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Life cycle sustainability assessment of non-beverage bottles made of recycled High Density Polyethylene

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ABSTRACT

The current plastic industry is associated with climate change, fossil fuel depletion and littering of plastic waste. To reduce these environmental impacts, companies and governmental bodies are increasingly adopting strategies based on the concept Circular Economy. However, circular decision-making is usually based on analyses that do not provide enough insights in every sustainability dimension, risking burden-shifting. In this study, environmental, economic and social life cycle assessment (LCA) techniques have been integrated into an overarching sustainability life cycle assessment (LCSA) to assess the impact of recycling High Density Polyethylene (HDPE) non-beverage bottles. The study assesses the impact in 11 environmental categories, the life cycle costs, and the social risks associated with the related economic sectors. An ad-hoc system expansion approach was developed to overcome the multifunctionality issue so commonly challenging in circular systems. The results indicate that using recycled HDPE leads to significant reductions in all the considered environmental categories. The economic analysis indicated that the material cost of recycled HDPE is slightly lower than for virgin HDPE, but the manufacturing costs are higher and highly dependent on the specific value chain. The social risks of recycling were found to be higher than for virgin plastic production, and mainly occurring outside the country where the recycling takes place (The Netherlands). Nevertheless, this analysis presents high uncertainty due to the heterogeneity in the recycling sector of the database. This study shows how the LCSA approach can be used to assess and compare the impacts and benefits of circular strategies and calls for further efforts to develop higher disaggregated social risk databases.

1. Introduction

Global production rates of plastics have increased from 0.5 Mt/y to over 367 Mt in 2020 (PlasticsEurope, 2021). The plastic industry is seen as unsustainable and associated with climate change, fossil fuel depletion and littering of plastic waste (Arena et al., 2003). The main application for plastic is packaging, whose short lifetime and disposable character leads to high production and disposal rates. It has been estimated that 32% of the globally produced plastic packaging is still leaking into the environment (Ellen MacArthur Foundation, 2016). In Europe, around 60% of the postconsumer plastic waste originates from packaging applications. Roughly 16% of all plastic waste collected in 2020 was still exported outside Europe, which impairs control over these waste streams. Only 34.6% of the plastic waste subjected to European management is being recycled, whereas 42% is subjected to energy recovery and 23.4% still ends up in landfills (PlasticsEurope, 2021).

To decrease global production rates of plastics, companies and governmental bodies are increasingly adopting strategies based on the Circular Economy (CE) concept. These CE strategies like recycling and reuse are often adopted without understanding their true sustainability. The impact of such strategies should be measured using science-based impact calculation methods. Life Cycle Assessment (LCA) is identified as the most suitable assessment method to determine the environmental impact of products (Elia et al., 2017) and helps decision-making in the context of the CE. Whereas many circularity metrics have appeared in the literature claiming to assess the level of circularity of products, most of these metrics are focused only on material circularity (or just a few sustainability themes), risking an evaluation that leads to burden shifting from material circularity to other sustainability impacts (Corona et al., 2019). The European Commission is calling for research into the

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life cycle impacts of alternative feedstocks for plastic production as part of the European Strategy for Plastics in a Circular Economy (Nessi et al., 2021). The potential environmental impacts of using alternative feedstock such as plastic waste instead of the virgin feedstock, should be investigated through LCA.

LCAs are often combined with costs analyses to determine viability (e.g. Morris, 2005; Kreiger et al., 2014). However, the social impact has often been neglected in LCA studies because the social dimension in LCAs is still in development. Incorporating the social dimension in the assessment of circular strategies is a challenge, as evidenced by the lack of publications assessing the social impact of circular products and services (Corona et al., 2019). Nevertheless, multiple databases and social and socio-economic indicators are being developed to determine social risks per sector and country, e.g., the Social Hotspots and PSILCA databases or the Social LCA guidelines (UNEP et al., 2020). Such techniques should be integrated into an overarching life cycle sustainability assessment (LCSA) to help clarify trade-offs between economic benefits and environmental or social burdens.

The issues associated with plastic recycling are broader than the environmental dimension only. It is generally accepted that sustainable development requires a balance between environmental, economic, and social aspects (Atilgan and Azapagic, 2016). A recent literature review concluded that the LCA scientific literature for recycled or biobased plastics is currently limited to the environmental dimension (Nessi et al., 2021). The range of environmental impact categories is also often narrow. For instance, Tonini et al. (2021) assessed the carbon footprint of recycled feedstock for rigid plastic packaging. Their cradle-to-grave analysis showed a better environmental performance for recycled plastic in rigid packaging than virgin alternatives. Despite the study being very detailed, it is limited to the carbon footprint. Another study included a broader range of environmental categories into their assessment and showed a better overall environmental performance of recycled High Density Polyethylene (HDPE) resin as opposed to virgin HDPE resin (Franklin Associates et al., 2018). But none of these studies analysed the economic and social performance of the systems. A few impact assessments have broadened the scope toward the social dimension, investigating the social issues around informal recycling of e-waste in India (Pandey and Govind, 2014), China (Li et al., 2011) and Pakistan (Umair et al., 2015), or different packaging waste collection systems in Instanbul (Yildiz-Geyhan et al., 2019). These studies found many social disadvantages due to informal packaging waste collection systems, especially in terms of health safety and security, human rights and working conditions. Milios et al. (2018) investigated the recycled plastic sector in Europe but focused only on the number of jobs created. Research investigating a broader perspective on the social performance of recycled plastic items in a European country is lacking.

Until now, no studies have taken a holistic approach to impact measurement of the application of recycled plastic in packaging. This research attempts to fill this gap by assessing the environmental, economic, and social impact of recycled HDPE in non-beverage bottles. These bottles are one of the most globally produced plastic packaging used in cosmetics, detergents and other chemical products (PlasticsEurope, 2021). The European plastic demand is highest for polyethene, used in producing such rigid packaging. In this study, the use of virgin and recycled HDPE in non-beverage bottles are compared using an integrated LCSA to identify trade-offs between economic benefits and environmental or social burdens. This study also attempts to demonstrate how LCSA offers a multidisciplinary framework suitable to assess circular strategies while exploring sustainability trade-offs instead of focusing only on material circularity or a single environmental indicator. Such a comprehensive multidisciplinary framework is necessary to avoid burden shifting from circular strategies (Corona et al., 2019).

2. Methods

The impacts of recycled HDPE in non-beverage bottles were explored and quantified through an LCSA combining environmental LCA, life cycle costing (LCC) and a social risk assessment based on the social LCA (S-LCA) framework. The LCA study was conducted according to the international standards ISO 14040 (ISO, 2006a) and ISO 14044 (ISO, 2006b). According to these standards, an LCA study consists of four iterative stages: Goal and Scope Definition, Life Cycle Inventory Analysis, Life Cycle Impact Assessment and Life Cycle Interpretation (ISO, 2006a; ISO, 2006b). The first two stages are described in sections 2.1 and 2.2, while the last two stages are described in sections 3 and 4 (Results and Discussion). The LCA was carried out using Simapro software (version 8.5.2). Sections 2.5, 2.6 and 2.7 explain how the environmental impact, economic viability and social risks are measured.

2.1. Goal and scope

This study aims to assess the environmental, economic, and social performance of non-beverage bottles made with 100% recycled HDPE and compare it to virgin HDPE bottles. Non-beverage bottles are defined as packaging products produced to contain, protect and carry chemical fluids. The functional unit (FU) of this LCA is defined as *carrying and containing 1 L of liquid while maintaining the quality of the liquid for at least five years*. A period of five years was chosen since this is the expiration time of most chemical fluids. Geographical boundaries were set in Europe, and the system is modelled with inventory data representative for the year 2018.

This study is a cradle-to-grave assessment, including all stages of the bottle life cycle, from raw material extraction to final product disposal. The use phase is excluded since there are no material and energy inputs or outputs during this phase. Tertiary packaging, i.e. cardboard boxes, trays, stretch foils and pallet top layers, is included in the analyses, but bottle labels and the energy, materials and costs to produce manufacturing machinery are excluded.

The product system and system boundaries of the bottle are visualised in Fig. 1. The left-side diagram represents the product flow of a single virgin bottle, while the right-side diagram represents the production flow of a virgin bottle followed by one recycling cycle. The study follows a system expansion approach, in which the primary bottle production and subsequent recycling cycles are included in the recycled scenario, and further referenced to one FU. This approach avoids allocating impacts between recycling loops, as recommended by the ISO 14040-44 standards (ISO, 2006a; ISO, 2006b), and is critical in the context of circular strategies because the impacts and benefits of recycling and using recycled content are well and consistently integrated. It allows for the integration of more than three recycling cycles, as opposed to the circular footprint formula (CFF) provided by the PEF guidelines, which applies an allocation approach that integrates only the previous and subsequent cycles of the product system. Although the system expansion approach is the preferred option for almost every LCA guideline, it is usually unsuitable for open-loop recycling (i.e. products that are recycled into other product systems with different functionality). The material of the investigated product is assumed to be recycled into a product with the same functionality, which makes the system expansion approach suitable for this study.

In the approach proposed by this study, the inventory of an X amount of recycled bottles plus the first virgin cycle is compared to the inventory of an X+1 amount of virgin bottles and normalized per FU,[1] allowing for a comparison of scenarios that provide the same amount of functional units (i.e. 1 FU = 1 bottle). The relationship between the recycling rate and the number of bottles produced after a specific amount of recycling

[1] According to the LCA method, inventory input and outputs are normalized per FU by dividing the inventory amounts by the number of FU that the inventory represents. E.g., if the FU is one bottle and the inventory of a process represents the production of three bottles, the process' inventory inputs and outputs are to be divided by three to obtain the values per bottle.

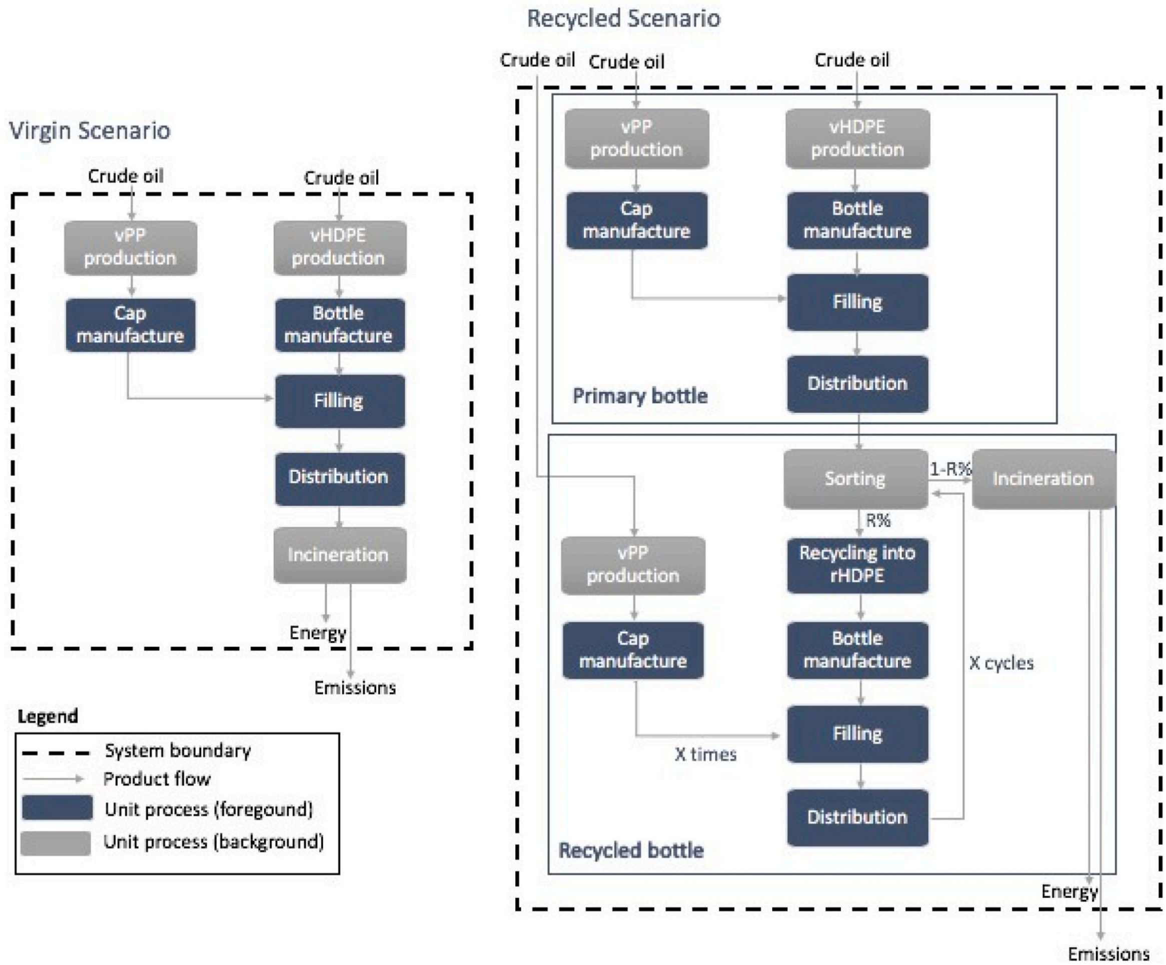

Fig. 1. Product system showing the packaging life cycle in a virgin and recycling scenario.

cycles needs to be defined to make this comparison.

This relationship is defined by Eq. (1), where the amount of bottles produced (n) after a specific amount of recycled cycles (i) is equal to the summation of the initial amount of primary bottles needed to produce one recycled bottle (1 /R) plus the successive amount of bottles produced after each recycling cycle. The Table A1 of Annex I includes a summary of the calculated n values considering different recycling rates and number of cycles, and an extended formulation of Eq. (1).

$$n = \frac{1}{R} + \sum_{n=1}^{i} R^{(n-1)} = \frac{\frac{1}{R} - R^{x}}{1 - R} \qquad [1]$$

n = Amount of bottles produced after i recycling cycles
R = Recycling rate
i = Amount of recycling cycles considered

The economic viability was addressed from a producer's point of view, in which material and resource purchase costs were compared. The social risks were assessed using the Product Social Impact Life Cycle Assessment database (PSILCA) version 2 (Eisfeldt and Ciroth, 2018). The plastic and recycling industry's potential social risks were identified and compared based on material costs (USD 2006) per bottle.

2.2. Life cycle inventory

Primary data for foreground processes were collected from a European supplier, and background processes were modelled using general data from EcoInvent database version 3 (Weidema et al., 2013). For the virgin scenario, foreground processes include bottle manufacture, cap manufacture, filling, and distribution. The production of HDPE and PolyPropylene (PP) and the incineration during the EoL are modelled as background processes (see Fig. 1). For the recycling scenario, foreground processes include bottle manufacture, cap manufacture, filling and distribution for the primary bottle. The production of HDPE and PP for the primary bottle is modelled as background processes. For the recycled bottle, foreground processes include HDPE recycling, bottle manufacture, cap manufacture, filling and distribution. PP production and the sorting and incineration after use are modelled as background processes.

Plastics that escape the recycling system are assumed to be incinerated for simplicity reasons. All other material is sorted and mechanically recycled. The recycling rate is assumed to be 40%, according to the relatively steady recycling rate for plastic packaging recycling in Europe from 2014 until 2019 (Eurostat, 2021). This rate as reported by Eurostat, represents Europe's average collection and sorting efficiency and excludes the recovery rate during recycling (i.e. recycling efficiency). The recycling efficiency depends on both material losses and quality losses. For example, material losses can occur during the granulation process, while quality losses can occur due to repeated extrusion (Antonopoulos et al., 2021). We define the recovery rate as the combination of the recycling rate and the recycling efficiency. Kawecki et al. (2021) used material flow analysis to calculate a 20%–24% European recovery rate for HDPE plastic in 2016 (as compared to 33% for PET). This recovery rate represents the actual share of recycled HDPE that flows back into the secondary market. More recent material flow analyses on the recovery rate of HDPE are unfortunately not available. It was decided to use the European average recycling rate of plastic packaging in the baseline calculations (40%) to represent material losses during collection, sorting and reprocessing, and integrate the potential effect of quality losses by limiting the number of times the material can be recycled. According to Oblak et al. (2015), recycled HDPE shows a deterioration of mechanical properties (such as hardness) after ten reprocessing cycles, i.e. through plastic extrusion, while other properties regarding thermal behaviour and durability show deterioration after 30 extrusion cycles. To integrate this material degradation, the baseline scenario was calculated by limiting to ten the number of recycling cycles (see Table A1 for the exact number of bottles produced after ten cycles with a 40% recycling rate). All relevant steps and processes are described in the following sections and summarised in Table 1. Sensitivity analyses on both the recycling rate and the number of recycling cycles were performed as defined in section 2.3.

2.2.1. HDPE production

The production of 58.2 g of virgin HDPE is modelled using the EcoInvent dataset for the average European production of HDPE granulate.

2.2.2. Bottle production

The HDPE granulate is transported over a distance of 505 km. At the bottle production site, this is mixed with 1.8 g of masterbatch colouring and blow-moulded into its shape using 0.08 kWh of grid electricity. Coloured HDPE remains in the machinery after a production cycle, referred to as scrap. This scrap is removed and transported to a granulation company to use for the subsequent production cycles. Both the transport and granulation process are included in the product system, and the obtained granulate material is subtracted from the HDPE input. The finished bottle component is packaged using 0.33 and 0.37 g of cardboard and stretch film. Transport to the filling site requires 420 km.

2.2.3. Cap production

The cap requires 10.24 g of virgin PP, which is transported over a distance of 30 km. This is mixed with 0.15 g of masterbatch at the cap production site and moulded into its shape using 0.02 kWh per cap. The scrap from this production process (2.89 g) is granulated and sold for recycling to other parties. The recycled PP material was assumed to follow a closed-loop system and modelled by subtracting it from the PP input (while accounting for electricity consumption of 0.6 kWh/kg PP for re-melting and granulating). A seal is placed mechanically, and 0.01 g of seal waste is created due to machinery flaws. The finished cap component is packed using 1.43 g cardboard and 0.007 g stretch film, and transported to the filling site over 450 km.

2.2.4. Bottle filling

The bottle and cap components are unpacked at the filling site, resulting in 1.76 and 0.38 g of cardboard and stretch film waste. The bottle is filled and sealed using 0.09 kWh in total. A substantial amount of cardboard (43 g), stretch film (0.69 g) and top plastic layer (0.35 g) are used for packaging of the final product. Transport from here to the distribution centre involves 737 km.

2.2.5. Distribution

Ten percent of the pallets are unpacked at the distribution site, resulting in low amounts of stretch film waste (0.003 g). Electricity consumption at the distribution centre is neglectable. The average transport distance to retailers is 209 km.

2.2.6. HDPE recycling

For one bottle, 57.90 g of postconsumer HDPE is sorted and transported over a distance of 190 km. Further sorting, cleaning and compatibilization activities result in 58.79 g of recycled HDPE, which is

Table 1
LCA inventory table for the bottle life cycle. Amounts are given per functional unit (FU). Certain amounts are not disclosed due to confidentiality agreements (marked with an X).

Process/material	Amount per FU
Bottle manufacture (60 g)	
HDPE	58.20 g
Transport virgin HDPE	23.22 kgkm
Electricity	0.08 kWh [a]
Yellow masterbatch	1.80 g
Cardboard	0.33 g
Stretch film	0.37 g
Transport scrap	1.83 kgkm
Scrap granulation	12.22 g
Transport bottle	104.33 kgkm
Scrap HDPE	0 g
Cap manufacture (7.5 g)	
PP	10.24 g
Transport PP	0.31 kgkm
Electricity	0.02 kWh
Green masterbatch	0.15 g
Seal	0.76 g
Cardboard	1.43 g
Stretch film	0.007 g
Transport cap	4.85 kgkm
Scrap PP	2.89 g
Seal waste	0.01 g
Cardboard waste	0.0001 g
Stretch film waste	0.0004 g
Filling	
Electricity	0.09 kWh [a]
Gas for internal transport	0.01 ml
In-between sites transport	3.62 kgkm
Cardboard	43 g
Stretch film	0.69 g
Top layer, low density	0.35 g
Cardboard waste	1.76 g
Stretch film waste	0.38 g
Distribution	
Transport filled bottle	942.39 kgkm
Electricity	0.00001 kWh
Stretch film	0.003 g
Transport retail	257.34 kgkm
Stretch film	0.003 g
Disposal	
Incineration bottle	60 g
Incineration cap	7.5 g
Only applicable for recycled scenario:	
HDPE recycling (58.20 g)	
Sorting HDPE	57.32 g
Transport HDPE	10.89 kgkm
Electricity	X kWh *
Heat from steam	X kJ
Water	X L
Additive 1	X g
Additive 2	X g
Transport rHDPE	9.83 kgkm
Waste water	X L

[a] Electricity consumption in warehouse excluded.

transported over 169 km. Material losses during disposal, transport and sorting were included in the study through the European recovery rate of 40%.

The end-of-life scenario for the bottle component was modelled as 60% incineration (36 g) and 40% recycled (24 g). The End-of-Life for the virgin bottle is modelled considering municipal incineration of the combination bottle and cap (67.5 g). Due to confidentiality agreements, the amount of electricity, steam, water, and additives used within the recycling process is not disclosed in the inventory tables.

2.3. Sensitivity analyses

The sensitivity of the study to the plastic packaging recycling rate (until the recycling gate) is explored because this rate is an important variable and differs per European country. The European average rate is chosen for this study (40%) but the recycling rate varies amongst European countries. Packaging recycling rates range from around 30%–75% throughout Europe (Eurostat, 2021). These recycling rates of 30% and 75% were included in the sensitivity analyses. This also aligns with the recycling target of 75% for packaging waste by 2030, communicated in the revised European waste proposal on packaging (European Commission, 2015). In that way, both geographical and temporal sensitivity of the results to the recovery rate were included.

The baseline is calculated by expanding the product system to include ten recycling cycles, as explained in section 2.2. This baseline does not account for any material, nor quality losses, and assumes that the same material can be recycled without material degradation. In reality, additional material may be lost during the recycling process, leading to less material becoming available to the secondary plastics market (Kawecki et al., 2021). Material losses happen directly due to loss during regranulation but also indirectly through the decrease in quality as a consequence of impurities, radiation, chemical contamination, or physical properties being affected (Schyns and Shaver, 2021). The distribution of recycled HDPE into the market is also challenging due to residual odours and lack of colour separation (Antonopoulos et al., 2021). To account for these losses, the sensitivity of the results to the number of recycling cycles is assessed by applying a cut-off in recycling cycles. When considering a recycling rate of 40%, most plastic is recovered after one recycling cycle, and the amount of material left after three cycles is already below 10% of the original recycled bottle. The sensitivity of the results to completing only one or three recycling cycles was also explored.

2.4. Data collection and quality

A total of nine companies provided primary data on the bottle life cycle, covering all life cycle steps in the product system. The origin of the data is kept confidential due to confidentiality agreements. The primary data was collected through preliminary interviews, company visits and questionnaires.

To ensure data validity, a mass balance calculation was performed. Incomplete mass balances were followed up with further consultation with the companies in the supply chain. For missing data on secondary and tertiary packaging materials, representative averages from the scenarios studied were used. In the case of site-specific materials, when a data point was only missing for one of the two product systems, this data point was considered identical for both scenarios.

2.5. Environmental impact assessment

The environmental impact of the product system was calculated using the ReCiPe Midpoint (H) V1.13 characterisation model. Combining the ReCiPe model and the general product category rules for packaging (EPD, 2019), the following midpoint categories were chosen: climate change, terrestrial acidification, freshwater eutrophication, marine eutrophication, photochemical oxidant formation (hereafter oxidant formation), ecotoxicity (land, freshwater and marine), human toxicity, natural land transformation, water depletion, mineral resource depletion and fossil fuel depletion. A hierarchic perspective was used based on the most common policy principles regarding time and exposure routes.

2.6. Economic viability assessment

Costs were collected from the bottle producers and translated to costs excluding taxes. Due to the low value of the bottle and the short time frame of bottle production, costs were not discounted to the present value.

Only gate-to-gate costs were included for both the virgin and recycled bottles (i.e. from the purchase of materials and resources to the bottle factory gate). The costs for filling, the cap production, and final product distribution are not shown in the analysis because these are not to be affected by recycled HDPE. The disposal costs depend on the local packaging waste management contribution, as determined by local funds. No taxes apply for the use of virgin plastic feedstock. For several European countries, taxes on non-reusable virgin plastic packaging will be implemented in the near future. For instance, Spain is considering a 0.45 €/kg tax for non-reusable plastic in Spain and United Kingdom a 0.2 GPB/kg tax for plastic packaging with less than 30% recycled content as of 2022. Companies in the Dutch packaging industry are required to pay a waste management contribution fee to contribute to the costs of recycling, but this fee strongly depends on the recyclability of the packaging and volumes being produced (Wtsglobal, 2022). A more stringent regulation of virgin feedstock use for plastic, may result in recycled HDPE being much more attractive for producers. Therefore, a hypothetical tax scenario of €0.45/kg of virgin plastic is considered in the results. The economic viability assessment only includes purchase costs for the bottle producer, as defined by equation (2). Table 2 shows the purchase costs for the virgin scenario. Table 3 shows the costs for the recycling scenario. A comparative analysis is given in the result section 3.2. To investigate if the blow moulding costs are affected by the different feedstocks, the results are also presented on a raw material level.

$$\text{Purchase costs} = Crm + Ce + Cl + Cps + Cpf + Ct + Cpw \quad [2]$$

Crm = Cost of Raw Materials.
Ce = Cost of energy consumption.
Cl = Cost of labour.
Cps = Cost of production site.
Cpf = Cost of production fees
Ct = Cost of transportation.
Cpw = Cost of production waste.

2.7. Social risk assessment

The generic social hotspots assessment was performed with SimaPro using the method provided with the PSILCA database. This method assigns different risk levels (ranging from no risk to very high risk) to the social indicators depending on the indicator's value. The risks of each activity are converted into *medium risk* values by applying characterisation factors based on Performance Reference Points (e.g. international

Table 2
Material purchase costs for bottle production associated with the use of virgin HDPE.

Process/material	Material purchase costs
Virgin HDPE	€0.083
Bottle manufacture	€0.103
Virgin plastic tax	€0.027
Total costs	€0.231

Table 3
Material purchase costs for bottle production associated with the use of recycled HDPE.

Process/material	Costs per FU
Recycled HDPE	€0.079
Bottle manufacture	€0.165
Virgin plastic tax	N/A
Total costs	€0.244

standards). The relevance of each unit process within the life cycle is reflected by the activity variable "worker hours", and therefore, the risk assessment results are given in the unit "medium-risk hours".

The subcategories provided by the PSILCA social LCIA method were classified into the subcategories and stakeholder categories of the S-LCA guidelines of UNEP/SETAC (UNEP-SETAC Life Cycle Initiative, 2009) according to the supporting document from the PSILCA database (Eisfeldt and Ciroth, 2018). The values were then aggregated assuming equal weighting for every subcategory to obtain final results per stakeholder category.

Social risks along the value chain were assessed by considering the production costs on raw material level for virgin and recycled HDPE. Since production costs were either not available or confidential, raw material purchase costs were used as a proxy. These costs were obtained in euros (€2018) and converted into USD 2006 using an inflation rate of 1.16 USD 2016/Euro 2018 (Pounsterling Live, 2022). This to make them compatible with the PSILCA database. The sectors "Manufacture of plastic products" and "Recycling industry" in the Netherlands were selected to represent the social risks of each product system, and the results are based on the inputs specified in Table 4.

3. Impact results

3.1. Environmental impact results

Table 5 shows the comparative environmental impact of the virgin and recycling scenario per midpoint impact category. The values obtained align with other studies investigating the impact of beverage plastic bottles. The climate change value obtained in this study is 0.479–0.589 kg CO2 eq. for an HDPE bottle weighing 67.5 g (with cap), which aligns with values reported in the literature ranging from 0.091 to 0.156 kg CO_2 eq for PET water bottles weighing 19.1 g on average (Tamburini et al., 2021). Table 5 also shows the relative impact of the two scenarios in percentage. The recycled bottle performs better in all chosen midpoint categories. The most significant improvements are found in the categories; climate change(-19%), freshwater eutrophication (−29%), ecotoxicity (−24%), water- (−19%), metal- (−22%), and fossil depletion (−18%). This better performance is due to the omission of oil extraction and processing into virgin HDPE, which avoids the impacts associated with these processes (i.e. climate change, acidification, oxidant formation, and fossil depletion). Human and ecotoxicity impacts are lower for the recycled bottle due to the less incineration at the EoL, resulting in less toxic gasses being released into the environment. The water depletion of the virgin scenario is higher due to the amount of water necessary to extract oil and process it into plastic.

Fig. 2 shows the contribution of the different life cycle activities the impact of one virgin bottle. The final impact of the virgin scenario is mostly dominated by the bottle production stage, followed by the distribution stage and filling (Fig. 2). Bottle production causes most of the impacts in the midpoint categories freshwater eutrophication (43%), water depletion (58%), metal depletion (44%) and fossil depletion (54%). Distribution has the highest contribution in the categories terrestrial acidification (47%), oxidant formation (54%), human toxicity (30%) and land transformation (68%). Filling contributes the most to the category marine eutrophication (52%), and the bottle incineration in climate change (31%) and ecotoxicity (48%).

Table 4
Estimated production costs per industry in USD 2006.

Industry	Production costs
Manufacture of rubber and plastic products/Industries/NL	$0.074
Recycling/Industries/NL	$0.071

Table 5
Characterised results of the virgin and recycling scenario per functional unit (i.e., one bottle) and their differences per impact category.

Midpoint impact category	Virgin scenario	Recycling scenario	Difference	Difference (%)
Climate change (kg CO2 eq/FU)	5.89E-01	4.79E-01	−109.16 g	−19%
Terrestrial acidification (kg SO2 eq/FU)	1.50E-03	1.38E-03	−0.12 g	−8%
Freshwater eutrophication (kg P eq/FU)	7.28E-05	5.15E-05	−0.02 g	−29%
Marine eutrophication (kg N eq/FU)	1.54E-04	1.46E-04	−0.008 g	−5%
Oxidant formation (kg NMVO/FU C)	1.92E-03	1.76E-03	−0.16 g	−8%
Ecotoxicity (kg 1,4-DB eq/FU)	2.68E-02	2.05E-02	−6.32 g	−24%
Human toxicity (kg 1,4-DB eq/FU)	1.52E-01	1.30E-01	−21.80 g	−14%
Land transformation (m^2/FU)	9.01E-05	8.66E-05	−0.003m2	−4%
Water depletion (m^3/FU)	3.36E-03	2.73E-03	−0.63m3	−19%
Metal depletion (kg Fe eq/FU)	2.19E-02	1.71E-02	−4.80 g	−22%
Fossil depletion (kg oil eq/FU)	1.98E-01	1.63E-01	−34.78 g	−18%

Fig. 3 shows the contribution of the different life cycle activities the impact of one recycled bottle. The final impact of the recycled scenario is mostly dominated by the distribution stage, followed by the bottle manufacture stage, and filling. Distribution contributes most to the midpoint categories climate change (33%), terrestrial acidification (51%), oxidant formation (59%), human toxicity (35%), land transformation (71%) and metal depletion (34%). Bottle production causes most of the impacts in the categories water depletion (48%) and fossil depletion (44%). Filling is scoring highest in the categories freshwater- and marine eutrophication (43–55%), and the bottle incineration in ecotoxicity (39%).

3.1.1. Sensitivity against recycling rate

The sensitivity of the results against the recycling rate is shown in Fig. 4. It shows that the higher the recycling rate, the higher the environmental benefits of the recycled scenario. Nevertheless, under a 30% recycling rate, the recycling scenario is still scoring better in all midpoint categories (e.g., −14% climate change, −22% freshwater eutrophication, and −18% ecotoxicity). This indicates that even in countries with lower recycling rates, the recycled scenario will result in environmental impact reductions. Under a 75% recycling rate, high reductions in impact are found in respect to the virgin scenario (e.g., −35% climate change, −55% freshwater eutrophication, and −45% ecotoxicity). High recycling rates result in more material being maintained each cycle, avoiding virgin HDPE production and incineration. Considering the steady increase in recovery rates among the countries of Europe, the environmental benefits of using recycled HDPE may be amplified in the future.

3.1.2. Sensitivity against quality losses

The results sensitivity to quality losses is given in Fig. 5. It shows that the less recycling cycles, the lower the environmental benefits of the

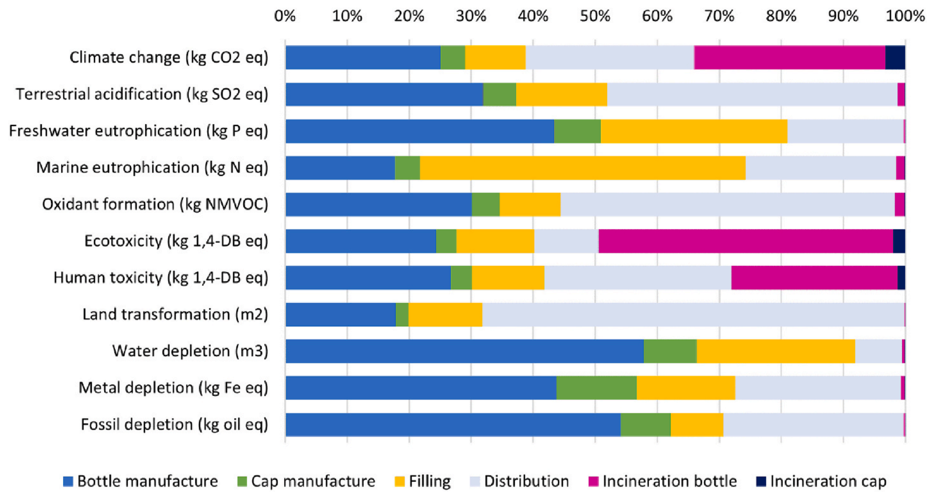

Fig. 2. Contribution of different life cycle activities to the final impact of the virgin scenario.

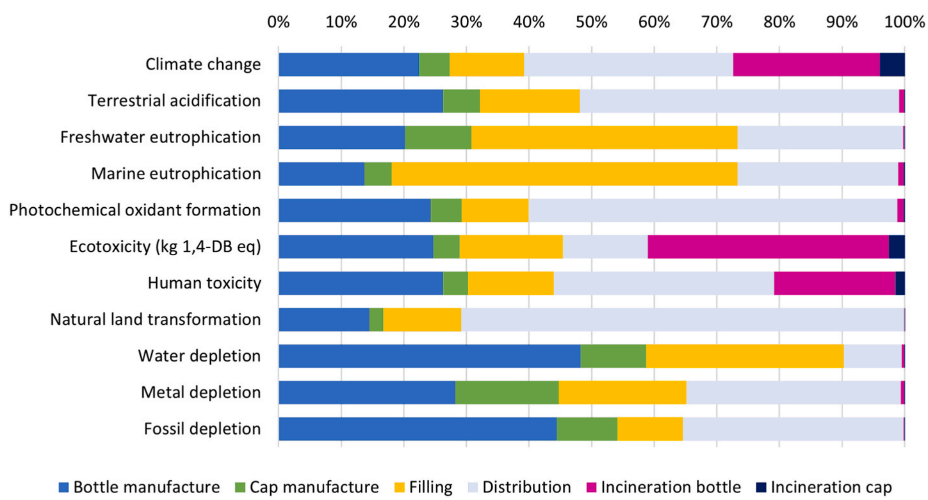

Fig. 3. Contribution of different life cycle activities to the final impact of the recycled scenario.

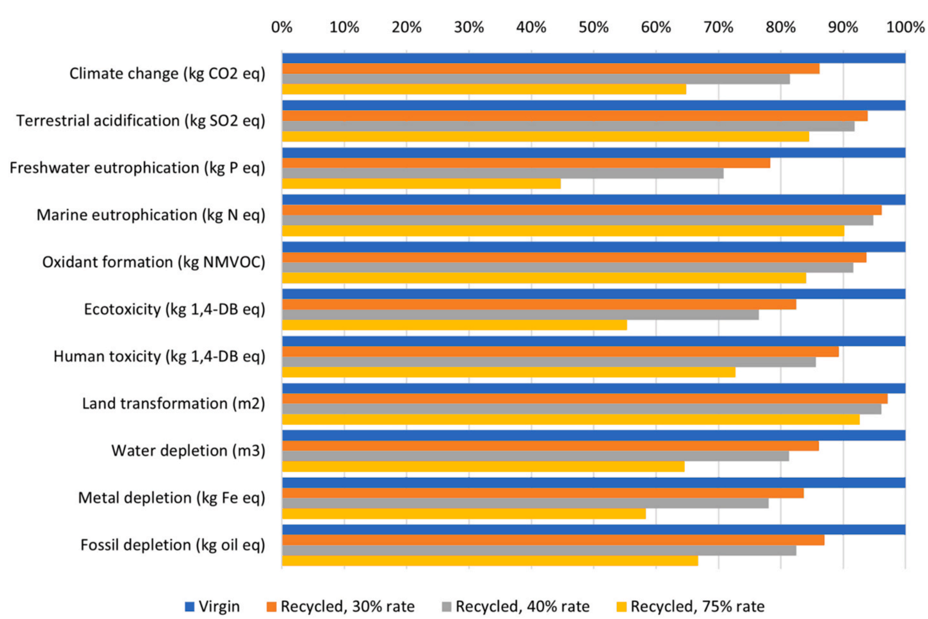

Fig. 4. Sensitivity results for low-end (30%) and high-end (75%) recycling rates.

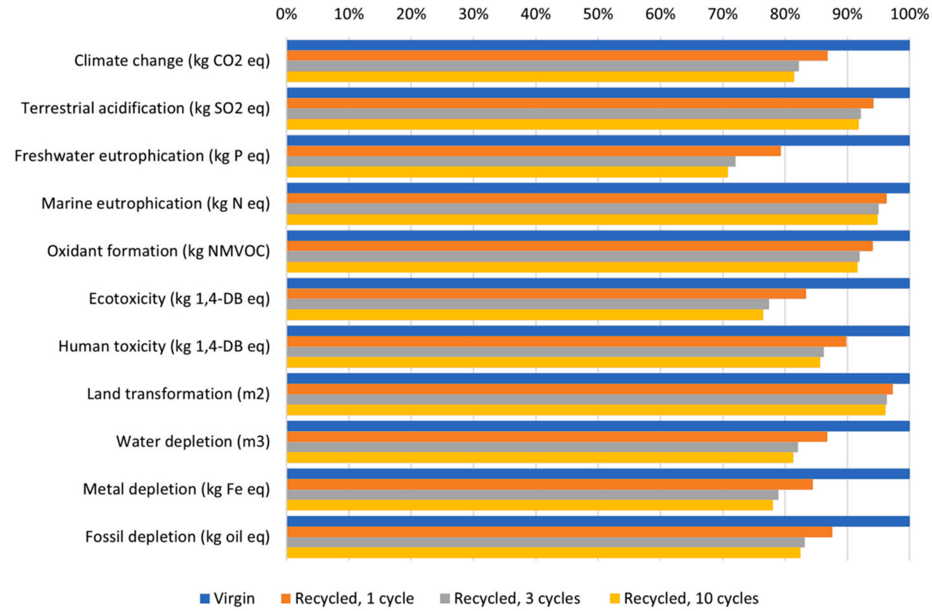

Fig. 5. Sensitivity of the results to fewer recycling cycles (one or three cycles).

recycled scenario. Despite only three recycling cycles being completed, the recycling scenario is scoring better in all midpoint categories (e.g., −18% climate change, −28% freshwater eutrophication, and −23% ecotoxicity). The recycling scenario is still scoring better in all categories when only one recycling cycle is completed (e.g., −13% climate change, −21% freshwater eutrophication, and −17% ecotoxicity). Although to a smaller degree, the benefits of recycling are still present when material or quality degradation is impairing the amount of recycling cycles. Most benefits are captured after completing at least three recycling cycles, indicating that quality losses have a strong effect on the environmental performance when less than three recycling cycles are completed.

3.2. Economic viability results

Fig. 6 and Fig. 7 show the life cycle costs of scenarios on material and product level. The purchasing cost of recycled HDPE is 6% lower than of virgin HPDE (0.079 versus 0.083 euro), but may vary per supplier. A hypothetical tax of €0.45/kg of virgin plastic would result in recycled HDPE being 40% less costly than virgin HDPE (0.079 versus 0.110 euro) (Fig. 6). The costs for bottle manufacture (blow-moulding) are 38% higher (0.165 versus 0.103 euro) (Fig. 7). The environmental inventory indicated that slightly more electricity (0.01 kWh) is consumed per bottle in the moulding process of the recycled bottle, pointing to a less energy-efficient process. The higher energy use could also be due to the

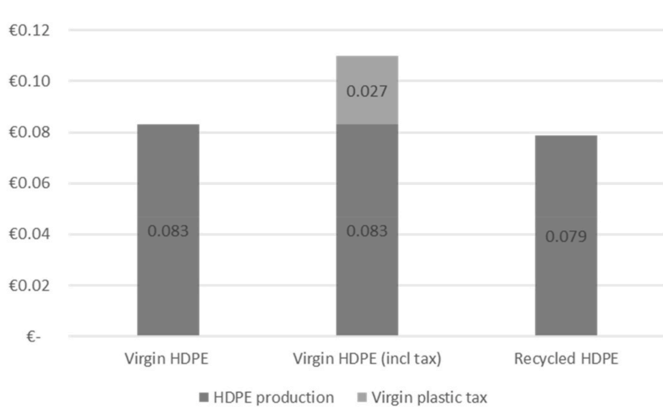

Fig. 6. Economic viability results on material level referenced to one FU.

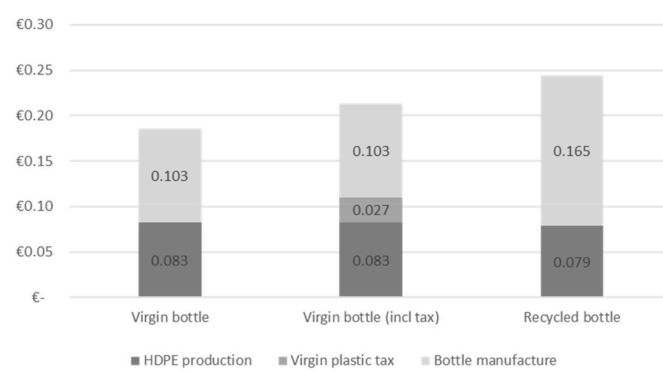

Fig. 7. Economic viability results per FU.

specific characterises of both production lines. Oblak et al. (2015) tested the mechanical properties of recycled HDPE and found that changes in melt pressure and extrusion torque of the recycled material lead to a more energy-demanding process when compared to virgin material. A hypothetical tax of €0.45/kg of virgin plastic would result in the total recycled bottle costs (product costs) being 13% higher (0.244 versus 0.213 euro). Without the virgin plastic tax the recycled bottle costs are found to be 24% higher for the recycling scenario (0.244 versus 0.186 euro) (Fig. 7).

3.3. Social risk results

The total social risks for the plastic product manufacture compared to the recycling industry are given per stakeholder category in Table 6. The results indicate higher social risks associated with the recycling sector than for the plastic products one in every stakeholder group. The differences in risk range from 16% to 236%, depending on the stakeholder group. Most of these risks occur in other sectors and countries than The Netherlands (i.e., indirect risks) because of the demand for goods and services from the Dutch sectors to other sectors in different parts of the world. Only 5% and 12% of the total life cycle risks occur directly in the Dutch plastics- and recycling sector respectively (i.e., direct risks). If we look only at the direct risks per sector, the risks of the plastic products manufacturing sector are actually higher than the risks in the recycling sector.

Table 6
Aggregated social risks per stakeholder for manufacturing $0.074 of plastic products and recycling $0.071 of products.

Stakeholder category	Unit	Recycled bottle		Virgin bottle	
		Total life cycle risks	Direct risks recycling sector	Total life cycle risks	Direct risks plastic product manuf. Sector
Worker	W med rh	0.73	7%	0.48	17%
Local communities	LC med rh	0.37	3%	0.28	7%
Society	SO med rh	0.18	0.1%	0.05	0.5%
Value chain actors	VCH med rh	0.44	6%	0.38	12%
Consumers	CO med rh	0.005	0.5%	0.004	0.1%

Most of the indirect risks of the recycling sector are happening in Indian sectors (at least 30% of the risks), followed by the Chinese and Pakistani sectors. These indirect risks are most probably caused by the informal recycling of e-waste happening in these countries, which is not necessarily related to the plastic recycling industry under study. Several papers have reported on the informal recycling in these countries, where unskilled workers are performing recycling through crude processes in India (Pandey and Govind, 2014), China (Li et al., 2011) and Pakistan (Umair et al., 2015) (e.g. manual dismantling, burning, dumping and dipping in acids to extract gold and other precious metals). In the case of the plastic products sector, most of the indirect risks occur in Dutch sectors, but also in Chinese and Indian sectors.

Fig. 8 shows the differences in direct and direct risks per subcategory. According to the PSILCA database, recycling $0.071 of plastics involves higher life cycle social risks than manufacturing $0.074 of plastic products for every subcategory except for workers' health and safety, local employment and migration. However, these risks are dominated by indirect risks taking place outside the Netherlands. Direct

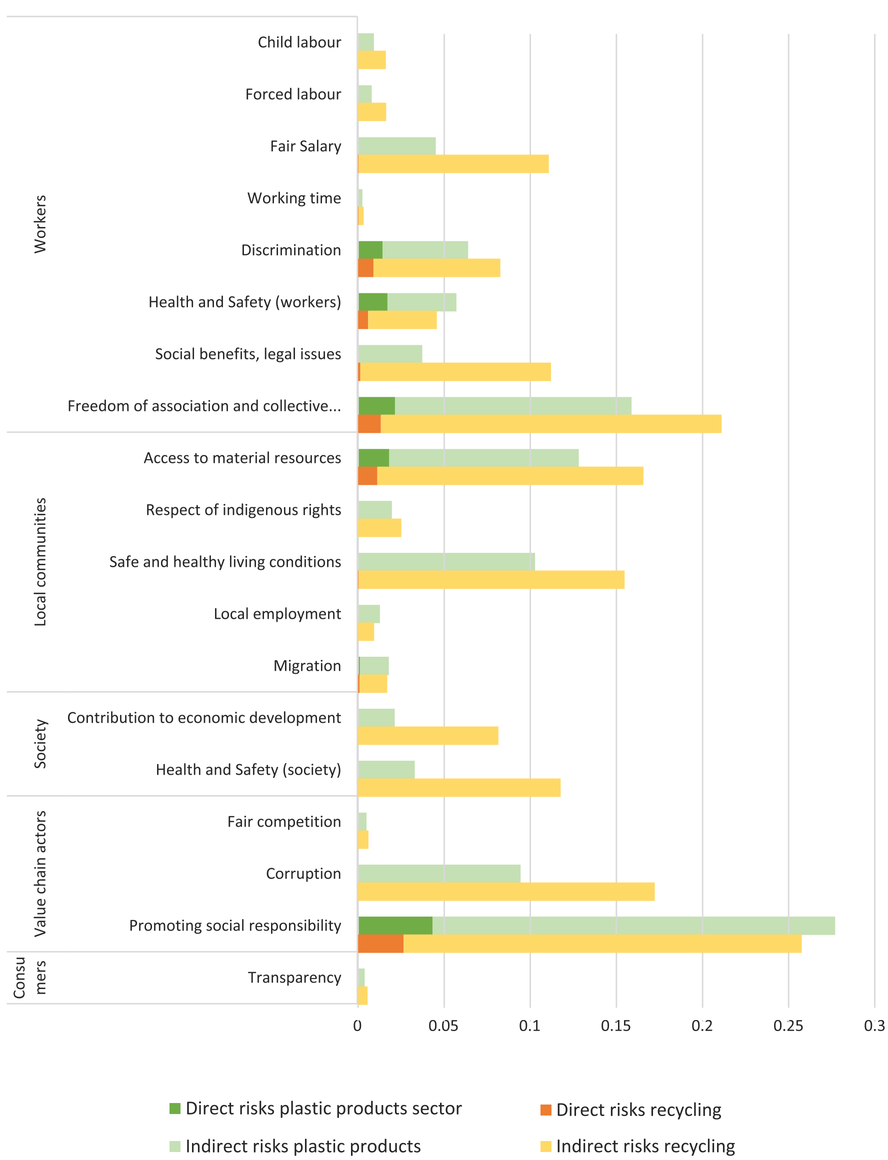

Fig. 8. Comparative social risks per subcategory for manufacturing $0.074 of plastic products and recycling $0.071 of material.

risks in The Netherlands are mostly found in the subcategories of discrimination, health and safety, freedom of association, access to material resources, and promotion of social responsibility. For each of these five subcategories, the risk in the plastic products industry is higher than the risks in the recycling industry. In the case of the virgin bottle sector, the presented risks are probably more representative than the recycling sector since it focuses on plastic products. Given the heterogeneity of the recycling sector in the PSILCA database, the indirect risks found can easily be related to other products than recycled plastic bottles. For a more accurate understanding of the social risks of recycling different products and materials, a higher disaggregation is needed for the recycling sector.

4. Discussion and conclusions

4.1. Discussion of results and their limitations

This study showed that when assuming a recycling rate of 75%, using recycled HDPE for plastic bottles instead of virgin HDPE leads to 35% lower impacts in climate change and a 7%–55% reduction in other environmental impact categories. This result aligns with other LCA studies that focus on recycled HDPE for packaging applications. Tonini et al. (2021) found a carbon footprint reduction of 32% when using 1 kg of recycled HDPE instead of 1 kg of virgin HDPE in rigid packaging applications (recycling rate of 60%). Another study found a 35% carbon footprint reduction for 1 kg postconsumer recycled HDPE resin as compared to 1 kg of virgin HDPE resin (Franklin Associates et al., 2018). This same research reported a 29% reduction in water consumption, a 24% reduction in acidification and an 18% reduction in oxidant formation. Only eutrophication showed an increase of 1%, which may be caused by a difference in the characterisation method.

The calculated purchase costs of recycled HDPE were lower than those of virgin HPDE, but the costs of moulding the recycled bottle were higher than for moulding the virgin bottle. The quality of recycled plastics strongly depends on the quality of sorting systems. Impurities may affect the physical properties of recycled plastic, which may cause difficulties when blow moulding (Brouwer et al., 2018). This highlights the importance of high-quality mono collection systems to enable similar physical properties and applications for recycled plastics (Thoden van Velzen et al., 2021). When produced in European countries where virgin feedstock taxes apply, recycled HDPE becomes less costly than virgin HDPE, showing the relevance of such environmental taxes.

The social risks analysis suggested higher social risks from the recycled bottle due to higher indirect risks in the Dutch recycling sector, probably due to the imports and exports of waste from countries like China, India and Pakistan, with high rates of informal recycling. This result also aligns with the available literature on the social impacts coming from the recycling sectors in these countries, as indicated in the introduction of this paper. Nevertheless, the comparative risk assessment was done through a simplistic approach, with a difference in resolution between the two sectors. The recycling sector is not focused on plastic recycling and covers the sector as a whole, so the indirect risks found in this study can easily be related to other recycled products than plastic bottles. Although this analysis could already indicate which social risks and opportunities might lay in both supply chains, a higher disaggregation of the recycling and waste management sectors is needed for a better understanding of social risks in circular supply chains. Social risk analyses are helpful in understanding where the social hotspots of a supply chain may lay but are not aimed at describing the actual social performance of specific companies in a supply chain. The analysis here performed should be considered a first step to mapping potential social impacts. For deeper and more precise insights, the analysis must be followed by an S-LCA investigating site-specific data on social performance.

Despite the comprehensiveness of this study and its alignment with the international standards de ISO 14040 and ISO 14044, some additional limitations were identified. For the environmental LCA, the virgin HDPE production and incineration processes were modelled with background data, which decreases the consistency of the data collection approach. Although time-consuming, the analysis would have been more accurate if site-specific data could have been obtained for these stages. Secondly, the midpoint category 'water depletion' within the ReCiPe characterisation method represents the total amount of water used and does not consider water scarcity in different geographic areas (Goedkoop and Huijbregts, 2013). The water scarcity in the regions under study is not expected to vary and would not lead to different conclusions as the ones found.

4.2. Assessment of circular strategies

A wide range of circularity metrics has been arising over the last years to understand how well circular strategies can be sustainably implemented in current human activities. Life cycle methodologies have been criticised for being too complex to be fully applied by companies and at the same time, not being able to consistently model recycling loops. Others consider LCA the best methodology to assess the environmental impacts (such as climate change impacts) of products and services (European Commission, 2013) and LCA is the most used methodology to assess the impacts of circular products and services (Corona et al., 2019). Newly developed metrics focused on material circularity can indicate to what extent circular strategies can be applied (e.g., the MCI) but are unable to indicate the effects on climate change, other environmental impacts, or social impacts. This study shows how the LCSA approach is suitable for assessing and comparing the impacts and benefits of circular strategies in every sustainability dimension. The adopted system expansion approach allows for fully integrating the properties of the recycled content and recycled output of the product. This provides a complete picture of the effects of circular strategies and overcomes the multifunctionality issue so commonly challenging in the LCA literature (Schrijvers et al., 2016). Recycled contents of 100% are recommended to achieve the highest environmental benefits, and used to model the recycled bottle. Although a sensitivity analysis on lower recycled content was not included in this study, such an analysis would lead to similar conclusions as the ones obtained from the sensitivity analysis provided on recycling rates. Less recycled content is expected to result in lower environmental benefits.

5. Conclusion

Due to the environmental issues associated with plastic production and incineration, companies are increasingly using recycled plastics under the assumption that this circular strategy improves the sustainability performance of their products and packaging. Although the European plastic demand is the highest for polyethene, little research has been conducted to measure the actual sustainability of recycled feedstock. In this study, environmental, economic and social impact measurement techniques have been integrated for the first time to assess the sustainability performance of recycled HDPE in the packaging life cycle.

This study shows that the use of recycled HDPE has the potential to significantly reduce not only the carbon footprint, but also a broad range of environmental impacts. These environmental benefits increase significantly when assuming higher recycling rates. Although quality losses is often a factor of uncertainty in the benefits of plastic recycling, this study has also shown that quality losses only significantly impair the environmental benefit when less than three recycling cycles are completed. From an environmental perspective, this study thus suggests that the use of recycled feedstock will have a positive contribution to the European Strategy for Plastics in a Circular Economy.

The environmental assessment was complemented with an LCC and social risk assessment to explore trade-offs between sustainability dimensions and avoid burden shifting. Although the purchase costs of recycled HDPE were found to be lower than that of virgin HPDE, such costs strongly vary per supplier and quality demand. An LCC from a producer perspective is valuable to assess feasibility for adopters but should be applied in short timeframes. For the first time, the scope of a life cycle impact assessment of recycled packaging was broadened towards the social dimension. The social risks of the recycling sector in general were found to be higher than for virgin plastic production, suggesting that recycling strategies may shift life cycle impacts from the environmental to the social dimension. However, due to the heterogeneity of the database used, these risks can be caused by recycling other materials than plastics.

The authors of this paper would like to call for further efforts to develop higher disaggregated databases of the recycling and waste management sectors, to better understand the social risks in circular supply chains. These risk assessments should be seen as a first scan to identify the type and magnitude social risks, and should be complemented with full S-LCA directed at these hotspots. Despite such methodological challenges, this study provides an example on how the LCSA approach can be used to assess the effects of circular strategies in the three sustainability dimensions.

CRediT authorship contribution statement

Marjolein Papo: Conceptualization, Methodology, Investigation, Writing – original draft, Visualization. **Blanca Corona:** Methodology, Validation, Resources, Writing – review & editing, 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

The data that has been used is confidential.

Annex I.

Table A1
Values of x (recycled bottles) and n (total amount of bottles) for different values of i (amount of recycling cycles) and recycling rates (30%, 40%, 75%)

Recycling rate	i = Amount of recycling cycles	X = Amount of recycled bottles produced	n = Total amount of bottles produced (recycled + virgin)
30%	0	0	3.333
30%	1	1.0	4.333
30%	3	1.390	4.723
30%	10	1.429	4.762
40%	0	0	2.50
40%	1	1.0	3.50
40%	3	1.560	4.060
40%	10	1.667	4.166
75%	0	0	1.333
75%	1	2.333	1.333
75%	3	3.464	2.313
75%	10	3.775	5.108

Simplification of Eq. 1 through a finite sum of geometrical series.

$$n = \frac{1}{R} + \sum_{n=1}^{x} R^{(n-1)} = \frac{1}{R} + \frac{1 - R^x}{1 - R} = \frac{1 - R^{x+1}}{R - R^2} = \frac{\frac{1}{R} - R^x}{1 - R}$$

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