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Research Paper

Management practices for compostable plastic packaging waste: Impacts, challenges and recommendations

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ABSTRACT

The EU Green Deal aims at solving the challenges related to plastic production, (mis-)use, and pollution. While the bioplastic industry is identified as one of the possible avenues to tackle the problem, bioplastic waste collection and management practices are still far from full-development and harmonisation. To inform policy makers on the best practices and their feasibility, this study quantifies environmental and economic impacts of compostable plastic packaging (CPP) waste management schemes by means of Life Cycle Assessment and Costing. Results show that, with respect to climate change and financial costs, the scheme leading to the highest benefits is collecting CPP with conventional plastic waste followed by mechanical sorting and recycling (saving ca. 306 kg CO_2eq. t^{-1} at a net income of 3.7 EUR t^{-1}). The second best option is collecting CPP with bio-waste followed by biological treatment (saving ca. 69 kg CO_2eq. t^{-1} at a cost of 197 EUR t^{-1}). Collecting CPP with conventional plastics followed by sorting and biological treatment is to be avoided. The trend on the other impact categories generally follows climate change. Ideally, closed loop is therefore preferred, but conditioned by (i) having high share of CPP in municipal waste (else sorting is economically unfeasible), (ii) good citizen's behaviour at source-segregation, and (iii) an established market for secondary material. Currently, overall benefits are limited by the low amounts, suggesting that the management choice could ultimately be based on rather simple technical and economic feasibility criteria while regulatory and management efforts should be focused on other waste streams with greater implications on environment.

1. Introduction

Plastic pollution is one of the most pressing environmental issues of the 21st century. The rapid growth in plastic production since the mid-20th century, accelerated by a global shift towards single-use containers in the packaging sector, resulted in an increase of plastic waste in municipal waste (MW) and littering in the natural environment. According to Geyer et al. (2017), in 2015 all plastic waste ever generated from fossil fuels worldwide (i.e., primary plastic) reached 5800 Mt, of which approx. 60% was either disposed in landfills or littered. Tackling plastics problems is one of the milestones of the European Green Deal (European Commission, 2019c), where, among the others, the bioplastic industry is identified as one of the avenues to tackle the problem. Nowadays the vast majority of global plastics produced annually (i.e., 367 Mt in 2020; Plastics Europe, 2021) are derived from fossil fuels and are non-biodegradable (i.e., it is estimated that it takes longer than 100 years to fully degrade in the environment). Only a negligible part of the actual total plastic production (i.e., 0.6%) is biodegradable (i.e., intended as materials that can be converted by microorganisms into natural substances without the need of additional additives, and in a time period that spans much lower than 100 years), albeit the global market for those products is expected to grow in the next years (European Bioplastics, 2021). Within biodegradable plastics, compostable plastics are considered as a subset with specific reference to their biodegradation in a compost system and time frame (Wojnowska-Baryla et al., 2020). In this study, the term compostable plastics is used indistinctly to

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biodegradable plastics since 99% of certified biodegradable products are either industrial or home compostable (Hann et al., 2020). The present study focuses on the most important market segment for biodegradable plastic production capacity (i.e., almost 50%) referred as compostable plastic packaging (CPP) that includes both rigid packaging (e.g., rigid fast food containers) and flexible packaging (comprising dedicated bags for bio-waste collection, shopping bags and other packaging (e.g., pre-packed fresh fruit bags)). It should be noted that flexible packaging includes shopping bags and bio-waste bags even if they are not packaging according to the definition under Article 3 of the Packaging and Packaging Waste Directive (Hann et al., 2020). According to European Bioplastics (2021), the main materials that compose the CPP market segment are polylactic acid (PLA), starch blends (TPS for thermoplastic starch), polyesters (such as polybutylene adipate terephthalate (PBAT), polybutylene succinate (PBS), and polycaprolactone (PCL)), and polyhydroxyalkanoate (PHA).

In view of the Circular Economy Action Plan (European Commission, 2020), which is at the heart of the EU Green Deal, and the expected growth in compostable packaging, it is important to provide guidance on how CPP waste should be managed within MW to minimise environmental and economic impacts. Currently, there is no consensus among stakeholders (i.e., EU Member States, producers, waste operators, NGOs) on how CPP waste should be both collected and treated. This results in different management approaches, even at local and regional scale, that give rise to confusion and scepticism among consumers. In other words, there are no clear and harmonised rules about which bin they should dispose CPP waste in. Concerning collection, CPP represents a negligible part of the total packaging waste generated at households and it is thus, depending on national approaches and legislation as well as on the local collection scheme, collected together with other waste streams, either with bio-waste, or other recyclable plastics, or residual waste. De Gisi et al. (2022) analyse the three disposal routes along with their pros and cons. As a consequence, clear messages to the consumer are often missing due to the lack of a consistent waste collection system across the region/country. Among EU countries, only Italy through Biorepack (the extended producer responsibility scheme for bioplastics), is promoting the collection of CPP along with the bio-waste fraction in line with the Italian legislation (legislative decree 152/2006 art 182). Other countries, such as Belgium, the Netherlands, France, Spain, Sweden and Germany (in the Bio-waste Ordinance of 2013 - Federal Law Gazette BGBl. I p. 658) support the use of certified compostable plastic bags only for the collection of bio-waste but tend to avoid the collection of CPP with bio-waste. For the remaining countries, no clear instructions are in place.

Concerning treatment, different options can be considered, from recycling to disposal. CPPs can be treated via mechanical recycling (MR) since most commercially available CPPs are thermoplastics, i.e., it can be melted and recycled. Despite being technically feasible (Cosate de Andrade et al. (2016); Soroudi & Jakubowicz (2013); Vu et al. (2020)), this option is currently not widely implemented as for CPP recycling it is necessary to achieve a completely pure waste stream, i.e., a mono-polymer stream. Being biodegradable, another treatment option for CPP waste is via organic recycling (composting and anaerobic digestion) along with bio-waste. Biodegradation is dependent on the degradation technique and the environment selected, so the optimal biodegradation route must be selected for each biodegradable bioplastic (Fredi & Dorigato, 2021). Finally, if not separately collected, CPPs end up in incinerators or landfills, even if these are the least preferred options in the waste hierarchy (European Commission, 2008). Notice that, even if feasible, chemically recycling of CPP is not herein considered due to its lower maturity relative to MR (Rosenboom et al., 2022).

To identify the management scheme for CPP leading to the lowest environmental impacts, Life Cycle Assessment (LCA) is the best positioned tool. Several review studies are found in the literature (Bishop et al., 2021; Ramesh & Vinodh, 2020; Spierling et al., 2020; Walker & Rothman, 2020) reporting results for LCAs on different bio-based polymers. Common to all these reviews is that they focus on specific plastic products and market applications thereof. Other studies narrow the focus on specific products or waste streams, mainly bags and PLA; Vinci et al. (2021) for example analyse the End-of-Life (EoL) options of bio- and fossil-based plastic bags used to collect bio-waste, and Maga et al. (2019) analyse the PLA-waste stream; Civancik-Uslu et al. (2019) model EoL scenarios for Spain for each plastic supermarket bag including both biodegradable and non-biodegradable polymers. Spierling et al. (2020) highlight that most of the studies are focused only on PLA, making it the only material for which there are results for all EoL options. We observe that little attention is devoted to the segregation and collection scheme and the implications of these stages on the downstream CPP treatments. There are only a few studies that consider a deeper focus on collection. Rossi et al. (2015) perform an LCA of EoL options for two biodegradable packaging materials (PLA and TPS) considering three types of logistic depending on the collection scheme. Gadaleta et al. (2022) combines environmental and economic assessment for three waste treatment routes for bioplastic waste when collected with organic, plastic and mixed waste streams. In the same line, Gadaleta et al. (2023) perform an LCA of EoL options for cellulosic bioplastics considering separately collection together with organic, plastic or mixed waste. However, they do not consider the whole market materials as addressed herein.

Drawing on the above description of the state-of-the-art, it is observed that limited results are available on the environmental and economic implications of CPP waste collection and management practices. To close the gap, this study aims to (i) quantify environmental and economic impacts of CPP waste management schemes, (ii) inform policy makers on the best practices and their viability by discussing key technical and governance criteria. To fulfil these objectives LCA and Life Cycle Costing (LCC) are used, complemented with consultations with industrial stakeholders and experts in the field.

2. Materials and methods

2.1. Environmental and economic assessment of CPP waste management pathways

2.1.1. Functional unit

LCA methodology is applied in accordance with ISO 14040/14044 standards (ISO, 2006b, 2006a). Besides, LCC methodology presented in Martinez-Sanchez et al. (2015) is applied to perform both the conventional and societal LCC, the former consisting in a traditional financial assessment and the latter also known as “welfare-economic” assessment that includes marketed goods along with the effects on the welfare of the society caused by externalities. The EASTECH software (Clavreul et al., 2014) is used to model scenarios and calculate the environmental and economic impacts. The functional unit (FU) is the management of one metric tonne (1 t) of CPP managed with the detailed composition described in Table 1 (further information in section 1 of the SM). The reference flow in each scenario is larger than one tonne (i.e., larger than

Table 1
Estimated share of each biodegradable plastic product (%) in the EU market by polymer and market segment for 2020. PBAT - polybutylene adipate terephthalate; PBS - polybutylene succinate; PLA - polylactic acid; TPS - thermoplastic starch; PHA – polyhydroxyalkanoate.

Material	Rigid packaging	Flexible packaging		
		Bio-waste bags	Shopping bags	Other packaging
PBAT	5.8	13.9	16.6	2.0
PBS	1.1	1.4	1.7	0.2
PLA	23.3	2.5	3.0	0.4
TPS	5.0	8.5	10.3	1.3
PHA	1.7	0.5	0.6	0.1

the FU) as explained in section 2.1.2.

2.1.2. System boundaries

This study performs an LCA and an LCC, including source-segregation at home, collection, transport, pre-treatment and further treatment/processing, material utilisation (e.g., on land), and disposal. While the FU is the management of 1 t of CPP, the reference flow that fulfils the FU is larger than one tonne because of the waste fractions that

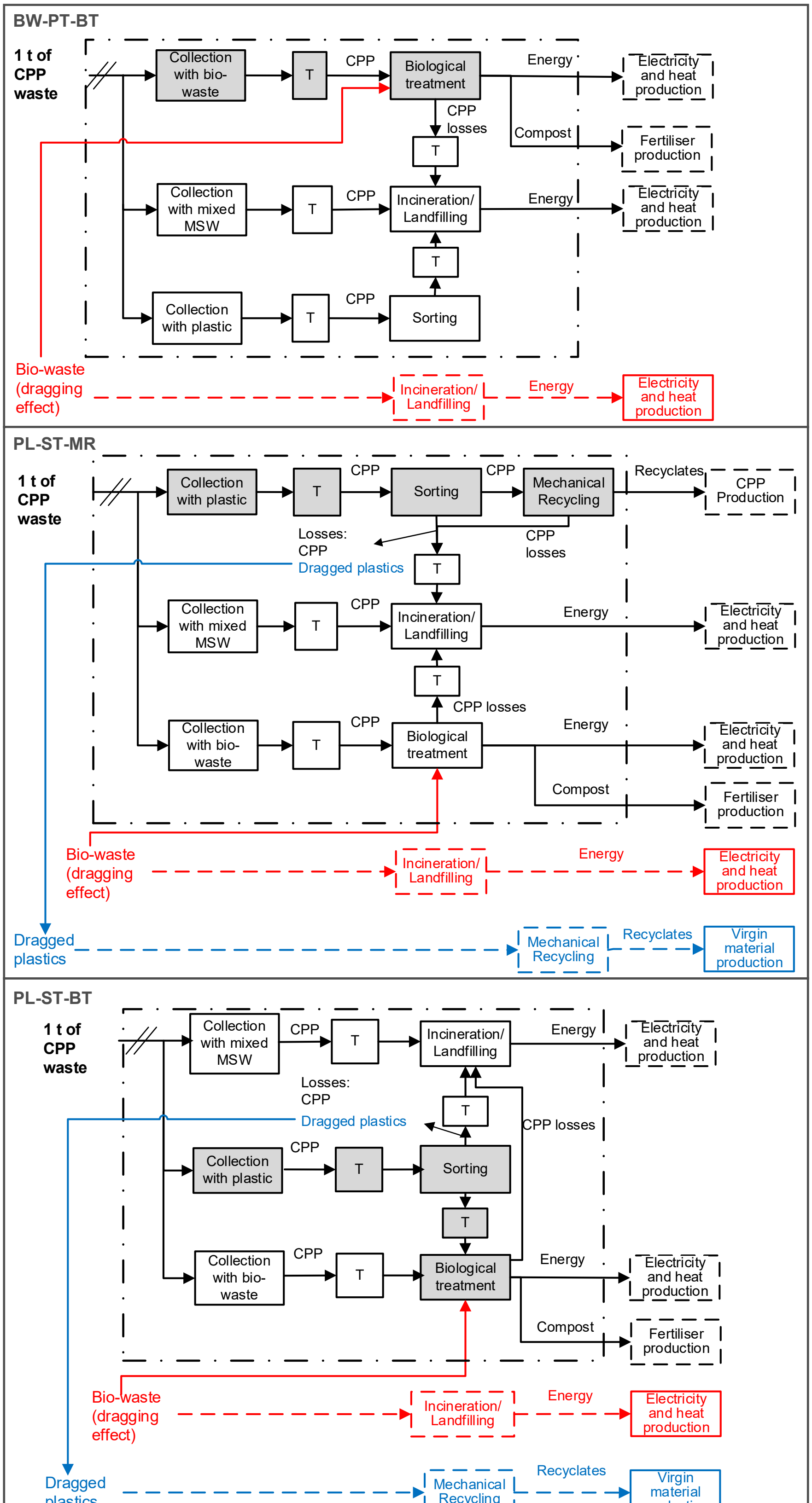

Fig. 1. Material flow for scenarios BW-PT-BT, PL-ST-MR, and PL-ST-BT. Note that the dash-dotted line represent the system boundaries and the shaded processes denote the recommended pathway for CPP. Solid rectangles and solid lines denote actual processes and flows, respectively, occurring in the scenario. On the other hand, dashed rectangles and dashed lines denote processes and flows, respectively that are no longer occurring due to the actual scenario. T refers to transport processes. The red and blue flows denote the system expansion to account for the effects on the bio-waste and plastic stream, respectively. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

are collected and treated along with CPP due to commingling, and which mass flow throughout the waste management system is affected by the presence of CPP. This reference flow accounts for all waste material fractions, different from CPP that are collected along with that one tonne of CPP, such as impurities following missortings. The quantity collected, as well as the composition, depends upon the specific management scheme (further information on SM – section 2). Besides, the system boundary of the assessment accounts for the consequential effects on other waste streams due to the management of CPP waste. This is reflected on the variation of the quantity of selected waste fractions sent to treatment depending on the commingling with CPP. In the case of bio-waste (further information on SM – section 3.1), for which a "dragging effect" is known whenever plastic packaging is sorted out at biological processing plants (i.e., there is a bio-waste portion attached to the packaging that is often removed together with the packaging during the screening pre-treatment), it is herein assumed that the use of CPP that is not screened out at the pre-treatment prior to biological treatment will avoid the dragging effect. This will lead to additional bio-waste entering the biological processing thus preventing bio-waste to end up incinerated or landfilled (see red elements in Fig. 1). In the case of conventional plastics (further info on SM – section 3.2), a dragging effect also occurs (i.e., along with the CPP separated at the sorting plant certain quantity of conventional plastic is also dragged together with it), and herein it is assumed that the quantity of conventional plastic dragged goes to incineration instead of being recycled (see blue elements in Fig. 1).

2.1.3. Impact categories

The impact assessment was performed for eight impact categories included in the product environmental footprint method (Zampori & Pant, 2019): climate change, ozone depletion, photochemical ozone formation, acidification, eutrophication terrestrial, eutrophication freshwater, freshwater eco-toxicity, and resource use for minerals and metal. Besides, economic impacts were also assessed through LCC using three main indicators, namely: the financial cost that accounts for the annualised cost of capital and operational expenditures, as well as revenues, the external cost that represents the monetisation of environmental emissions, and the societal cost that is the sum of financial costs (expressed as shadow price, subtracting taxes) and external costs.

2.1.4. Scenarios definition

Three scenarios that differ for the collection scheme implemented and the subsequent treatment technology are herein investigated (see Fig. 1). The first scenario (named BW-PT-BT) considers that CPP is collected along with the bio-waste (BW) and is treated through a screening pre-treatment (PT) preceding biological treatment (BT) combining anaerobic digestion and composting (intended as industrial composting), as this is a growing technology in the EU (for example in Italy according to ISPRA (2021). This scenario reflects the fact that CPP is in principle designed to be disposed through BT, and therefore this would be the most logic pathway. This specific scenario assumes that BT facilities agree to receive all types of CPP (i.e., flexible and rigid) when it is assured a high purity of the stream (with a certain limit on the contamination with conventional plastic packaging). The remaining two scenarios (named PL-ST-MR and PL-ST-BT) consider that CPP is collected along with other plastics (PL), and treated after sorting (ST) either through mechanical recycling (MR) or BT. These scenarios reflect that CPP is *de facto* a plastic material and commingling it with conventional plastics may be the most intuitive choice for the citizen.

2.1.4.1. Common elements and assumptions in all scenarios. It is important to include the consumer's behaviour in the collection phase. It is assumed that most of the CPP is disposed along with the recommended fraction for each scenario (either bio-waste or plastics) since consumers are instructed to do so, but based on the experience with other fractions, it is considered that some part is misplaced accidentally or purposely (named as missorting ratio, representing the % of products misplaced). It is assumed that the general missorting ratio is 36% (Rousta et al., 2016), ending either in the residual waste or the not recommended fraction for each scenario (assuming 50/50 for that), being their final fate in both cases either incineration or landfilling (assuming 70% and 30%, respectively) (relevant data for the modelling of incineration and landfilling processes are shown in the SM – section 4.2). Note that lower missorting ratios (around 10%) are reported in literature (van Velzen et al., 2019), and this is addressed in a sensitivity analysis. Relevant data for the modelling of collection and transport processes are shown in Tables S4-S6 in the SM.

Dedicated compostable bags are used to collect the bio-waste fraction, and thus it is assumed that 100% of the compostable bags are collected with this fraction. Likewise, the shopping bags made of CPP will mostly be used to collect bio-waste (62% according to Biorepack, 2022) just after being employed for carrying grocery shopping, being their main function. The remaining 38% of shopping bags is not reused for collecting bio-waste, and is assumed to be disposed along with plastic (13%) or mixed MW (25%) (Biorepack, 2022), assuming that is also common practice to reuse these bags to collect those fractions.

As mentioned earlier, whenever CPP is treated via BT and not rejected in the pre-treatment, an additional quantity of bio-waste is also modelled to be treated as a result of avoiding the "dragging effect". For this study, the dragging effect is considered as 130% for rigid packaging and 250% for flexible packaging, i.e., per each kg of rigid and flexible packaging that is not rejected, 1.3 kg and 2.5 kg of bio-waste, respectively, are treated also via BT (Consorzio Italiano Compostatori, 2020) (further information on how this quantity is calculated for each scenario can be found in SM – section 3.1). Finally, electricity and heat consumed (and substituted) in the scenarios are modelled according to the EU mix of 2020 (Keramidas et al., 2021) (see Table S12 in SM).

2.1.4.2. Scenario BW-PT-BT: CPP is collected with bio-waste and treated biologically after pre-treatment. In this scenario (see Fig. 1), consumers are instructed to dispose CPP along with BW. First, the collected fraction passes through a pre-screening step where materials that should not enter the process (i.e., some impurities) are discarded assuming a 100% efficiency, while the positive efficiency (η), i.e., the material that enters the anaerobic digestion process, equals 70% (Nessi et al., 2020) for both rigid and flexible packaging. This means that the losses that go to incineration/landfill equal to 30% of the CPP input at this step. Then, the material enters the anaerobic digestion process where energy is produced, and the digestate produced is redirected to the composting process. Note that non-degraded CPP material after composting is recirculated to the process. Further information on the modelling of the biological treatment process can be found in section 4.3 of the SM.

2.1.4.3. Scenario PL-ST-MR: CPP collected with plastics and treated via mechanical sorting and recycling. In this scenario (see Fig. 1), consumers are instructed to dispose CPP along with other plastics. The collected material is transported to the sorting facilities that remove impurities and separate plastic materials by polymer. From a detection and classification point of view, sorting bioplastics is not an easy task. Manual separation is not feasible since visual identification is not possible. There are many technologies for sorting plastic packaging waste, being the most referenced one NIR (near infrared) sorting systems that can achieve an accuracy of 86%-98% (Alaerts et al., 2018; Chen et al., 2021; Muller et al., 2014) (further details in section 4.4 of the SM). In this study it is assumed that the sorting plant has an efficiency of 97% for rigid materials and 70% for flexible (i.e., sorting positively). The material sorted negatively (i.e., 100% minus the efficiency) is sent to incineration. Lower sorting efficiencies are addressed in a sensitivity analysis. Further details for the modelling of sorting are in section 4.4 of the SM.

Once CPP material is separated from conventional plastics, additional sorting stages are required to separate CPP generating five pure streams (one per CPP type) for further processing through MR (Chen et al., 2021). Thus, the plant will present four additional NIR with the same efficiency assumed for this study. The MR process used to recycle conventional plastic waste can also be used for CPP consisting of the following steps: grinding, washing, optional drying, re-melting and re-granulating (further information in section 4.5 of the SM). The recycled pellets can be processed using all common technologies of plastics conversion (Muller et al., 2014). Recycled polymer granulate is assumed to replace virgin granulate of the same material, whose primary production burdens are credited to the system. The efficiency of the MR step is considered to be 96% for CPP (based on PLA MR inventory presented by Cosate de Andrade et al., 2016). The substitution ratio for virgin material (based on the price of virgin and recyclates obtained) for PLA is 48.5% according to Maga et al. (2019). For the rest of materials, a general substitution ratio of 50% is used as proposed by Hottle et al. (2017).

As mentioned before, during the sorting process, apart from the quantities of CPP that are not positively sorted and thus sent to incineration, certain quantity of conventional plastics are lost due to the presence of the CPP in the input material, i.e., reducing the sorting rate due to missorting or contamination (Antonopoulos et al., 2021). For this study, that quantity is estimated according to the impurity rate (6%) reported by Beeftink et al. (2021) when sorting PLA from a mix plastic stream.

As a reminder, bio-waste bags and most of the shopper bags that are reused to collect bio-waste are obviously collected along with that fraction. Thus, they are sent directly to the BT assuming the total acceptability of those products in the plant. As for the share of CPP different from bio-waste and shopper bags, which is wrongly collected with bio-waste, it is assumed that it is fully rejected at the entrance of the BT since it is considered as impurity (consumers are instructed to sort them with plastics).

2.1.4.4. Scenario PL-ST-BT: CPP collected with plastics and treated via biological treatment after mechanical sorting. This scenario is similar to PL-ST-MR explained in section 2.1.4.3, but it assumes that BT is used to valorise the CPP stream resulting after the general sorting process. Herein, the CPP stream is sorted and transported to the BT plant instead of MR (see Fig. 1), where it is assumed that the BT plant fully accepts to treat that stream (low-rejection – see details in section 2.1.4.2) as otherwise the CPP stream would have been sent directly to incineration. This CPP sorted at the sorting plant is not mixed with bio-waste at the pre-screening step passing directly to the anaerobic digestion process, and therefore does not have any influence on the bio-waste dragging effect (bio-waste is dragged while pre-screening plastics).

2.1.5. *Uncertainty and sensitivity analysis*

A parameter uncertainty analysis is conducted following the approach suggested in Bisinella et al. (2016). The uncertainty type assigned is triangular and the range assigned to the parameters is either based on literature or assumed +/- 20% (further information in Table S11, SM). Besides, four individual sensitivity analyses are conducted on four key framework assumptions to illustrate their effect on the results one-at-the-time. First, it is analysed the influence of the CPP flow acceptance at the BT plant. According to the European Compost Network (2019), not all bio-waste recycling facilities give unconditioned access to all CPP. This sensitivity scenario considers a low acceptance (high reject rate) at the pre-screening step, occurring when plants are not equipped to deal with CPP, except for bio-bags, or when there are too many impurities in the input. The second is to test the influence of the upcoming political framework concerning the use of a low-carbon energy mix (i.e., year 2050) (see Table S12 in SM), the ban on certain waste disposal technologies such as landfill, and specific economic instruments to be implemented (i.e., incineration tax of 100 EUR t^{-1} CO_2). The third is on consumer's behaviour to test how a higher compliance with the segregation instructions affects the ranking of the results. Finally, the fourth is on efficiencies at the sorting plant (i.e., sorting positively CPP), reducing drastically the assumed value to 40% for rigid materials, and 20% for flexible in line with values from Brouwer et al. (2019).

3. Results

3.1. *Mass and energy flows*

Main mass and energy flows of the three scenarios are presented in Table 2. Along with that, the quantity of recycled material is calculated for each scenario. According to calculation rules stated in the Commission implementing decisions 2019/1004 (European Commission, 2019a) and 2019/665 (European Commission, 2019b), results show that PL-ST-BT leads to a recycling rate of 66%, followed by PL-ST-MR (57%) and finally BW-PT-BT (51%).

3.2. *Environmental assessment results*

The LCA results for the eight impact categories selected are shown in Fig. 2. The contribution breakdown presents the following aggregation: (i) collection and transport including all operations involved; (ii) biological treatment with use-on-land including pre-treatment, anaerobic digestion, composting, and screening post-composting; (iii) incineration; (iv) landfilling; (v) energy recovery from biological treatment; (vi) energy recovery from incineration; (vii) material recovery including compost; (viii) mechanical sorting and recycling; (ix) bio-waste dragging effect; and (x) plastic dragging effect.

With respect to climate change impacts (Fig. 2a), the scenario leading to the highest benefits was PL-ST-MR (-306 kg CO_2eq. t^{-1}), followed by BW-PT-BT (-69 kg CO_2eq. t^{-1}), and, finally, PL-ST-BT (51 kg CO_2eq. t^{-1}) that was the only one contributing with net burdens. Across the three scenarios, the main contributions to the burdens (calculated as percentages of the total impact) for PL-ST-MR, BW-PT-BT, and PL-ST-BT came from incineration (16%, 15%, and 17%, respectively), and biological treatment and use-on-land (15%, 24%, and 25%, respectively). Specifically, the burdens from incineration were driven by fossil CO_2 contained in PBAT material, while the ones from biological treatment and use-on-land were associated with CH_4 emissions from composting and N_2O from use-on-land. On the other hand, the main contributions to the savings (also calculated as percentages of the total impact) for PL-ST-MR, BW-PT-BT, and PL-ST-BT were both energy recovery from incineration (16%, 18%, and 16%, respectively) and energy recovery from biological treatment (12%, 17%, and 18%, respectively), due to displacement of conventional electricity and heat. Material recovery, and the bio-waste dragging effect also contributed significantly to the savings but their influence was different depending on the scenario. The savings from additional bio-waste recovery were maximum in scenario BW-PT-BT (amounting to 11%) and minimum in PL-ST-MR (amounting to 7%). As for material recovery, the maximum savings were achieved in scenario PL-ST-MR (amounting to 23%), while the minimum in PL-ST-BT (amounting to 6%). The contributions from other processes such as collection and transport, albeit not negligible, were nevertheless small relative to the others.

The impacts on the other categories (Fig. 2c-h), except for ozone depletion, generally followed a similar trend to that of climate change with respect to the ranking of the scenarios (PL-ST-MR, followed by BW-PT-BT, and PL-ST-BT) and also the impact contributions. For photochemical ozone formation, the results ranged from −1.2 mol H + eq. t^{-1} (PL-ST-MR) to 0.01 mol H + eq. t^{-1} (PL-ST-BT); for acidification from −1.7 mol N eq. t^{-1} (PL-ST-MR) to 0.14 mol N eq. t^{-1} (PL-ST-BT); for eutrophication terrestrial from −0.1 kg N eq. t^{-1} (PL-ST-MR) to 7.3 kg N eq. t^{-1} (BW-PT-BT); for eutrophication freshwater from −0.16 kg P eq. t^{-1} (PL-ST-MR) to −0.1 kg P eq. t^{-1} (PL-ST-BT); for freshwater eco-

Table 2
Mass (kg) and energy (kWh) flows in the three scenarios modelled. FU: Functional Unit; IMP: Impurities from collection stage; BT: Biologically treated; INC: Incinerated; LF: Landfilled; MR: Mechanically recycled; PD: Plastic dragging (i.e., plastic dragged and sent to incineration); BW: Bio-waste dragging (i.e., bio-waste entering the biological treatment due to the avoidance of the dragging effect); Recycled output: Material output from recycling. Note that the mass balance is only closed in this table for CPP.

	Mass flow at collection		Mass flow at BT		Mass flow to incineration/landfill		Sorting output		Recycling output		Energy recovery output		
	CPP	IMP	CPP	BW (dragging)	CPP INC	CPP LF	CPP	PD (dragging)	Compost	Plastic	BT	INC	LF
BW-PT-BT	1000	47	511	1079	377	112	–	–	328	–	1024	1131	132
PL-ST-MR	1000	114	327	819	317	111	246	19	206	246	702	1115	126
PL-ST-BT	1000	114	656	892	249	95	–	19	385	–	984	920	99

toxicity from −23760 CTUe t^{-1} (PL-ST-MR) to −15821 CTUe t^{-1} (PL-ST-BT); and, finally, for resource use for minerals and metals from −0.003 kg Sb eq. t^{-1} (PL-ST-MR) to −0.001 kg Sb eq. t^{-1} (PL-ST-BT). For the above-mentioned impact categories, the main contributions to the impacts were on the positive side (i.e., burdens) mainly from biological treatment and use-on-land (up to 57% for eutrophication terrestrial in the BW-PT-BT and PL-ST-BT). On the other hand, the main contributions to the savings were from energy recovery (mainly for scenarios where CPP is treated through BT) from both biological treatment that achieved up to 36% for eutrophication freshwater, and from incineration that achieved up to 29% also for eutrophication freshwater, and material recovery for the PL-ST-MR accounting between 25% for eutrophication terrestrial and 65% for freshwater eco-toxicity. Further, the bio-waste dragging effect influenced greatly the savings of photochemical ozone formation (between 10% for PL-ST-MR and 15% for BW-PT-BT) and eutrophication terrestrial (between 7% for PL-ST-MR and 10% for BW-PT-BT), but led to burdens in eutrophication freshwater, freshwater eco-toxicity, and resource use for minerals and metals. The contribution from other processes were nevertheless small compared to the others for all scenarios and impact categories analysed.

The results obtained for ozone depletion (Fig. 2b) did not follow the general trend seen for climate change. In this case, BW-PT-BT presented the highest savings (-8E-5 kg CFC-11 eq. t^{-1}), followed by PL-ST-BT (-6E-5 kg CFC-11 eq. t^{-1}), and finally PL-ST-MR (-4.5E-5 kg CFC-11 eq. t^{-1}). Herein, the main contributions to the burdens were related to the landfilling process, mainly due to emissions from the oxidation in the top cover of the landfill (between 30% for PL-ST-BT and 34% for PL-ST-MR). On the other hand, the main contributions to the savings were related to the bio-waste dragging effect (between 34% for PL-ST-MR and 42% for PL-ST-BT), mainly due to the avoided emissions from the oxidation of bio-waste in the top cover of the landfill, and energy recovery from incineration (11% for the three scenarios analysed). For PL-ST-MR, the material recovery contributed ca. 7%, mainly due to the substitution of PLA.

3.3. Life Cycle Costing results

The results of the LCC are shown in Fig. 3. The scenario with the lowest financial costs (see Fig. 3a) was PL-ST-MR with −3.7 EUR t^{-1} (i.e., a net income), followed by BW-PT-BT and PL-ST-BT with a net cost of 197 and 296 EUR t^{-1}, respectively. The main contributions to the financial costs came from collection and transport equalling 175 EUR t^{-1} for BW-PT-BT (20% of the whole cost) and ca. 228 EUR t^{-1} for both PL-ST-MR and PL-ST-BT (17% and 26% of the whole cost, respectively), as well as biological treatment with use-on-land summing to 275 EUR t^{-1}, 196 EUR t^{-1}, and 264 EUR t^{-1} for BW-PT-BT (32%), PL-ST-MR (15%), and PL-ST-BT (30%), respectively. For the scenario PL-ST-MR, other two processes contributed significantly to the financial costs, namely sorting and recycling process as well as the revenues from material recovery, accounting for 137 EUR t^{-1} (11%) and −367 EUR t^{-1} (28%), respectively.

Fig. 3b shows the external costs resulting in net savings for the environment of −35 EUR t^{-1} for PL-ST-MR, −25 EUR t^{-1} for BW-PT-BT, and −12 EUR t^{-1} for PL-ST-BT. Fig. 3c shows the societal cost, resulting from summing financial and external costs, which follows a similar trend to that of climate change. The external costs represent 4%, 11% and 90% of the societal cost for PL-ST-BT (285 EUR t^{-1}), BW-PT-BT (173 EUR t^{-1}), and PL-ST-MR (-38 EUR t^{-1}), respectively.

3.4. Parameter uncertainty and sensitivity analyses results

In general, across all the impact categories assessed, the ranking between the scenarios was maintained even when accounting for uncertainty variation around the net result (Figs. 2-3). Generally, there was a slight overlap for scenarios BW-PT-BT and PL-ST-BT for climate change, acidification, terrestrial eutrophication, freshwater eutrophication, freshwater ecotoxicity and resource use, as well as for financial cost. The only impact category that presented a clear overlap among the three scenarios was ozone depletion. The main contributions to the uncertainty across all environmental impact categories (accumulating more than 95%) were given by four parameters, namely the efficiency of heat production at incineration (correlated with the parameter for electricity production), the parameter that distributes the percentage of residues going to landfilling, the pre-screening efficiency of CPP at the biological treatment plant, and the dragging effect factor for flexible packaging. For the financial costs, the main contributions to the overall uncertainty were the total cost for collection as well as the electricity and heat price. For the PL-ST-MR scenario, apart from those parameters already mentioned, another main contribution was the capital expenditure of the NIR. The detailed results of the analytical uncertainty propagation may be consulted in section 4.6.3 of the SM.

The results of the three sensitivity analyses are shown in Fig. 4a-b for climate change and societal cost, respectively. The first sensitivity analysis performed on the influence of the CPP acceptance at the biological treatment plant, showed that a high rejection worsened the impact on climate change by 72% relative to the default scenario, while costs showed negligible variations. The second sensitivity analysis on the influence of the upcoming political framework (low carbon energy system, no landfill, and tax on incineration) showed higher climate change impacts for all scenarios analysed relative to the default calculation. Impacts from landfilling obviously disappeared and from incineration increased. But the net increase of emissions relative to the default scenario was due to the decrease in savings from energy recovery as the energy system will be less C-intensive. The third sensitivity analysis on the consumer's behaviour showed that better sorting behaviour of citizens had almost no effect for BW-PT-BT, but significant consequences on the performance of PL-ST-MR (increase of savings) and PL-ST-BT (increase of burdens), accompanied with an increase in recycled material (9–13%). This incurred in higher revenues when the CPP are sent to mechanical recycling (PL-ST-MR). Finally, the fourth one on sorting efficiencies, in line with the sensitivity analysis on consumer's behaviour, showed that lower sorting efficiencies at the sorting plant almost had no effect for BW-PT-BT. However, it had significant consequences on the performance of PL-ST-MR (increase in burdens) and PL-

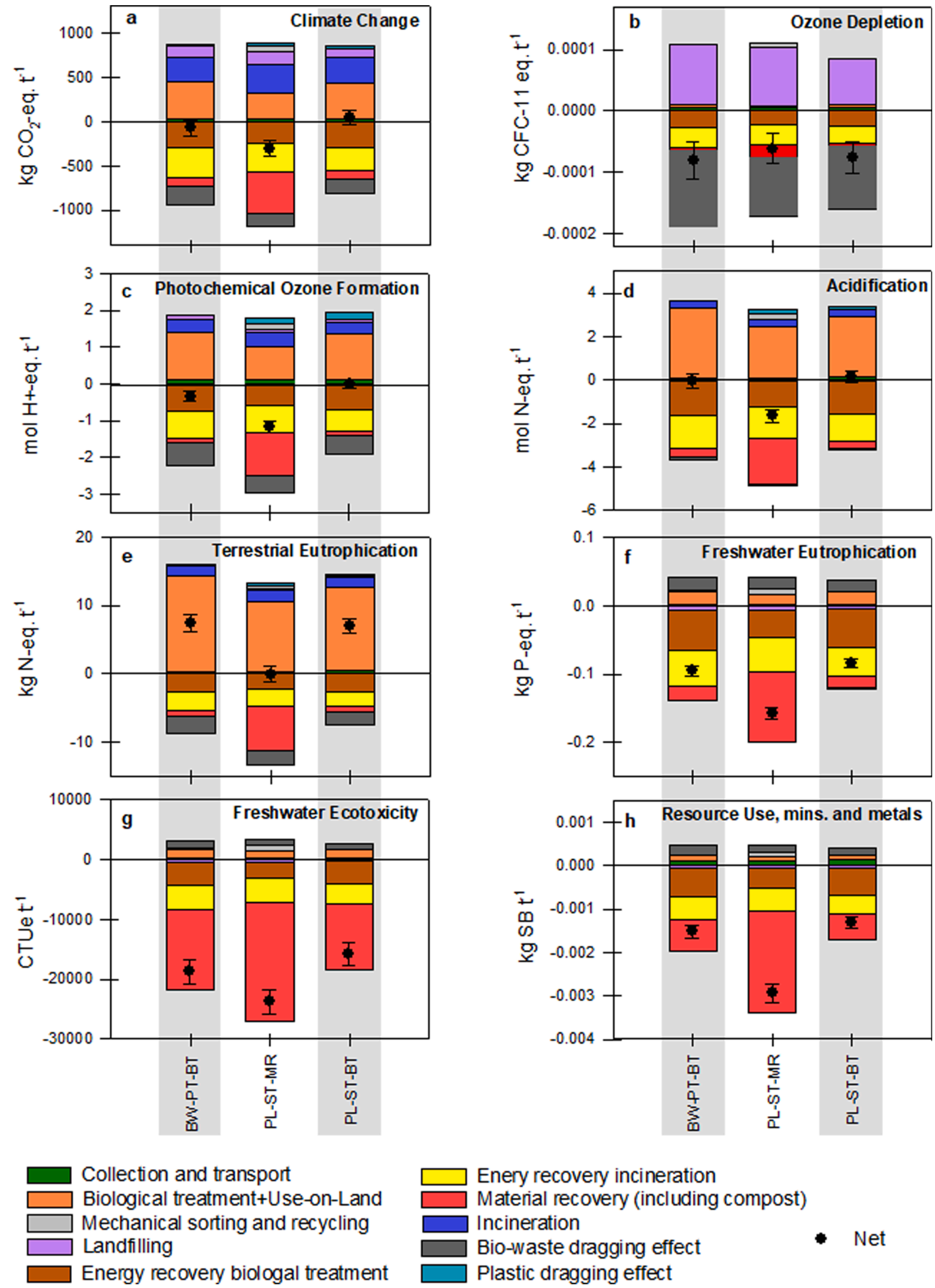

Fig. 2. Characterised LCA results per tonne of CPP managed with breakdown of the contributions. Values above zero represent burdens, while values below zero represent savings. The final net impact, per each individual category, is the sum of burdens and savings and is represented with a black circle. The error bars represent the standard deviation of the net result.

ST-BT (increase in savings), accompanied with a decrease in recycled material (around 18%) for both. This incurred in higher costs for PL-ST-MR due to lower material recovery revenues.

4. Discussion

4.1. Main assumptions and limitations of the study

The scenarios assessed rely on several assumptions and face important limitations that need to be considered. Concerning the material modelling, for the sake of simplicity, the different materials were considered as standalone polymers conforming the products; however, in the real market some of them are blended, such as TPS and PBAT. Besides, some technological and market data were based on experiments done for PLA and extrapolated to the other materials. Lack of real scale studies about NIR separation with low levels of CPP as a percentage of conventional plastics introduces additional uncertainty, and better data are desirable for more accurate modelling and results. Furthermore, future studies should also focus on the actual quality of the secondary material produced from mechanical recycling and the actual substitutability of the virgin bioplastic counterparts (Tonini et al., 2022). In this study, we assume that the substitution equalled to 48–50% in the default calculation. A sensitivity analysis showed that a substitution factor below 19% would make this scenario worse than biological treatment.

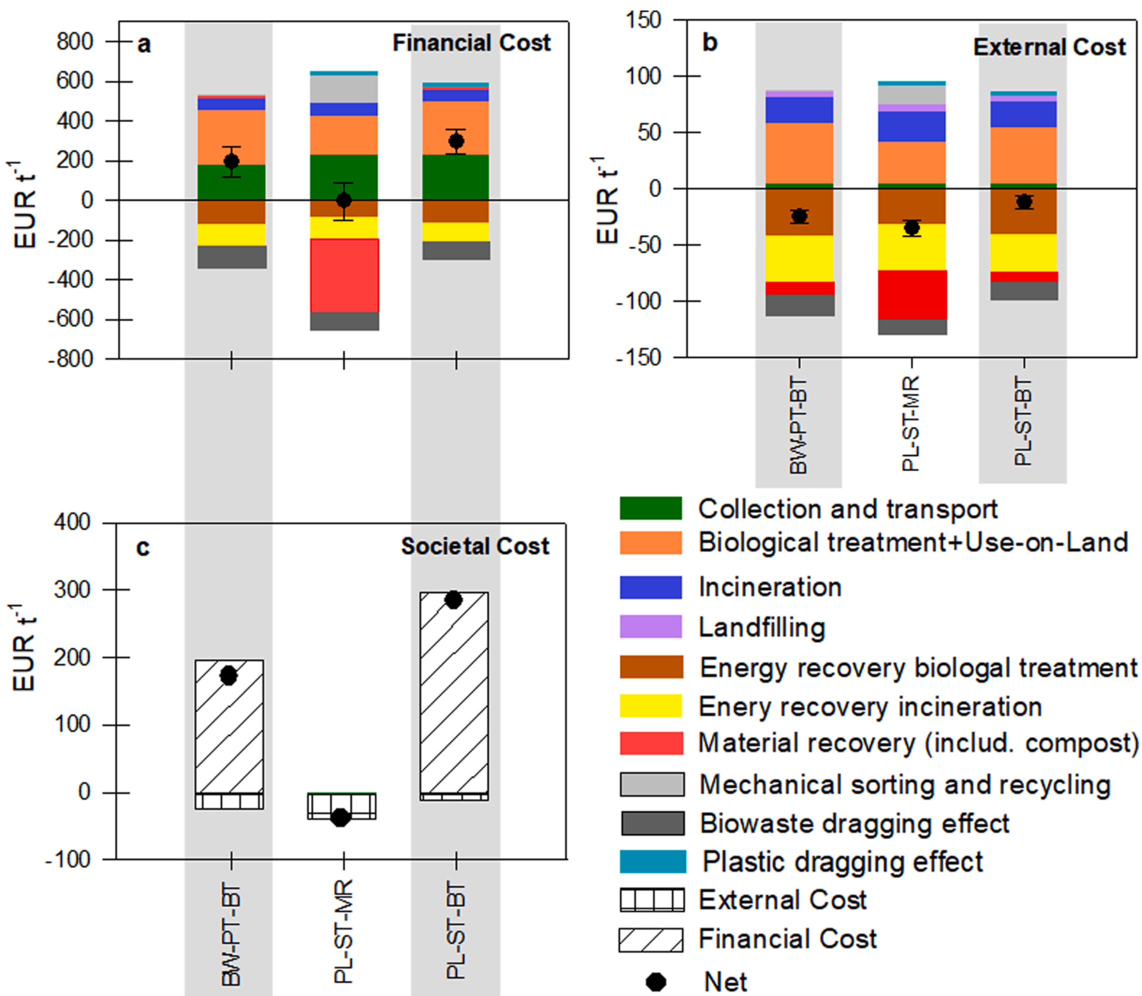

Fig. 3. LCC results per tonne of CPP managed with breakdown of the contributions: a) financial costs; b) external costs, c) societal costs. Values above zero represent costs, while values below zero represent revenues. The final net result, per each individual category, is the sum of costs and revenues and is indicated with a black circle. The error bar represent the standard deviation of the net result.

In the same line, further studies should include chemical recycling as a promising option for close-loop recycling (Payne & Jones, 2021), as well as analyse the possible competition with mechanical recycling for the plastic material in the market. For the consumer's behaviour, in the default calculations it is assumed that citizens are able to recognise CPPs and behave accordingly to the mandated instructions for separate waste collection. This was tested in a sensitivity analysis, which showed that the ranking did not change. While the former can be achieved via appropriate certification and labelling schemes (Allison et al., 2021), the latter can be obtained through different policy instruments that help citizens to sort properly (e.g., economic incentives). However, the efficiency of these instruments is not yet fully understood (Cristóbal et al., 2022).

For the cost of mechanical recycling, a key assumption is that the capital expenditure of the NIR needed to separate CPP from conventional plastics when commingled together is calculated assuming a share of CPP in the mix of plastic packaging equal to 6%, based on Beeftink et al. (2021). Assuming a share of CPP in the packaging mix of 0.6% results in a financial cost of ca. 676 EUR t^{-1} for the scenario involving MR, making it economically unfeasible. However, the 6% assumption seems well-supported by future outlooks (European Bioplastics, 2021; Spierling et al., 2018).

4.2. How do these results relate to the broader MW management impacts?

In 2020, according to Eurostat (2023), 240 Mt of MW were generated in the EU27. According to Albizzati et al. (2023), the impacts on climate change of MW management in the EU range between −148 kg CO_2eq. t^{-1} and 740 kg CO_2eq. t^{-1}, with a weighted average across EU27 of 218 kg CO_2eq. t^{-1}. As for MW management cost, Albizzati et al. (2023) indicate a range between 207 and 412 EUR t^{-1} across EU27, with an average of 310 EUR t^{-1}. It derives that the annual impact of MW management in the EU27 is around 52 Gt CO_2eq., and the cost 73 billion EUR. According to the results and under the assumption that plastic packaging is 5.5% of the total MW and that CPP increases up to 6% in the packaging mix (i.e., 0.33% of total MW leading to a total of 0.78 Mt of CPP annually), the annual impact of CPP management would be, in the worst case, 40 kt CO_2eq. (i.e., 0.08% of the total MW management impact on climate change). The annual financial cost of managing CPP waste would be in the worst case 232 million EUR (i.e., 0.3% of the total MW management cost). This suggests that the choice of one option over the other has (overall) limited environmental and economic consequences at the broader level of MW management. This, in turn, suggests that the CPP management practice could be rather based on technical and economic feasibility, and that regulatory and management efforts could be instead focused on other waste streams that have greater implications from an environmental and economic perspective.

Further to this, one should consider the EU recycling targets set out

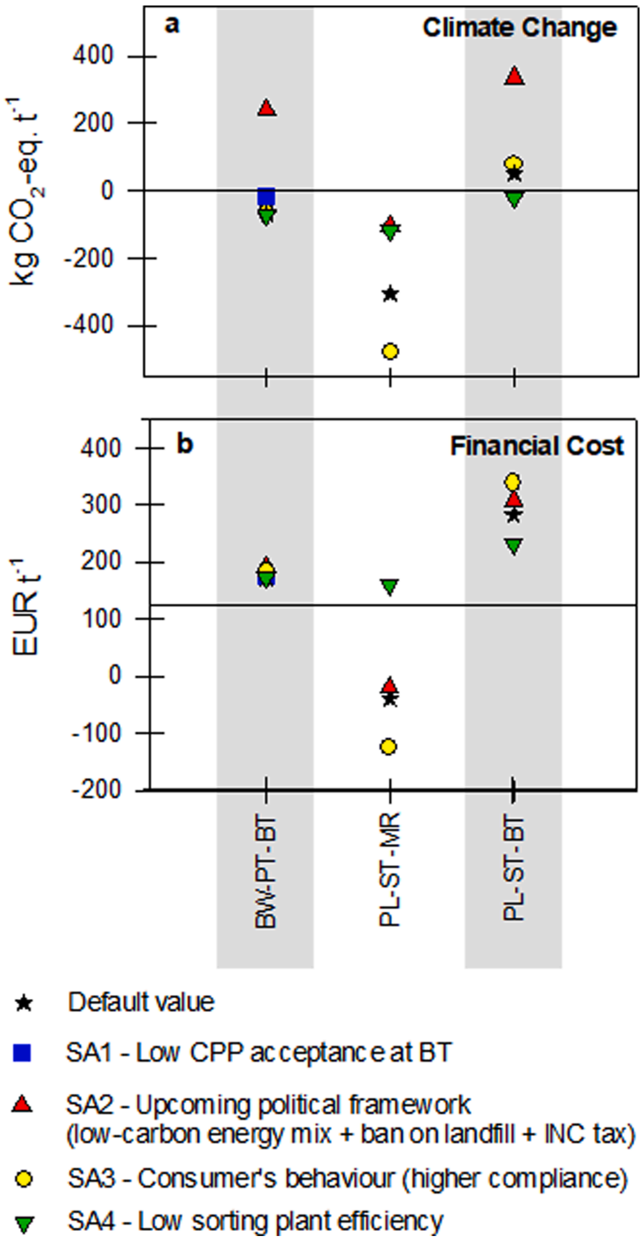

Fig. 4. Net results obtained for climate change and societal costs for the default scenario and the four sensitivity analyses performed. Notice that SA stands for sensitivity analysis, BT for biological treatment, CPP for compostable plastic packaging, and INC for incineration.

by the revised EU Directive 2008/98/EC on waste (European Commission, 2008). It is important to note that, as herein modelled, the use of CPP for packaging will avoid the bio-waste dragging effect (assuming that this packaging would have been otherwise made using conventional plastics), and, thus, increases the total recycling rate. For instance, in the scenario where CPP waste is collected and treated with bio-waste, it is estimated that ca. additional 1 t of bio-waste is sent to recycling (instead of landfill or incineration) per t of CPP managed, avoiding in total 2 kg of waste/capita going to landfill and increasing the overall recycling rate by ca. 0.4% points additional relative to a scenario in which CPP and dragged bio-waste go to landfill or incineration.

4.3. Challenges and barriers

The CPP value chain comprises many different actors with many challenges and barriers and sometimes opposing interests. This section briefly summarises challenges and barriers for some of them..

4.3.1. The view of CPP producers

According to the CPP producers, these products are generally designed to be collected with bio-waste, and they should preferably be used for applications where they are contaminated with food waste. In case they are also used for non-food-contact applications, collection with plastics for mechanical recycling should be preferred (clean streams). A clear identification of the two situations using pictograms could be useful. Producers (Buijzen et al., 2020) acknowledge that the most convenient EoL option for bioplastics (after prevention, reduction and reuse) is mechanical recycling followed by chemical recycling and biological treatment. Nowadays, it is technically possible to separate CPPs in sorting plants but it is not cost-efficient due to their relatively small share in the mix of plastic packaging. Preconditions to justify additional investments are a higher share of bio-based plastics in the recycling stream and an established market for the resulting secondary materials. However, the actual position of some packaging recycling stakeholders (e.g., CEFLEX and RECYCLASS) in their design-for-recycling guidelines is that flexible CPPs, even when present at low levels, are expected to cause disruptions in the mechanical recycling process and negatively affect the quality and value of the final recyclate (CEFLEX, 2020).

4.3.2. The view of biological treatment operators

According to the European Compost Network (European Compost Network, 2019), CPP can be differentiated into two categories with different acceptability level at the plants. On the one hand, a group with great acceptance from operators including compostable products that ease citizens in collecting bio-waste separately (e.g., compostable bags) that eventually increase the capture rate of bio-waste while reducing the amount of impurities. On the other hand, a group with poor acceptance from operators (e.g., catering packaging and complex compostable packaging), especially from composting plants due to the actual layout and the material flow management (e.g., pre-treatment before the composting process). However, in Member States like Italy, where compostable items must be collected commingled with bio-waste by law, most composting and digestion plants accept all types of CPP and are adjusting the pre-treatment to minimise the reject rate. It is important to notice that from a legal perspective there is a challenge for biological treatment operators since there is no specific waste code for CPP, so they may not be accepted by composting/digestion plants because of licensing issues.

4.3.3. Considerations on consumer's behaviour

Efficient collection and sorting largely determine the efficiency of waste management systems. This implies that consumer's behaviour is a key factor. To increase participation, labels and pictograms play an important role (Wojnowska-Baryla et al., 2020). On the other hand, there are concerns that consumers might misunderstand biodegradability or compostability claims as a 'licence to litter', but evidence supporting or refuting these concerns is scarce (Hann et al., 2020). It is also not clear whether citizens will get the message right, and behave consequently, when purchasing a compostable item and being told to deliver it together with conventional plastics, if this is the preferred management practice by the authority. When it comes to foster a behavioural change related to improve recycling and reducing contamination, Kaufman et al. (2020) pointed out that this can be induced firstly by relieving constraints (e.g., offering more frequent collection and smaller waste-collection containers to relieve limited space) to make correct recycling easier. The authors also highlight that a very important role is played by the consistency and predictability across regions of what can be recycled. In this respect, the fact that CPP are quite spread and accepted in some Member States or specific treatment plants, and are seen as an unwanted material in others, increases the sense of uncertainty by citizens and contamination as a side effect.

4.3.4. Considerations at system level and policy making

The system-level lock-in that results from well-established traditional systems of producing and consuming can inhibit the implementation of new technological solutions despite their apparent environmental and economic advantage, thus hindering the transformation towards a circular economy (Aminoff & Sundqvist-Andberg, 2021). The plastic industry is dependent on conventional plastics and presents several lock-ins across the different steps of the value chain (Bauer et al., 2022) that can only be solved with a simultaneous co-evolution of the technological and institutional systems. In this line, the European Commission, with the main aim to ensure packaging re-use and recycle and acknowledging the sustainability challenges and trade-offs that CPP presents, proposed a policy framework on the use of biodegradable and compostable plastics (European Commission, 2022) that restricts the suitable applications for CPP but also leaves open the possibility to extend that list when justified.

5. Conclusion and recommendations

The results indicate that, ideally, collection with plastics for closed-loop mechanical recycling is the best option, both from an environmental and a cost perspective. However, the practical implementation of this option is conditioned by the economic feasibility of the NIR technology at the sorting, the establishment of a secondary material market, and proper citizen's behaviour in the segregation phase. To this, clear labelling and sorting instructions for citizens are a precondition along with awareness-raising and information campaigns at national level to increase participation to the source-segregation overall. Collecting and treating compostable plastic with bio-waste appears as the second best option. This practice is implemented by law and promoted by extended producer responsibility schemes in Italy and would profit from the experience already in place in this Member State. Achieving a low reject rate of compostable plastic packaging at the screening pre-treatment of biological facilities is key to achieve the environmental performances illustrated in this study. Thus, promoting the acceptance of these products at such plants is a precondition for the success of this management practice. Collecting compostable plastic packaging with plastics followed by separation with NIR at sorting and subsequent biological treatment appears as the worst option and should be avoided. Considering the broader municipal solid waste management in the EU, the results further suggest that the choice of one option over the other has overall limited environmental and economic consequences. This, in turn, suggests that the decision could be rather based on technical and economic feasibility while focusing instead the efforts on waste streams that have greater implications from an environmental and economic perspective.

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.

Data availability

Data will be made available on request.

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Disclaimer.

The views expressed in the article are the sole responsibility of the authors and in no way represent the view of the European Commission and its services.

Appendix A. Supplementary material

Supplementary data to this article can be found online at https://doi.org/10.1016/j.wasman.2023.08.010.

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