Journal of Cleaner Production 319 (2021) 128580

Contents lists available at ScienceDirect

Journal of Cleaner Production

journal homepage: www.elsevier.com/locate/jclepro

Environmental life cycle assessment of the incorporation of recycled high-density polyethylene to polyethylene pipe grade resins

Ioan-Robert Istrate [a,b], Rafael Juan [b,c], Mario Martin-Gamboa [b,*], Carlos Domínguez [b,c], Rafael A. García-Muñoz [b,c], Javier Dufour [a,b]

[a] Systems Analysis Unit, IMDEA Energy, Avda. Ramón de La Sagra 3, 28935, Móstoles, Madrid, Spain
[b] Chemical and Environmental Engineering Group, Rey Juan Carlos University, 28933, Móstoles, Madrid, Spain
[c] LATEP, Polymer Technology Laboratory, Rey Juan Carlos University, Tulipán St., 28933, Móstoles, Madrid, Spain

ARTICLE INFO

Handling editor: Bin Chen

Keywords:
Carbon footprint
Circular economy
High-density polyethylene (HDPE)
Life cycle assessment
Plastic pipe
Plastic recycling

ABSTRACT

Plastic recycling involves a range of potential environmental benefits, from curbing landfill and incineration rates to the reduction of greenhouse gas emissions. However, the main challenge is to find applications where recycled plastic can successfully provide the same functionality as the replaced virgin plastic. Particularly, the incorporation of recycled high-density polyethylene (HDPE) to polyethylene (PE) pipe grade resins is a great challenge that is not currently being implemented in the manufacture of pressure pipes. In this study, life cycle assessment (LCA) is applied to quantitatively evaluate the potential environmental impacts from producing PE pipe grade resins from recycled HDPE blended with virgin HDPE. The LCA involves four HDPE waste feedstocks (crates/caps, packaging/detergency bottles, post-consumer industrial containers, and automobile fuel tanks) and two PE pipe grades (PE80 and PE100). Moreover, different allocation approaches that affect the LCA of plastic recycling, namely the cut-off approach and the Circular Footprint Formula, were investigated. The recycled content was found to largely determine the LCA results. In this regard, the production of PE80 quality from the pure HDPE waste feedstocks (such as automobile fuel tanks and post-consumer industrial containers) allows a higher recycled content, thus resulting in lower impacts. Compared with a 100 % virgin resin, these two scenarios show 80 % and 53 % less carbon footprint if the waste feedstock is considered burdens free (cut-off allocation). These percentages however decrease to 32 % and 20 % if the impacts and benefits are shared according to the Circular Footprint Formula. These trends were similarly observed for most of the impact categories evaluated, such as, acidification and fossil resources. The robustness of these results is supported by error propagation via Monte Carlo simulation.

1. Introduction

The production of plastics has increased twentyfold since the 1960s, and it is expected to double again over the next 20 years with the subsequent generation of million tonnes of plastic litter (Ellen MacArthur Foundation, 2017). Each year around 300 million tonnes of plastic waste is generated globally (Geyer et al., 2017). The largest proportion of this plastic waste is currently incinerated and/or disposed of in landfills, while recycling still plays a minor role. Geyer et al. (2017) estimated that only 9 % of all plastics produced between 1950 and 2015 have been recycled. In the United States, the rate of plastic recycling barely reached 6.2 % by 2015 (Di et al., 2021). At European level, from the post-consumer plastic waste collected in 2018 about a 33 % was recycled, 43 % was incinerated for energy recovery, and the rest was disposed of in landfills (Plastics Europe, 2020). The relatively low rates of plastic recycling observed worldwide represents a major environmental concern, primarily due to microplastic contamination (Mitrano and Wohlleben, 2020). The global amount of plastic entering the ocean in 2010 was estimated between 5 and 13 million tonnes (Jambeck et al., 2015). Lau et al. (2020) estimated that the global amount of macro- and microplastic entering aquatic systems can double by 2040 with respect to 2016. Overall, the environmental issues related to plastic waste will be exacerbated in the future driven by an increased generation (Lebreton and Andrady, 2019).

The circular economy concept is increasingly seen as a potential solution to the plastic waste problem. One of the measures included in

* Corresponding author.
E-mail addresses: mario.mgamboa@urjc.es (M. Martin-Gamboa), carlos.dominguez@urjc.es (C. Domínguez).

https://doi.org/10.1016/j.jclepro.2021.128580
Received 11 May 2021; Received in revised form 20 July 2021; Accepted 5 August 2021
Available online 6 August 2021
0959-6526/

the European Strategy for Plastics in a Circular Economy is the use of alternative feedstocks, such as plastic waste, for plastic production (European Commission, 2018a). The potential environmental benefits of using waste feedstocks for plastic production (i.e. plastic recycling) range from curbing landfilling and incineration rates to the reduction of fossil fuels consumption and the associated GHG emissions (Lazarevic et al., 2010). However, these potential environmental benefits strongly depend on the quality of the recycled plastic and, more precisely, on its ability to provide the same functionality as the replaced virgin plastic (Faraca et al., 2019). The quality of the recycled material is also the limiting factor to ensuring market competitiveness compared to virgin plastics, which is reflected in the relatively low recycled content achieved by the most advanced economies. For instance, the average recycled content of plastics produced in the United States in 2015 was just 4.2 % (Di et al., 2021). Within the European context, the recycled content of polyethylene terephthalate (PET) bottles that entered the market reached 20 % in Austria (reference year 2013) and 30 % in Germany (reference year 2017) (Schmidt et al., 2020). A main challenge to the plastic waste problem is to find strategies where recycled plastic can successfully substitute virgin plastic while maintaining functionality.

Among all the plastics demanded every year, polyethylene (PE) is one of the most abundantly produced plastics worldwide. Around 110–120 million tonnes of PE are produced annually, with almost the same amount of waste generated (Geyer et al., 2017). Nowadays recycled PE is used in the construction sector or packaging (Grigore, 2017; WRAP, 2019), whereas other applications have been discarded due to higher requirements. Particularly, the incorporation of recycled high-density polyethylene (HDPE) into PE pipe grade resins is a great challenge that is not currently used in the pipe industry. In this sense, very little research has been conducted in recent years. Poduška et al. (2019) evaluated the possibility of introducing recycled HDPE in PE pressure pipes combining layers of recycled HDPE with virgin HDPE layers. Recently, Juan et al. (2020) evaluated the potential use of post-consumer recycled HDPE from different sources in the manufacture of the entirely PE pressure pipes with the aim to enhance the circular economy by replacing part of the conventional HDPE raw material with recycled one for closing the loop on HDPE for pipe application.

Life cycle assessment (LCA), with its holistic perspective, is a tool that can prevent circular economy strategies from shifting environmental burdens between supply chain stages, between impact categories, and/or between countries (Peña et al., 2020). A number of studies have used LCA to compare the potential environmental impacts of commonly used materials for pipe applications, such as HDPE, polyvinyl chloride (PVC), low-density polyethylene (LDPE), cast iron, steel, copper, and fibre reinforced polyester (FRP); among others Hajibabaei et al. (2018) and Xiong et al. (2020). However, little attention has been paid to the environmental impacts of incorporating recycled plastic into pipes. For instance, Nguyen et al. (2020) found that a pipe grade resin made with post-consumer recycled HDPE blended with virgin HDPE generates on average 50 % fewer GHG emissions compared to virgin HDPE resin. The Plastics Pipe Institute (2021) showed that a corrugated pipe made with 50 % post-consumer recycled HDPE reduces the climate change impact by 13 %, the ozone layer depletion by 8 %, and the impact on acidification by 6 % compared to a virgin HDPE pipe. All in all, the existing literature suggests environmental benefits related to the incorporation of recycled HDPE into pipes. Nevertheless, there is still a lack of comprehensive LCA studies, especially on assessing recycled HDPE from different sources, such as households and/or industry.

The incorporation of recycled materials into new products gives rise to a key LCA methodological aspect and large source of discrepancy, namely the handling of the impacts associated with the recycled material (Nessi et al., 2020). Typically, the recycled material does not carry environmental burdens from its previous life cycle. Furthermore, the impact of collection and recycling is fully allocated to the recycled material (Schrijvers et al., 2016). This approach, often called cut-off, has been criticised for its inefficiency on fostering recycling since the life cycle that provides the waste feedstock cannot claim any recycling credits (Koffler and Finkbeiner, 2018). An alternative solution is the Circular Footprint Formula, a standardised approach, developed in the frame of the Product Environmental Footprint initiative, to allocate impacts and benefits from materials through multiple life cycle stages (European Commission, 2018b). So far, the Circular Footprint Formula has been applied in a limited number of practical applications related to plastic recycling; e.g. see Tonini et al. (2021).

The goal of this study is to quantitatively evaluate the potential environmental life cycle impacts in the production of PE pipe grade resins from recycled HDPE blended with a virgin HDPE pipe grade. The LCA involves four HDPE plastic waste feedstocks (crates/caps, packaging/detergency bottles, post-consumer industrial containers, and automobile fuel tanks) and two HDPE pipe grade qualities (PE80 and PE100). The environmental performance of the evaluated resins was compared against that of a 100 % virgin HDPE resin in order to verify their potential lower environmental burdens. The scientific significance of this work can be summarised in the following three points. First, the study involves a range of plastic waste feedstocks from specific sources. This is a novelty compared with previous studies, which assessed a generic post-consumer recycled HDPE flow. Second, the technical-based recycled contents used in the present LCA derive from experimental work carried out by some of the authors (Juan et al., 2020). As far as we are aware, little work has been carried out in using practical substitution ratios within the plastic LCA literature (Rigamonti et al., 2020; Viau et al., 2020). Third, we investigate different allocation approaches that affect the LCA of plastic recycling, namely the cut-off approach and the Circular Footprint Formula. This is an increasing relevant discussion in the transition to a circular economy; see Nessi et al. (2020) and Tonini et al. (2021).

2. Material and methods

2.1. Scope and functional unit

The function of the system under analysis is to produce PE pipe grade resins from recycled HDPE blended with virgin HDPE. The functional unit of the analysis was defined as 1 kg PE pipe grade resin. The recycled HDPE has been obtained from the following four waste feedstocks: (I) injection products such as crates and caps (WF1), (II) blow moulding goods such as packaging and detergency bottles (WF2), (III) post-consumer industrial containers (WF3), and (IV) automobile fuel tanks (WF4). Crates/caps and packaging/detergency bottles are typically contained in the packaging waste stream collected separately from households. Consequently, these two options represent waste feedstocks with a certain level of impurities obtained from mixed waste streams. In contrast, post-consumer industrial containers and automobile fuel tanks can be obtained from sources (e.g. industry and car scrapping) as very pure waste feedstocks.

Each of the four waste feedstocks has been evaluated for the production of both PE80 and PE100 pipe grade qualities, resulting in a total of eight scenarios for PE pipe grade resin production. Table 1 describes the scenarios assessed along with the recycled content interval allowed for assuring the functionality of each grade quality (Juan et al., 2020). The recycled contents are based on experimental work carried out by the authors. During the experimental work, each recyclate was blended with raw pipe grade resin at different recycled contents, in order to evaluate their feasibility in the manufacture of pressure pipes. Results obtained show that blending process was effective for all systems analysed at any recycled contents. Molecular and mechanical characterisation was performed to determine the operational interval for all the blends studied. Short-term mechanical properties such as yield strength and flexural modulus for all blend systems guarantee the minimum requirements for PE pipe grades, following a linear trend, as observed in other works with the incorporation of recycled PE (Cecon et al., 2021). Moreover, special

Table 1
Scenarios for polyethylene (PE) pipe grade resin production assessed in the current study. The upper values of the recycled content intervals were used as baseline values in the current study.

Scenario	Grade quality	Waste feedstock	Recycled content (%)
PE80-WF1	PE80	Crates/caps	5–23
PE80-WF2	PE80	Packaging/detergency bottles	9–37
PE80-WF3	PE80	Post-consumer industrial containers	25–68
PE80-WF4	PE80	Automobile fuel tanks	31–96
PE100-WF1	PE100	Crates/caps	0–5
PE100-WF2	PE100	Packaging/detergency bottles	0–9
PE100-WF3	PE100	Post-consumer industrial containers	0–25
PE100-WF4	PE100	Automobile fuel tanks	0–31

attention was paid to two key properties required in pipe applications: Slow Crack Growth (SCG) and Rapid Crack Propagation (RCP). Both properties were sensitive to the recycled content and their behaviour with the increasing content in recycled material is far from the linearity obtained previously for other properties. These results were used to establish the maximum recycling content for each pipe grade. The upper values reported in Table 1 were used as baseline recycled contents, while the influence of the interval was assessed through sensitivity analysis (Section 3.3).

Common PE pipe grade includes PE80 and PE100. Both designations are based on the long-term strength of the materials, named as the minimum required strength (MRS), in accordance with ISO 12162, being 8.0 MPa for PE80 and 10.0 MPa in the case of PE100. Both pipe grades are made from HDPE, being the name and type of material used for reference. The most important factors are the strength of the material together with the Standard Dimensional Ratio (SDR), which are used to determine the pressure and pipe required for the application. It should be noted that, in general, PE80 and PE100 grades can be used with the same functionality for certain applications, such as water distribution with pressures up to 16 bar, sewers, outfall pipes, industrial pipes and also gas pipe for natural gas distribution network with pressures up to 4 bars. However, other applications with higher pressure requirements, where PE80 grade cannot fulfil the same functionality than PE100 grade, are demanded.

The LCA follows an attributional approach, meaning that average market data for the European context has been used to model the life cycle inventory (LCI). In general, the data used to model the foreground

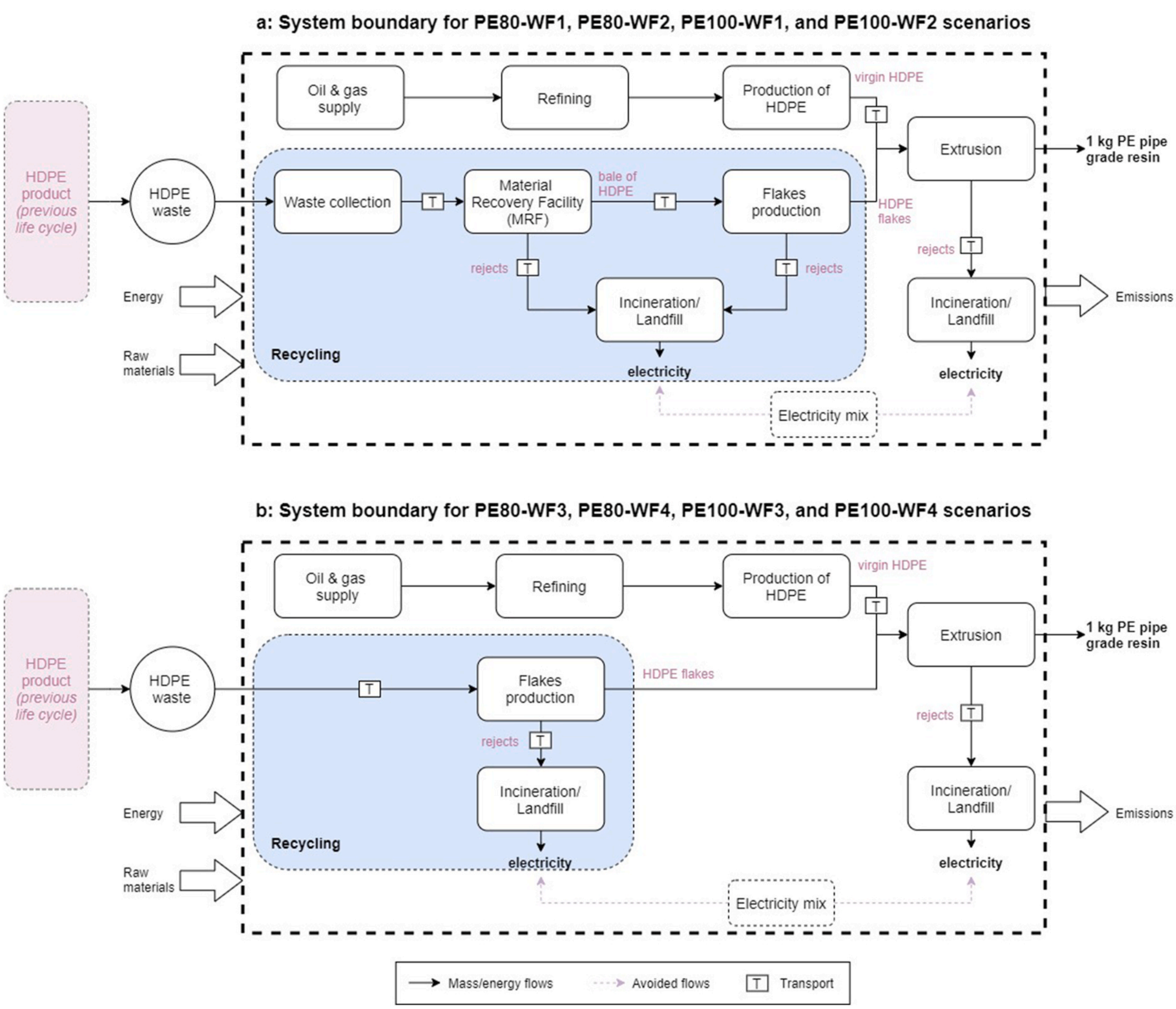

Fig. 1. System boundaries for the production of polyethylene (PE) pipe grade resins from recycled high-density polyethylene (HDPE) blended with virgin HDPE. (a) Scenarios of resin production from crates/caps (PE80-WF1 and PE100-WF1) and packaging/detergency bottles (PE80-WF2 and PE100-WF2). (b) Scenarios of resin production from post-consumer industrial containers (PE80-WF3 and PE100-WF3) and automobile fuel tanks (PE80-WF4 and PE100-WF4). The description of the scenarios can be found in Table 1.

processes, i.e. the processes directly involved in the production of the PE pipe grade resin, were retrieved from the literature. Data for the background processes, i.e. energy and raw materials supply, were taken from the Ecoinvent database v3.7.1 using the cut-off system model (Wernet et al., 2016). The list of Ecoinvent datasets used is summarised in Table S1 in the Supplementary material. The electricity required within the foreground system has been regionalised by using the 2020 Spanish electricity mix composed by 28 % natural gas, 22 % wind, 22 % nuclear, 13 % hydropower, 6 % photovoltaic, 4 % other renewables, 2 % coal, and 3 % others (Ministerio para la Transición Ecológica y el Reto Demográfico, 2020). The influence of this choice was addressed by sensitivity analysis (Section 3.3).

2.2. System boundaries

Fig. 1 shows the system boundaries defined following a “cradle-to-factory gate” approach. Fig. 1a schematically illustrates the system boundaries for the scenarios of PE pipe grade resin production from crates/caps (PE80-WF1 and PE100-WF1) and packaging/detergency bottles (PE80-WF2 and PE100-WF2). The system boundaries encompass the following life cycle stages: (I) collection of waste from households, (II) sorting of waste at a material recovery facility (MRF) and distribution of HDPE bales, (III) mechanical recycling into HDPE flakes, (IV) production and distribution of virgin HDPE, (V) blending of recycled and virgin materials in an extruder to produce pipe grade resin, and (VI) management of rejects through incineration with electricity production and landfilling. Similarly, Fig. 1b schematically illustrates the system boundaries for the scenarios of PE pipe grade resin production from post-consumer industrial containers (PE80-WF3 and PE100-WF3) and automobile fuel tanks (PE80-WF4 and PE100-WF4). Given the high purity of these waste feedstocks, the MRF stage is not herein considered being the waste feedstocks collected and directly recycled into HDPE flakes. The other life cycle stages remain the same. Overall, the environmental burdens related to infrastructure have been included as far as possible, whereas the stages of pipe manufacture, distribution, installation, use, and end-of-life were assumed the same for all scenarios and therefore excluded.

2.3. Allocation issues

Two allocation issues arise in the system under analysis. The first one is related to multifunctionality. In addition to producing PE pipe grade resins, the system under analysis also produces electricity through the incineration of the rejects (Fig. 1). This multifunctionality was solved by means of substitution. In consequence, the electricity fed into the grid was assumed to substitute an equivalent amount of electricity generated by the Spanish mix, and the PE pipe grade resin system is credited for the avoided environmental burdens (Fig. 1). Substitution is a valid approach to solve multifunctionality in attributional LCA when the system under study interacts with other systems and no large-scale consequences are expected (European Commission, 2010, 2012, 2010; Laurent et al., 2014).

The second allocation issue is related to handling the impacts associated with the recycled HDPE. In the current study two approaches have been considered, namely the cut-off approach and the Circular Footprint Formula. In the cut-off approach, the recycled HDPE does not bear burdens from the previous life cycle but only the impacts of recycling. In the second approach, the Circular Footprint Formula is applied (European Commission, 2018b), according to which the LCI per functional unit (EI) is calculated as follows:

$$EI = R_1 \cdot \left[A \cdot E_R + (1-A) \cdot E_V^{\circ} \cdot \frac{Q^o}{Q_v^o}\right] + (1-R_1) \cdot E_V + E_{EXTR} \quad (1)$$

where:

- R_1 is the recycled content per functional unit;
- A is a factor used for allocating the burdens and benefits (i.e., credits) between supplier and user of recycled materials;
- E_R is the LCI of recycling per functional unit;
- E_V° is the LCI of the virgin HDPE used in the original product per functional unit (the original products are crates/caps, packaging/detergency bottles, post-consumer industrial containers, automobile fuel tanks);
- Q^o is the quality of the HDPE flakes;
- Q_v^o is the quality of the virgin HDPE used in the original product;
- E_V is the LCI of the virgin HDPE incorporated into the pipe grade resin per functional unit;
- E_{EXTR} is the LCI of the extrusion stage per functional unit (including the management of the extrusion rejects and the substitution of electricity)

The LCI of recycling is calculated as follows:

$$E_R = E_{CT} + E_{MRF} + E_{FLK} + E_{REJ} - C_{ER} \quad (2)$$

where:

- E_{CT} is the LCI of waste collection and transportation per functional unit;
- E_{MRF} is the LCI of the MRF per functional unit;
- E_{FLK} is the LCI of HDPE flakes production per functional unit;
- E_{REJ} is the LCI of the management of flake production rejects per functional unit;
- C_{ER} is the LCI of the substituted electricity per functional unit.

Note that the end-of-life component in the Circular Footprint Formula is excluded since the management of the pipe at its end-of-life is not considered in this work. The value of the parameter A was assumed equal to 0.5, which is the default value for plastic materials (European Commission, 2018b). This implies that the recycled HDPE bears 50 % of the LCI of recycling and 50 % of the LCI of the virgin material used in the product that supplies the waste feedstock. The quality ratio between the HDPE flakes and the virgin HDPE (Q^o/Q_v^o) was assumed 0.9 for the flakes produced from packaging waste feedstocks (WF1 and WF2) and 1 for the flakes produced from pure waste feedstocks (WF3 and WF4). The 0.9 is the recommended value for packaging HDPE waste (European Commission, 2018b). On the other hand, it is assumed that the purity and quality of the HDPE flakes produced from WF3 and WF4 is similar to that of the virgin HDPE (i.e. quality ratio of 1).

2.4. Life cycle inventory

Table 2 summarises the main parameters used to model the LCI, their baseline values, and the assumed distributions used in the uncertainty analysis (Section 2.5). The following paragraphs describe the main assumptions behind the LCI.

2.4.1. Waste feedstock collection/transportation

Transport distances were obtained from the literature for the European context (Table 2). The collection of crates/caps (WF1) and packaging/detergency bottles (WF2) is assumed to be carried out by a 21 t diesel truck for an average distance of 30 km. The bales of HDPE obtained at MRF as well as the WF3 and WF4 are assumed to be transported for 300 km by a 32 t diesel Euro 6 truck to the mechanical recycling facility.

2.4.2. Material recovery facility

The MRF sorts the packaging waste stream into a number of output streams which consist of a single material type (e.g. PET, HDPE, LDPE, and mix plastic). The sorted materials are then baled and sold to recyclers. In the current study the bales of HDPE consist of crates/caps

Table 2
Parameters, baseline values, and ranges used to model the life cycle inventory of the production of polyethylene (PE) pipe grade resin from recycled high-density polyethylene (HDPE) blended with virgin HDPE. The value of some parameters is specific for the type of waste feedstock (WF) used. WF1: crates/caps; WF2: packaging/detergency bottles; WF3: post-consumer industrial containers; WF4: automobile fuel tanks. LHV: lower heating value.

Parameter	Units	Baseline	Distribution	Reference(s)
Waste feedstock collection/transportation				
Collection distance (WF1 and WF2)	Km	30	Uniform (14,55)	Belboom et al. (2013); Giugliano et al. (2011); Iriarte et al. (2009); Jeswani et al. (2021); Lausselet et al. (2016); Turner et al. (2016)
Distribution distance (bales of HDPE, WF3, and WF4)	Km	300	Uniform (50,640)	Abejón et al. (2020); Andreasi Bassi et al., 2020, 2017; Ardolino et al. (2017); Bovea et al. (2010); Cimpan et al. (2015b); Faraca et al. (2019); Giugliano et al. (2011); Jeswani et al. (2021); Rigamonti et al. (2014); Shen et al. (2010); Tascione et al. (2016); Tonini et al. (2021); Turner et al. (2016)
Material recovery facility (MRF)				
Sorting efficiency of HDPE	%	77.5	Triangular (57,98,77.5)	Bishop et al. (2020)
Purity of HDPE bales	%	92.8	Triangular (87.4,96.4,92.8)	Cimpan et al. (2015a)
Electricity	kWh/kg bale	0.07	Triangular (0.03,0.21,0.07)	Andreasi Bassi et al., 2020
Diesel	MJ/kg bale	1.12E-04	Triangular (6.51E-05,5.16E-04,1.12E-04)	Andreasi Bassi et al., 2020
Steel wire	g/kg bale	4.72	Triangular (3.87,7.23,4.72)	Andreasi Bassi et al., 2020
Waste preparation facility	unit/kg bale	2.21E-09	Lognormal (2.21E-09,0.59)	Ecoinvent database v3.7.1
Flakes production				
Yield (bale of HDPE)	%	88	Triangular (64.1,93.9,88)	Andreasi Bassi et al., 2020
Yield (WF3 and WF4)	%	95	Triangular (90,100,95)	Authors' assumption
Wastewater	kg/kg feedstock	2.60	Triangular (1.25,5.25,2.60)	Equal to input water
Particulate matter <2.5 um	kg/kg feedstock	1.97E-05	Lognormal (1.97E-05,0.25)	Haupt et al. (2018)
Particulate matter, 2.5–10 um	kg/kg feedstock	1.28E-05	Lognormal (1.28E-05,0.25)	Haupt et al. (2018)
Electricity	kWh/kg feedstock	0.56	Triangular (0.18,0.76,0.56)	Andreasi Bassi et al., 2020
Diesel	MJ/kg feedstock	0.04	Triangular (0.01,0.88,0.04)	Andreasi Bassi et al., 2020
Water	kg/kg feedstock	2.60	Triangular (1.25,5.25,2.60)	Andreasi Bassi et al., 2020
Sodium hydroxide	g/kg feedstock	0.23	Triangular (0.22,0.25,0.23)	Andreasi Bassi et al., 2020
Other chemicals, inorganic	g/kg feedstock	2.71	Triangular (2.35,3.41,2.71)	Andreasi Bassi et al., 2020
Waste preparation facility	unit/kg flake	2.00E-09	Lognormal (2.00E-09,0.59)	Ecoinvent database v3.7.1
Extrusion				
Yield	%	95		Authors' assumption
Water, air	m^3/kg resin	5.26E-03	Lognormal (5.26E-03,0.25)	Ecoinvent database v3.7.1
Water, water	m^3/kg resin	8.31E-02	Lognormal (8.31E-02,0.25)	Ecoinvent v3.7.1
Heat, natural gas	MJ/kg resin	0.08	Triangular (0.04,0.14,0.08)	Andreasi Bassi et al., 2020; Ecoinvent database v3.7.1
Heat, other than natural gas	MJ/kg resin	0.47	Triangular (0.20,0.80,0.47)	Andreasi Bassi et al., 2020; Ecoinvent database v3.7.1
Carbon black (WF1, WF2, WF3)	kg/kg resin	0.023	Triangular (0.020,0.025,0.023)	Authors' assumption
Carbon black (WF4)	kg/kg resin	0.01	NA	
Water, natural origin	m^3/kg resin	1.36E-02	Lognormal (1.36E-02,0.35)	Ecoinvent database v3.7.1
Lubricating oil	kg/kg resin	1.42E-04	Lognormal (1.42E-04,0.35)	Ecoinvent database v3.7.1
Plastic film	kg/kg resin	3.55E-05	Triangular (1.62E-05,5.48E-05, 3.55E-05)	Andreasi Bassi et al., 2020
Sulfuric acid	kg/kg resin	9.66E-04	Triangular (9.66E-04,1.56E-02, 9.66E-04)	Andreasi Bassi et al., 2020
Steel wire	kg/kg resin	8.20E-07	Triangular (3.75E-07,1.26E-06,8.20E-07)	Andreasi Bassi et al., 2020
EUR pallet	unit/kg resin	1.42E-06	Triangular (6.58E-07,2.19E-06,1.42E-06)	Andreasi Bassi et al., 2020
Infrastructure	unit/kg resin	1.42E-09	Lognormal (1.42E-09,0.59)	Ecoinvent database v3.7.1
Management of rejects				
Distance to incineration/ landfilling	Km	50	Uniform (16,70)	Abejón et al. (2020); Andreasi Bassi et al., 2017; Bovea et al. (2010); Cimpan et al. (2015b); Jeswani et al. (2021)
Share of incineration	% mass	63	NA	Plastics Europe, 2019
Share of landfilling	% mass	37	NA	Plastics Europe, 2019
Net electrical efficiency of incineration	% of reject LHV	12	Uniform (12,0.94)	Istrate et al. (2021)
LHV of rejects from MRF	MJ/kg wet waste	37.72	Uniform (37.72,0.18)	Götze et al. (2016)
		33.96	Uniform (33.96,0.49)	Götze et al. (2016)

(continued on next page)

Table 2 (*continued*)

Parameter	Units	Baseline	Distribution	Reference(s)
LHV of rejects from flakes production	MJ/kg wet waste			
LHV of rejects from extrusion	MJ/kg wet waste	37.72	Uniform (37.72,0.18)	Götze et al. (2016)

(WF1) and packaging/detergency bottles (WF2), respectively. The baseline sorting efficiency of HDPE was assumed equal to 77.5 %, i.e. 22.5 % of the input HDPE is rejected (Bishop et al., 2020). The bale of HDPE has an average purity of 92.8 % (Cimpan et al., 2015a). The main inputs to the MRF are electricity and diesel for daily activities and steel wire for bale preparation (Table 2).

2.4.3. Flakes production

The mechanical recycling of HDPE into flakes involves standard activities such as sorting, shredding, and washing (Haupt et al., 2018). The process starts with sorting the input feedstock in order to remove any unwanted material. The sorted material is then shredded to produce flakes, which are washed and dried. The washing process often involves caustic wash with sodium hydroxide and other washing chemicals in order to remove paper labels and additives (Franklin Associates, 2011). The technical yield –mass of flakes produced per unit of input feedstock– was assumed 88 % for HDPE bales and 95 % for pure waste feedstocks (WF3 and WF4). Due to the higher purity of WF3 and WF4, much less material will be lost during the mechanical recycling. The consumption of electricity, diesel, water, and washing chemicals is summarised in Table 2.

2.4.4. Extrusion

Recycled HDPE flakes are blended in an extruder with virgin HDPE to produce a PE pipe grade resin. The ratio between the recycled and virgin materials is specific to each scenario (Table 1). The technical yield of extrusion –mass of resin produced per unit of input materials– was assumed equal to 95 %. The impacts of the virgin HDPE were obtained from the Ecoinvent dataset *"market for polyethylene, high density, granulate [GLO]"*. This dataset represents the global mix of HDPE production from oil and gas and includes average transport distances for its distribution. The main inputs into the extrusion stage, in addition to the recycled and virgin materials, are heat and carbon black (Table 2). Heat is required to achieve the extrusion temperature, which ranges from 185 to 240 °C (Juan et al., 2020). An average consumption of 0.177 MJ heat/kg resin was assumed (Andreasi Bassi et al., 2020) with 15 % of heat from natural gas and 85 % from other sources, according to the figures reported in the Ecoinvent dataset *"extrusion, plastic pipe"*. The impact of carbon black is also considered, as it is an important UV stabilizer used to prevent plastic pipe degradation. Plastic pipe standards for PE pipes establish that the content of carbon black is ranged between 2 and 2.5 wt% (ISO, 2012, 2010), therefore, an average of 2.3 wt% carbon black content has been considered for raw PE pipe resins. Furthermore, the inventory includes the consumption of plastic film, sulfuric acid, steel wire, and EUR pallet needed for packaging the resin (Table 2). Finally, the impacts per kg of a 100 % virgin resin have been modelled as 0.977 kg of virgin HDPE plus 0.023 kg of carbon black.

2.4.5. Management of rejects

The rejects generated by the MRF, flakes production, and extrusion are assumed to be 63 % incinerated with electricity production and 37 % disposed of in landfill (Plastics Europe, 2019). All the rejects are assumed to be transported 50 km to the final disposal site by a 32 t diesel Euro 6 truck. The impacts of incineration and landfilling were retrieved from the Ecoinvent database v3.7.1. The incineration of plastic rejects produces electricity following common practices in Spain. The net electrical efficiency of incineration was assumed 12 ± 0.94 % with respect to the lower heating value (LHV) of the input rejects (Istrate et al., 2021).

2.5. Uncertainty analysis

The uncertainty associated with the results due to parameter uncertainty was addressed by error propagation via Monte Carlo simulation with 1000 iterations. For each iteration, parameters were randomly sampled according to their probability distribution and the results were computed. The results produced by the Monte Carlo simulation have been statistically evaluated in order to provide a level of confidence behind conclusions. The probability distribution assigned to each parameter can be found in Table 2. The uncertainty associated with the background processes is taken directly from the Ecoinvent database v3.7.1.

2.6. Impact assessment

The LCA modelling has been carried out using Brightway2 (Mutel, 2017), an open source framework for LCA in Python, and the Activity Browser (Steubing et al., 2020), a graphical user interface to Brightway2. The life cycle impact assessment (LCIA) has been performed using the midpoint impact assessment methods recommended in the framework of the Environmental Footprint (EF) developed by the European Commission (Fazio et al., 2018). The EF method was recommended for plastic LCA due to its ability to cover all planetary boundaries (Bishop et al., 2021). Table S2 in the Supplementary material lists the impact categories included in this study, their units, and the recommended LCIA method.

3. Results and discussion

3.1. Mass balance

The mass balances per kg of PE pipe grade resin for the eight scenarios are summarised in Table 3. Depending on the scenario, producing 1 kg of PE80 resin requires between 0.04 and 0.79 kg virgin HDPE, while producing 1 kg of PE100 resin requires between 0.72 and 0.98 kg virgin HDPE. The PE80 resin produced from automobile fuel tanks (PE80-WF4) has by far the lowest demand of virgin HDPE (0.04 kg/kg resin) due to the high allowed recycled content (up to 96 %). The overall rate of rejects ranges from 8.28 % to 18.37 % for the PE80 resin and from 6.23 % to 8.64 % for the PE100 resin. The higher rate of rejects observed for the PE80 resin is primarily due to the higher allowed recycled content, which results in a higher mass of waste feedstock processed through the system. Overall, the production of PE resins from pure HDPE waste feedstocks (i.e. PE80-WF3, PE80-WF4, PE100-WF3, and PE100-WF4) achieves the lowest demand for virgin HDPE as well as the lowest rate of rejects. This represents a dual benefit for the stakeholders involved in the PE resin supply chain, since fewer resources are spent in the acquisition of virgin materials and the management of rejects by third parties, thus improving the material efficiency of the manufacturing process.

3.2. Life cycle impact assessment

Fig. 2 shows the LCIA of the production of PE80 and PE100 pipe grade resins from recycled HDPE blended with virgin HDPE according to the proposed scenarios. Impacts were calculated using the two

Table 3
Mass balance per kg of polyethylene (PE) pipe grade resin for the evaluated scenarios. The mass of waste feedstock includes weight of impurities in the incoming material. The description of the scenarios can be found in Table 1.

Flow	PE80-WF1	PE80-WF2	PE80-WF3	PE80-WF4	PE100-WF1	PE100-WF2	PE100-WF3	PE100-WF4
PE pipe grade resin (kg)	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Virgin HDPE (kg)	0.79	0.65	0.33	0.04	0.98	0.94	0.77	0.72
Recycled HDPE (kg)	0.24	0.38	0.70	1.00	0.05	0.09	0.26	0.32
Waste feedstock (kg)	0.34	0.55	0.74	1.05	0.07	0.13	0.27	0.34
Total rejects (kg)	0.16	0.22	0.09	0.10	0.07	0.09	0.06	0.07
Reject rate (%)	13.78	18.37	8.28	9.57	7.06	8.64	6.23	6.53

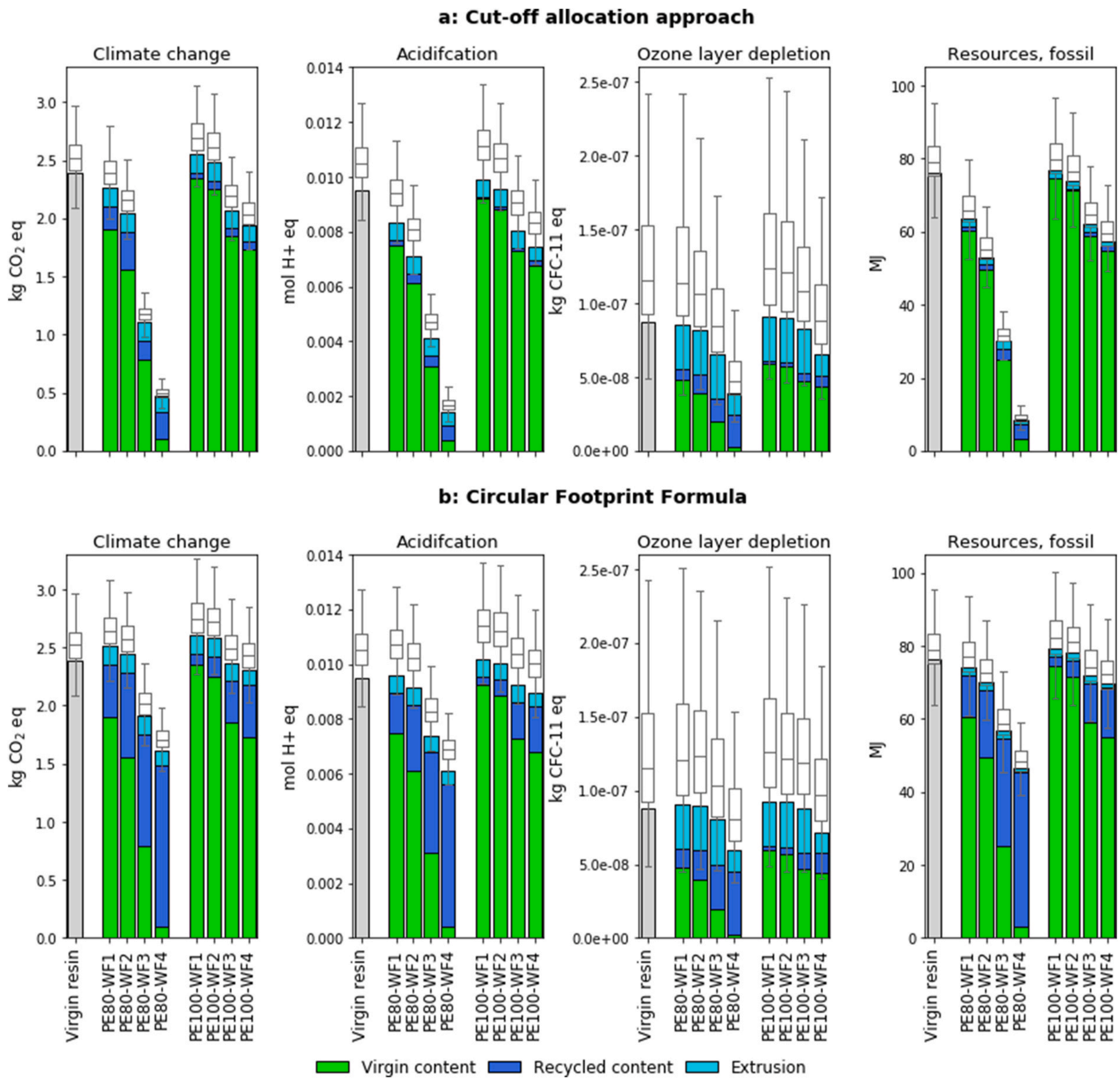

Fig. 2. Life cycle impact assessment of the eight scenarios for the production of 1 kg polyethylene (PE) pipe grade resin from recycled high-density polyethylene (HDPE) blended with virgin HDPE. Impacts were calculated using two allocation approaches: cut-off (a) and Circular Footprint Formula (b). The stacked bar chart shows the results based on the baseline parameters. The boxplots show the 25th percentile, median value, and 75th percentile values from the Monte Carlo simulation. The whiskers show the minimum and maximum values excluding outliers. The grey bar represents the impacts of a 100 % virgin resin (for benchmarking). The baseline results significantly depart from the median since parameters were generally modelled with skewed distributions. The description of the eight scenarios can be found in Table 1.

allocation approaches, namely cut-off (Fig. 2a) and Circular Footprint Formula (Fig. 2b). The stacked bar chart shows the contribution of the virgin content production, recycled content production, and extrusion to the total impacts, calculated with the baseline parameters. The boxplots represent the distribution of the impacts based on the Monte Carlo simulation. For benchmarking purposes, the grey bar represents the impacts of a 100 % virgin resin. It was observed that the impact categories generally follow a similar trend. Therefore, in order to streamline the discussion, the focus in this section is placed on the following four impact categories: climate change, acidification, ozone layer depletion, and fossil resources. Figures for other impact categories can be found in Section S3.1 in the Supplementary material.

3.2.1. Climate change impact

Under the cut-off allocation approach (Fig. 2a), the baseline climate change impact ranges from 0.47 to 2.27 kg CO_2 eq/kg for the PE80 resin and from 1.94 to 2.55 kg CO_2 eq/kg for the PE100 resin. The production of PE80 resin from post-consumer industrial containers (PE80-WF3) or automobile fuel tanks (PE80-WF4) achieve the lowest carbon footprint (0.47 and 1.11 kg CO_2 eq/kg, respectively). On the other side, the production of PE100 resin from crates/caps (PE100-WF1) or packaging/detergency bottles (PE100-WF2) achieve the highest carbon footprint (2.55 and 2.49 kg CO_2 eq/kg, respectively). These results are largely explained by the ratio between the virgin and recycled contents of the resin. The virgin content has a notable contribution to the impact, largely due to the production of ethylene. It should be noted that 1 kg of virgin HDPE generates 2.40 kg CO_2 eq, while 1 kg of recycled HDPE flakes generates between 0.86 kg CO_2 eq (WF1/WF2) and 0.23 kg CO_2 eq (WF3/WF4). The flakes produced from WF3/WF4 generate approximately 73 % less impact than those produced from WF1/WF2 due to the higher purity of the waste feedstocks, thus avoiding the need for additional sorting at MRF and reducing the mass of rejects to incineration. The environmental impacts associated with the recycled HDPE flakes are further elaborated in Section S3.2 in the Supplementary material.

One of the reasons for the lower impact observed for the resins with higher recycled content is that the waste feedstock used for recycled flakes production is considered burdens free under the cut-off allocation approach. In other words, the waste feedstock does not carry any burden from the extraction and processing of the raw materials used for its production. This assumption clearly fosters the use of the recycled material (i.e. the production of PE pipe grade), but it does not give any credit to the producer of the first life cycle product (Weidema, 2017). This can be seen as unfair by some stakeholders (Ekvall et al., 2020). When using the Circular Footprint Formula (Fig. 2b), the climate change impact increases up to 1.62–2.51 kg CO_2 eq/kg for the PE80 resin and up to 2.31–2.61 kg CO_2 eq/kg for the PE100 resin. As can be seen, the contribution of the recycled content to the impact increases notably –especially in the case of WF3 and WF4– as a consequence of bearing part of the burdens from the product that supplies the waste feedstock. Overall, the use of the Circular Footprint Formula instead of the cut-off approach increases the carbon footprint between 11 % and 246 % for the PE80 resin and between 2 % and 19 % for the PE100 resin. However, the allocation choice does not alter the ranking order of the scenarios under evaluation.

Compared to the 100 % virgin resin, all the evaluated scenarios but PE100-WF1 and PE100-WF2 result in lower carbon footprint under the cut-off allocation approach. Nevertheless, when using the Circular Footprint Formula, only the PE80 and PE100 resins produced from WF3 and WF4 show a lower carbon footprint compared to the virgin resin. The Monte Carlo simulation reveals that uncertainties are substantial (boxplots in Fig. 2). Despite the large spread of the results, some conclusions can be drawn with a high level of confidence. The PE80-WF3 and PE80-WF4 scenarios perform better than the virgin resin in more than 95 % of the Monte Carlo iterations regardless the followed allocation approach (Fig. 3). The PE80-WF2, PE100-WF3, and PE100-WF4 scenarios perform better than the virgin resin in more than 75 % of the iterations (the 75th percentile) under the cut-off approach. However, when using the Circular Footprint Formula these scenarios perform worst that the virgin resin in more than 75 % of the iterations. Furthermore, the PE80-WF1, PE100-WF1, and PE100-WF2 scenarios perform worse than the virgin resin in more than 75 % of the iterations regardless the followed allocation approach.

Considering the influence of allocation and parameter uncertainty, it can be concluded with strong confidence that the production of PE80 resin from pure waste feedstocks (PE80-WF3 and PE80-WF4) result in lower climate change impact compared to the 100 % virgin resin. Furthermore, it can be also concluded with strong confidence that the use of automobile tank fuel (WF4) outperforms the use of post-consumer industrial containers (WF3) as waste feedstock. The potential climate benefit of using more heterogeneous feedstocks, such as crates/caps and packaging/detergency bottles, is more uncertain. In these cases, alternative applications which allow plastic waste with less quality should be explored; however, this is out of the scope of the present study.

3.2.2. Acidification

The impact on acidification follows a similar trend to that of climate change (Fig. 2). The largest contributor to the impact is the virgin content due to the production of ethylene. Consequently, the lowest impact is achieved for the scenarios with the highest recycled content, i.e. PE80-WF3 and PE80-WF4. The allocation approach has the same

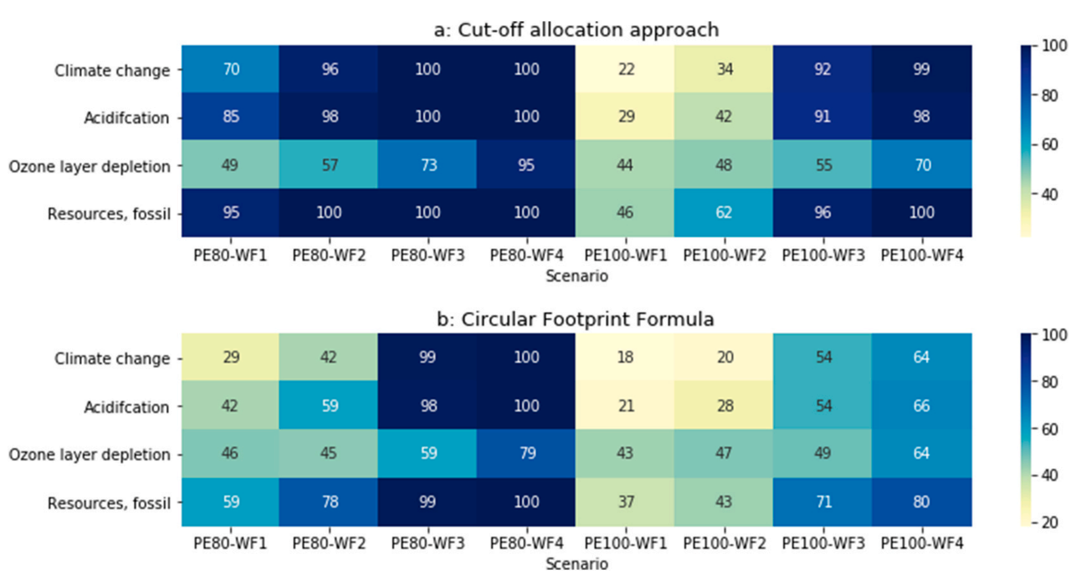

Fig. 3. Percentage of Monte Carlo iterations in which the eight scenarios for the production of 1 kg polyethylene (PE) pipe grade resin from recycled high-density polyethylene (HDPE) blended with virgin HDPE are environmentally preferred over 1 kg of 100 % virgin HDPE resin. Impacts were calculated using two allocation approaches: cut-off (a) and Circular Footprint Formula (b).

influence on the results as for climate change. Furthermore, the Monte Carlo simulation results reveal a similar trend as for climate change (Fig. 3). Therefore, it can be concluded with strong confidence that the PE80-WF3 and PE80-WF4 scenarios have a lower impact on acidification compared to the other evaluated scenarios and the 100 % virgin resin.

3.2.3. *Ozone layer depletion*

Ozone layer depletion shows a slightly different trend compared with climate change and acidification (Fig. 2). On the one hand, the extrusion stage has a significant contribution to the impact caused by the carbon black used as additive. On the other hand, the uncertainties associated with this impact category are significantly larger than that of climate change and acidification. Indeed, the large spread of the results does not allow to clearly concluding the preference of the proposed scenarios over the 100 % virgin resin. For example, the PE80-WF4 scenario performs better than the virgin resin in more than 95 % of the Monte Carlo runs iterations under the cut-off allocation approach, but in 79 % of the iterations under the Circular Footprint Formula. All the remaining scenarios generally perform worse than the virgin resin in 75 % or more of the Monte Carlo runs.

3.2.4. *Resources, fossil*

The depletion of fossil resources reflects the trend observed for climate change and acidification, where higher recycled content results in lower impact. The baseline results under the cut-off allocation approach range from 8 to 63 MJ/kg for the PE80 resin and from 57 to 77 MJ/kg for the PE100 resin (Fig. 2a). When using the Circular Footprint Formula, these figures increase up to 47–74 MJ/kg for the PE80 resin and up to 70–79 MJ/kg for the PE100 resin (Fig. 2b). The virgin content bears the largest burden, mainly due to the energy consumed for ethylene production. In this regard, 1 kg of virgin HDPE leads to the depletion of 76 MJ fossil resources, while 1 kg of recycled HDPE flakes leads to the depletion of about 3.97 MJ (WF1/WF2) and 4.14 MJ (WF3/WF4). Considering the influence of allocation and parameter uncertainty, it can be concluded with strong confidence that the PE80-WF3 and PE80-WF4 scenarios result in a lower depletion of fossil resources compared to the 100 % virgin resin. Furthermore, the PE80-WF2 and PE100-WF4 scenarios result in a lower impact than the virgin resin in more than 75 % of the Monte Carlo iterations regardless the followed allocation approach.

3.2.5. *Other impact categories*

The trends observed for climate change, acidification, and fossil resources depletion are repeated for most of the other non-toxic impact categories, such as freshwater eutrophication, marine eutrophication, terrestrial eutrophication, photochemical ozone formation, respiratory inorganics, dissipated water, land use, and depletion of mineral and metal resources. For these categories, the PE80-WF3 and PE80-WF4 scenarios generally result in the lowest impact compared with both

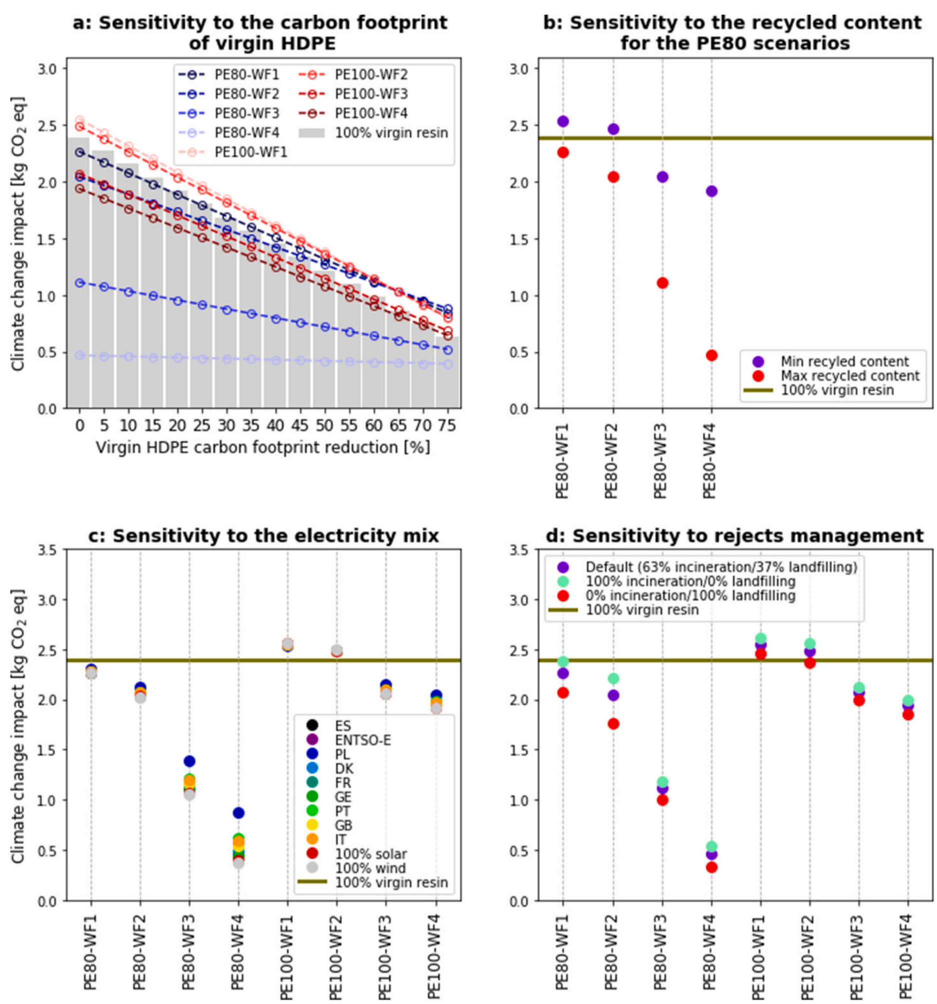

Fig. 4. Sensitivity of the carbon footprint of the proposed scenarios to four key assumptions that particularly influence the LCA results: (a) the carbon footprint of virgin high-density polyethylene, (b) the allowed share of recycled content, (c) the electricity mix used within the foreground system, and (d) the management of rejects. Results are based on the baseline parameters using the cut-off allocation approach.

the remaining proposed scenarios and the 100 % virgin resin. The exception is the impact on ionising radiation. For this category, the impact of the scenarios is directly proportional to their allowed recycled content. This trend is caused by the electricity consumed for recycled HDPE flakes production. The rationale behind this fact is the selected Spanish electricity mix, which share of nuclear is 22 %. Finally, the toxic impact categories, such as freshwater ecotoxicity, human toxicity with carcinogenic effects, and human toxicity with non-carcinogenic effects, also reflect the trend observed so far (Fig. S3 and Fig. S4 in the Supplementary material). However, the uncertainty associated with those categories is substantial and hamper any robust conclusion (Fig. S7 in the Supplementary material).

3.3. Sensitivity analysis

Fig. 4 shows the sensitivity of the carbon footprint of the proposed scenarios to four key assumptions that particularly influence the LCA results: (I) the carbon footprint of virgin HDPE (Fig. 4a), (II) the allowed share of recycled content in the PE80 pipe grade resins (Fig. 4b), (III) the electricity mix used within the foreground system (Fig. 4c), and (IV) the management of rejects (Fig. 4d).

The impacts of the virgin HDPE were retrieved from the Ecoinvent database v3.7.1, representing the current context for plastic production. However, the transition towards renewable energies can significantly reduce the carbon footprint of plastics (Zheng and Suh, 2019). For example, Daniel Posen et al. (2017) found that the life cycle climate change impact of U.S. plastics could be reduced by 50–75 % under a 100 % renewable energy scenario. In this regard, Fig. 4a shows the sensitivity of the results to a reduction in the carbon footprint of virgin HDPE. As can be seen, the difference between the eight scenarios narrows as the carbon footprint of the virgin HDPE is reduced. The PE80-WF3 and PE80-WF4 scenarios outperform the 100 % virgin resin even if the carbon footprint of virgin HDPE decreases by 75 %. In contrast, the remaining scenarios perform worse than the 100 % virgin resin after certain point. For instance, the PE80-WF1 and PE80-WF2 scenarios are outperformed by the 100 % virgin resin after a 30 % and 45 % reduction in the carbon footprint of virgin HDPE. Since the transition to an increasingly renewable energy system is a fact, these results suggest that the incorporation of recycled HDPE into new products could deliver the higher climate benefits in the short-to medium-term. In this sense, policies are currently needed to encourage the use of recycled plastics while progressing in the decarbonisation of the economy.

Fig. 4b shows the carbon footprint of the PE80 scenarios for the minimum and maximum recycled content allowed in order to meet the grade quality (see Table 1). The recycled content especially affects the carbon footprint of PE80-WF3 and PE80-WF4 scenarios, due to the wider interval allowed for these resins. However, these scenarios present a more favourable carbon footprint than the 100 % virgin resin even under the minimum recycled content. On the other hand, the PE80-WF1 and PE80-WF2 scenarios outperform the 100 % virgin resin under the maximum recycled content, but not under their minimum recycled content. It should be noted that the PE100 scenarios have not been assessed since the minimum recycled content is 0 % and the maximum recycled content was used as baseline value.

The sensitivity to the impact of the electricity used within the foreground system has been tested by evaluating some of the national electricity mixes available in the Ecoinvent database 3.7.1 as well as 100 % electricity from solar photovoltaics and wind turbines (Fig. 4c). As can be seen, the results are not particularly affected by the impact of the electricity consumed by the processes directly involved in the production of the recycled blends. The carbon footprints slightly increase only when assuming the electricity mix of Poland due to the higher share of coal. In this sense, the advance towards national renewable electricity mixes at a European and global level will allow reducing this slight variability.

Finally, Fig. 4d shows the sensitivity of the carbon footprint to different scenarios for the management of rejects. The management of the rejects was found to have a significant contribution to the impacts of recycled HDPE flakes, largely due to the incineration of plastic rejects (see Section S3.2 in the Supplementary material). In this regard, the environmental credits due to the avoided electricity were found negligible in comparison with the emissions generated by incineration. Therefore, two additional scenarios have been assessed in addition to the default scenario (based on 63 % incineration and 37 % landfilling), one based on 100 % incineration with electricity production and another based on 100 % landfilling. Overall, the management of rejects has a negligible influence on the results for most of the scenarios but for PE80-WF1 and PE80-WF2. These two scenarios achieve the highest rate of rejects and therefore the management of rejects acquires greater importance. In this regard, avoiding the incineration of rejects in favour of landfilling can further reduce the carbon footprint in these scenarios. However, increasing the landfilling of plastic rejects can exacerbate other environmental issues such as microplastic pollution.

4. Conclusions

This study quantitatively evaluated the potential environmental life cycle impacts of producing polyethylene (PE) pipe grade resins from recycled high-density polyethylene (HDPE) blended with virgin HDPE. The life cycle assessment (LCA) involved four HDPE waste feedstocks (crates/caps, packaging/detergency bottles, post-consumer industrial containers, and automobile fuel tanks) and two pipe grades (PE80 and PE100). Taking into account the influence of key choices and parameter uncertainty, it can be concluded with strong confidence that the production of PE80 resin from automobile fuel tanks or post-consumer industrial containers results in lower environmental impacts compared to the remaining scenarios and a 100 % virgin resin. PE80 grades allow a higher recycled content compared with PE100 grades, thus reducing the demand for virgin HDPE and the associated impacts. Moreover, the use of homogeneous waste feedstocks, such as automobile fuel tanks and post-consumer industrial containers, reduces the impact associated with the supply of recycled HDPE. The lower environmental impacts obtained for PE80 grades containing recycled HDPE could justify its use over PE100 grades, especially for applications with lower requirements like gravity sewer pipelines. Besides, the good results obtained for automobile fuel tanks and post-consumer industrial containers open the window to combine both feedstocks to obtain a single recycled HDPE. This could be important in the future, when the demand of recycled resins increases, and the quality of plastic recovered from waste becomes pivotal for incorporating these recyclates into more restrictive industries such as plastic pipe manufacturing. Overall this study may constitute a starting point for those stakeholders who want to environmentally verify this type of products, increasing the confidence and competitiveness of strategies based on the incorporation of plastic waste to virgin material.

Regarding methodological issues, this study delved into two key aspects that particularly influence the LCA of products that incorporate recycled plastic, namely the recycled content and the allocation approach. The recycled content was found to largely determine the environmental performance of PE pipe grade resin made from recycled HDPE. The allowed recycled contents used in this study were derived from experimental work and vary from 5 % up to 96 %, depending on the waste feedstock and the grade quality. This variation results in an average variation of 46 % in the LCA results. Thus, in order to reduce the uncertainty associated with the potential environmental benefits of incorporating recycled plastic, it is recommended to go in-depth into the allowed recycled content related to both the waste feedstock used and the characteristics of the final product. Overall, the incorporation of recycled plastic produced from heterogeneous waste feedstocks, such as household packaging waste, generally allows lower recycled contents. The influence of the allocation approach to handle the impacts associated with the recycled plastic was illustrated by applying both the cut-off approach and the Circular Footprint Formula. The Circular

Footprint Formula tends to increase the environmental impact of the product that incorporates recycled plastic as a consequence of bearing part of the burdens from the product that supplies the recycled plastic. In the current study, the allocation approach followed does not alter the ranking order of the resin scenarios, but seriously compromise the comparison with the 100 % virgin resin.

CRediT authorship contribution statement

Ioan-Robert Istrate: Conceptualisation, Methodology, Formal analysis, Investigation, Writing – original draft, Visualisation. **Rafael Juan:** Investigation, Writing – original draft. **Mario Martin-Gamboa:** Conceptualisation, Methodology, Formal analysis, Writing – review & editing, Supervision. **Carlos Domínguez:** Writing – review & editing, Supervision. **Rafael A. García-Muñoz:** Conceptualisation, Writing – review & editing, Supervision. **Javier Dufour:** Methodology, Writing – review & editing, Supervision.

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.

Acknowledgment

This research has been supported by the Spanish Ministry of Science, Innovation and Universities through the project REDEFINERY (RTI 2018-097227-B-I00). Additionally, Dr. Martín-Gamboa would like to thank the Regional Government of Madrid for financial support (2019-T2/AMB-15713).

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jclepro.2021.128580.

References

Abejón, R., Laso, J., Margallo, M., Aldaco, R., Blanca-Alcubilla, G., Bala, A., Fullana-i-Palmer, P., 2020. Environmental impact assessment of the implementation of a Deposit-Refund System for packaging waste in Spain: a solution or an additional problem? Sci. Total Environ. 721, 137744 https://doi.org/10.1016/j.scitotenv.2020.137744.

Andreasi Bassi, S., Boldrin, A., Faraca, G., Astrup, T.F., 2020. Extended producer responsibility: how to unlock the environmental and economic potential of plastic packaging waste? Resour. Conserv. Recycl. 162, 105030 https://doi.org/10.1016/j.resconrec.2020.105030.

Andreasi Bassi, S., Christensen, T.H., Damgaard, A., 2017. Environmental performance of household waste management in Europe - an example of 7 countries. Waste Manag. 69, 545–557. https://doi.org/10.1016/j.wasman.2017.07.042.

Ardolino, F., Berto, C., Arena, U., 2017. Environmental performances of different configurations of a material recovery facility in a life cycle perspective. Waste Manag. 68, 662–676. https://doi.org/10.1016/j.wasman.2017.05.039.

Belboom, S., Digneffe, J., Renzoni, R., Germain, A., Léonard, A., 2013. Comparing technologies for municipal solid waste management using life cycle assessment methodology : a Belgian case study. Int. J. Life Cycle Assess. 18, 1513–1523. https://doi.org/10.1007/s11367-013-0603-3.

Bishop, G., Styles, D., Lens, P.N.L., 2021. Environmental performance comparison of bioplastics and petrochemical plastics : a review of life cycle assessment (LCA) methodological decisions. Resour. Conserv. Recycl. 168, 105451 https://doi.org/10.1016/j.resconrec.2021.105451.

Bishop, G., Styles, D., Lens, P.N.L., 2020. Recycling of European plastic is a pathway for plastic debris in the ocean. Environ. Int. 142, 105893 https://doi.org/10.1016/j.envint.2020.105893.

Bovea, M.D., Ibáñez-Forés, V., Gallardo, A., Colomer-Mendoza, F.J., 2010. Environmental assessment of alternative municipal solid waste management strategies. A Spanish case study. Waste Manag. 30, 2383–2395. https://doi.org/10.1016/j.wasman.2010.03.001.

Cecon, V.S., Da Silva, P.F., Vorst, K.L., Curtzwiler, G.W., 2021. The effect of post-consumer recycled polyethylene (PCRPE) on the properties of polyethylene blends of different densities. Polym. Degrad. Stabil. 190, 109627 https://doi.org/10.1016/j.polymdegradstab.2021.109627.

Cimpan, C., Maul, A., Jansen, M., Pretz, T., Wenzel, H., 2015a. Central sorting and recovery of MSW recyclable materials: a review of technological state-of-the-art, cases, practice and implications for materials recycling. J. Environ. Manag. 156, 181–199. https://doi.org/10.1016/j.jenvman.2015.03.025.

Cimpan, C., Rothmann, M., Hamelin, L., Wenzel, H., 2015b. Towards increased recycling of household waste : documenting cascading effects and material ef fi ciency of commingled recyclables and biowaste collection. J. Environ. Manag. 157, 69–83. https://doi.org/10.1016/j.jenvman.2015.04.008.

Daniel Posen, I., Jaramillo, P., Landis, A.E., Michael Griffin, W., 2017. Greenhouse gas mitigation for U.S. plastics production: energy first, feedstocks later. Environ. Res. Lett. 12, 034024 https://doi.org/10.1088/1748-9326/aa60a7.

Di, J., Reck, B.K., Miatto, A., Graedel, T.E., 2021. United States plastics: large flows, short lifetimes, and negligible recycling. Resour. Conserv. Recycl. 167, 105440 https://doi.org/10.1016/j.resconrec.2021.105440.

Ekvall, T., Björklund, A., Sandin, G., Jelse, K., Lagergren, J., Rydberg, M., 2020. Modeling Recycling in Life Cycle Assessment. Swedish Life Cycle Center, Gothenburg.

Ellen MacArthur Foundation, 2017. The new plastics economy: rethinking the future of plastics [WWW Document]. URL. http://www.ellenmacarthurfoundation.org/publications.

European Commission, 2018a. A European Strategy for Plastics in a Circular Economy, COM(2018) 28 Final. Brussels.

European Commission, 2018b. PEFCR Guidance Document - Guidance for the Development of Product Environmental Footprint Category Rules (PEFCRs), Version 6.3.

European Commission, 2012. Life Cycle Indicators for Waste Management: Development of Life Cycle Based Macro-Level Monitoring Indicators for Resources, Products and Waste for the EU-27. European Commission, Joint Research Centre, Institute for Environment and Sustainability.

European Commission, 2010. International Reference Life Cycle Data System (ILCD) Handbook - General Guide for Life Cycle Assessment - Detailed Guidance. Publications Office of the European Union, Luxembourg.

Faraca, G., Martinez-Sanchez, V., Astrup, T.F., 2019. Environmental life cycle cost assessment: recycling of hard plastic waste collected at Danish recycling centres. Resour. Conserv. Recycl. 143, 299–309. https://doi.org/10.1016/j.resconrec.2019.01.014.

Fazio, S., Castellani, S., Sala, V., Schau, S., Secchi, E., Zampori, M., 2018. Supporting Information to the Characterisation Factors of Recommended EF Life Cycle Impact Assessment Method. European Commission, Ispra. https://doi.org/10.2760/671368.

Franklin Associates, 2011. Life Cycle Inventory of 100% Postconsumer HDPE and PET Recyled Resin from Postconsumer Containers and Packaging.

Geyer, R., Jambeck, J.R., Law, K.L., 2017. Production, use, and fate of all plastics ever made. Sci. Adv. 3, 19–24. https://doi.org/10.1126/sciadv.1700782.

Giugliano, M., Cernuschi, S., Grosso, M., Rigamonti, L., 2011. Material and energy recovery in integrated waste management systems. An evaluation based on life cycle assessment. Waste Manag. 31, 2092–2101. https://doi.org/10.1016/j.wasman.2011.02.029.

Götze, R., Pivnenko, K., Boldrin, A., Scheutz, C., Astrup, T.F., 2016. Physico-chemical characterisation of material fractions in residual and source-segregated household waste in Denmark. Waste Manag. 54, 13–26. https://doi.org/10.1016/j.wasman.2016.05.009.

Grigore, M.E., 2017. Methods of recycling, properties and applications of recycled thermoplastic polymers. Recycling 2, 1–11. https://doi.org/10.3390/recycling2040024.

Hajibabaei, M., Nazif, S., Tavanaei Sereshgi, F., 2018. Life cycle assessment of pipes and piping process in drinking water distribution networks to reduce environmental impact. Sustain. Cities Soc. 43, 538–549. https://doi.org/10.1016/j.scs.2018.09.014.

Haupt, M., Kägi, T., Hellweg, S., 2018. Life cycle inventories of waste management processes. Data Br 19, 1441–1457. https://doi.org/10.1016/j.dib.2018.05.067.

Iriarte, A., Gabarrell, X., Rieradevall, J., 2009. LCA of selective waste collection systems in dense urban areas. Waste Manag. 29, 903–914. https://doi.org/10.1016/j.wasman.2008.06.002.

ISO, 2012. EN 12201-1: Plastics Piping Systems for Water Supply, and for Drainage and Sewerage under Pressure — Polyethylene (PE) — Part 1. General.

ISO, 2010. EN 1555-1: Plastics Piping Systems for the Supply of Gaseous Fuels — Polyethylene (PE) — Part 1: General.

Istrate, I.-R., Galvez-martos, J.-L., Dufour, J., 2021. The impact of incineration phase-out on municipal solid waste landfilling and life cycle environmental performance : case study of Madrid , Spain. Sci. Total Environ. 755, 142537 https://doi.org/10.1016/j.scitotenv.2020.142537.

Jambeck, J.R., Ji, Q., Zhang, Y.-G., Liu, D., Grossnickle, D.M., Luo, Z.-X., 2015. Plastic waste inputs from land into the ocean. Science (80-.) 347, 764–768.

Jeswani, H., Krüger, C., Russ, M., Horlacher, M., Antony, F., Hann, S., Azapagic, A., 2021. Life cycle environmental impacts of chemical recycling via pyrolysis of mixed plastic waste in comparison with mechanical recycling and energy recovery. Sci. Total Environ. 769, 144483 https://doi.org/10.1016/j.scitotenv.2020.144483.

Juan, R., Domínguez, C., Robledo, N., Paredes, B., García-Muñoz, R.A., 2020. Incorporation of recycled high-density polyethylene to polyethylene pipe grade resins to increase close-loop recycling and Underpin the circular economy. J. Clean. Prod. 276 https://doi.org/10.1016/j.jclepro.2020.124081.

Koffler, C., Finkbeiner, M., 2018. Are we still keeping it “real”? Proposing a revised paradigm for recycling credits in attributional life cycle assessment. Int. J. Life Cycle Assess. 23, 181–190. https://doi.org/10.1007/s11367-017-1404-x.

Lau, W.W.Y., Shiran, Y., Bailey, R.M., Cook, E., Stuchtey, M.R., Koskella, J., Velis, C.A., Godfrey, L., Boucher, J., Murphy, M.B., Thompson, R.C., Jankowska, E., Castillo, A.

C., Pilditch, T.D., Dixon, B., Koerselman, L., Kosior, E., Favoino, E., Gutberlet, J., Baulch, S., Atreya, M.E., Fischer, D., He, K.K., Petit, M.M., Sumaila, U.R., Neil, E., Bernhofen, M.V., Lawrence, K., Palardy, J.E., 2020. Evaluating scenarios toward zero plastic pollution. Science (80- 369, 1455–1461. https://doi.org/10.1126/SCIENCE.ABA9475.

Laurent, A., Clavreul, J., Bernstad, A., Bakas, I., Niero, M., Gentil, E., Christensen, T.H., Hauschild, M.Z., 2014. Review of LCA studies of solid waste management systems - Part II: methodological guidance for a better practice. Waste Manag. 34, 589–606. https://doi.org/10.1016/j.wasman.2013.12.004.

Lausselet, C., Cherubini, F., Serrano, A., Becidan, M., Hammer, A., 2016. Life-cycle assessment of a Waste-to-Energy plant in central Norway : current situation and effects of changes in waste fraction composition. Waste Manag. 58, 191–201. https://doi.org/10.1016/j.wasman.2016.09.014.

Lazarevic, D., Aoustin, E., Buclet, N., Brandt, N., 2010. Plastic waste management in the context of a European recycling society: comparing results and uncertainties in a life cycle perspective. Resour. Conserv. Recycl. 55, 246–259. https://doi.org/10.1016/j.resconrec.2010.09.014.

Lebreton, L., Andrady, A., 2019. Future scenarios of global plastic waste generation and disposal. Palgrave Commun 5, 1–11. https://doi.org/10.1057/s41599-018-0212-7.

Ministerio para la Transición Ecológica y el Reto Demográfico, 2020. Plan Nacional Integrado de Energía y Clima 2021-2030. Gobierno de España.

Mitrano, D.M., Wohlleben, W., 2020. Microplastic regulation should be more precise to incentivize both innovation and environmental safety. Nat. Commun. 11, 1–12. https://doi.org/10.1038/s41467-020-19069-1.

Mutel, C., 2017. Brightway: an open source framework for life cycle assessment. J. Open Source Softw. 2, 236. https://doi.org/10.21105/joss.00236.

Nessi, S., Bulgheroni, C., Konti, A., Sinkko, T., Tonini, D., Pant, R., 2020. Comparative Life Cycle Assessment (LCA) of Alternative Feedstock for Plastics Production - Part 1. European Commission, Ispra.

Nguyen, L.K., Na, S., Hsuan, Y.G., Spatari, S., 2020. Uncertainty in the life cycle greenhouse gas emissions and costs of HDPE pipe alternatives. Resour. Conserv. Recycl. 154, 104602 https://doi.org/10.1016/j.resconrec.2019.104602.

Peña, C., Civit, B., Gallego-schmid, A., Druckman, A., 2020. Using life cycle assessment to achieve a circular economy. Life Cycle Initiat 1–2.

Plastics Europe, 2020. Plastics – the facts 2020 [WWW Document]. URL. https://www.plasticseurope.org/en/resources/market-data.

Plastics Europe, 2019. Plastics – the facts 2019 [WWW Document]. URL. https://www.plasticseurope.org/en/resources/market-data.

Plastics Pipe Institute, 2021. Life cycle assessment of north American stormwater pipe systems [WWW Document]. URL. https://plasticpipe.org/pdf/tr-53-2021.pdf.

Poduška, J., Dlhý, P., Hutař, P., Frank, A., Kučera, J., Sadílek, J., Náhlík, L., 2019. Design of plastic pipes considering content of recycled material. Procedia Struct. Integr. 23, 293–298. https://doi.org/10.1016/j.prostr.2020.01.102.

Rigamonti, L., Grosso, M., Møller, J., Sanchez, V.M., Magnani, S., Christensen, T.H., 2014. Environmental evaluation of plastic waste management scenarios. Resour. Conserv. Recycl. 85, 42–53. https://doi.org/10.1016/j.resconrec.2013.12.012.

Rigamonti, L., Taelman, S.E., Huysveld, S., Sfez, S., Ragaert, K., Dewulf, J., 2020. A step forward in quantifying the substitutability of secondary materials in waste management life cycle assessment studies. Waste Manag. 114, 331–340. https://doi.org/10.1016/j.wasman.2020.07.015.

Schmidt, S., Laner, D., Van Eygen, E., Stanisavljevic, N., 2020. Material efficiency to measure the environmental performance of waste management systems: a case study on PET bottle recycling in Austria, Germany and Serbia. Waste Manag. 110, 74–86. https://doi.org/10.1016/j.wasman.2020.05.011.

Schrijvers, D.L., Loubet, P., Sonnemann, G., 2016. Developing a systematic framework for consistent allocation in LCA. Int. J. Life Cycle Assess. 21, 976–993. https://doi.org/10.1007/s11367-016-1063-3.

Shen, L., Worrell, E., Patel, M.K., 2010. Open-loop recycling: a LCA case study of PET bottle-to-fibre recycling. Resour. Conserv. Recycl. 55, 34–52. https://doi.org/10.1016/j.resconrec.2010.06.014.

Steubing, B., de Koning, D., Haas, A., Mutel, C.L., 2020. The Activity Browser — an open source LCA software building on top of the brightway framework. Softw. Impacts 3, 100012. https://doi.org/10.1016/j.simpa.2019.100012.

Tascione, V., Mosca, R., Raggi, A., 2016. Optimizing the environmental performance of integrated waste management scenarios by means of linear programming: a case study. J. Clean. Prod. 112, 3086–3096. https://doi.org/10.1016/j.jclepro.2015.10.016.

Tonini, D., Schrijvers, D., Nessi, S., Garcia, P., Jacopo, G., 2021. Carbon footprint of plastic from biomass and recycled feedstock : methodological insights. Int. J. Life Cycle Assess. 26, 221–237. https://doi.org/10.1007/s11367-020-01853-2.

Turner, D.A., Williams, I.D., Kemp, S., 2016. Combined material flow analysis and life cycle assessment as a support tool for solid waste management decision making. J. Clean. Prod. 129, 234–248. https://doi.org/10.1016/j.jclepro.2016.04.077.

Viau, S., Majeau-Bettez, G., Spreutels, L., Legros, R., Margni, M., Samson, R., 2020. Substitution modelling in life cycle assessment of municipal solid waste management. Waste Manag. 102, 795–803. https://doi.org/10.1016/j.wasman.2019.11.042.

Weidema, B.P., 2017. In search of a consistent solution to allocation of joint production. J. Ind. Ecol. 0 1–11. https://doi.org/10.1111/jiec.12571.

Wernet, G., Bauer, C., Steubing, B., Reinhard, J., Moreno-Ruiz, E., Weidema, B., 2016. The ecoinvent database version 3 (part I): overview and methodology. Int. J. Life Cycle Assess. 21, 1218–1230. https://doi.org/10.1007/s11367-016-1087-8.

WRAP, 2019. Plastics market situation report 2019 [WWW document]. URL. https://wrap.org.uk/resources/market-situation-reports/plastics-2019#download-file.

Xiong, J., Zhu, J., He, Y., Ren, S., Huang, W., Lu, F., 2020. The application of life cycle assessment for the optimization of pipe materials of building water supply and drainage system. Sustain. Cities Soc. 60, 102267 https://doi.org/10.1016/j.scs.2020.102267.

Zheng, J., Suh, S., 2019. Strategies to reduce the global carbon footprint of plastics. Nat. Clim. Change 9, 374–378. https://doi.org/10.1038/s41558-019-0459-z.