Environmental benefits of material-efficient design: a hybrid life cycle assessment of a plastic milk bottle

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Abstract

Over a million plastic bottles are bought around the world every minute, 20,000 every second. While there is agreement that single-use plastics should be phased out, records of plastic pollution point in the opposite direction. Until the phasing-out of single-use plastics materialises, there is an immediate opportunity to mitigate the environmental impacts of plastic use through better design, that is to improve material efficiency without affecting function or performance. In this article, we focus on a standard plastic milk bottle, which we redesign based on a previously developed shape factor for the sustainability of forms, achieving a ~13% material reduction. Environmental benefits are evaluated globally through hybrid life-cycle assessment (LCA) both in the 10 countries in which the original plastic bottle is used and the other 22 countries upstream in the supply chain. Results obtained through hybrid LCA are 17.3% higher than those obtained through process-based LCA. Environmental benefits arising in the 22 countries in the upstream supply chain are twice as much as those observed in the 10 countries where the original bottle is used. Overall, our findings show a potential annual reduction of ~1.9 Mt CO_2e, equivalent to approximately 415,000 cars taken off the road, thus proving the substantial and viable benefits that can be achieved by material efficiency through redesign.

Keywords: process-based; input-output; hybrid; life cycle assessment (LCA); plastic; waste; HDPE; optimisation; design.

1. Introduction

The Brand Audit Report 2020 from the Break Free From Plastic (BFFP) initiative has identified Coca-Cola, PepsiCo, Nestlé and Unilever as the world's top plastic polluters (BFFP, 2020). All of them are signatories to the New Plastics Economy Global Commitment—an initiative by the Ellen MacArthur Foundation (EMF) in collaboration with the United Nations Environment Programme (UNEP). So far this initiative has not led to significant pollution reductions. Specifically, the EMF/UNEP 2020 report on the global commitment (EMF and UNEP, 2020) shows only a 1.9% share of reusable plastic packaging, a mere 0.1% increase from the previous year. At a time when plastic pollution has turned into a "plastic pandemic" (Duer, 2020), it is clear that single-use items are unlikely to leave our lives (and economy) any time soon.

While this and other global initiatives (e.g. the Single Use Plastics Elimination Initiative from The Travel Corporation; Global Plastics Action Partnership) bourgeon and progress, yielding tangible behavioural shifts and restructuring global and local supply chains, an immediate option for short-term action is to investigate the possibilities offered by material efficiency, such as minimising the amount of plastic used in a product without affecting function or performance. Allwood (2018) presents an effective framework for delivering resource efficiency and distinguishes between techno-optimistic and physically realisable options. The techno-optimistic include, for instance, the highly marketed circular economy as well as carbon capture and storage (CCS) technologies, but he shows that these fail to meet the scale of the challenge (Allwood, 2018). Conversely, the physically realisable options include avoiding over-design and extending a product's life. In the case of single-use plastics, the latter is an oxymoron, and therefore avoiding over-design is the chiefly promising solution. Designing products with less material input is one of the four major strategies for reducing material demand identified by Allwood et al. (2011).

In this article we do a deep dive into the material efficiency of plastic bottles, and more specifically on milk bottles made from high-density polyethylene (HDPE), such as those shown in Figure 1. The choice is due to the widespread use of this design in many countries and the suitability in terms of overall shape and size to undergo a redesign exercise with material efficiency in mind whilst keeping realistic shapes as outputs. Additionally, milk is a daily product consumed by millions around the world, thus representing a significant problem area.

Figure 1: Examples of milk bottles considered for, and optimised in, this research (pictures taken from online supermarkets' websites). From right to left, these are bottles found in Israel, South Africa, New Zealand, and Thailand. The full list of countries surveyed is given in the Supplementary Information.

We evaluate an alternative (optimised) design for the bottles' shape, which provides the same function to the user in terms of volume of milk contained while leaving supply chains and retailing mostly unaffected, thus minimising disruptions to adjust to the new design. Our optimisation is based on a previously developed scale-independent shape factor for sustainable forms (D'Amico and Pomponi, 2019) that allows to benchmark alternative shapes against the theoretically absolute optimum. The selected improved design is then translated into environmental impacts through an advanced hybrid life-cycle assessment (LCA). The results obtained represent the main contribution of the article, which unfolds as follows. In the next section, we review previous works that are useful within the scope of this research. Methods follow where we present in detail the LCA elements of this work (i.e. process-based, input-output assisted hybrid, and path-exchange hybrid) and the approach used to quantify material savings. These are followed by the results; discussion and conclusions end the paper.

2. Previous works

One of the first quantifications of resource requirements, emissions and waste flows (now recognised as a partial ancestor of LCA) was carried out by one of the world's top plastic polluters, Coca-Cola, in 1969 (Guinée et al., 2011). Since then, industry and scholarly interest in environmental impact assessment of plastic packaging has steadily increased. This work does not focus on the broader area of environmental performance and impacts of plastic products but rather on the smaller area around plastic bottles. However, even though a systematic review of LCAs of plastic bottles is beyond the scope of this work, we could locate only few studies[1] focused on HDPE bottles and underpinned by a cradle-to-grave life cycle perspective. The most transparent, and therefore useful, contributions are listed below to both understand manufacturing processes and supply chains and to identify additional data sources to compare our own findings.

Lehmann et al. (2005) focused on a comparative LCA (using SimaPro) of HDPE and polyethylene terephthalate (PET) for bottling applications in the Swedish context. Their functional unit (FU) was 1 kg of polymeric material, under the assumption that the same quantity of PET and HDPE is used in a bottle. They adopt an approach based on 'ecopoints' for environmental performance (called endpoints in LCA terminology). This however lacks transparency as it does not offer quantities of environmental impacts across different categories making results hard to interpret. In terms of carbon dioxide (CO_2) only, their results show an emissions intensity value for HDPE of 6.13 kg CO_2/FU. The authors state that the average bottle has an assumed weight of 53 g, thus fitting approximately 19 bottles in each kilogram of polymeric material. This coarse normalisation seems to suggest emissions values at the bottle level for HDPE of 0.322 kg CO_2/bottle.

A similar comparative focus can be found in the work of Scipioni et al. (2013) who analysed a laminated carton and an HDPE bottle through a process-based LCA with a cradle-to-grave system boundary in a seemingly European context. The weight of the HDPE bottle is not given (though its capacity is 1 L), and the authors report a mean greenhouse gas (GHG) emissions value of 0.262 kg CO_{2e}/bottle. The lack of full disclosure in background data, inventory and

[1] A search on Web of Knowledge for "life cycle assessment" AND "HDPE" AND "bottle" as a Topic (which searches within title, abstract, and author keywords) across all years and all indexes returned only 7 entries. More literature exists on polyethylene terephthalate (PET) bottles, which are not however the focus of this research.

system processes – often typical of process-based LCAs – limits the comparability of the numerical findings available in the existing literature.

Outside Europe, Singh et al. (2011) carried out a process-based, cradle-to-grave LCA in the context of the USA focusing on a functional unit of 1 gallon of HDPE-packaged milk, finding GHG emissions in the range of 1.27 – 1.81 kg CO_{2e}/FU (i.e. 0.335 – 0.477 kg CO_{2e}/L of packaged milk). More recently, Treenate et al. (2017) undertook a process-based, cradle-to-grave LCA of HDPE bottling, focused on Thailand, following the ISO 14040 standard. Their functional unit was one HDPE bottle weighing 35 g, but the authors also offer results per kg of HDPE pellets. They consider two end-of-life scenarios and obtain results of 0.89 (recycling) and 5.48 (incineration) kg CO_{2e}/kg of HDPE pellets. Under the same assumption of milk bottle weight used above (53 g), these would translate into 0.046 and 0.288 kg CO_{2e}/bottle. The relatively low value obtained for recycling is caused by the LCA assumption of attributing the recycling benefit to the original product. This is still a debated issue in the LCA of products, though some conclusive clarity comes from the European standard EN 15978 (Module D). According to this, benefits and loads occurring beyond a product's life cycle should be reported separately and not attributed to the product being assessed. With this in mind, the 0.046 kg CO_{2e}/bottle is an outlier among the values retrieved in the literature as it comes from unusual interpretation of LCA approaches and lacks compliance with some of the existing standards.

Outside academia, WRAP (2010, as reported in Treenate et al. 2017), also considered two potential end-of-life scenarios for HDPE disposal (recycling and incineration) and reported values of 2.74 and 4.7 kg CO_{2e}/kg of HDPE pellets respectively, which would translate into 0.144 and 0.247 kg CO_{2e}/bottle. These values conclude the scant data available in published literature on HDPE bottles.

The studies reviewed in this section show that, despite the ubiquitous nature of HDPE plastic bottles, the academic literature is not as rich as one would expect. Despite its scarcity, and even with the methodological variation highlighted above, there seems to be good consistency in the data with the cradle-to-grave average of the figures above at 0.3 kg CO_{2e}/1L bottle, which is not far from most individual contributions (lowest and highest values excluded as outliers).

3. Methods

3.1 Shape factor and material efficiency

Like for any container, the main function of a bottle is to carry a substance. Therefore, its material efficiency can be quantified by looking at the material requirement (in the bottle-making process) per unit of substance being carried. The volume of material required to make one bottle corresponds to the product between the bottle's surface area, and the average wall thickness. If we momentarily ignore such thickness (i.e. take it equal to unity) then we could account for material efficiency simply as the ratio between the bottle's surface area, S, and the substance volume, corresponding to the bottle's inner volume, V. Clearly such a compactness metric, S/V, makes a strong case for having fewer bottles of bigger size in circulation, since this would reduce material requirement (bottle's surface) per unit volume of carried substance. This, of course, ignores the fact that bigger bottles also require thicker walls for structural reasons. However, if we limit ourselves to the case of a fixed-volume bottle, then it does make sense to look for optimisations of its shape such that material requirements can be reduced by just minimising its surface area. In more formal terms, the optimisation criteria can be stated as to find a bottle's shape that minimises the surface area (optimisation objective) while subject to a fixed inner volume (optimisation constraint). Therefore, an appropriate metric to measure material efficiency of different shapes (all having the same inner volume) can be set as the ratio between the surface area of the bottle's shape (S) and surface area of a sphere (S_{sphere}) of volume V, that is also the bottle's inner volume:

$$Shape\ factor = \frac{S}{S_{sphere}(V)} \quad ; \qquad S_{sphere}(V) = \sqrt[3]{36\pi V^2}$$

For this study, we selected the shape of a 2-pint (1.136 L) milk bottle commonly found in UK stores. Its weight and components are given in the Supplementary Information. This is shown in Figure 2(a), along with the optimised design (Figure 2(b)) which requires 12.8% less material than the original, and the theoretically optimal spherical shape for which a 30% material reduction is shown. This last percentage represents the absolute limit (corresponding to a shape factor = 1) that could be theoretically achieved in terms of material efficiency for this specific 2-pint bottle design. While the proposed optimised bottle design only achieves a moderate 12.8% material reduction (therefore less than half the theoretical limit) its cylindrical body with tapered neck is also a common design for a milk bottle in other countries and for other liquids (e.g. freshly squeezed fruit juices). This should assure us that transportability supply chains and customer's usability should be minimally affected by the proposed change in design.

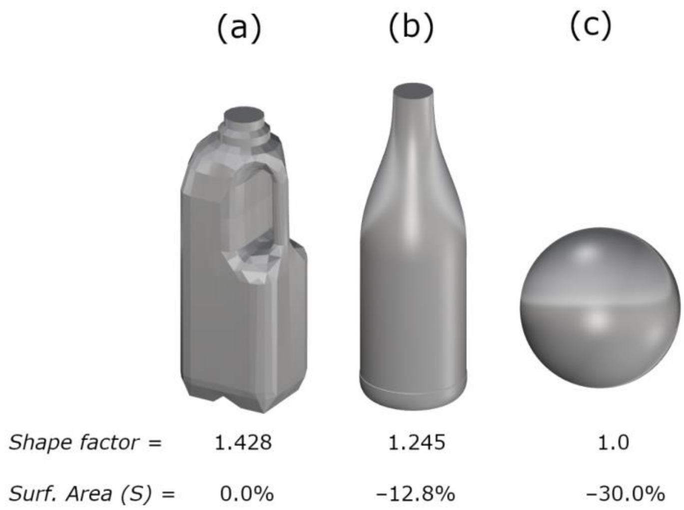

Figure 2: Shape factors and corresponding material (surface area) reductions of three shapes having internal volume V = 0.001136 m^3, i.e. 2 pints (1.136 L) volume capacity. (a) A handle jar is the reference design; (b) A cylindrical bottle is the optimised design; (c) A sphere is theoretically optimal.

In this work, we apply hybrid LCA, which is an approach integrating process analysis (PA) and multi-region input-output (MRIO) analysis ((Bullard et al., 1978; Heijungs and Suh, 2002; Moskowitz and Rowe, 1985; Suh et al., 2004; Suh and Nakamura, 2007)). Both PA and MRIO suffer from inherent limitations: PA generally focuses on high-quality, high-accuracy specific data for the process or product under examination but accounts for very little of the upstream impacts occurring in the supply chain behind the specific object of analysis, thus resulting in a *truncation error*. MRIO does not suffer from this due to its inherent completeness achieved by the use of data on the whole economy, but it cannot grasp the granularity required by the millions of products and processes that make up the economy, thus resulting in an *aggregation error*. Hybrid LCA combines the best of two worlds, in that i) PA is used to assess the functional unit under consideration and its immediate upstream processes with specificity and accuracy, whilst ii) MRIO analysis completely and exhaustively covers the remaining higher-order upstream processes. Due to this, hybrid LCA is not straightforwardly comparable with traditional (process-based) LCAs, as the methods, system boundaries and governing equations are different and it generally results in higher impacts due to the greater completeness offered by including upstream impacts in the assessment. On a debate about the merits and shortfalls of process and IO analysis see (Pomponi and Lenzen, 2018; Yang et al., 2017). However, hybrid LCA is well suited to the problem being studied here since (1) HDPE bottles can be characterised by high quality process data in terms of material types and quantities as well as manufacturing processes and (2) plastic pellets are coming from global supply chains, thus justifying the use of MRIO background data to accurately grasp impacts.

3.2 Process LCA

The process-based LCA adopts the functional unit of a single-use standard milk bottle that holds 1 litre of milk, under a linear, cradle-to-grave approach. A standard milk bottle includes the HDPE plastic bottle and cap, the aluminium seal and the label (LDPE film). The system boundaries include raw material extraction and processing, manufacturing, end-of-life waste management and benefits, and the transport between each stage, as shown in Figure 3. Note that the use phase of the milk bottle is not included in this analysis as it is assumed to be consistent across the scenarios. Table 1 details the life-cycle inventory (LCI) for the standard milk bottle, i.e. before optimisation, and includes the weight of the materials that are used and processed as well as the transport distances, in tonne-kilometres.

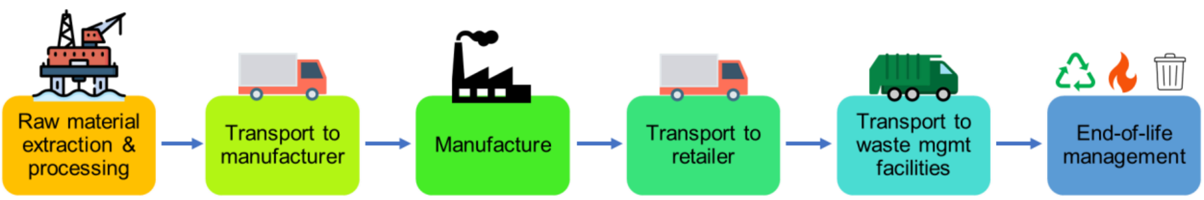

Figure 3: Life cycle stages considered in the process-based Life Cycle Assessment (LCA)

The process-LCA element of this research was conducted using the SimaPro software (v9.0.0.49) equipped with the ecoinvent database (v3.5). Table 1 shows the generic ecoinvent processes used in this analysis; country/region-specific processes were used where possible to give more accurate input data for substitution into the input-output analysis. Ten countries/regions were assessed using the best available data in the ecoinvent database and country-specific electricity carbon intensity factors and transport distances. In terms of raw material extraction, manufacture, and waste processing, ecoinvent only has processes for five main regions for the materials under evaluation: Europe (RER), the United States (US), Switzerland (CH), 'Rest of World' (RoW) and 'Global' (GLO). The RoW processes represent the production volumes that cannot be accounted for by specific countries when looking at total GLO production. Table 2 provides a breakdown of the countries considered and the associated ecoinvent region used. A full account of the LCI and processes used is provided in the Supplementary Information.

Table 1: Cradle-to-grave life cycle inventory, including generic ecoinvent processes. Country specific processes are used for the life cycle impact assessment.

Material/process	*Amount*	*Unit*	*Generic ecoinvent process*
HDPE bottles	26.2	g	*Polyethylene, high density, granulate*
HDPE rigids (bottle lid)	1.5	g	*Polyethylene, high density, granulate*
Aluminium (seal)	0.2	g	*Aluminium, primary, ingot*
LDPE film (label)	0.8	g	*Packaging film, low density polyethylene*
PE manufacture	28.5	g	*Blow moulding*
Aluminium manufacture	0.2	g	*Sheet rolling, aluminium*
Transport, refrigerated lorry	4.3	kgkm	*Lorry with refrigeration machine, 7.5-16 ton, EURO4, R134a refrigerant, cooling*
Municipal waste collection service	4.3	kgkm	*Municipal waste collection service by 21 metric ton lorry*
Recycling – Mixed Recycling Facility	12.2	g	*Treatment of waste polyethylene, for recycling, unsorted, sorting*
PE recycling process	11.9	g	*Polyethylene, high density, granulate, recycled*
Aluminium recycling process	0.2	g	*Treatment of aluminium scrap, post-consumer, prepared for recycling*
PE incineration	11.3	g	*Treatment of waste polyethylene, municipal incineration*
PE landfill	5.3	g	*Treatment of waste polyethylene, sanitary landfill*

Table 2: Countries considered in the process LCA and the regions associated with the ecoinvent processes.

Country	*ecoinvent region used*
Australia	*RoW/GLO*
Israel	*RoW/GLO*
Malaysia	*RoW/GLO*
New Zealand	*RoW/GLO*
South Korea	*RoW/GLO*
South Africa	*RoW/GLO*
Thailand	*RoW/GLO*
Ukraine	*RoW/GLO*
UK	*RER/CH/GLO*
USA	*US/RoW/GLO*

The LCIs for the 10 regions were used to conduct the life-cycle impact assessment (LCIA) to determine the Global Warming Potential (GWP), i.e. GHG emissions, of the functional unit in each region. The LCIA method used was the IPCC GWP 100a as it is internationally recognised and robust, containing climate change factors from IPCC with a timeframe of 100 years (IPCC, 2014).

The upstream impacts consist of the sum of the raw material extraction, manufacturing, and transport impacts of the components of the functional unit, i.e. HDPE, LDPE, and aluminium. The end-of-life impacts include the processing and transportation of the functional unit from a mix of waste management activities, as well as the benefits associated with material and energy recovery. The split assumes that 42% of the functional unit is recycled, 39.5% is sent for energy recovery (incineration), and 18.5% is sent to controlled landfill (includes landfill gas utilisation) (Plastics Europe, 2019). In the case of incineration and landfill, the energy recovered is credited to the system as a negative impact, offsetting fossil fuel use. For recycling, the benefits of recycling the materials, i.e. offsetting raw material production, are considered along with the burdens associated with preparing materials for recycling, i.e. transport, sorting, and processing.

3.3 Input-output-assisted hybrid LCA

The objective of this work is to change a particular transaction – plastic products as an input into dairy products – in an entire global supply-chain network. To this end, we use the Path Exchange method for hybrid LCA (Lenzen and Crawford, 2009). This method builds on pioneering work by Treloar (1997), and has been used by Baboulet and Lenzen (2010) for evaluating the environmental performance of a university, and by Wiedmann et al. (2011) for assessing an emerging wind power sector in the UK. The basis of this method is environmentally-extended multi-region input-output (EE-MRIO) analysis. There are numerous descriptions of this method and its constituents in the literature, so we will offer here a complete but brief explanation of our approach. The interested reader is referred to seminal work on multi-region input-output (MRIO) analysis (Isard, 1951; Leontief, 1953; Leontief and Strout, 1963); on the foundations of environmentally-extended IO analysis (Forssell, 1998; Leontief and Ford, 1970); and on modern MRIO frameworks (Tukker and Dietzenbacher, 2013).

Let **T** be an $N \times N$ matrix describing the *intermediate* transactions between N supplying and N receiving sectors[2] of an economy. Let $\mathbf{Y}$ be an $N \times M$ matrix describing the *final* transactions between N supplying sectors, and M final demand agents[3]. Then, total economic *output* ($N \times 1$) is $\mathbf{x} = \mathbf{T}\mathbf{1}^{\mathbf{T}} + \mathbf{Y}\mathbf{1}^{\mathbf{Y}}$, where $\mathbf{1}^{\mathbf{T}} = \{\underbrace{1,1,\ldots,1}_{N}\}$ and $\mathbf{1}^{\mathbf{Y}} = \{\underbrace{1,1,\ldots,1}_{M}\}$ are summation operators. Writing intermediate transactions as $\mathbf{T} = \mathbf{A}\mathbf{x} \Leftrightarrow \mathbf{A} := \mathbf{T}\hat{\mathbf{x}}^{-1}$, with the hat (^) symbol denoting vector diagonalisation, defines the $N \times N$ *input coefficients* matrix **A**. Total output can now be written as $\mathbf{x} = \mathbf{A}\mathbf{x} + \mathbf{Y}\mathbf{1}^{\mathbf{Y}} \Leftrightarrow \mathbf{x} = (\mathbf{I} - \mathbf{A})^{-1}\mathbf{Y}\mathbf{1}^{\mathbf{Y}}$, with **I** being an $N \times N$ identity matrix ($I_{ij} = 1 \forall i = j, I_{ij} = 0 \forall i \neq j$), and $(\mathbf{I} - \mathbf{A})^{-1}$ being the *Leontief's inverse* **L**. So far, the input-output system $\{\mathbf{x}, \mathbf{T}, \mathbf{Y}\}$ is expressed in purely monetary terms. Environmental extension uses a so-called environmental *satellite account* **Q** ($1 \times N$), from which environmental *intensities* $\mathbf{q} := \mathbf{Q}\hat{\mathbf{x}}^{-1}$ can be determined. Total environmental load, Q, can then be described as $Q = \mathbf{q}\mathbf{x} = \mathbf{q}(\mathbf{I} - \mathbf{A})^{-1}\mathbf{Y}\mathbf{1}^{\mathbf{Y}} =: \mathbf{m}\mathbf{Y}\mathbf{1}^{\mathbf{Y}}$, where the $1 \times N$ vector **m** holds so-called environmental *multipliers*.

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[2] Extracting (primary) industries such as agriculture, forestry, fishing and mining; manufacturers (secondary) such as food, paper, wood, textiles, fuels, metals, machines and equipment; utilities such as electricity, gas and water; and services (tertiary) such as construction, transport, communication, trade, hospitality, finance, government administration, public health, entertainment and personal care.

[3] Households, the government, the capital sector, and changes in inventories.

IO data are derived from the full version of the Eora MRIO database (Lenzen et al., 2012; Lenzen et al., 2013) covering the global economy. Supply chain pathways have been automatically extracted through an algorithm developed in Python 3.9 that identified and stored sector pairs within the milk-plastic wider supply chains. Pairs matching the keys "M/milk*" and "D/dairy" on the one hand and "P/plastic" on the other were first identified and then further refined by excluding those containing non-pertinent sectors such as "raw", "cattle", "untreated", or "sheep". This ensured we would focus on cow milk for human consumption. As an outcome, we obtained 113 sectors across 32 countries (full list in the Supplementary Information). There is a possibility that other countries should have been included but the relevant paths were not available in the MRIO database. To maintain consistency in our method and only use one underlying dataset we did not conduct a desk based research for other countries that could also use the same bottle.

3.4 Path exchange

The idea pursued in the Path Exchange method for hybrid LCA is that the IO system can be represented by an infinite assembly of *structural paths*: $\mathbf{m} = \mathbf{q} + \mathbf{qA} + \mathbf{qA}^2 = \cdots$, or in scalar notation: $m_k = q_k + \sum_j q_j A_{jk} + \sum_{ij} q_i A_{ij} A_{jk} + \cdots$, where each term $q_i A_{ij} \ldots A_{mn}$ represents an individual structural path. For example, $\mathbf{q}$ holds GHG emissions intensities, and let i be crude oil and gas extraction, j refining, k basic chemicals, l basic plastics, m plastic bottles, and n treated milk, then the product $q_i A_{ij} A_{jk} A_{kl} A_{lm} A_{mn}$ describes the GHGs emitted during venting and flaring at rigs extracting oil and gas, transported to refineries to be turned into basic petrochemicals, used as the feedstock for basic plastics, which are then turned into milk bottles.

IO output databases offer annual snapshots of actual economies. To assess altered or future production systems (see (Wiedmann et al., 2011)), the Path Exchange allows replacing individual environmental intensities, q_i, or input coefficients, A_{ij}, with alternative values, q_i^* and A_{ij}^* (for examples see (Lenzen and Crawford, 2009)). The methodological hybridisation lies in the fact that these alternative values are sourced from detailed product or process knowledge. In this work, we conducted consecutive numerical experiments in which we replaced the input coefficients, $A^c_{\text{plastics,milk}}$, for particular countries c, with $A^{c,*}_{\text{plastics,milk}} = A^c_{\text{plastics,milk}}(1-\gamma^c)$, where γ^c is the proportional reduction in material input achieved through advanced milk packaging (as shown in the Methods section). We then re-calculated $\mathbf{m}^* = \mathbf{q}(\mathbf{I} - \mathbf{A}^*)^{-1}$, and determined relative GHG emissions savings $\Delta\mathbf{m}^* = (\mathbf{m}^* - \mathbf{m})\widehat{\mathbf{m}}^{-1}$.

4. Results

4.1 Process-based results

Our process-based analysis focused on the full life cycle of both the optimised and standard bottles, with the exclusion of the use stage. This is because the reduction in material does not allow a store to stock more milk bottles in a refrigerated shelf, and as such the impacts linked to the use phase (i.e. energy required for the refrigerators and related emissions) are considered identical. Conversely, material inputs, transportation and end-of-life activities all differ depending on which bottle is being considered and results are shown in Table 3. As explained in Section 3, for end-of-life management, we adopt a mix of options based on data from Plastics Europe (2019). However, to model this mix, each of the options (i.e. recycling, incineration and landfill) had to be modelled individually and then allocated their share, thus we also offer results for each of the end-of-life options in the Supplementary Information.

Results for the standard bottle range from 188 to 260 gCO_2e, whereas for the optimised bottle the range is 166 to 230 gCO_2e. These results align sufficiently well with those retrieved in existing studies and shown in our literature review. Reductions average at 25.6 gCO_2e/bottle, with the maximum reduction corresponding to an 11.5% decrease (following a 12.8% decrease in material requirements). In both cases, pre-use impacts (e.g. A1 to A4) are about half of the total (48.8%) thus reinforcing the importance and far-reaching consequences of design choices as well as end-of-life management on future impacts.

Table 3: Cradle-to-grave results, in g CO_2eq, for the process-based LCA for the countries directly covered. For enhanced clarity, stages refer to the life cycle schematisation generally used for an Environmental Product Declaration (EPD) according to the standard EN 15804. Detailed results for other end-of-life scenarios (100% recycled, 100% incinerated, 100% landfilled) are given in the Supplementary Information.

		HDPE bottles	HDPE rigids (non bottles)	Aluminium	PE film	HDPE manufacture	PE film manufacture	Aluminium manufacture	Transport, refrigerated lorry	Municipal waste collection service	End-of-life	IPCC GWP 100a (total - Mixed EoL scenario)
	Stages	A1, A2	A1, A2	A1, A2	A1, A2	A3	A3	A3	A4	C2	Mixed	**g CO_{2eq}**
STANDARD BOTTLE	Australia	56	3	4	2	44	1	0	1	6	93	210
	Israel	55	3	4	2	44	1	0	1	6	114	231
	Malaysia	54	3	4	2	44	1	0	1	6	96	211
	New Zealand	57	3	4	2	44	1	0	1	6	142	260
	South Korea	53	3	4	2	44	1	0	1	6	109	223
	South Africa	56	3	4	2	44	1	0	1	6	70	188
	Thailand	54	3	4	2	44	1	0	1	6	119	234
	Ukraine	55	3	4	2	44	1	0	1	6	116	233
	UK	56	3	2	2	26	1	0	1	5	116	213
	USA	55	3	2	2	38	1	0	1	6	114	223
OPTIMISED BOTTLE	Australia	49	3	4	2	39	1	0	1	5	83	186
	Israel	48	3	4	2	39	1	0	1	5	101	204
	Malaysia	47	3	4	2	39	1	0	1	5	85	187
	New Zealand	50	3	4	2	39	1	0	1	5	125	230
	South Korea	46	3	4	2	39	1	0	1	5	96	198
	South Africa	49	3	4	2	39	1	0	1	5	62	166
	Thailand	47	3	4	2	39	1	0	1	5	105	207
	Ukraine	48	3	4	2	39	1	0	1	5	103	206
	UK	49	3	2	2	23	1	0	1	5	103	188
	USA	48	3	2	2	33	1	0	1	5	101	197

In addition, to also offer results for other impacts we used the CML-IA baseline (v3.06) LCIA method. In terms of GWP, the sensitivity of the results boils down to two parameters: the HDPE material in the bottle (58%) and the HDPE manufacturing (27%) with all the other materials and processes accounting for less than 15% overall. These two are the hotspots throughout all impact categories in fact, averaging at 31% and 50% respectively of the contribution to the overall impacts of the product system. Noteworthy are also the contributions of aluminium (for the impact categories abiotic depletion, 18%, and marine aquatic ecotoxity, 12%) and of the municipal waste collection services (19% contribution in the ozone layer depletion impact category). In terms of uncertainty analysis we did carry out a MonteCarlo analysis in SimaPro for both the IPCC GWP 100a and CML Baseline impact assessment methods. This has been done for the UK and USA LCIs as these were the most diverse in terms of processes. It is now offered in full for both countries as part of the supplementary files.

4.2 Results at scale: process-based vs. hybrid

To scale up results and evaluate what impact optimising the design of milk bottles would have in the countries we covered, we had to follow two different approaches depending on our starting point. The IO-based hybrid approach has been described in Section 3.4. For the process-based analysis, we use statistics from the Food and Agriculture Organization (FAO) of the United Nations (FAO, 2020) on milk consumption per country (in tonnes), and transform it to litres assuming an average milk density of 1.03 kg/L. We also base our scaling-up on a study from the University of Minnesota (Thraen et al., 1974) stating that fluids represent 45% of the total milk consumed, and that 74% of these are consumed by households. This is an important limitation of our PA, as this is a single, US-based datapoint nearly four decades old, but we could not locate better data. Results from this comparison at scale are shown in Figure 4, offered as a g CO_2e/$ figure.

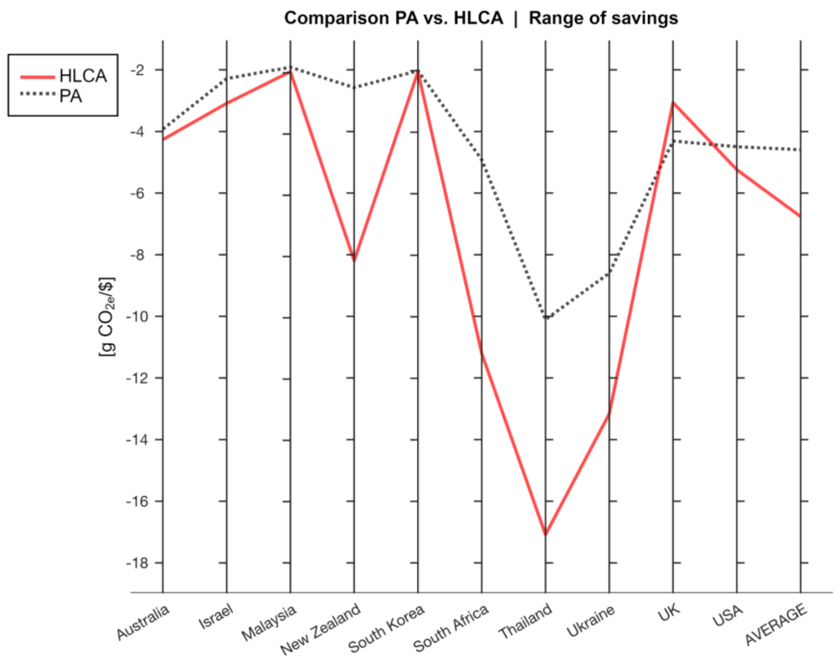


Figure 4: Country level results: process-analysis (PA) vs. input-output-assisted hybrid LCA (HLCA)

Figure 4 shows, for the most part, an expected trend: hybrid LCA results are higher than those obtained with PA due to the more complete coverage offered by IO assisted hybrid analyses. There are two exceptions to this trend: South Korea and the UK, although in both cases the results obtained with the two methods are relatively close to one another. For South Korea, the slightly higher PA value is due to a different approach used in the scaling up as the country has a milk/dairy consumption surprisingly high for an East Asian country. For this reason we retrieved average monthly expenditure for milk and eggs for South Korea (Statista, 2020), the average consumption of milk per person (CLAL, 2020), and the average cost ($2.17) of 1 L of milk in the country (NUMBEO, 2020). This bottom-up estimate on realistic figures yielded an overall value when scaled up for the whole country higher the FAO stat data. For the UK instead, our explanation for its lower value in the HLCA is twofold and boils down to both a technical reason and a limitation. Firstly, the UK's direct intensity is low, presumably because of wind and nuclear in its energy system, that regularly account for ~ 22% and 17% respectively of the UK's grid. Secondly, the UK has a very disaggregated IO table within the Eora database, showing many individual plastic sectors and, as such, **A** and **L** values underlying plastic-to-milk supply chains are low. This disaggregation allows for a tighter focus on supply chain data and excludes irrelevant other industries that might appear aggregated in other countries' tables.

The UK case shows two interesting points. GHG reduction linked to lower energy demand due to design optimisation might not be that significant in a country where the energy grid is being steadily decarbonised, unless that country heavily relies on global supply chains still operating on carbon intensive grids. As such our suggested optimisation should not be seen as a one-size-fits-all solution but rather framed in specific national contexts to evaluate its effectiveness. As a second point, we notice that countries with higher carbon intensities might mask the effect of disaggregated tables (like the US) so that carbon intensity and (dis)aggregation get somewhat mixed and are not univocally discernible without further digging into the background data.

4.3 Overall emissions saving potential

Notwithstanding the limitations of our hybrid approach discussed above, HLCA does produce more comprehensive results that cover upstream impacts both in national ($HLCA_{10}$) as well as global supply chains ($HLCA_{global}$) as one would expect, which is evident from Figure 5.

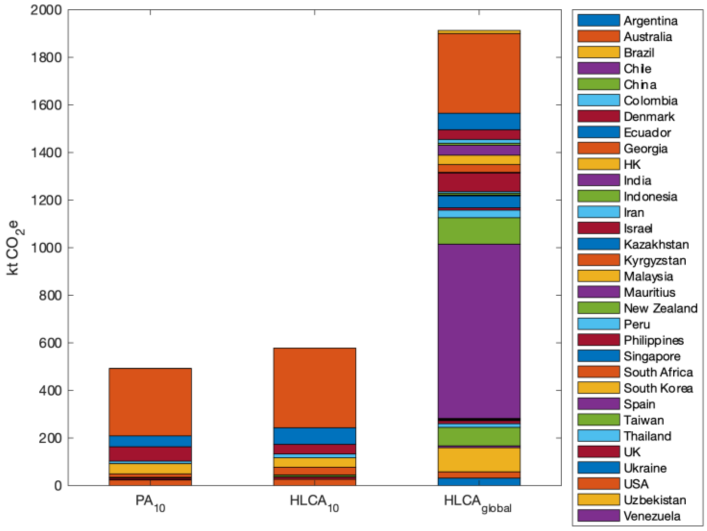
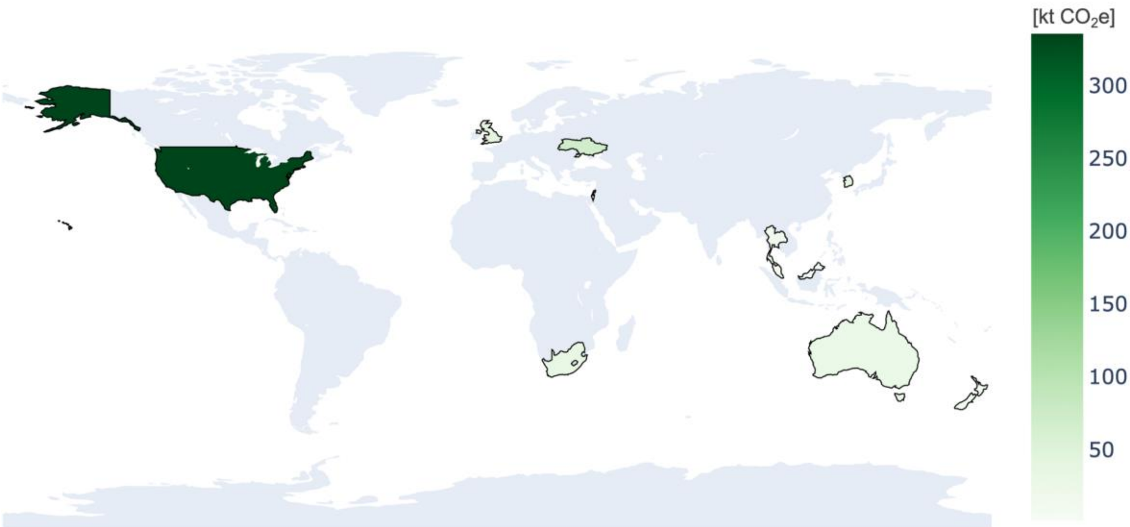


Figure 5: Overall results for the three approaches. PA_{10} refers to process analysis (PA) scaled up for the 10 countries we cover individually. $HLCA_{10}$ refers to input-output assisted hybrid life cycle assessment (HLCA) solely limited to the national supply chains for those same 10 countries we cover individually. $HLCA_{global}$ instead takes into account savings occurring upstream in the global supply chains. Detailed results for each country are given in the Supplementary Information.

Focusing solely on the 10 countries we cover directly in our analysis, the total obtained through HLCA is 17.3% higher than the total obtained through PA. When considering the additional savings occurring in upstream supply chain layers that are captured through IO-assisted hybridisation, PA only accounts for 25.7% of the impacts captured through HLCA with a global focus. Both these values are in line with observed magnitude of the truncation error introduced by PA (Lenzen, 2001). Since our two sets of results are arrived at from two different methodological paths and underlying data, we see this as an indication of the robustness of our findings. World maps with the savings for the 10 countries directly covered in our analysis and then combined with the additional 22 countries in which we observe emissions when adopting a global focus are visually shown in Figure 6(a) and Figure 6(b) respectively. Full numerical results obtained for these countries from the HLCA are also available in the Supplementary Information.



(a)

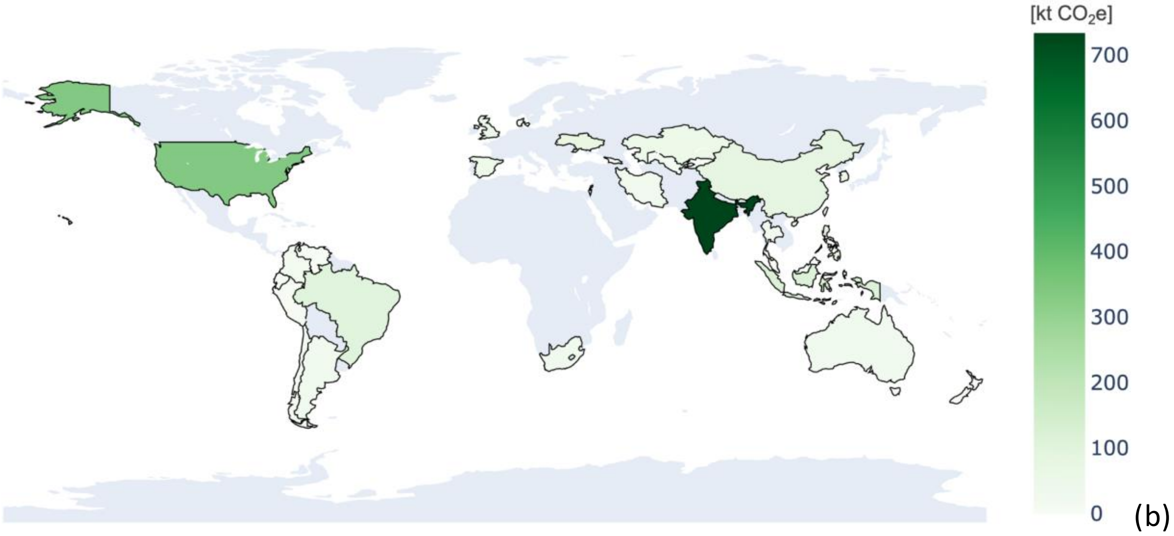


Figure 6: Map view for the ten countries directly covered in our analysis (a) and the additional 22, for a total of 32 countries (b) in which we observe an emissions reduction due to material-efficient design of the milk bottle analysed in this research.

From Figure 5 and Figure 6 it can be appreciated, for instance, the role of China and the Philippines (countries not covered within the ten we include directly) as top global HDPE producers or the contribution of several Latin American countries (again not directly covered) to the dairy sector in the US. None of these savings are captured through the process-based approach as they do not happen strictly within the product's system boundaries but further upstream in the supply chain.

4.4 Limitations

The map shows the many countries currently not covered. In some cases this is genuinely correct, for instance since a more optimal shape for milk bottles is already in use (e.g. Italy). In many others, however, the omission is a limitation of our research. This boils down to two main reasons. Firstly, the number of countries originally covered (32) was dependent on matching keywords when looking for sectoral pairs automatically, as explained in Section 3.3. Sector labelling, and what is included in them, can therefore have obscured sectors and countries that should have been included in our analysis but that currently are not. Secondly, the PA side of our analysis is even more limited, to 10 countries only. As shown in the Supplementary Information we have modified processes, energy mixes, and transportation distances to reflect national contexts as best we could. Nevertheless, the original processes used and available in ecoinvent are those of Switzerland, Europe (average), United States, Global, and Rest of the World. A full market analysis on plastic milk bottles across the 195 UN Member countries would likely result in a significantly higher number of countries that can be included to explore a truly global potential for emissions savings and as such this is an interesting area for further research.

A further limitation comes from the necessity to interpret FAOSTAT data to convert milk consumption (in tonnes per each country) into litres of milk consumed by households. The example of South Korea presented in Section 4.2 shows that different estimates through bottom-up realistic data can lead to much higher consumption values. In our case, the one datapoint we retrieved to identify the liquid fraction of milk consumed, and within this, the share consumed by households, is from an old piece of research and entirely US based. Surely national contexts will vary significantly, and this assumption might have introduced significant errors in our scaling up of process-based data. This limitation is somewhat mitigated by our dual approach, with HLCA results showing emissions savings which are higher (as expected) but still within the same order of magnitude of those identified through PA. Having country level accurate data for liquid milk consumption would help significantly in removing this limitation and it can be easily implemented in our analysis as and when such data becomes available.

Further, we only focus on GHG emissions reduction as an environmental impact category when it is likely that further significant environmental benefits would be observed if additional categories were included. This is easily achieved in a PA approach by simply selecting one of the many multi-category impact assessment methods, and we offer this in the Supplementary Information (e.g. abiotic depletion, human / aquatic / terrestrial toxicity, etc.). However, a smooth integration of such categories into HLCA is less straightforward and remains an area of active and further research.

A third and final limitation comes from the well-known aggregation issue of IO tables. We have seen this in our work for the UK, as explained in Section 4.2. There is ongoing work on ameliorating the smooth integration of PA and MRIO data (e.g. Agez et al., 2020) into HLCA and this is expected to continue and eventually reconcile the two as fully and realistically as possible. In the meantime, if we accept the current standpoint that PA potentially underestimates and IO-based hybridisation potentially overestimates and position ourselves at the average of the two methods, then we observe an annual potential saving of 1.2 Mt CO_2e/year.

5. Conclusions

Single use plastics is a global environmental issue and cause for concern. Bottles are one of the main forms of single use plastics. While they can be nearly fully recycled, this does not often happen, and recycling is increasingly seen as somewhat of a failure in the waste hierarchy as we aspire towards a circular economy. Realistically though, single use plastic bottles are so ubiquitous that they will not disappear from our lives any time soon and as such any mitigation opportunity to lower the environmental burdens they create should be reaped.

In this article we focus on single-use milk plastic bottles, which we redesigned through a material-efficient perspective to achieve optimal form without changing capacity (i.e. volume). The redesign resulted in a 12.8% material reduction, whose environmental benefits we evaluate in the form of greenhouse gas (GHG) emissions reduction over the full life cycle of the milk bottle. We adopted two common methods in life cycle assessment (LCA): a process-based analysis (PA) and an input-output-supported hybridisation (HLCA). Due to information availability and limitations in our methods and the data that support them, we only cover 10 countries directly, with a further 22 included in the upstream layers of the supply chain that HLCA allows us to capture. Broadening the scope of our analysis, resolving some methodological barriers in including more impact categories in HLCA, partnering with milk bottle manufacturers, and surveying users to validate the design proposed in this paper are all interesting avenues for further research. The latter point is useful in evaluating any potential obstacles to implementation of the new design such as crates, packing, handling, and filling as well as assessing the general inertia and resistance to change from both the consumers as well as the manufacturers' sides.

When implemented at scale, the annual emissions reduction estimated through PA totals 492 ktCO_2e while HLCA points to an overall annual reduction of 578 ktCO_2e, with a focus limited to the 10 countries covered, and of 1913 ktCO_2e when the focus broadens to include savings occurring in upstream, global supply chains. In the ongoing debate around the accuracy of PA versus HLCA we do believe the HLCA yields more comprehensive results due to the completeness of the background data. However, if we remain agnostic on this debate, and simply average the reductions identified by the two methods (PA and global HLCA), our optimised design can lead to annual emissions savings of 1.2 Mt CO_2e. This is not trivial since it represents a significant benefit coming from material efficiency through redesign when other material efficiency approaches (e.g. material savings by thinning HDPE bottles) have already been exploited to the full and almost beyond the limit of usability. Our findings also do not require reconfiguration of global supply chains (although localised manufacturing and reduced transportation could generate additional benefits), thus removing another obstacle to implementation. It is hoped that our results, while arising from a theoretical yet feasible redesign, can support industry-based initiative in the plastic and milk sectors to reduce their environmental footprint.

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