the end-of-life phase (OECD, 2001), thereby also providing incentives for more recycling-friendly plastic packaging (OECD, 2018). Although different EPR systems have been implemented globally (Monier et al., 2014), EPR often includes an environmental fee that producers and importers pay to have their products managed through a producer responsibility organization (PRO), namely, a "collective entity set up by producers or through legislation, which becomes responsible for meeting the recovery and recycling obligations of the individual producers" (Monier et al., 2014). However, little consensus exists about how best to distribute costs and responsibilities between the involved stakeholders (Monier et al., 2014), and existing EPR and take-back systems have yet to demonstrate genuine improvements in product design and plastic recyclability (Watkins et al., 2017). The question is, can EPR and take-back systems offer the incentives needed to attain the EU's high recycling targets?

Plastic recycling is characterized by several challenges: low material

Abbreviations: CAPEX, capital investment expenditure; EU, European Union; EPR, Extended producer responsibility; FU, Functional unit; GWP, Global warming potential; HDPE, High-density polyethylene; LCA, Life cycle assessment; LDPE, Low-density polyethylene; MPO, Mixed polyolefin; MRF, Material recovery facility; OPEX, operating expense; PMFP, Particular matter formation potential; PET, polyethylene terephthalate; PP, Polypropylene; PRO, Producer responsibility organization; wPE, Weighted person equivalent

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densities involving high collection costs (WRAP, 2007), a variety of different polymers (e.g. PET, HDPE, PP) with distinctive chemical structures, properties (e.g. thermos-plasticity, bio-degradability, density) and additives (Villanueva and Eder, 2014), and a product design based on product functionality rather than considerations regarding waste sorting and recycling (NewInnoNet, 2016). Most often, post-consumer plastic is often contaminated with relatively high levels of impurities, potentially affecting downstream resource quality and recycling processes (Eriksen et al., 2018). Furthermore, the economic viability of plastic recycling is often limited, due to i) a lack of stable market demands (EC, 2018a) (e.g. due to concerns about material quality and/or stability of supply), ii) competition with low-cost virgin alternatives and iii) complex value chains involving a variety of stakeholders within the recycling system (NewInnoNet, 2016; Villanueva and Eder, 2014). A consistent evaluation of these recycling systems is needed, to identify the most appropriate system configurations and to ensure the best possible level of sustainability.

Several studies have evaluated the sustainability of plastic waste management. From an environmental perspective, life cycle assessment (LCA) has been applied to plastic waste, comparing recycling versus disposal (Arena et al., 2003; Chilton et al., 2010; Huysman et al., 2015), different collection rates (Rigamonti et al., 2014) and different recycling technologies (Shen et al., 2010; Shonfield, 2008). However, most of the studies included only one or few environmental impact categories (Hestin et al., 2015) - meaning that environmental trade-offs may not be identified - or are interested in quantifying environmental burdens of the current plastic waste management (Ferreira et al., 2017; Haupt et al., 2018) and not the effects of its upgrading. A detailed mass balance and LCA of Austrian plastic waste was performed by Van Eygen et al., (2018), but was not followed by an economic assessment. From an economic perspective, the majority of studies only analyzed collection costs (Greco et al., 2015; Groot et al., 2014; Jaeger and Rogge, 2014), sorting costs (Cimpan et al., 2016) or provided an environmental life cycle costing based on rather simplified modeling approaches and life cycle inventory (Feil et al., 2016; Pressley et al., 2015; van Velzen et al., 2013). Even the more complete environmental life cycle costing (e.g. Faraca et al., 2019a) did not quantify the effects of specific initiatives for individual stakeholders, nor identified bottlenecks in the value chain. When the focus was placed on analyzing the efficiency and the application of the extended producer responsibility of packaging material (e.g. Cunha et al., 2014; Ferreira da Cruz et al., 2014; Rigamonti et al., 2015a), it was not possible to extrapolate any conclusion for plastic since all material fractions were aggregated. These studies thereby fail to include a stakeholder perspective when evaluating circular economy systems where cooperation is of particular importance (Schaubroeck et al., 2019). As such, no studies in the literature to date have provided a consistent approach for evaluating these multi-stakeholder recycling systems that are so essential for promoting a circular economy.

The present study differed from the already published work for several reasons: a detailed data collection to combine national and European data was performed; a detailed mass balance to track each polymer and the impurities along all the value chain was modeled; several waste management alternatives and modeling choices were compared; midpoint and endpoint environmental impact categories were quantified and combined in a single indicator; and the economic sustainability of the system was evaluated from the perspective of different stakeholders.

The overall aim of this study is to provide a consistent framework for evaluating the environmental and economic impacts of selected packaging plastic waste management solutions, including EPR. Based on an Italian case study, this is addressed through the following specific objectives: i) establish an assessment framework that covers the regulatory, environmental and economic spheres relative to individual stakeholders in the recycling value chain; ii) apply this framework to a selection of alternative options for managing plastic packaging waste generated by Italian households; iii) identify limitations associated with current plastic waste management and existing EPR systems; and (iv) provide recommendations for improving the sustainability of plastic packaging waste management while ensuring appropriate incentives for individual stakeholders. Household plastic packaging waste was selected due to its heterogeneity and considerable current attention, shown by the high recycling targets set by the EU on this type of waste (EUROPEN, 2016). Moreover, Italy was selected based on its long history of EPR implementation (Watkins et al., 2017), a source-segregated plastic collection rate in line with the European average (EC, 2019; PlasticsEurope, 2019), and a well-established plastic recycling industry that absorbed 13.9% of the total European converters demand in 2018 (PlasticsEurope, 2019).

2. Materials and methods

2.1. Case study: management of household plastic packaging in Italy

The case study covers plastic packaging waste management-related activities from the point of waste generation (collection from bins) to marketing the processed flakes/granules ready to be used in new products (Fig. 1). The stakeholders directly involved in the value chain are: i) the municipality, ii) the plastic packaging producer responsibility organization (PRO), iii) the material recovery facility (MRF), and iv) the recyclers of clear PET, light blue PET, mixed-color PET, HDPE, PP, films and mixed polyolefins. In Italy, producers and importers of plastic packaging pay an environmental fee to the PRO per ton of plastic packaging produced/imported. The environmental fee was 188 EUR/t in 2017 (Corepla, 2018a). Citizens sort generated plastic packaging waste into a separate bin, together with some unwanted impurities (e.g. non-plastic materials, labels, residuals in uncleaned containers). Municipalities are responsible for organizing and implementing the collection of source-segregated plastic waste, either as a single-stream (only plastic) or co-mingled with metals (Corepla, 2018a), and they receive a financial compensation from the PRO that decreases with increasing content of impurities (303.8 EUR/t plastic packaging received ANCI and Corepla, 2014). All non-sorted packaging plastic ends up in mixed waste, which is also managed by the municipality. The PRO is responsible for transferring part of the environmental fee to the municipality, setting purity targets for the sorted material, and sampling/analyzing the waste. For simplicity, all plastic was assumed to be collected as a single-stream (e.g. one bin for plastic), corresponding to 75% of plastic collection schemes in Italy (Corepla, 2018a). After collection, the material is transported to the MRFs, represented by private companies contracted by the PRO to sort the materials into seven primary fractions (bales), namely, clear PET bottles, light blue PET bottles, mixed-color PET bottles, HDPE, PP, soft packaging (film), and mixed polyolefin (MPO). The MRF is characterized by sorting efficiencies (i.e. the kg of targeted polymer that the plant can correctly sort in the corresponding bale) and purity of the bales (i.e. kg of un-wanted contaminations in the bale) and its residues are sent to disposal. Being the legal owner of the sorted plastic, the PRO sells the bales through open auctions, and recyclers buy these bales and sort, shred, wash, extrude, and pelletize the materials for the production of plastic flakes (in the case of non-food-grade PET) or granules (in all other cases). The technical yields quantify the efficiency of the recyclers, which is the kgs of flakes/granules per kg of entering bale. The flakes/granules are subsequently sold to converters for manufacturing the final plastic products.

2.2. Management alternatives

Five management alternatives for managing plastic packaging waste were evaluated: A1 and A2 represent two baseline systems, each with different collection systems, while alternatives A3–A5 represent three options for future improvements. Alternatives A3-A5 are based on a

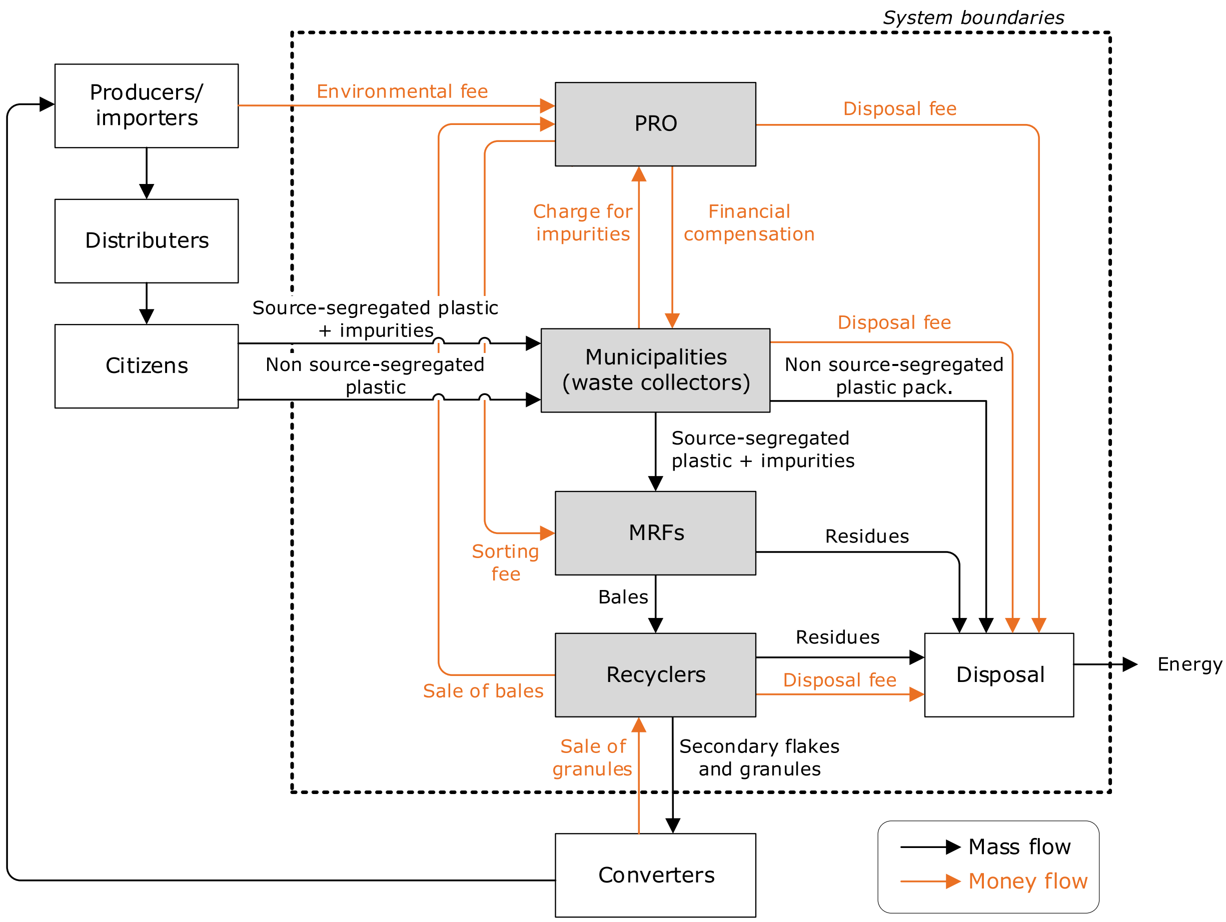

Fig. 1. System boundaries (dotted line), mass flows (black solid lines), and monetary flows (orange solid lines) relevant for the Italian packaging plastic waste management. Dark gray boxes represent the stakeholders included in the study: municipality, producer responsibility organization (PRO), material recovery facilities (MRFs), and recyclers. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

door-to-door collection (namely A1), as this system is more frequently adopted when implementing new separate collection systems in Italy (Bain and Company, 2013).

The five management alternatives were:

- A1) Door-to-door collection: The waste management system represented the Italian waste management in 2017 assuming that all the source-segregated plastic was collected with a door-to-door collection. The collection efficiency was 62% ± 9% (ISPRA, 2017) and the share of impurities in the bin was 10% ± 2.5% of the collected materials. The source-segregated plastic was sent directly to the MRF characterized by average sorting efficiencies and both the bales and the secondary material were sold in the market with average market prices in 2017 (see Appendix A).
- A2) Street collection: Similar to alternative A1 but involving street collection instead of door-to-door. Collection efficiency was assumed as being identical to A1, albeit with a higher share of impurities of 30% ± 5% reflecting available data (see Appendix A). Contrary to A1, municipalities involved a pre-sorting plant before the MRF to reduce the content of impurities and thereby received larger financial compensation from the PRO (Guerrini et al., 2016). The other parts of the system were identical to A1.
- A3) Improved recyclability for PET bottles: All PET bottles sold on the market were assumed to be clear unpigmented, and the type of additives, labels, glues, etc. was compatible with plastic recycling systems (APR, 2020; EPBP, 2020). This was reflected by higher PET sorting efficiencies in the MRF, higher technical yields in the PET recycling plants, and higher market values for PET bales and PET flakes relative to A1.
- A4) Improved recyclability for PET, HDPE, PP, and films: All plastic packaging products sold on the market were designed for improved recyclability, and the choice of additives, labels, glues, etc. was compatible with plastic recycling systems (APR, 2020; EPBP, 2020). For all polymers, higher sorting efficiencies, higher technical yields in the recycling plants, and higher market values of bales and flakes/granules relative to A1 were assumed.
- A5) Deposit system: A deposit system for PET bottles was introduced as an alternative to the collection in A1, similar to the well-established Danish deposit system which achieves a return rate of 91% (Dansk Retursystem, 2019). The deposit system was assumed to be operated directly by the PRO, without involving other stakeholders. The introduction of the deposit system caused a higher cost of the door-to-door collection of the remaining plastic and a higher market price for the PET bales.

2.3. Modeling approach and scope

The assessment framework was established in accordance with LCA standards (ISO, 2006a,b), thereby keeping goal and scope, life cycle inventory, and system boundaries consistent across alternatives and assessments. The functional unit (FU) was “the management of 1000 kg of household plastic packaging waste, having a fractional composition of 9% clear PET bottles, 13% light blue PET bottles, 9% mixed-color PET bottles, 2% opaque and sleeve-labeled PET bottles, 3% PET trays, 10% HDPE, 7% PP, 28% soft packaging, and 19% other polymers” based on Italian data (Conte, 2016). Impurities found alongside the source-segregated plastic (43% non-plastic packaging, 4% metals, 21% other combustibles, 32% fines Albetti et al., 2012) were assessed by i) including all waste management activities as an integral part of the alternatives, and ii) subtracting the counter-factual management (i.e. waste management of the mixed waste if the impurities were correctly thrown in the mixed waste bin). The geographical scope of the case study was Italy, and the temporal scope was 2017–2030. The modeling was carried out with the software EASETECH (Clavreul et al., 2014), while Excel and STAN (Cencic, 2016), a software that performs uncertainty propagation and data reconciliation, were used for material flow analysis to ensure balanced and consistent material flows throughout the system.

Based on the definition of goal and scope, relevant stakeholders within the value chain were identified (Fig. 1), material and monetary flows were quantified (Fig. 1), and alternatives were modeled for comparison (Section 2.2). The regulatory, environmental, and economic spheres were evaluated by i) recycling rate calculation, ii) life cycle assessment and iii) multi-stakeholder economic cost-benefit analysis, respectively.

2.3.1. Recycling rate

The regulatory sphere was represented by EU recycling targets, the key performance indicator for member states (EC, 2018c,b, 2008). European legislation included the concept of recycling rates at the beginning of the ‘90 s. However, its definition has been changing with time. The Directive 2008/98/EC (EC, 2008) included the objective of “preparing for re-use and recycling” 50% of the generated paper, metal, plastic, and glass waste from households by 2020. The Directive allowed countries to report directly the collected material or the outputs from MRFs if there were no significant losses (EC, 2011). With the new Directive 2018/851 (EC, 2018c) and 2018/852 (EC, 2018b), the European Commission amended the Waste directive 2008/98/EC and the packaging and packaging waste Directive 94/62/EC and it stated that “the calculation of the recycling targets should be based on the weight of municipal waste which enters recycling” (recycling operations defined as the recycling of waste into products, materials or substances) excluding all the losses of materials due to sorting.

The recycling rates were determined at several points in the value chain to highlight the difference in the results: i) Weight of the source-segregated plastic, including both plastic packaging and impurities ($Rec_{coll} = kg_{collected}/1000\ kg_{packaging\ plastic}$); ii) mass sold as bales after sorting ($Rec_{bale} = kg_{bales}/\ 1000\ kg_{packaging\ plastic}$); iii) quantity of targeted packaging plastic in the bales without impurities ($Rec_{bale\ pack} = kg_{targeted\ plastic\ packaging\ in\ the\ bales}/1000$ kg packaging plastic); and iv) kg of flakes/granules produced by the recyclers ($Rec_{granule} = kg_{granules}/1000\ kg_{packaging\ plastic}$).

The recycling rate required by the actual EU legislation (Rec_{EU}) should include the sorting steps of the recyclers but not the physical losses due to shredding and extrusion, meaning that it should be higher than Rec_{Bales} and lower than $Rec_{Granules}$. However, the Rec_{EU} was calculated as a simple average between Rec_{Bales} and $Rec_{Granules}$, since plastic recyclers have been traditionally reporting only the overall efficiency (kg of secondary material sold in the market per kg of material entering the plant) and not the efficiencies of the single reprocessing steps.

2.3.2. Life cycle assessment

The LCA adopted a consequential approach to model the potential effects of future decisions, reflected by the individual management alternatives (Ekvall et al., 2016). For this reason, system expansion was applied to address multi-functional processes. For example, the produced flakes and granules substituted virgin material that would otherwise have been produced applying a value-corrected substitution factor as it is commonly done in consequential LCAs (Van Eygen et al., 2018a), which is defined as the market price ratio between secondary and primary granules. In all alternatives, PET HDPE, and PP substituted virgin fossil polymers; in alternatives A1, A2, A3, and A5, granules produced from soft plastic and mixed polyolefin MPO bales were assumed to replace outdoor furniture otherwise made of wood and cast iron (Van Eygen et al., 2018a); and in alternative A4, the quality of granules from soft plastic was assumed to be high enough to replace virgin PP and LDPE. The energy produced by incineration substituted marginal energy production, calculated as the growing technologies (Ekvall and Weidema, 2004), based on the Italian data provided in the GECO2018 report (Keramidas et al., 2018): Marginal electricity (6% biomass, 18% hydro, 1% natural gas, 28% solar, 28% wind, 19% CHP) and space heating (3% electrical boilers, 49% biomass, 21% hydrogen, and 27% central heating). Energy from the biomass was assumed responsible for also 0.32 fossil CO_2/kg_{wood}, due to the indirect land-use change (Faraca et al., 2019b). Industrial heating consisted of 50% coal, 48% biomass, and 2% hydrogen. Each process included capital goods, transport, ancillary materials, energy consumption, and direct and indirect emissions (Appendix A).

ReCiPe 2016 v1.1 (Huijbregts et al., 2017) was applied as an LCA collection to express the results as a single indicator: end-point impact categories for the three areas of protection (human health, ecosystems, and resources) were quantified, followed by normalization, weighting, and aggregation (Laurent et al., 2019; PRé Consultants, 2001).

2.3.3. Economic analysis

The economic analysis involved a multi-stakeholder approach for calculating a separate cost-benefit analysis for each stakeholder (Soltani et al., 2016) to identify potential imbalances and/or economic losses (Freeman, 1984). Table 1 provides an overview of financial costs and revenues for the selected stakeholders. The cost-benefit analysis of the MRF and the seven recycling plants were calculated similarly to Cimpan et al. (2016):

Table 1
Financial burdens and benefits of the included stakeholders. MRF: material recovery facility; CAPEX: capital expenditure; OPEX: operating expense.

Stakeholder	Responsibilities	Outputs	Costs	Benefits
Municipality	Collection	Source-segregated plastic	Collection costs (door-to-door or street) Administration costs Pre-sorting (A2)	Financial compensation from the PRO
PRO	Ensuring recycling rates	N/A	Financial compensation to municipalities Service fee to MRFs Quality analysis of the MRF's bales OPEX Disposal of MRF's residues	Environmental fee from companies Sale of bales to recyclers
MRF	Sorting	Bales	CAPEX OPEX	Service fee from the PRO
Recyclers	Sorting, shredding, compacting, washing, melting, etc.	Flakes or granules	CAPEX OPEX Purchase of the bales from the PRO Disposal of residues	Sale of flakes/granules

- The capital expenditure (CAPEX) included the annualized building and equipment costs together with the project and installation costs. Annualized CAPEX was calculated as the total CAPEX multiplied by the capital recovery factor that is equal to $\frac{i*(1+i)^n}{[(1+i)^n-1]}$, where i is the interest rate and n the lifetime of the plants.
- The operating expense (OPEX) included the maintenance and the insurance of the building and the equipment, the energy (electricity and heat), the ancillary material consumption (e.g. diesel), and the personnel salaries.

All economic data were normalized for Italy for 2017, using purchasing power parities (World Bank, 2019) and inflation rates (fxtop.com/, 2019).

2.4. Uncertainty analysis

Due to the inherent variations in most data, the around 300 parameters describing the model (e.g. transfer coefficients, emissions, consumptions, costs) were defined as probability distributions (normal, uniform, or triangular) rather than as fixed values. The propagation of the uncertainty in the mass balance was performed in Excel and STAN, allowing to have consistent transfer coefficients for all the fractions from the waste generation until the production of secondary material (i.e. sorting efficiencies, technical yields and cross-contaminations in each bale).

The uncertainty in the data was propagated with Monte Carlo analysis (1000 samples), and the parameters contributing the most to the uncertainty of the results were identified by global sensitivity analysis (Bisinella et al., 2016). In Section 3, all results are presented as the average of the Monte Carlo iterations plus/minus the associated standard deviation. Data distributions were identified based on an extensive literature review addressing all foreground data (see Appendix A for references), including around 45 reference sources for material balances, 50 references for the environmental assessment, and 25 references for the economic assessment. Background data were retrieved from the ecoinvent v3.5 “consequential” database (Wernet et al., 2016).

2.5. Life cycle inventory

While this paragraph summarizes the life cycle inventory of the study, a detailed description of all data, references, and assumptions is available in Annex A. Residual waste collection, door-to-door and street source-segregated plastic collection and deposit systems are modeled using literature data for diesel consumption (Ferreira et al., 2017; Larsen et al., 2009; Nilsson and Christensen, 2011; Rigamonti et al., 2014; Van Eygen et al., 2018a) and Italian collection costs (D'Onza et al., 2016; ISPRA, 2017; Utilitalia and Bain and Company, 2018). The mass balance of the MRF was modeled by combining information on different sorting efficiencies for each polymer found in the literature (Arena et al., 2003; Axion Consulting, 2009; Cimpan et al., 2016; Conte, 2016; Giugliano et al., 2011; Haig et al., 2015; Perugini et al., 2005; Pressley et al., 2015; Rigamonti et al., 2014; Shen et al., 2010; Shonfield, 2008; Turner et al., 2015; Van Eygen et al., 2018b) and the quality requirements of the Italian plastic PRO for plastic bale (Corepla, 2018b). Data for the ancillary materials and energy consumption for the pre-sorting plant, the MRF, the simplified sorting (used in the deposit system) and recycling plants for each of the polymers were based on literature (Arianna Ambiente, n.d.; Axion Consulting, 2009; Chilton et al., 2010; Cimpan et al., 2016; Conte, 2016; Franklin Associates, 2011; Haupt et al., 2018; McDougall et al., 2001; Perugini et al., 2005; Pressley et al., 2015; Rigamonti et al., 2014; Torregrossa et al., 2005; Turner et al., 2015; Van Eygen et al., 2018b). The capital and operational expenditures of sorting and recycling plants were calculated based on operating plants in Europe (Axion Consulting, 2009; Cimpan et al., 2016; Conte, 2016; Faraca et al., 2019a; Martinez-Sanchez et al., 2016; Pressley et al., 2015; Pringle and Barker, 2004). The costs of incineration and landfilling were specific of the Italian context and included incineration and landfill taxes (EEA, 2013; Moretto and Favot, 2017), while the ancillary material consumption and emissions to the environment were based on French, Danish and Italian plants (Beylot and Villeneuve, 2013; Møller et al., 2013; Turconi et al., 2011). Specific Italian data were used for the costs and revenues of the PRO (ANCI and Corepla, 2014; Corepla, 2018a, 2015), cost of personnel (Federambiente, 2012), electricity (Eurostat, 2019), heat (Eurostat, 2018), diesel (Statista, 2018). Prices of plastic bales, secondary and primary material were built on market data (Camera di commercio di Milano, n.d.; Corepla, 2018a, 2013, 2012; PIE, n.d.; Plasticker.de, n.d.; Popovic, 2017; WRAP, 2017).

2.6. Scenario analysis

Seven scenarios were modeled for each plastic packaging waste management alternative (A1-A5). Two scenarios (a-b) evaluated key assumptions regarding waste management:

a) Production of food-grade PET granules was maximized rather than PET flakes based on Van Eygen et al. (2018): 54% of the entering material is sorted out as higher quality PET and can be recycled in food-grade granules, while the remaining 46% has lower quality and is used for flakes production. The energy and ancillary material consumption were modified accordingly. The food-grade PET granules substituted food-grade virgin PET, instead of amorphous PET;
b) Collection rate was increased to 83% (i.e. the highest collection rate reported in Italy (ISPRA, 2017));
c) Four additional scenarios (c-g) evaluated key modeling choices:
d) The composition of the generated packaging plastic waste was based on the manual sorting from Conte (2016) that included higher contents of PET (from 33% to 50%) and HDPE (from 10% to 17%) and lower contents of soft packaging (from 28% to 13%) and other polymers (from 19% to 9%);
e) Soft packaging and mixed polyolefin were assumed to substitute thermal plaster instead of wood and cast iron;
f) Virgin plastic was modeled following the inventory described in Franklin Associates (2011) instead of the one provided by ecoinvent v3.5 (Wernet et al., 2016). The energy utilized was updated with the marginal electricity and heat used for all the other processes;
g) Average electricity for 2017 based on the GECO2018 report (Keramidas et al., 2018) was used instead of marginal electricity;
h) Average space and industrial heat for 2017 based on the GECO2018 report (Keramidas et al., 2018) was assumed instead of marginal space and industrial heat.

3. Results and discussion

3.1. Recycling rate

The recycling rates reported in Table 2 were calculated based on the mass balances modeled for the different alternatives (see Fig. 2 for A1 and Appendix A for the other alternatives). By comparing Rec_{coll} and Rec_{bale}, it is evident that only about 60% of the collected materials in A1 and 40% in A2 were sold as bales, reaching 70% in A3 and A4 with higher sorting efficiencies. The amount and quality of the source-segregated and collected plastic waste may reflect the involvement of the population, but in itself, it provides little information about the environmental savings due to recycling because it does not take into account the processing losses. Rec_{coll} in A2 was relatively high (89%) because of high levels of impurities in the collected waste, and consequently, Rec_{bale} was only 37%. Overall, none of the individual alternatives (different collection rates in A1 and A2, recyclability improvement in A3 and A4, or the deposit system in A5) reached the 50% EU

Table 2
Recycling rates for the five alternatives and the scenarios a, b, and c, expressed as average ± standard deviation. Rec_{EU} is a simple average between Rec_{bale} and $Rec_{granule}$. A4-scenario b (improved recyclability and higher collection rate) is the only case where the recycling rate was higher than the 55% target for 2030 (EC, 2018b).

Indicator	Alternatives A1	A2	A3	A4	A5
Rec_{coll}	70% ± 9%	88% ± 11%	70% ± 9%	70% ± 9%	77% ± 6%
Rec_{bale}	43% ± 7%	37% ± 10%	44% ± 7%	50% ± 7%	48% ± 4%
$Rec_{bale\ pack}$	40% ± 6%	34% ± 10%	41% ± 6%	47% ± 6%	46% ± 4%
$Rec_{granule}$	34% ± 5%	29% ± 8%	35% ± 5%	43% ± 6%	40% ± 3%
Rec_{EU}	38% ± 4%	33% ± 7%	40% ± 4%	46% ± 5%	44% ± 3%
Rec_{EU} – scenario a	38% ± 4%	33% ± 4%	39% ± 4%	46% ± 5%	44% ± 3%
Rec_{EU} – scenario b	51% ± 3%	46% ± 8%	53% ± 3%	63% ± 1%	52% ± 1%
Rec_{EU} – scenario c	46% ± 5%	40% ± 8%	47% ± 5%	53% ± 5%	54% ± 2%

recycling target for 2025 (EC, 2018b). However, almost all the alternatives with a collection rate equal to 83% exceeded the 50% target, and by combining a high collection rate with improved recyclability of all polymers (A4-scenario b) it was possible to achieve even the 55% EU target for 2030 (EC, 2018b). This suggests that existing plastic recycling systems in Europe primarily based on EPR implementations are insufficient, and that design improvements are most likely needed to reach the legislative targets.

3.2. Life cycle assessment

Fig. 3 (I, II) shows the aggregated results of the LCA in weighted person equivalent (wPE), where all impact categories are normalized and weighted to provide a single score. The characterized mid-point and end-point results for all the impact categories and area of damage are reported in Appendix B. The largest contributors to the single end-point indicator were global warming potential (GWP) and particulate matter formation potential (PMFP) shown in Fig. 3, III and IV, respectively. Therefore, the following discussion is focused on these two categories only. Plastic waste management generally represented an environmental burden (positive score; see Fig. 3), unless the collection rates were maximized and the material losses minimized. The environmental impacts were similar for both baseline alternatives, i.e. 1E−02 ± 3.6E−03 wPE (A1, door-to-door collection) and 1.3E−02 ± 5.9E−03 wPE (A2, street collection), where A1 resulted better than A2 in more than 70% of the Monte Carlo iterations. Compared to A1, improving the recyclability only of PET bottles (A3), including a deposit system (A5), and improving the recyclability of all plastic products (A4), decreased the environmental impacts by 40%, 64% and 103%, respectively, due to the larger quantities of virgin plastic being substituted. Alternatives A4 and A5 scored almost always better than A1 (in 96% and 94% of the Monte Carlo iterations) independently from the uncertainty of the parameters. GWP represented a net burden particularly due to large direct emissions of fossil CO_2 from incinerating the non-recycled plastic waste (2.5 ± 0.009 t CO_2 per t of plastic packaging incinerated). On the other hand, PMFP always provided net savings (negative results), albeit the contribution from virgin plastic was higher than from energy production. As such, increased plastic incineration means higher GWP impacts and smaller PMFP savings, indicating that measures aiming at maximizing recycling efficiency, reducing the presence of impurities in the collected waste, and minimizing physical losses in the recycling chain should be prioritized. PET contributed to 62% and 80% of the savings coming from material substitution for GWP and PMFP in A1, respectively. This reflects that energy demands for virgin PET production are higher than for the other polymers (Franklin Associates, 2011), and PET is the most abundant polymer in plastic packaging waste (Andreasi Bassi et al., 2017). The uncertainties associated with GWP and PMFP (indicated in

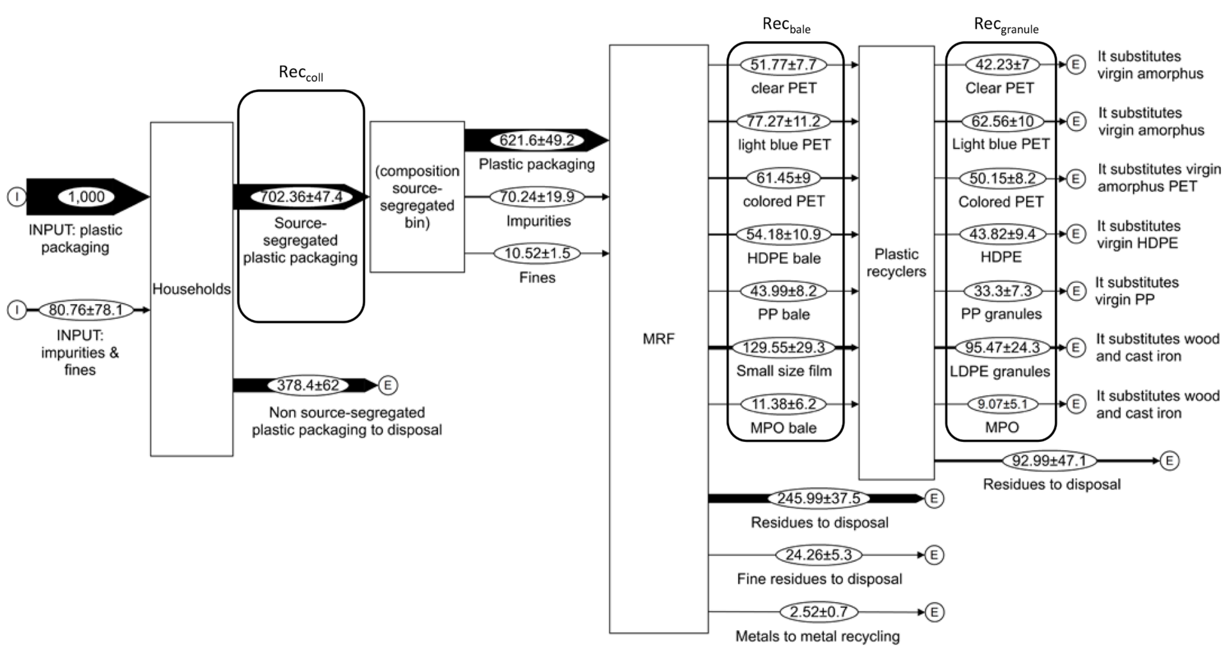

Fig. 2. Results of the mass balance of alternative A1 and the corresponding recycling rates. Note that the input called "impurities & fines" was modeled subtracting the counter-factual management (collection of the residual waste and disposal).

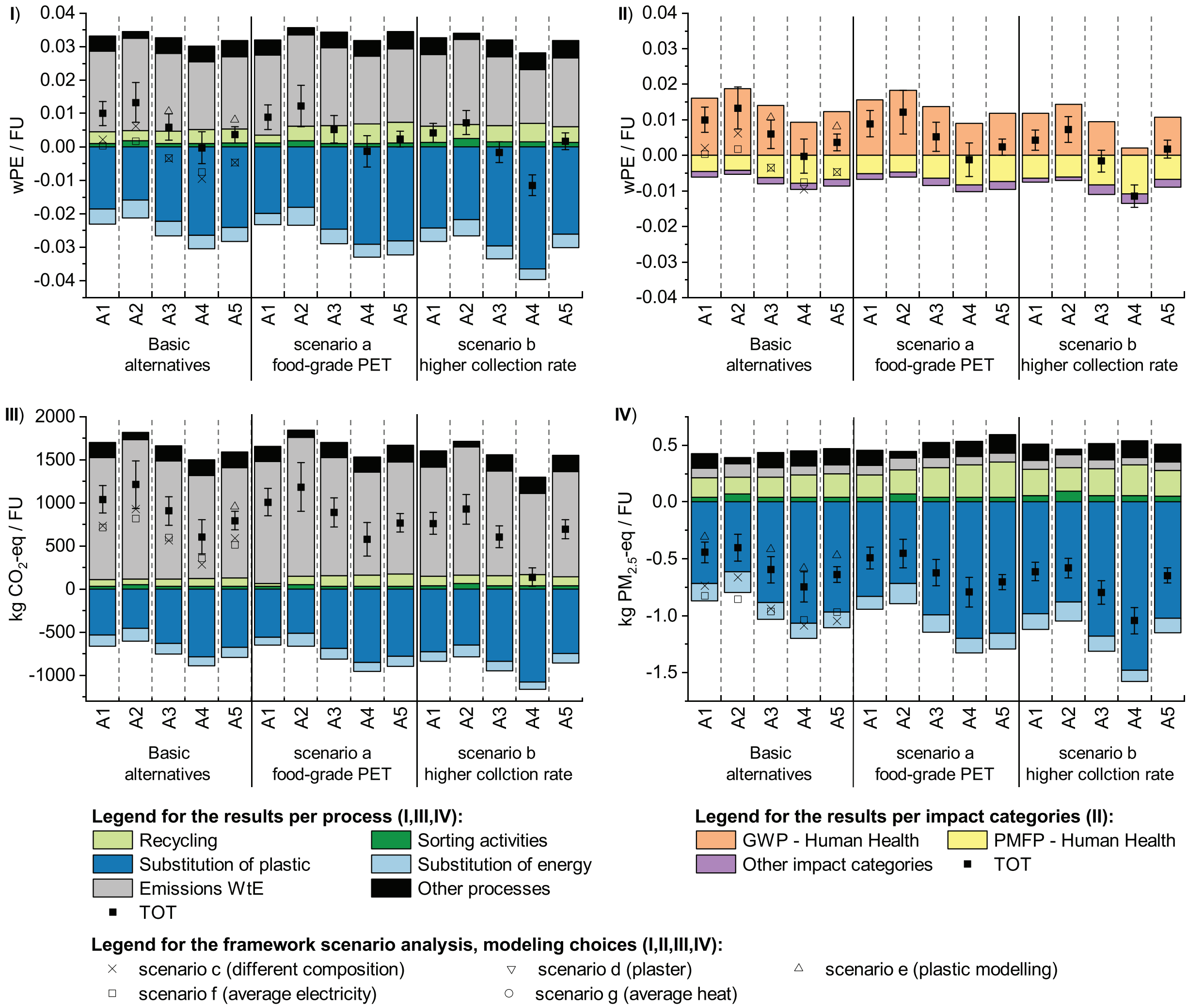

Fig. 3. Single end-point indicator grouped per impact category (I) and per process (II) and midpoint results of GWP (III) and PMFP (IV) for the five alternatives and the two framework scenarios regarding waste management (a, b). The other framework scenarios (c, d, e, f, and g) are shown for the basic alternatives only if outside the uncertainty interval. All results are shown with the average and their standard deviation. wPE: weighted person equivalent; FU: functional unit; GWP—Human Health: Global warming potential on the human health area of protection; PMFP-HH: particular matter formation potential on the human health area of protection. .

Fig. 3 as standard deviations) were caused mainly by the variability related to the collection rate (and of the pre-sorting efficiency in A2), and to a lesser extent to the fuel consumption in the collection phase, the energy recovery efficiency, and the NO_x emissions from the incinerator. More details relating the global sensitivity analysis results are in Appendix A.

The scenario analysis confirmed the ranking of the management alternatives in most situations: A4 (recyclability improvement for all products) was always the best alternative, and A5 (deposit) was the second-to-best alternative in all scenarios involving a constant collection rate, except for scenario b, where A3-scenario b (PET bottle recyclability improvement and a high collection rate) was better than A5-scenario b (deposit and a high collection rate). Finally, the door-to-door collection always appeared slightly better than street collection.

The environmental single scores decreased by 12% and by 57% when recycling into food-grade PET granules was maximized (A1-scenario a vs A1) and when higher collection rates were achieved (A1-scenario b vs A1). However, the uncertainty propagation revealed that there is, statistically, no difference between recycling PET into food-grade and amorphous PET (baseline alternative vs scenario a); conversely, increasing the collection rate (scenario b vs baseline alternatives) results in a better environmental performance between 80%% and 97%% of the Monte Carlo iterations.

Although the overall ranking was not affected, the results demonstrated the critical importance of the waste composition (Bisinella et al., 2017), in that the environmental impacts were 79% lower with the adjusted composition (A1-scenario c). In contrast, only a 4% difference was found when changing the assumption of material substitution for mixed polyolefin (A1-scenario d). A 33% higher single score was obtained when modeling virgin plastic substitution with a different dataset (A1-scenario e). Modeling heat substitution as an average heat mix (A1-scenario g) decreased the results by only 8%, while changing assumptions for electricity modeling (A1-scenario f) decreased the results by 97%, which is in accordance with previous LCA studies (Andreasi Bassi et al., 2017; Faraca et al., 2019b).

3.3. Multi-stakeholder economic benefits

Fig. 4 provides an overview of financial revenues (negative values) and losses (positive values) based on the economic analysis of the individual stakeholders. In the following, A5 is discussed separately, due to the different features of the deposit system. Detailed results of the cost-benefit analyses for all stakeholders in all alternatives and scenarios can be found in Appendix B.

The municipality experienced losses of 189–197 EUR/FU in A1, A2, A3, and A4 (to be paid by Italian citizens through a waste tax). The

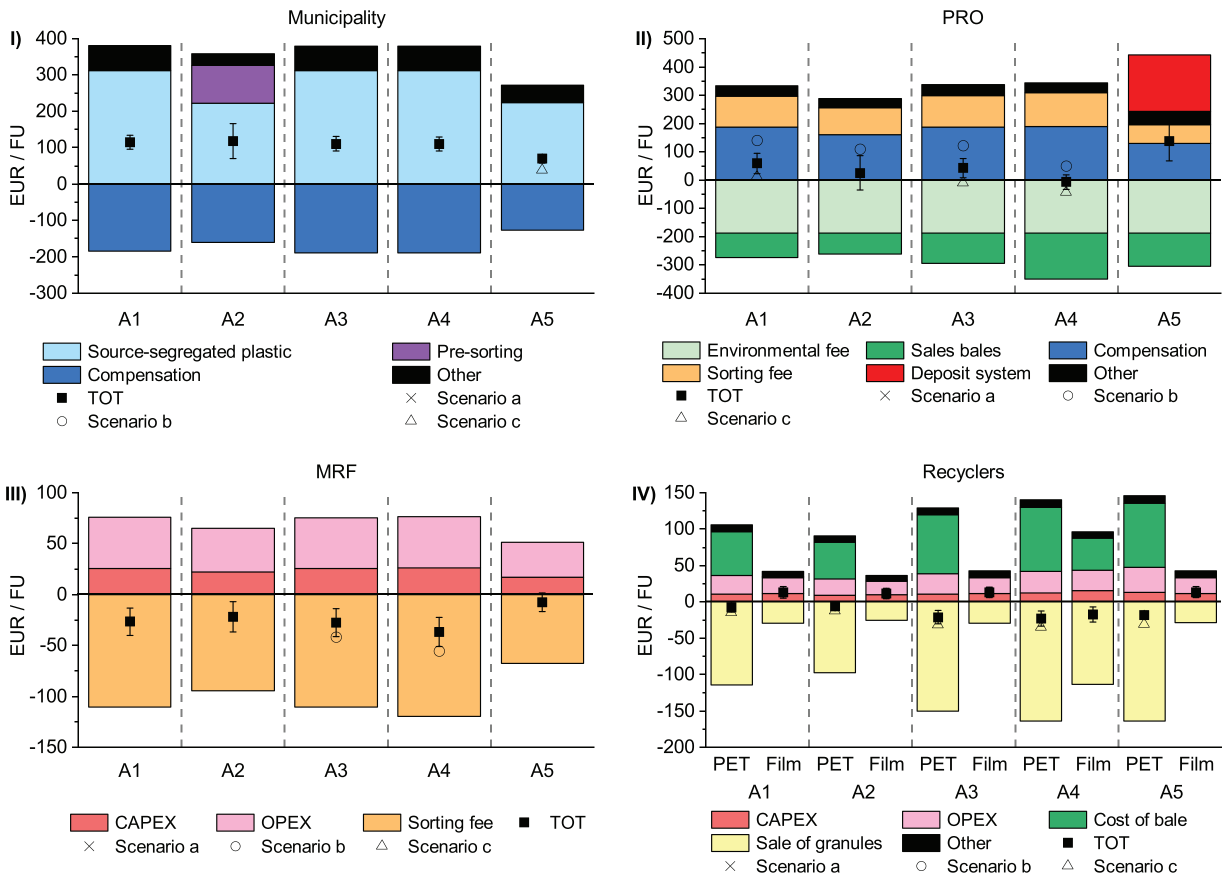

Fig. 4. Mean and standard deviation of the economic evaluation for the municipality (I), the PRO (II), the material recovery facility (III), and recyclers (IV). The results of the framework scenario were shown only when outside the uncertainty range. Due to space limitations, only PET and flexible packaging (film) recyclers were shown, where PET indicates the sum of the clear, light blue, and mixed colors PET recyclers. CAPEX: capital expenditure; OPEX: operating expenses. FU: functional unit. Detailed results are in Appendix B.

financial compensation from the PRO covered only between 60 and 70% of the costs related to the collection and management of the source-segregated plastic packaging waste in all alternatives, while municipalities additionally had to cover the costs for managing the non-source-segregated plastic packaging material. This observation is in accordance with reports for other materials in other countries (Ferreira da Cruz et al., 2014; Rigamonti et al., 2015b). Contrary to the LCA, the results of the economic assessment demonstrate that the difference between A1 and A2 is very small for the municipalities, effectively indicating little incentive for municipalities to reduce the level of impurities in the source-segregated plastic bin. The profits of the MRF were around 30 EUR/FU in A1 and A2, a value directly proportional to the collection rate. Small variations between A1 and A2 were observed for the municipality and the MRF, but not for the PRO and recyclers. The results showed economic losses for the PRO in A1 (58 ± 35 EUR/FU) and A2 (24 ± 60 EUR), in accordance with recent financial reports (Corepla, 2018a, 2017), demonstrating that increasing collection rates resulted in increasing financial losses for the PRO. Overall, the environmental fee from industry represented over 65% of the PRO's income, which achieved financial revenues only in A4, representing a situation with higher and more stable market prices for bales.

The recyclers were the weakest link in the chain. The aggregated results of the cost-benefit analysis for all recyclers involving all polymers demonstrated no practical difference between A1 and A2 (total revenues were between −3.7 and −4.6 EUR/FU), whereas revenues tripled in A5 (−15 EUR/FU) and A3 (−18 EUR/FU) and increased 13-fold in A4 (−65 EUR/FU). These results reflect the larger quantities, more homogenous properties, and higher economic value of the materials entering the system in A3 and A4. For PET and HDPE recyclers, the main expense was purchasing the plastic bales, followed by operational activities, while for PP, film, and MPO recyclers, the major cost was found in operational activities. Less important factors were transport costs for the bales and the disposal of residues. Recycling soft plastic and mixed polyolefin was particularly critical, with financial losses of around 100 EUR/t soft plastic input and 150 EUR/t polyolefins input, respectively. In fact, post-consumer film is rarely collected separately from households (EC, 2018d; Recoup, 2014a; Villanueva and Eder, 2014), and it is a challenging product (CEFLEX consortium, 2018; Horodytska et al., 2018; Plastics Recyclers Europe, 2018) because of multi-polymer composition (Eco-emballage, 2012; Horodytska et al., 2018), significant content of glues, inks, and coating (Eco-emballage, 2012; Plastic Recyclers Europe, 2018a), and its negative consequences on the sorting efficiencies of other plastic streams (Axion Consulting, 2009; Recoup, 2014b,a). Film recycling was economically feasible (negative net results, see Fig. 4) only in A4, where major design efforts were assumed to ensure virgin PP and LDPE substitution of the recycled granules.

Based on uncertainty propagation and Monte Carlo calculations, the probability of generating a profit for the individual polymers was determined. Recycling clear and light-blue PET, HDPE, and PP appeared profitable in more than 90% of cases, recycling of HDPE in 80% of cases, and recycling of mixed-color PET in 35% of cases, while soft plastic and mixed polyolefin represented a net economic loss in more than 95% of cases. This is consistent with the recognized lack of European market demand for low-quality recycled plastic and the consistent export of plastic waste to Asia. This suggests that higher environmental fees should be paid by producers of less recyclable materials (soft and mixed plastics) and poorly designed materials. The recyclers were associated with the largest uncertainty among all stakeholders, with a standard deviation between 51% for PP and over 300% for PP (in A1). The results were particularly sensitive to technical yields in recycling (i.e. how many kgs of granules are produced per kg of bale entering the plant), contributing to about 60–73% of the total uncertainty, with the remaining uncertainty associated with price variations for bales and granules and the collection rate. The relative uncertainty of the result was smaller in A3 and A4 compared to A1 and A2, reflecting the modeled reduced variability of the technical yield and of the market price with improved product recyclability.

Only three of the scenarios (a, b, c) affected the results of the economic analysis. Recycling PET into food-grade granules (scenario a) provided no clear economic advantage for the recyclers, due to higher capital investments. Increasing collection rates (scenario b) only increased the material quantities received by each stakeholder: both revenues and losses increased correspondingly. Evaluating a different plastic composition (scenario c), improved the financial situation for the PRO, MRF, and recyclers in all alternatives. This demonstrates that the entire EPR system is financially vulnerable to changes in plastic waste composition, for example, due to changes in consumer behavior, producers, and prevention initiatives. This further illustrates that regularly reviewing and adjusting financial incentives for the improved management and recycling of plastic waste is essential.

By introducing a deposit system (A5), municipalities could reduce their costs to around 50 EUR/t of collected waste (even with higher costs for collecting the remaining plastic), as the financial responsibility for PET bottle collection would be associated with the PRO. The MRF's profits were reduced to 16 ± 8 EUR/FU due to lower throughput, while the PET recyclers would generate larger profits through higher technical yields and the market value of the now cleaner materials. Compared to the baseline, the PRO observed a loss of 137 ± 68 EUR/FU in A5, thus reflecting a situation with the PROs being fully responsible for costs associated with collection, management, and sorting, while the costs for plastic packaging waste collection were shared with the municipalities in the other alternatives. Naturally, financial responsibility for the deposit system may be allocated differently; however, the results demonstrate that environmental fees from producers have to be increased in the case of deposit systems.

3.4. Recommendations for plastic waste management

3.4.1. Data quality and availability

This study indicates that limited data availability reduces the basis for system improvements, as also pointed out in previous LCA studies (Andreasi Bassi et al., 2017). Indeed, considerable differences in data quality and data availability were observed: more environmental data were available than economic data (45 studies versus 20), and more studies addressed the costs of collection and sorting in contrast to recyclers (10 studies versus 3). Data published by the European PROs were often aggregated, with the least accessible data representing the deposit systems. Legislators are thus highly recommended to address data transparency when establishing requirements for EPR and deposit systems, as data availability limits the basis for system improvements, and also in the interest of enhancing monitoring of implemented measures e.g. regarding material composition, level of impurities, correlation between impurities, and type of collection, material flows between stakeholders.

3.4.2. The role of EPR

The PRO is a critical stakeholder in the waste value chain and may absorb the majority of risks, due to its direct connection to all the other stakeholders. However, based on the results herein, further development of the EPR is paramount to reaching the European recycling targets by 2025, while minimizing the environmental burdens prevalent in plastic packaging management. First, environmental fees should be increased to reflect a product's recyclability and the existence of a market for secondary material, in agreement with recent developments in Italy (Corepla, 2019) and France, including the introduction of economic incentives for "recycling-friendly" product designs. Second, financial compensation provided to municipalities should also be redefined to support increased collection of the highest possible material quality. While deposit systems can bring both economic and environmental improvements compared to the baseline, their implementation should be carefully integrated with existing EPRs and plastic waste management systems, to guarantee financial robustness and stability throughout the value chain.

3.4.3. The challenges of European recyclers

Environmental and economic performances have the potential to improve significantly, but only when involving recycling-oriented product designs that would minimize the amount of rejects from the MRF and enable marketing of a more homogenous quality (A3 and A4). However, although significant efforts are carried out to enhance the recycling of PET bottles (EPBP, 2020), PET trays, HDPE, and PP (Plastics Recyclers Europe, 2018), the growing use of engineered plastic packaging may counteract these efforts (Eco-emballage, 2012), as also indicated by the results (A1-A2 versus A3-A4).

The results reveal that only between 15% (in A2) and 36% (in A4) of generated plastic packaging waste can be transformed profitably into flakes and granules, thereby highlighting a systemic weakness in the plastic recycling system, namely the lack of an independent and stable demand for secondary plastic products. Without improving the profitability of plastic recycling, large amounts of the plastic waste generated may be shipped to low- and middle-income countries (Ellen MacArthur Foundation, 2015; Plastic Recyclers Europe, 2018b) with lower environmental standards and cheaper labor, without creating the local jobs and the cleaner industrial activities envisioned by the "plastic circular economy" (Ellen MacArthur Foundation, 2015).

3.4.4. Reflections on the assessment framework

The choice of the three criteria to evaluate plastic waste management alternatives (recycling rate, LCA, and multi-stakeholder economic evaluation) gave a comprehensive understanding of the strengths and weaknesses of the studied system. While the assessment framework was implemented for plastic packaging, it may potentially be applied to any waste fraction. Similar analyses are recommended for those waste fractions where collection, sorting, and market conditions are strongly regulated by legislation, for example, other packaging materials, batteries, end-of-life vehicles, and WEEE (Monier et al., 2014). Such analyses can help legislators to introduce or improve EPR policies, to individuate which costs are carried by which stakeholder, to identify possible bottlenecks, to increase local job creation, and to decide when it is necessary to intervene to reduce specific environmental impacts. However, a large amount of data is needed to obtain detailed and reliable results in terms of environmental and economic assessment, and applying the assessment framework to contexts with very little available data (e.g. developing countries) may be very time demanding and bring to limited results. Finally, the majority of life cycle impact assessments does not include any impact for the plastic dispersed in the environment that are less relevant in Europe compared to other geographical contexts.

4. Conclusions

The regulatory, environmental and financial implications of five different plastic packaging waste management alternatives and 7 scenarios for each alternative were analyzed to identify potential bottlenecks in the recycling chain and thereby suggest possible improvements of the system. By simultaneously increasing the source-segregated plastic collection rate and the recyclability of all plastic products, the Italian plastic packaging system would reach a 63% recycling rate and would reduce the environmental impacts of 200% shifting from being a burden to a saving. The environmental performance was proportional to the amount of plastic recycled as flakes and granules, rather than the amount collected from households, clearly illustrating that collecting material that will be discarded later during the following sorting steps has no environmental benefit. This demonstrates that it is essential to account for the quality of the source-segregated plastic. The results of the study lead to several recommendations for future regulations: the data quality needs to be improved especially for the data published by PRO and deposit systems; the new definition of recycling rate of the European Union appears to be consistent with the environmental results (higher rate higher benefits), but the collection rate and the quantity of impurities in the source-segregated plastic bin can add some important information regarding the efficiency of the system. Also, extended producer responsibility policies (EPR) are crucial to reach environmental and economic sustainability because they are the connection link between the different stakeholders and can influence all of them either with directive incentives (as with the municipalities) or with the quality requirements for the bales (as with the MRF and the recyclers). Finally, recyclers are the weakest key of the chain because they are the only completely private institutions that have to deal with market variation and fixed operational costs. Even with a high collection rate, recycling certain polymers would not be economically profitable in Europe: without both a stronger support to create better input quality (as for flexible packaging or multi-polymeric products) and a stronger market demand by plastic converters, there is a risk that plastic waste initially collected for recycling would be either send to disposal or exported to low- and middle-income countries.

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CRediT authorship contribution statement

Susanna Andreasi Bassi: Conceptualization, Formal analysis, Methodology, Investigation, Software, Visualization, Writing - original draft. **Alessio Boldrin:** Conceptualization, Methodology, Validation, Writing - review & editing, Supervision, Funding acquisition. **Giorgia Faraca:** Methodology, Visualization, Investigation, Validation, Writing - review & editing. **Thomas F. Astrup:** Conceptualization, Methodology, Supervision, Writing - review & editing, Funding acquisition.

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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Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.resconrec.2020.105030.

Appendices

- Appendix A: Detailed information on background information on the Italian case study, goal and scope, alternatives and scenario modeling, life cycle inventory data, results of the mass balance and results of the global sensitivity analysis.
- Appendix B: Detailed results of the LCA (characterized mid-point results, characterized end-point results, aggregated end-point results in the three areas of protection, normalized end-point results in person equivalent, and weighted single score in weighted person equivalent), the economic analysis (per functional unit and per ton) and recycling targets.

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