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Hard-to-recycle plastics in the automotive sector: Economic, environmental and technical analyses of possible actions.

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ABSTRACT

The use of plastics in the automotive industry is favoured by their relatively low cost, but a sustainable treatment at their end of life is still challenging. The objective of this study is to contribute to the identification of best practices to increase the recovery rate of plastic materials from end-of-life vehicles (ELVs). European regulations for ELVs foresee that the reuse/recovery and reuse/recycling had to be increased to a minimum of 95% and 85% of the vehicle weight respectively by 2015. Three areas with room for possible improvement were identified in this study: the dismantling phase, the recycling processes, and the material recovery from automotive shredder residues (ASRs) as solid recovered fuels (SRFs). The economic feasibility of recovering specific plastic components from ELVs was assessed using a criterion based on the cost of dismantling, recycling and disposal of the components, as well as the environmental costs of the processes. Based on the results, disassembly and recycling could be cost-effective for a disassembly time below 180 s and a component mass above 600 g. For the recycling processes, the Life Cycle Assessment (LCA) methodology was applied to evaluate the environmental impacts of recycling HDPE from fuel tanks, polyamides PA6/PA66 and PET from automotive components. As the climate change indicator is concerned, Tthe LCA study showed that the impact for 1 kg of these secondary raw materials is respectively of 0.83, 0.16/0.17 and 2.17 kg CO_2 eq, obtained from these fractions resulting more sustainable than the respective virgin materials. Electricity consumption was among the main contributors to the potential environmental impacts. The characterization process of ASRs was conducted to assess their compliance to certain types of SRFs. According to the results of the industrial tests, the treatment facility can recover only around 74% of an ELV. The characteristics of ASRs were compliant to be assimilated to a SRF. This study showed that the amount of plastics recoverable from ELVs has the potential to increase thus facilitating the fulfilment of EU recovery targets.

1. Introduction

In 2020, global virgin plastics production almost reached 367 million tonnes, of which 55 million tonnes in Europe. The European plastics industry had a turnover of more than 330 billion euros in 2020. An amount of 29.5 million tonnes of plastic waste were collected in the EU27 + 3 in order to be treated. 34.6% of this amount was recycled, 42% sent to energy recovery, 23.4% landfilled. The third biggest end-use market for plastics in Europe is the automotive industry, with around 9% share of demand. In 2018, around 80% of recycled plastic produced in Europe re-entered in the European economy in order to manufacture new products. Of this amount, 3% was used in the automotive industry (Plasticseurope, 2022).

The use of plastics in the automotive industry is favoured by the relatively low cost of production (in comparison with other materials), which further discourages their recycling. Worldwide, regulations were set to prevent vehicle waste by reducing hazardous substances, designing with disassembly, re-using and recycling, and increasing the use of recyclable materials (Anthony and Cheung, 2017). The waste hierarchy provides that components must be first evaluated for their reuse (i.e. used again for the same purpose), then for been recycled (i.e. removing materials from the waste stream and using them as raw materials to create new products) and finally for the recovery of energy. In Europe, as of 2015, the End of Life Vehicle (ELV) European Directive 2000/53/EC (recently modified by Directive, 2018/849) for ELVs foresees that the reuse and recovery had to be increased to a minimum of 95% of the vehicle weight by 2015. Within the same time limit, the reuse

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List of abbreviations

ASRs	automotive shredder residues
ATF	authorized treatment facility
BHET	bis-hydroxy-ethylene-terephthalate
DEM	disassemblability evaluation method
DM	Ministerial Decree
ELV	end of life vehicles
EPDM	ethylene propylene diene monomer rubber
ETS	emission trading scheme
GHG	greenhouse gases
HDPE	high density polyethylene
LCA	life cycle assessment
MEG	monoethylene glycol
PA	polyamide (nylon)
PA6	polyamide 6
PA66	polyamide 66
PE	polyethylene
PES	polyether sulfone
PET	polyethylene terephthalate
POM	polyoxymethylene (acetal)
PP	polypropylene
PU	PUR, polyurethane
PVC	polyvinyl chloride
SRF	solid recovered fuels
VOC	volatile organic carbon

and recycling had to be increased to a minimum of 85% of the vehicle weight. In 2018, the average reuse and recycling rate of ELVs in the EU stood at 87.3%. However, this result has been achieved thanks to eleven EU Member States which reported reuse and recycling rates above 90.0%, while most European countries still fail to comply with the mentioned Directive.

End of life vehicles (ELVs) are usually subjected to three treatment stages: decontamination, disassembly, and shredding (which includes crushing and material sorting). Plastic materials recovery may be obtained both by means of a separation before the dismantling operation or from automotive shredder residues (ASRs) after the comminution operation. The reuse and recycling process following the raw material recovery will be simpler and more effective in case of the separation before the demolition operation. Plastics recycling during ELV treatment is complex and the methods used are presently insufficiently selective, leading to substantial loss. Such inefficiency is a consequence of a variety of economic and technical challenges that discourage recycling (Vogt et al., 2021). At present, only the heaviest and easiest to remove components are recovered. Unfortunately, most of the remaining plastic parts in the vehicle are relatively small and hard to remove. An important aspect is also the complexity of individual components. A high number of sub-components increases the probability of having a heterogeneous material, which hinders the recycling process. Finally, recycled materials can only be used if they have exactly the same properties of the virgin material (European Automobile Manufacturers Association (ACEA) and European Association of Automotive Suppliers (CLEPA), 2018).

The production, consumption and disposal of automotive plastic components mainly generate undesired impacts on the environment and the economy. Some of these impacts, such as waste management, impose direct economic costs, while others impose indirect costs related to the deterioration of the environment and human health. These latter are usually considered externalities, as they are not included in the price of virgin plastic (European Environment Agency (EEA), 2021). Costs induced by plastics not currently accounted for in the market price include: the cost of greenhouse gas (GHG) emissions, health costs, waste management costs and costs of a poor end-of-life management. Within each cost dimension, there are some elements that are quantifiable and some that are currently not (Afrinaldi and Mat Saman, 2008) (Dalberg Advisors, 2021).

Significant progress has already been made to improve the mechanical recycling of plastics, with recycled quantities of plastic waste having doubled in Europe since 2006 (Volk et al., 2021). The act of recovering and recycling secondary materials is, in general, thought to be a 'good thing' but there are relatively few analyses, which monitor existing or proposed recycling schemes to find out if they really produce any environmental benefits (Turner et al., 2015) (Gu et al., 2017). For the treatment of ELVs, it is necessary to assess whether the recovery processes actually lead to a net economic and environmental benefit, in order to avoid the impacts outweighing the benefits due to the availability of secondary raw materials. The objective of this study was therefore to contribute to the identification of best practices to increase the recovery rate of plastic materials from ELVs, by assessing the technical-economic feasibility of recycling certain components or fractions and quantifying the environmental impacts of recycling processes of certain critical plastic components. To this end, three areas with room for possible improvement were identified in this study: the dismantling phase, the recycling processes, and the material recycle from shredder residues for solid recovered fuels (SRFs) production. Analyses have been carried out using different specific methodologies and tools, which, according to the authors, best address the specific problems of the selected areas.

Among the main challenges of the dismantling phase, there is its economical sustainability: often, the dismantling of small components is uneconomical, even when the recyclability rate of the component is high. Therefore, feasibility of recycling specific plastic components from ELVs was assessed using an economic criterion based on the cost of dismantling, recovery and disposal of the components, as well as the environmental cost of the processes.

For recycling processes, it is important to define if recycling represents an environmental sustainable solution even when components are of difficult recyclability or have to be treated with not well-established technologies. In this context, this paper applies the Life Cycle Assessment (LCA) methodology to evaluate the environmental impacts of recycling HDPE from fuel tanks, polyamides PA6/PA66 and PET from automotive components. This allows to avoid the shifting of environmental impacts from the ELV waste treatments to the recycling.

As the material recycle is concerned, this study focuses on the plastic separated from the automotive shredder residues (ASRs), which is generally considered a waste. The aim of the study is to evaluate if the shredded plastic can be classified as a solid recovered fuels (SRFs) according to the Italian regulations. Therefore, in positive case, ASRs would allow to increase the share of an ELV to be recycled as material, thus contributing to the achievement of 85% target fixed by EC Directive 2000/53/EC. To this aim, this study developed a characterization process of ASRs to assess if it is compliant with the requirements of DM 14/02/2013, n. 22, that regulates the cessation of the waste status of certain types of solid recovered fuels (SRFs).

In this paper, methodology and results of each of the three analysis stages are presented separately, then comprehensively discussed in light of the general purpose of the study.

2. Methodology

2.1. *Analysis of the economic and environmental cost of dismantling and recycling plastic components*

In order to increase the recycling of plastic component, the performed operations must be sustainable and represent a potential economic advantage for the dismantler. It is therefore necessary to determine the optimal stage of disassembly, when all economically valuable components are retrieved (Gerrard and Kandlikar, 2007). The objective of this stage of analysis was thus to assess the feasibility of dismantling and recycling certain plastic components from disused vehicles. Feasibility was assessed using an economic criterion based on the cost of dismantling, recycling and disposal of the components, as well as the environmental cost of the processes.

Economic criteria focusing on ELV disassembly have been presented since the late Nineties. The metrics used in the proposed methodologies can be generally divided into two categories: absolute metric such as time and cost, energy for disassembly and entropy for disassembly, and relative metrics such as design effectiveness (Go et al., 2011). In 1993, the Disassemblability Evaluation Method (DEM) was developed as a quantitative measurement of the ease with which a product could be disassembled (Kroll et al., 1996). DEM provided a "Disassemblability Evaluation Score" based in a 100-point scale. McGlothin and Kroll (1995) introduced the spread sheet-like chart. Using this method, disassembly difficulties were categorised into accessibility, positioning, force, additional time and special. Gupta and Isaacs (1997) defined profit functions based on a series of costs and revenues of material removed by the disassembler. Other methods based on disassembly time were presented by Yi et al. (Hwa-Cho Yi et al., 2003) and Kongar and Gupta (2006). Lee et al. proposed detailed guidelines to determine the optimal level of disassembly of end-of-life products (Lee et al., 2001).

This study was based on the cost of dismantling. In addition, the concept of environmental costs linked to the life cycle of components was introduced in the economic evaluation. The environmental costs considered were the cost of greenhouse gas (GHG) emissions, waste management costs, and costs of poor end-of-life management (Adelodun, 2021). The study started with the identification of the components potentially most suitable for the effective dismantling in the field. This assessment was obtained by means of dismantling tests carried out in collaboration with project partners (Stellantis Group and Centro Recuperi e Servizi S.p.A) during the period 2019–2021. Fig A.1 (Appendix A) shows the selected components. These components were the input data used for the cost analysis. The approximate weight and the main materials each component is made of are reported in Table A.1.

Specifically, the costs were compared considering two options.

1) disassembly and recycling;
2) disposal of the dismissed component and production of a new part from virgin raw material.

Fig. 1 shows a diagram of the compared alternative solutions, indicating the boundaries of the analysis and the costs and emissions that have been accounted for in the calculation. The study boundaries were limited to the production of the base material only, without calculating the cost of producing the finished component. This is because the objective of the comparison was to assess the different origins of the production materials (recycled and non-recycled), rather than the final cost of producing the parts. The reported costs therefore do not refer to the finished part, but to the raw material needed to produce the part.

The total cost of dismantling and partial recovery C of a generic component was calculated as:

$$C = C_{opt1}\ (m_{rec}, t_{dis}) + C_{opt2}\ (m_{norec}) \tag{1}$$

$$m_{tot} = m_{rec} + m_{norec} \tag{2}$$

Where C_{opt1} is the cost of the dismantling of the component (function of dismantling time t_{dis}), and recycling of the portion m_{rec} (amount that is recovered); C_{opt2} is the cost of the disposal and the production of new material referred to the portion m_{norec} of the component (amount that is not recovered); m_{tot} is the mass of the component, listed in Table A.1.

The different cost elements which were considered in the calculation of C_{opt1} and C_{opt2}, and the related data sources, are reported in Table 1 and Table 2.

2.2. Life Cycle Assessment of HDPE from fuel tanks, polyamides PA6/PA66, PET-PUR multilayer material

When dealing with recycling processes, especially using new techniques or technologies, it is fundamental to quantify if and in which measure the recycling process is more environmental sustainable than the alternative scenarios (use of primary materials, disposal of the end-of-life object). If it is true that the recycling of plastic materials is currently well established, there are still some components that result critical, and which, at the same time can make the difference to achieve the recycling targets set by the European Commission. This study focused on the environmental performances of innovative recovery technologies developed by the partners of RECIPLAST project. Specifically, the technologies allow the recycling of HDPE from vehicle tanks, Polyamides PA6/PA66 and PET-PUR multilayer materials. The environmental analyses were developed with the Life Cycle Assessment

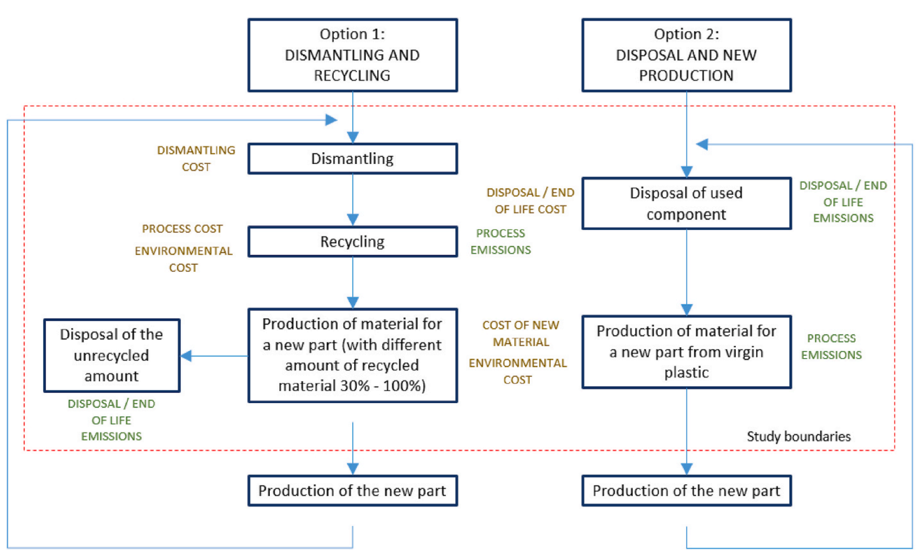

Fig. 1. Operational alternatives comparison.

Table 1
List and sources of cost elements considered in the analysis for option 1 (dismantling and recovery, C_{opt1}). Cost factors are referred to year 2021.

Cost element	Data source	Notes
Dismantling cost	Own tests + Italian Directorial decree n. 23 of 3 April 2017 (Italian Ministry of Labour, 2017)	Calculated as the product of dismantling time and the average gross cost of workers (30 €/h)
Cost of the recycling process	Cost factors (€/t): PA,PP, PET, HDPE, 400; EPDM, PUR 500; PE, 350	Information collected from RECIPLAST project partners. Data of EPDM and PUR must be considered with caution, as the recycling processes of these material are not yet consolidated. For PES and POM it was not possible to define a cost. The components made of these materials were thus excluded from the study.
Cost of GHG emissions from the recycling process	Emission factors ($kgCO_{2eq}$/kg): PA 1.98 (Solvay Company, 2021), PP 0.763 (Bora et al., 2020) PET 0.73 (European Union, 2022) EPDM 0.76 (Magnusson and Mácsik, 2017), PE 0.598 (Econinvent, 2022) HDPE 0.86 (Istrate et al., 2021a) PUR 0.644 (Marson et al., 2021). CO_2, 85 €/t (ETS market, average of June 2022)	Calculated as the product of process emission factor and unitary cost of CO_2.
Direct costs of disposal of the unrecovered material	Information collected from RECIPLAST project partners Cost factor:	Assumed average value of 290 €/t
Cost of GHG emissions due to the disposal of the unrecovered material	European Environmental Agency, report "Greenhouse gas emissions and natural capital implications of plastics (including biobased plastics)" (European Environment Agency (EEA), 2021)	Calculated as the product of the mass of material sent for disposal, the emission factor ($kgCO_{2eq}$/kg) of the disposal process and the unit cost of the CO_2 emitted. The emission factor of the disposal process is a representative value of the end-of-life emissions of non-recovered materials in the EU, which include collection, transport and final disposal (landfill or incineration). This value was defined as 1.73 $kgCO_{2eq}$/kg, according to the data reported by the European Environmental Agency.

(LCA) methodology, standardized by ISO 14040-44 (The International Standards Organisation, 2006a, 2006b). Impact analyses were performed with the CML-IA baseline method (version 3.05) and all the available impact categories were analysed (global warming, abiotic depletion, fossil abiotic depletion, ozone layer depletion, human toxicity, fresh water aquatic ecotoxicity, marine aquatic ecotoxicity, terrestrial ecotoxicity, photochemical oxidation, acidification, eutrophication). Calculations were supported by the LCA Software SimaPro 8.5 and by background data of the Ecoinvent 3.4 database.

2.2.1. LCA of recycling of HDPE fuel tanks

The main obstacle to the recycling of vehicles HDPE fuel tanks is the strong odor and the VOC contamination due to the use phase of the tank.

Table 2
List and sources of cost elements considered in the analysis for option 2 (production of new material, C_{opt2}). Cost factors are referred to year 2021.

Cost element	Data source	Note
Direct costs of disposal of the unrecovered material	Same as option 1	–
Cost of GHG emissions due to the disposal of the unrecovered material	Same as option 1	–
Market price of virgin material	Cost factors (€/t): PA 2700 PP 1800 PET 1150 EPDM 1900 PE 1750 HDPE 1500 PUR 3400	Information collected from RECIPLAST project partners, Plasticfinder.it (Plasticfinder, 2022), Plastiker.de, (Plasticker, 2022). Prices were referred to October 2021.
Cost of GHG emissions from the production process of the virgin material	Emission factors ($kgCO_{2eq}$/kg): PA 6.4 Ecoprofile (Plastics Europe, 2022b) PP 1.63 Ecoprofile (Plastics Europe, 2022b) PET 2.1 Ecoprofile (Plastics Europe, 2022b) EPDM 3.67 EU Environmental Footprint Database (European Union, 2022) PE 1.8 Ecoprofile (Plastics Europe, 2022b) HDPE 1.8 Ecoprofile (Plastics Europe, 2022b) PUR 4.2 Ecoprofile (Plastics Europe, 2022b) Cost factor:CO_2, 85 €/t (ETS market, average of June 2022)	GHG emissions defined on a cradle-to-gate basis.

To best of authors' knowledge, the vehicle tank is currently not recycled by any company. The innovative extrusion process studied during the project uses a co rotating twin-screw extruder with degassing points combined with the injection of water as medium for desorbing the organic contaminants. Further details of this process have been recently published (Monti et al., 2022).

Results of the impact assessment are given for the functional unit of 1 kg of recycled HDPE. The analysis included the processes from the grinding of waste tanks to the production of HDPE granulate. For each process, the consumption of materials and energy was considered, as well as waste treatments and emissions. The scheme in Fig. 2 summarizes the processes included in the analysis. The inventory is mostly composed of primary data, provided by Maris SpA company in year 2022, with exception of data for the grinding and washing of the tank, which are secondary data, obtained from a recent scientific article (Istrate et al., 2021b). Tables 1–6 of the Supplementary Material provide the specific life cycle inventories.

2.2.2. LCA of recycling of polyamides PA6 and PA66

Polyamide, compared to other plastics, is not easily recyclable, mainly because of its low temperature of melting, which hinders the decontamination of pollutants. In this case, Maris SpA, partner of the RECIPLAST project, developed a the Evorec Plastic Plus process, which consists in the coupling of a single screw extruder with a system for loading and treating the incoming material (grinding and dehumidification) and the co-rotating twin screw extruder. Therefore, the combination of these two technologies in a single machine and in a single step enables the recycling of materials having a high level of contamination, which was difficult with previous technologies (chemical, mechanical or thermal recycling; Alberti et al., 2019; La Mantia et al., 2002;

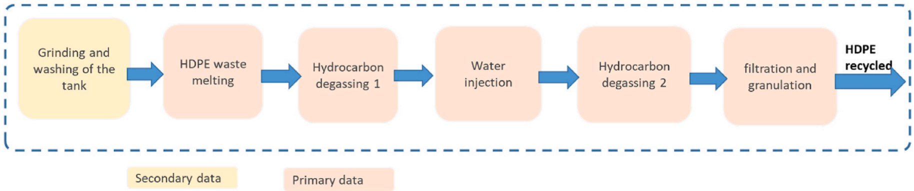

Fig. 2. System boundaries of the HDPE recycling. Indication of primary and secondary data sources is provided as well.

Mondragon et al., 2020; Ozmen et al., 2019).

The functional unit was 1 kg of polyamides PA6/PA66 granulate. The employed technology was the same for both the analysed polyamides, but with differences in the energy consumption. Fig. 3 summarizes the system boundaries of the study, which included the processes from the waste grinding to the production of PA6/PA66 granulate. As it can be noticed, the entire process was divided into two sub-processes. For both of them, primary data of year 2022 were provided by the companies that have developed the process. Table s7 and 8 of the Supplementary Material provide the specific inventory used for the assessment.

2.2.3. LCA of recycling of PET-PUR multilayer materials

Multilayer materials such as PET-PUR present difficulties for the separation of the different layers. A recent article (de Mello Soares et al., 2022) provides a deep overview on the current available technologies for multilayer materials recycling, dividing into high-performance recycling technologies, chemical recycling and downcycling. The partner Garbo SpA of the RECIPLAST project developed a technology based on a chemical reaction, which transform PET into an intermediate product called BHET (bis-hydroxy-ethylene-terephthalate). This latter is subsequently purified and used again for the PET production. The process is presented in (Garbo srl, 2022). The scheme in Fig. 4 shows the system boundaries of this process, whose data were all directly collected from the partner Garbo srl. As can be noticed, the PET-PUR material undergoes a solvolysis in MEG, which dissolves the polyurethane part and 15% by weight of the PET fraction. The remaining 85% of PET remains solid and can be removed from the solution to be treated separately. Two co-products are obtained: (i) PET-PUR in MEG, which is sold to an external company for the production of polyols; (ii) the PET impregnated fabric, which will undergo further treatments in order to obtain recycled PET granules. An economic allocation was introduced to divide the impacts among the two co-products, considering the economic values provided by Garbo srl of 500 €/t for PET impregnated fabric and 100 €/t for PET-PUR in MEG. Tables 9–11 of the Supplementary Material provide the specific inventory.

2.3. ASR analysis

A sample (28 May 2020) of light ASRs was collected from the Centro Recuperi e Servizi ELV authorized treatment facility (ATF) of Settimo Torinese (Metropolitan Turin Area, NW Italy). The ATF has a treatment capacity of 123,200 t/y that is sufficient to accommodate and treat all the ELVs dismissed in the Turin province plus an amount of white goods (washing machines, refrigerators and other large electrical household appliances). The sample was collected during an industrial test that involved the shredding and treatment of ELVs only. At the end of the test all the separated fractions were weighted and the light ASRs was found to be 23.10% b.w. of the shredded ELVs.

The sampling operation was carried out, in agreement with UNI EN ISO 21645:2021 (Italian Standardization Body, 2021) rule on the waste generated from the aspiration performed onto the main shredder of the shredding plant of the ATF. The sample underwent a product composition analysis through manual sorting. The plastic separated from the other ASR components (namely foam rubber, textile, rubber, metals and particles with dimensions of less than 10 mm) was subjected to a particle size analysis and a sink-float separation, by using water ($\rho = 1$ g/cm^3) as a separating medium. The floated fraction, that was deemed the most interesting also for other processes intended to material recovery (Ruffino et al., 2021), was quartered and a sub-sample was ground to sizes <1 mm to further characterization.

The assimilation of the plastic contained into the light ASRs to a SRF, according to DM 14/02/2013 n. 22, required the compliance with three parameters, namely heating value, and chlorine and mercury content, and with the content of a number of metals (namely Sb, As, Cd, Cr, Co, Mn, Ni, Pb, Cu, Tl and V).

The heating value was determined in a calorimetric bomb onto three replicates of a sample of 1.00 ± 0.05 g. For the determination of chlorine and metals, six replicates (0.15 ± 0.01 g each) were subjected to a two-stage acid digestion, with sulphuric acid in the first stage and nitric acid in the second stage. The acid mixture, after filtration (Whatman 542, 2.7 μm retention size) was analysed for chloride (iron-mercury thiocyanate method with spectrophotometric determination at 463 nm) or metal (ICP-OES Perkin Elmer Optima 2000 DV; Perkin Elmer, 2022) determination.

3. Results

3.1. Cost analysis of dismantling options

The results of the cost analysis, calculated according to equations (1) and (2), are shown in Table 3, considering for option 1 a "limit" assumption of 80% recovery and recycling of the source material ($m_{rec} = 0.8\ m_{tot}$). For option 1 (dismantling of components), the purely operational costs range between 0.1 € and 9.6 €/component, depending on the material, dismantling time and mass of the component. If environmental externalities are also taken into account, the cost of components is between 0.11 € and 10.1 €. By reducing the share of recovered material, costs increase by 105%–168% for 50% recovery ($m_{rec} = 0.5\ m_{tot}$), and by 109%–236% for 20% recovery ($m_{rec} = 0.2\ m_{tot}$). The inclusion of environmental cost items, albeit to a limited extent, helps to reduce the cost increase. For option 2 (without component disassembly), the purely economic costs range between 0.20 € and 37.6 €, depending on the market price of the material and the mass of the component. Considering also the environmental factors, the cost of the components is between 0.23 € and 42.8 €. In this case, excluding market price factors, the costs (both economic and environmental) are linearly proportional

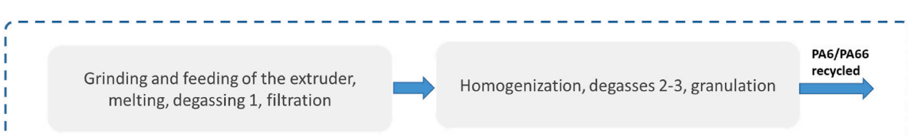

Fig. 3. System boundaries of the polyamides PA6 and PA66 recycling.

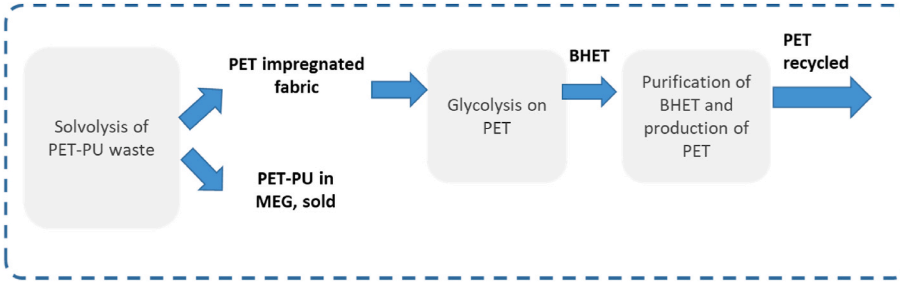

Fig. 4. System boundaries of the PET-PUR multilayer recycling.

Table 3
Comparison of costs with and without the environmental component (values in € referred to year 2021).

Component	Material	WITHOUT environmental costs			WITH environmental costs		
		Option 1 (dismantling and 80% recycling)	Option 2 (NO dismantling)	Difference	Option 1 (dismantling and 80% recycling)	Option 2 (NO dismantling)	Difference
Airbag	PA	3.01	3.29	**−0.28**	3.31	4.05	**−0.74**
Kick plate	PP	2.80	0.84	**1.96**	2.84	0.95	**1.89**
Luggage guard	PP	1.93	1.21	**0.72**	1.99	1.38	**0.61**
Hatbox	PP	1.12	3.14	**−2.02**	1.28	3.56	**−2.28**
Seatbelts	PET	1.34	2.59	**−1.25**	1.55	3.19	**−1.64**
Wheel cover	PP	1.63	4.39	**−2.76**	1.86	4.99	**−3.13**
Headlights	PA	3.19	2.24	**0.95**	3.39	2.76	**0.63**
Headlights	PP	3.05	1.57	**1.49**	3.14	1.78	**1.35**
Air filter and filter cover	PP	1.44	3.14	**−1.69**	1.60	3.56	**−1.96**
Window gasket	EPDM	1.30	2.19	**−0.89**	1.44	2.65	**−1.21**
Door gasket	EPDM	1.13	2.63	**−1.50**	1.30	3.18	**−1.88**
Glass scraper gasket	EPDM	0.42	0.44	**−0.02**	0.45	0.53	**−0.08**
Radiator sleeve	EPDM	0.29	0.44	**−0.15**	0.32	0.53	**−0.21**
Handle	PA	1.14	2.09	**−0.95**	1.33	2.58	**−1.24**
Central cabinet	PP	1.49	2.09	**−0.60**	1.60	2.38	**−0.78**
Air inlet cover	PP	1.63	1.78	**−0.15**	1.72	2.02	**−0.30**
Door panel	PP	3.71	6.27	**−2.56**	4.04	7.13	**−3.09**
Bumper	PP	6.30	13.59	**−7.29**	7.01	15.44	**−8.44**
Wheel arch	POM	n.d.	n.d.	**n.d.**	n.d.	n.d.	**n.d.**
Wheel arch	PES	n.d.	n.d.	**n.d.**	n.d.	n.d.	**n.d.**
Pillar	PP	0.80	0.84	**−0.04**	0.84	0.95	**−0.11**
Sun shield	PE	0.18	0.49	**−0.31**	0.21	0.56	**−0.36**
Wheel guard	PP	2.09	1.67	**0.42**	2.18	1.90	**0.28**
Seats	PUR	16.11	37.64	**−21.53**	17.58	42.78	**−25.20**
Seats	PP	5.75	3.55	**2.20**	5.94	4.04	**1.90**
Fuel tank	HDPE	7.63	14.86	**−7.23**	8.61	17.35	**−8.74**
Washer fluid tank	PE	2.41	1.22	**1.19**	2.47	1.40	**1.07**
Battery tray	PP	1.07	0.21	**0.86**	1.08	0.24	**0.85**

to the mass of material.

Table 3 also shows the comparison between the two considered operational alternatives (with and without dismantling and recycle). Negative values indicate an advantage of the first solution over the second, i.e. that it is more convenient to recycle the material. Conversely, positive values indicate an advantage of the second solution over the first, i.e. that it is not worth recovering the material. Values close to zero indicate that the two options are equivalent in terms of cost. For ease of visualisation, to the values in Table 3 three colours have been assigned: green for negative cost deltas (10 components), yellow for limited cost deltas (less than 1 €, 12 components), and red for positive cost deltas (4 components). The majority of delta costs are therefore rather limited.

The most favorable cases are bumpers, tank and seats. Bumpers are components that are usually recovered, as they can be dismantled quite quickly. The fuel tank is a good candidate, although to date there is still the problem of eliminating the fuel smell. The seats are also good candidates, but in this case the result found is influenced by two main factors. The first one is that PUR recovery has no structured market at present, and the cost and emission factors of the recovery process are not consolidated and therefore they should be evaluated with caution. The second uncertainty factor is due to the disassembly time of the seats: being composed of several materials and varying according to the vehicle, the disassembly cost could indeed be higher than that found in this study (Marson et al., 2021). Similarly to PUR, the results for EPDM components have also to be evaluated with caution, for the same reasons (Magnusson and Mácsik, 2017).

The least favorable results are represented by the headlights, the bumper and the rigid part of the seats. These components are all characterised by high disassembly times (>300 s). The introduction of environmental costs into the calculation tends to favor the recovery and recycling of the component.

Fig. 5 shows the cost difference as a function of disassembly time for polypropylene components only (14 components out of 28). A trend towards an alignment of the points can be discerned which can be approximated by a power relationship (Fig A.2 and Table A.2, see Appendix). If this approximation is taken into account, it can be seen that the delta cost equal to zero corresponds to a disassembly time of about 180 s. The two outliers in Fig. 5 represent those components that have limited disassembly time and high mass, i.e. bumpers (6500 g; 180 s;

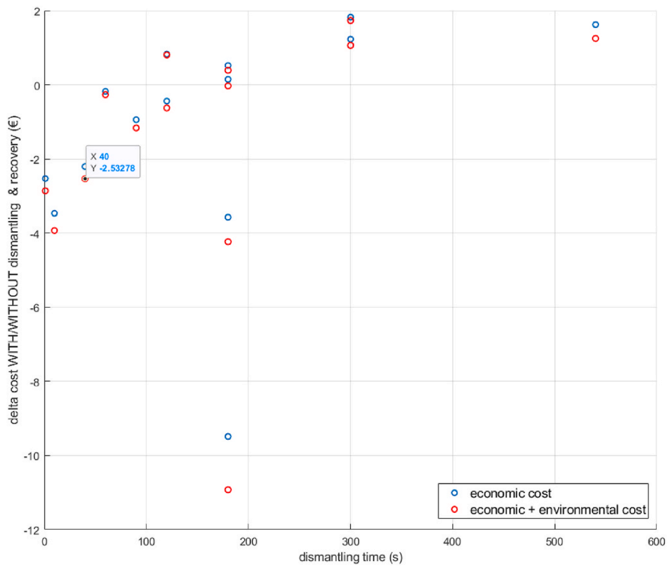

Fig. 5. Cost difference as a function of disassembly time.

−10.9 €) and door panels (3000 g; 180 s; −4.2 €) (Table A.1). Fig. 6 shows the cost difference as a function of component mass. Also in this case, it is possible to identify a tendency towards an alignment of the points which, for polypropylene components, is linear as a function of mass (Fig A.2 and Table A.2). If this approximation is taken into account, it can be seen that the delta cost value becomes negative for mass values of the component greater than 600 g. In this case, seats (1700 g, 540 s; 1.6 €) represent an outlier point as despite their high mass, their high disassembly time influences negatively on their cost delta.

3.2. Life cycle impact assessment (LCIA)

3.2.1. LCIA of recycled HDPE from fuel tanks

The inventory data summarized in the Supplementary Material was used to create the LCA model of recycled HDPE. The impact analysis was performed with the CML-IA baseline method. Table 4 lists the impact values related to the production of 1 kg of recycled and virgin HDPE.

With reference to the climate change impact category, Fig. 7 shows the contribution of the sub-processes associated with the production of

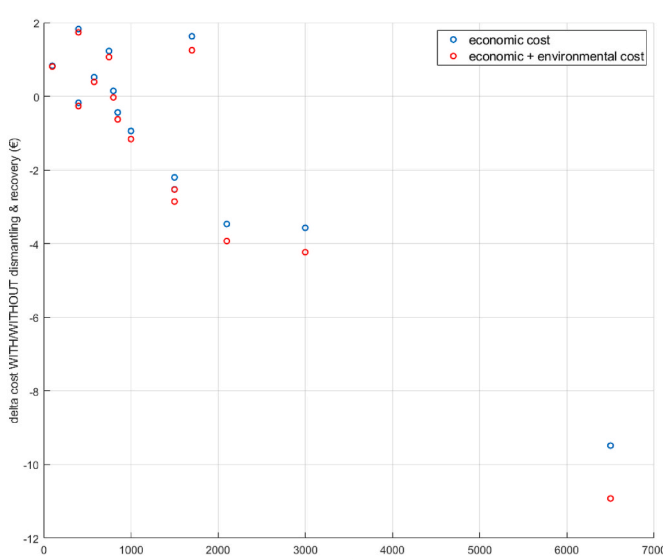

Fig. 6. Cost difference as a function of the mass of the component.

Table 4
Potential environmental impacts of 1 kg of recycled HDPE from fuel tanks and 1 kg of virgin HDPE.

Impact category	Unit	Impact of 1 kg of recycled HDPE granulate	Impact of 1 kg of virgin HDPE granulate
Abiotic depletion	kg Sb eq	1.64E-06	4.32E-08
Abiotic depletion (fossil fuels)	MJ	9.55E+00	6.63E+01
Global warming (GWP100a)	kg CO_2 eq	8.25E-01	2.00E+00
Ozone layer depletion (ODP)	kg CFC-11 eq	9.49E-08	1.11E-09
Human toxicity	kg 1.4-DB eq	2.82E-01	9.57E-02
Fresh water aquatic ecotox.	kg 1.4-DB eq	4.12E-01	1.31E-01
Marine aquatic ecotoxicity	kg 1.4-DB eq	8.27E+02	7.05E+02
Terrestrial ecotoxicity	kg 1.4-DB eq	5.18E-03	1.24E-04
Photochemical oxidation	kg C_2H_4 eq	1.78E-04	6.25E-04
Acidification	kg SO_2 eq	5.16E-03	6.54E-03
Eutrophication	kg PO_4^{-} eq	1.34E-03	5.45E-04

recycled HDPE. This analysis shows that 94% of the impact on climate change is due to the electricity consumed during the process. The grinding and tank washing phase affects 31%, although this data has a higher uncertainty as it is derived from secondary data. Among the processes carried out by Maris SpA, the greatest contribution is given by the energy used by the extruder resistances (20% of the total) and by the main engine (18% of the total). The virgin HDPE produced in Europe (Ecoinvent dataset named "Polyethylene, high density, granulate (RER)"), has an impact on climate change of 2 kg CO_2 eq./kg (Table 4), which means that recycled HDPE can save 60% of the potential impacts on climate change. However, it has to be noticed that for other impact categories (abiotic depletion, ozone layer depletion, human toxicity, ecotoxicity, eutrophication) the virgin HDPE has higher environmental performances.

3.2.2. LCIA of recycled polyamide PA6 and PA66

Table 5 lists the potential impacts of 1 kg of PA6 and PA66 granulate. Results are provided for both granulate obtained with the recycling process described in the previous paragraph and average granulate produced in Europe (with reference to Ecoinvent datasets "Nylon 6 {RER}| production" and "Nylon 6-6 {RER}| production").

A contribution analysis was carried out to identify which processes have the greatest impacts. Analyzing the impacts of PA6 on all the indicators present in the CML-IA baseline method (Fig. 8), it emerges that for almost all impact categories, the first macro-process (grinding and feeding of the extruder, melting, degassing 1, filtration), is responsible for the greatest impacts. Its contribution varies between 28% (for the Abiotic depletion category) and 62% (for the abiotic depletion (fossil fuel), global warming and ozone layer depletion categories). The remaining part of the impacts is due to the energy used by the Maris SpA process, in particular the energy used by the main engine and the cutter. Similar considerations apply to PA66.

Impacts on climate change of average Nylon 6 and Nylon 6-6 produced in Europe respectively result of 9.22 and 8.23 kg CO_2 eq./kg, therefore higher than the recycled PA6 and PA66. Also for the other impact categories (with exception of the ozone layer depletion indicator), the process developed by Maris SpA results being significantly more sustainable.

3.2.3. LCIA of recycled PET granulates

Table 6 lists the potential impacts of 1 kg of recycled PET granulates,

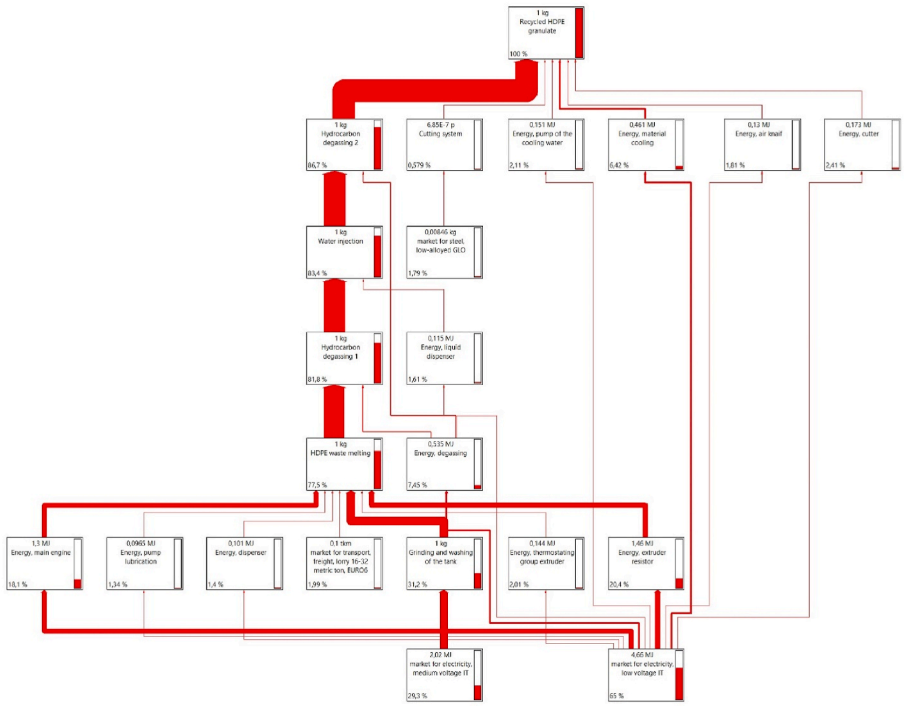

Fig. 7. Chart of the potential impact on climate change of 1 kg of recycled HDPE, from fuel tank (visualisation cut-off: 1%). This chart provides: (i) the quantity of each input for the production of 1 kg of recycled HDPE, in the top part of each box; (ii) the cumulative impact (as a percentage of the total impact) in the bottom-left of each box; (iii) arrows connecting the processes, whose dimension is proportional to the impact on climate change.

Table 5
Potential environmental impacts of 1 kg of recycled and virgin polyamide PA6 and PA66.

Impact category	Unit	Recycled PA6	Average PA6	Recycled PA66	Average PA66
Abiotic depletion	kg Sb eq	1.99E-07	6.52E-05	1.80E-07	2.85E-06
Abiotic depletion (fossil fuels)	MJ	1.80E+00	1.04E+02	1.97E+00	1.12E+02
Global warming (GWP100a)	kg CO_2 eq	1.56E-01	9.22E+00	1.70E-01	8.23E+00
Ozone layer depletion (ODP)	kg CFC-11 eq	1.78E-08	5.36E-09	1.94E-08	2.42E-09
Human toxicity	kg 1.4-DB eq	5.89E-02	4.75E-01	5.67E-02	4.24E-01
Fresh water aquatic ecotox.	kg 1.4-DB eq	5.96E-02	4.31E-01	5.78E-02	3.27E-01
Marine aquatic ecotoxicity	kg 1.4-DB eq	1.44E+02	2.19E+03	1.52E+02	1.60E+03
Terrestrial ecotoxicity	kg 1.4-DB eq	7.00E-04	9.38E-04	6.64E-04	6.71E-04
Photochemical oxidation	kg C_2H_4 eq	3.31E-05	1.39E-03	3.57E-05	1.37E-03
Acidification	kg SO_2 eq	9.75E-04	2.97E-02	1.06E-03	2.93E-02
Eutrophication	kg PO_4^{--} eq	2.40E-04	6.10E-03	2.58E-04	7.62E-03

with reference to the process described in the previous paragraph. Impacts of 1 kg of the average production of PET granulate in Europe (Ecoinvent dataset "Polyethylene terephthalate, granulate, amorphous {RER}| production") are provided as well.

In addition, Fig. 9 shows the contribution of the sub-processes is in terms of CO_2 eq. for the PET recycling. As it can be noticed, the impacts on climate change are mainly due to the heat (total 32%) and electricity (total 16%) used during the process, and the MEG consumed (26%). There are no impacts due to the incoming plastic material as the latter derives from a waste. As a result, the process could be further improved by recovering the MEG to a greater extent and using a greater share of energy from renewable sources.

As for the previous analyses, also for this material, for all the analysed indicators with the exception of the ozone layer depletion category, the average PET granulate results having higher impacts than the recycled PET here analysed.

3.3. ASR analysis

The composition of the sample of light ASR is shown in Fig. 10. It can be seen that foam rubber and heavy textile were the two most abundant products in the sample, accounting for approx. 46% and 24% by weight (b.w.), respectively. The amount of plastic was approx. 12% b.w. The sizes of the plastic product ranged from 15 to 250 mm, with $D_{10} = 50$ mm, $D_{50} = 120$ mm and $D_{90} = 230$ mm. The results of the sink-float separation carried out at 1 g/cm^3 revealed that 62% of the plastic extracted from the light ASR had a density of less than 1 g/cm^3. This result was in line with that of a previous characterization carried out on two samples of light ASR collected from the same ATF (Ruffino et al., 2021). In that case the amount of plastic with a density of less than 1 g/cm^3 was approx. 55%.

The results of the characterization aimed to verify the assimilability of the light plastic fraction, extracted from the light ASR, to a SRF are shown in Table 7. The assimilability requires the compliance of the waste product with the three parameters that are deemed to be able to describe the compatibility with commercial (i.e. the heating value),

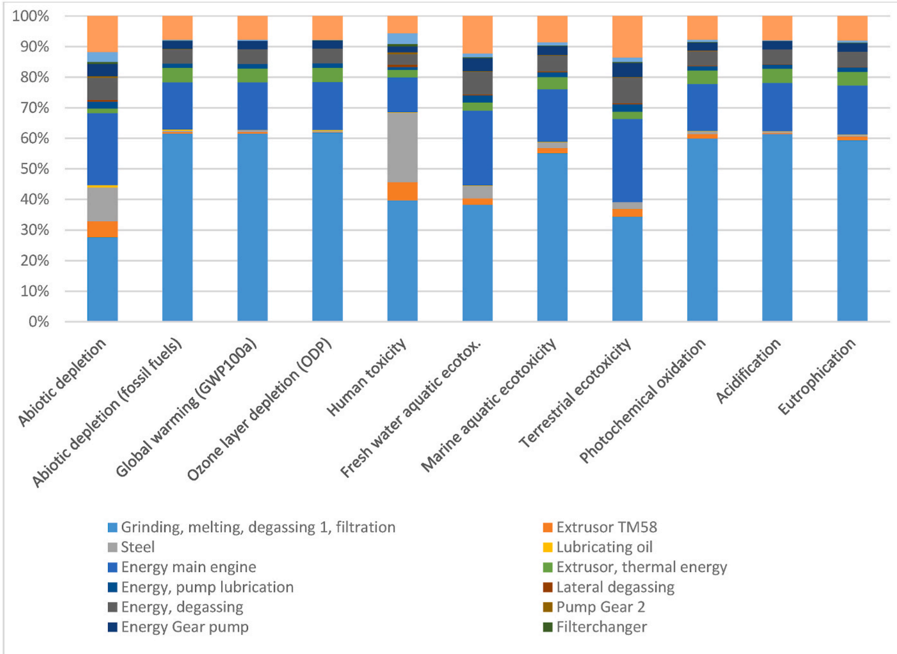

Fig. 8. Relative contribution of sub-processes to potential impacts of the recycling of PA6.

Table 6
Environmental impacts of 1 kg of recycled and virgin PET granulates.

Impact category	Unit	Recycled PET	Average PET
Abiotic depletion	kg Sb eq	3.15E-06	1.17E-05
Abiotic depletion (fossil fuels)	MJ	3.20E+01	6.60E+01
Global warming (GWP100a)	kg CO2 eq	2.17E+00	3.02E+00
Ozone layer depletion (ODP)	kg CFC-11 eq	1.78E-07	1.30E-07
Human toxicity	kg 1.4-DB eq	4.71E-01	1.45E+00
Fresh water aquatic ecotox.	kg 1.4-DB eq	3.01E-01	7.44E-01
Marine aquatic ecotoxicity	kg 1.4-DB eq	1.04E+03	2.77E+03
Terrestrial ecotoxicity	kg 1.4-DB eq	2.03E-03	4.06E-03
Photochemical oxidation	kg C2H4 eq	3.67E-04	6.78E-04
Acidification	kg SO2 eq	5.65E-03	1.15E-02
Eutrophication	kg PO4— eq	1.80E-03	3.41E-03

process (i.e. the chlorine content) and environmental (i.e. the mercury content) requisites. Furthermore, the compliance with a number of metals is required.

It can be seen from Table 7 that the heating value of the light plastic was more than adequate (34 MJ/kg vs. 20 MJ/kg) for the assimilation to a SRF. The process of sink-float separation allowed to remove plastics with a density of more than 1 g/cm^3 such as PVC, thus avoiding a chlorine contamination of the SRF, as testified by the very low chlorine content found in the plastic sample. The content of mercury and some other metals (namely arsenic, lead, thallium and vanadium) was below the detection limits of the ICP-OES. The content of the remaining metals was detected and it proved to be below the threshold values fixed by DM 14/02/2013, n. 22.

4. Discussion

This study considered three operational areas (dismantling, recycling and material recovery) with a single objective, namely maximising the recycle of plastic materials from ELVs. For the purposes of an evaluation, it is appropriate to consider the results obtained first separately, then jointly.

The results of ELV disassembly analysis showed, for both operational alternatives, a variability of costs as a function of the disassembly time of the component and the mass of the component. The costs of option 1, which involves disassembly and recovery of the component, are also strongly influenced by the share of material that is recovered and recycled downstream of disassembly operation. The costs of option 2, which involves the disposal of the component and the production of a new material, are linearly proportional to the mass of the part. The comparison of the two operational options made it possible to calculate the cost difference and to give indications as to whether or not disassembly and recycling of the components is feasible. The analysis of the alignment of cost deltas as a function of disassembly time and component mass (for PP components only) established that disassembly and recycling could tend to be cost-effective for a disassembly time below 180 s and component mass above 600 g. This study also reported data for materials whose recycling processes are still in the experimental phase (PUR, EPDM), or concerning multi-material components (seats, gaskets). It is recommended to use those results with due caution as they require further in-depth studies. The introduction of environmental costs into the calculation, although not leading to significant changes in cost differences, contributed to shift the result in favor of component dismantling and recycling. This means that the consideration of the environmental costs for the production, use, dismantling and recycling of plastic materials, in addition to the already considered economic operating costs, could influence the assessment of the feasibility of recovering disused components.

This analysis was inherently affected by several sources of uncertainty, mainly due to market constraints or variability of the production or recycling processes. To characterize such an uncertainty, an analysis was conducted assuming the following factors.

- Cost of the materials recycling process varying in the range 300–500 €/t for PA, PP, and PET and in the range 250–450 €/t for HDPE and PE.
- Cost of CO2 varying between 85 and 100 €/t;
- Disposal costs varying between 280 and 300 €/t;
- Market cost of virgin material variable by ±10%.

The analysis was conducted by creating a script with Matlab software and processing a very large number (10^5) of cost calculations. In each calculation, a random value to the parameters was assigned, extracted from the indicated ranges. It was assumed that the probability

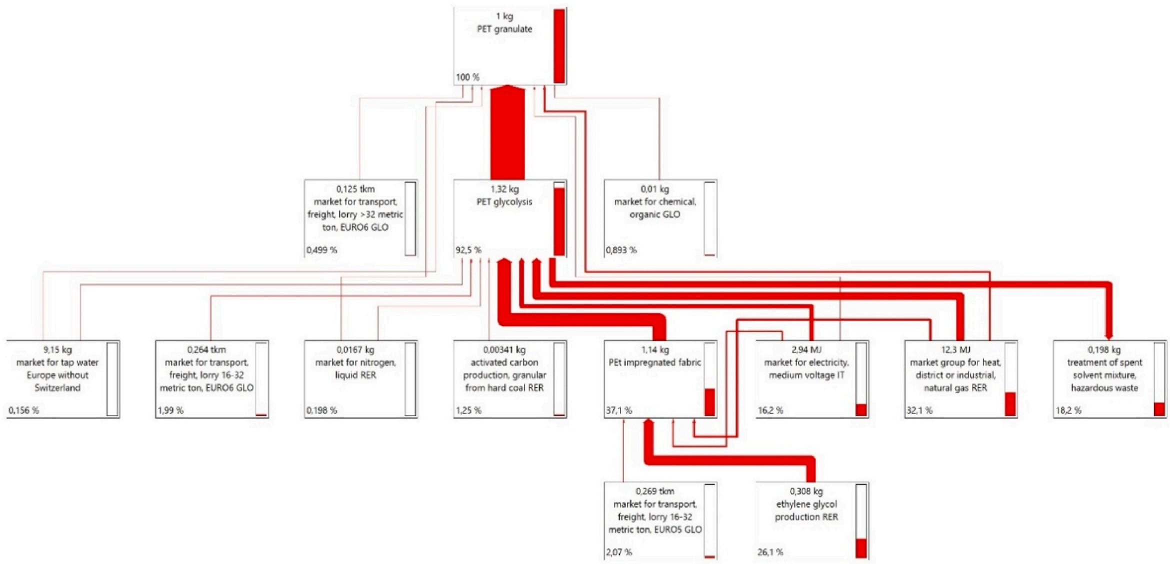

Fig. 9. Chart of the potential impact on climate change of 1 kg of recycled PET (visualisation cut-off: 0.1%). This chart provides: (i) the quantity of each input for the production of 1 kg of recycled HDPE, in the top part of each box; (ii) the cumulative impact (as a percentage of the total impact) in the bottom-left of each box; (iii) arrows connecting the processes, whose dimension is proportional to the impact on climate change.

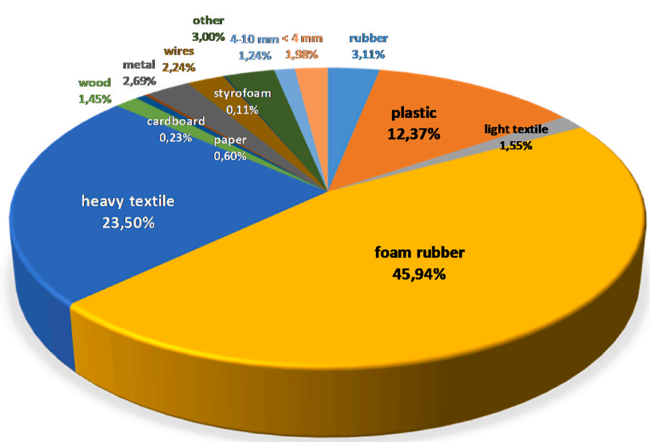

Fig. 10. Results of the product composition analysis carried out on the light ASR sample.

Table 7
Results of the characterization of the light ASR sample.

Parameter	Sample	Threshold values
Heating value (MJ/kg)	34.0 ± 1.2	20
Chlorine (% s.s.)	$<1.7\bullet10^{-3}$	0.6
Hg (mg/MJ)	<0.01	0.03
Sb (mg/kg)	14.2 ± 2.5	50
As (mg/kg)	<1.8	5
Cd (mg/kg)	0.715 ± 0.556	4
Cr (mg/kg)	22.4 ± 15.4	100
Co (mg/kg)	8.27 ± 14.3	18
Mn (mg/kg)	12.7 ± 5.4	250
Ni (mg/kg)	7.22 ± 3.40	30
Pb (mg/kg)	<1.4	240
Cu (mg/kg)	11.8 ± 4.9	500
Tl (mg/kg)	<1.5	5
V (mg/kg)	<1.2	10

distribution of the values within the intervals was uniform. The result is shown in Table A.3, in terms of the cost range and variation below and above the central value. For option 1, the lower variation was between 1 and 14%, while the upper variation was between 2 and 18%. The variation was higher for components with higher mass and lower disassembly time. For option 2, the lower variation was between 7 and 8%, and the upper variation was between 9 and 10%. This result indicates that the cost estimate for option 1 is subject to greater uncertainty, related mainly to the cost of recycling processes.

From the LCA study it emerged that for the recycling of HDPE from fuel tanks, polyamides PA6/PA66 and PET are more sustainable than the respective virgin materials. In addition, the electricity used is among the main contributors to the potential environmental impacts, especially for the indicator on climate change. As a consequence, the use of energy with a high percentage of renewable sources could further decrease the impacts of the secondary raw materials considered in this study. In addition, the impact of recycled PET could further decrease by recycling a greater amount of MEG.

A detailed study provided information also for the assimilation of the plastic extracted from ASRs to a SRF. According to the results of the industrial test mentioned in Section 2.3, the ATF considered in this study can generate an amount of light ASR in the order of 30,000 t/y, that is approx. 23–25% of the shredded ELVs. The mass balance carried out at the end of the industrial test revealed that the separation operations carried out in the ATF can recycler only approx. 74% of an ELV (see Table 8), 11% less than the value (85%) fixed by Directive (2000)/53/EC.

Plastic materials in the light ASR accounted for approx. 12%, thus 3800 t/y, and the light fraction of plastic ($\rho < 1$ g/cm^3) was in the order

Table 8
Amounts of the valorizable or waste products separated at the ATF during the industrial test.

Proler, ferrous scraps	69.03%
Copper wires	1.07%
Small zorba (<20 mm), non-ferrous miscellaneous	1.80%
Large zorba (>20 mm), non-ferrous miscellaneous	2.30%
Total of the recovered fractions	**74.20%**
Light ASRs	23.10%
Heavy ASRs	2.70%

of 2300 t/y. This study demonstrated that the characteristics of that fraction of plastic were fully in compliance with the requirements of DM 14/02/2013, n. 22, thus permitting the assimilation to a SRF. This practice can contribute to the achievement of the goal of 85% material recycling stated by EC Directive 2000/53/EC with an amount of approx. 2% (1.9%). However, this practice alone is not sufficient to the achievement of the above-mentioned goal and other solutions must be found to increase the share of material recycling in an ELV.

In an overall assessment of the obtained results, this study showed that there is room for improvement in the amount of plastics recoverable from ELVs, and that these materials are potentially suitable for assimilation into SRF. Despite of this, the achievement of EU targets remains difficult. Looking at the dismantling phase as a possible phase for improving the recovered quota, it was confirmed that the recyclability of a component at this stage is driven by strictly economic factors. In particular, the cost of labour and the mass of recyclable quantity determine the feasibility of the operation. In addition to these, there are other factors that may contribute but were not considered in this study, such as those related to component design (e.g. assembled materials). The results of this study can complement the most recent findings on the impacts of ELVs and the sustainability of the automotive supply chain in general, also considering other materials and components. Tarrar et al. (2021) recently published a review paper in which practical challenges of improving vehicle end-of-life management were investigated. They reported a complex inter-relationship among all component sectors, highlighting four main areas of improvement: plastics recycling, batteries recycling, investment/ownership structures, and the workforce.

Considering the environmental aspects, this study showed, for the reported processes, that the recycling of plastic components of the automotive sector is cleaner than the use of virgin materials, and environmental impacts could be even lower by using energy with a higher rate of renewables during the recycling process. In the perspective of a reduction of the carbon footprint of the automotive life cycle, possible design solutions for the reuse or recycling of plastic components, or their replacement by more easily disassemblable materials, should be evaluated at the scale of the whole vehicle, under a general environmental and economic methodology (Spreafico, 2021).

5. Conclusion

This study investigated ELVs best practices to increase the recycling rate of plastic materials, by assessing the technical-economic feasibility of recycling certain components or fractions and quantifying the process environmental impacts of certain critical plastic components.

The main conclusion of this study is that improving the environmental compatibility of plastics recycling processes in the automotive sector is a valid approach not only for reducing GHG emissions but also for achieving EU recovery targets. Specifically, this study highlighted two key aspects: (i) plastic recycling can be considered a sustainable solution also for components that are currently scarcely recycled (such as fuel tanks) and (ii) it results significant to evaluate the potential progressive internalisation of external environmental costs, which are currently not accounted for in the market. The presented results must be read in the light of the limitations of this study deriving from the various assumptions that have been made. Cost analyses were made based on a limited set of dismantling tests, including only B-segment cars. The applied emission and cost factors may rapidly change in time due to the evolution of emission, commodity, and energy markets. Similarly, the LCA study was based on the innovative recovery technologies, which present peculiarities due to the specific materials and components that are treated.

Increasing the recycling rate of materials is a complex process that must be supported by involving a variety of stakeholders: car manufacturers, dismantlers, recycling companies, administrations. All these subjects must work on the definition of a unified methodology so that the European objective is reached and exceeded.

CRediT authorship contribution statement

Marco Ravina: Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Software, Supervision, Visualization, Writing – original draft, Writing – review & editing. **Isabella Bianco:** Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Resources, Software, Visualization, Writing – original draft, Writing – review & editing. **Barbara Ruffino:** Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Supervision, Validation, Visualization, Writing – original draft, Writing – review & editing. **Marta Minardi:** Data curation, Formal analysis, Investigation, Software, Visualization, Writing – original draft. **Deborah Panepinto:** Conceptualization, Formal analysis, Methodology, Project administration, Resources, Supervision, Validation, Visualization. **Mariachiara Zanetti:** Conceptualization, Formal analysis, Funding acquisition, Methodology, Project administration, Resources, Supervision, Validation, Visualization.

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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Appendix B. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jclepro.2023.136227.

Appendix A

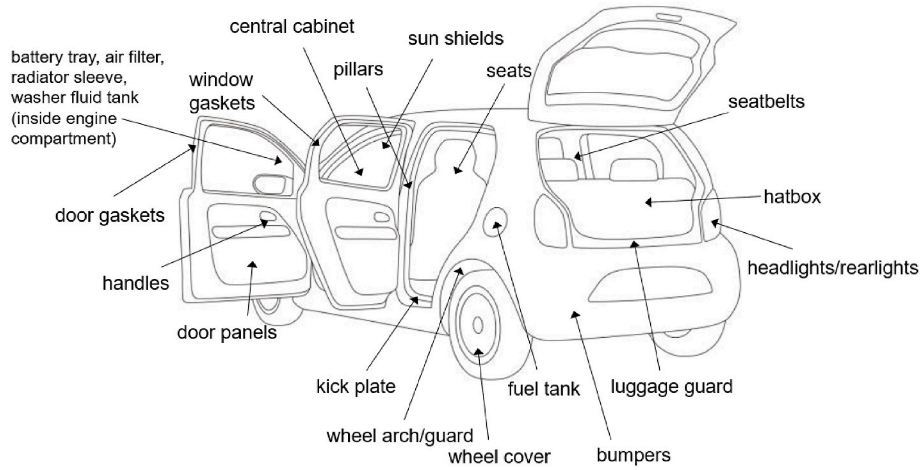

Fig. A.1. Representation of the components considered in the analysis

Table A.1
Components considered in the analysis

Component	Material	Mass (g)	Dismantling time (s)
Airbag	PA	1100	240
Kick plate	PP	400	300
Luggage guard	PP	580	180
Hatbox	PP	1500	1
Seatbelts	PET	1800	30
Wheel cover	PP	2100	10
Headlights	PA	750	300
Headlights	PP	750	300
Air filter and filter cover	PP	1500	40
Window gasket	EPDM	1000	55
Door gasket	EPDM	1200	15
Glass scraper gasket	EPDM	200	30
Radiator sleeve	EPDM	200	15
Handle	PA	700	60
Central cabinet	PP	1000	90
Air inlet cover	PP	850	120
Door panel	PP	3000	180
Bumper	PP	6500	180
Wheel arch	POM	350	150
Wheel arch	PES	150	150
Pillar	PP	400	60
Sun shield	PE	240	2
Wheel guard	PP	800	180
Seats	PUR	10200	540
Seats	PP	1700	540
Fuel tank	HDPE	8300	240
Washer fluid tank	PE	600	240
Battery tray	PP	100	120

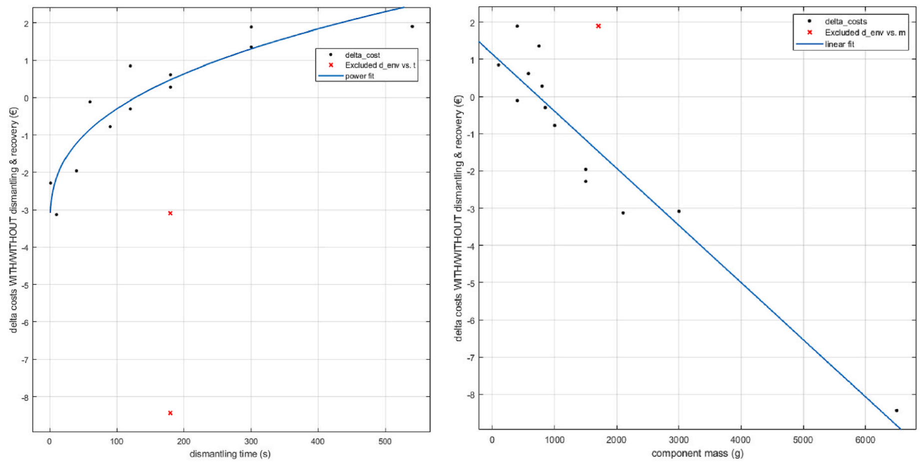

Fig. A.2. Data fitting for cost difference as a function of the dismantling time (left) and mass of the component (right).

Table A.2
Fitting parameters for cost difference as a function of the dismantling time and mass of the component.

x = dismantling time	x = component mass
General model Power2:	Linear model Poly1:
f(x) = a*x^b + c	f(x) = p1*x + p2
Coefficients (with 95% confidence bounds):	Coefficients (with 95% confidence bounds):
a = 0.9863 (−2.15, 4.123)	p1 = −0.001917 (−0.002233, −0.0016)
b = 0.3048 (−0.119, 0.7285)	p2 = 1.138 (0.4369, 1.839)
c = −4.625 (−9.295, 0.04548)	
Goodness of fit:	Goodness of fit:
R-square: 0.8	R-square: 0.9417
RMSE: 0.8845	RMSE: 0.8455

Table A.3
Cost variability and uncertainty estimation (values in € referred to year 2021)..

Component	Material	Option 1 (dismantling and 80% recycling), cost range	Lower-higher variation with respect to mean	Option 2 (NO dismantling), cost range	Lower-higher variation with respect to mean
Airbag	PA	3.16–3.50	7.6%–9.8%	2.98–3.55	4.4%–5.8%
Kick plate	PP	2.79–2.89	8.0%–9.7%	0.76–0.91	1.7%–1.8%
Luggage guard	PP	1.92–2.07	8.0%–9.7%	1.10–1.32	3.4%–4.0%
Hatbox	PP	1.11–1.48	8.0%–9.7%	2.85–3.41	13.6%–15.5%
Seatbelts	PET	1.37–1.77	7.0%–8.7%	2.37–2.76	11.9%–13.9%
Wheel cover	PP	1.62–2.14	8.0%–9.7%	3.99–4.77	12.9%–15.1%
Headlights	PA	3.29–3.52	7.6%–9.8%	2.03–2.42	2.9%–3.9%
Headlights	PP	3.05–3.24	8.0%–9.7%	1.43–1.70	3.0%–3.0%
Air filter and filter cover	PP	1.43–1.80	8.0%–9.7%	2.85–3.41	10.7%–12.8%
Window gasket	EPDM	1.40–1.50	7.5%–9.5%	1.99–2.36	2.8%–4.1%
Door gasket	EPDM	1.25–1.37	7.6%–9.5%	2.39–2.83	3.5%–5.7%
Glass scraper gasket	EPDM	0.44–0.46	7.6%–9.5%	0.40–0.47	2.6%–1.8%
Radiator sleeve	EPDM	0.31–0.33	7.6%–9.5%	0.40–0.47	2.1%–4.1%
Handle	PA	1.24–1.46	7.6%–9.8%	1.90–2.26	6.7%–9.5%
Central cabinet	PP	1.48–1.73	8.0%–9.7%	1.90–2.27	7.5%–8.3%
Air inlet cover	PP	1.62–1.83	8.0%–9.7%	1.62–1.93	5.8%–6.6%
Door panel	PP	3.69–4.45	8.0%–9.7%	5.70–6.81	8.7%–10.1%
Bumper	PP	6.25–7.87	8.0%–9.7%	12.35–14.76	10.8%–12.3%
Wheel arch	POM	n.d.	n.d.	n.d.	n.d.
Wheel arch	PES	n.d.	n.d.	n.d.	n.d.
Pillar	PP	0.79–0.89	8.0%–9.7%	0.76–0.91	5.6%–6.3%
Sun shield	PE	0.19–0.25	7.9%–10.2%	0.45–0.53	10.5%–17.9%
Wheel guard	PP	2.08–2.29	8.0%–9.7%	1.52–1.82	4.4%–4.8%
Seats	PUR	16.87–18.50	8.3%–9.8%	34.07–40.90	4.0%–5.2%
Seats	PP	5.74–6.17	8.0%–9.7%	3.23–3.86	3.4%–3.8%
Fuel tank	HDPE	7.69–9.70	7.7%–10.2%	13.53–16.19	10.7%–12.6%
Washer fluid tank	PE	2.43–2.58	7.9%–10.2%	1.11–1.34	1.7%–4.4%
Battery tray	PP	1.07–1.10	8.0%–9.7%	0.19–0.23	0.6%–1.7%

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