Potential trade-offs between eliminating plastics and mitigating climate change: An LCA perspective on Polyethylene Terephthalate (PET) bottles in Cornwall

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Abstract

The aim of this study is to investigate whether eliminating plastics entirely under existing waste infrastructure and management practices could have an adverse effect on climate change, using a case study on the hypothetical substitution of Polyethylene Terephthalate (PET) with glass as the material for bottling liquids in the domestic sector in Cornwall, England. A life cycle environmental impacts-based model was created using high resolution local data on household waste and current management practices in combination with Life Cycle Assessment (LCA) datasets. The model allows users to define key system parameters such as masses of materials, transport options and end-of-life processes and produces results for 11 environmental impact categories including the Global Warming Potential (GWP). The results from the application of this model on the case study of Cornwall have shown that the substitution of PET with glass as the material for bottling under the current waste infrastructure and management practices could lead to significant increases in GWP and hinder efforts to tackle climate change. A sensitivity analysis of the glass/PET mass ratio suggests that in order to achieve equal GWP the glass bottles need to become approximately 38% of the weight they are now. Increasing the recycled content and decreasing losses during the recycling processes could also help lower the GWP by

18.9% and 14.5%, respectively. This model can be expanded further to include more types of plastics and other regions to evaluate designs of new regional circular economy with less plastics waste and pollution. Our study suggests that it is necessary and crucial to consider the specific waste infrastructure and management practices in place and use science-based models that incorporate life cycle thinking to evaluate any solutions to plastics pollution in order to avoid problem shifting.

Keywords: circular economy, LCA, plastics, waste management, decision support

1 Introduction

Plastic products play a major role in our modern society due to their many useful attributes such as durability, lightweight, flexibility, electrical and thermal insulation, water and air impermeability and low costs. It is projected that following the same use patterns, 12,000 million tonnes of plastic waste will have been discarded in landfills or the natural environment by 2050, which is more than double the estimated 5,800 million tonnes of plastic waste ever generated from virgin sources up to 2015 (Geyer et al., 2017). Therefore, it is necessary to develop a circular economy approach to plastics that addresses the accumulation, impact and costs in the environment without compromising their use for multiple high value purposes.

In recent years, there are a growing number of local community-led “plastics free” initiatives in the UK, particularly the South West of England. One of the most obvious and practical options for these initiatives is to substitute plastics with other materials. However, whether efforts to eliminate plastics by material substitution can lead to negative impacts on other key environmental goals such as mitigating climate change needs to be carefully evaluated as it depends on a wide range of factors.

Polyethylene Terephthalate (PET) is a type of plastics widely used in packaging, particularly for non-alcoholic drinks, and can be easily eliminated and substituted by other established alternatives such as glass. However, several context specific factors can influence the climate impact of substituting PET with glass as a packaging material for drinks. On one hand, glass is much heavier than PET and higher energy consumption for transportation and production is expected. On the other hand, recycling rates for glass are usually higher than those for plastics, which are affected by consumer recycling behaviours as well as local waste infrastructure and management practices.

Studies comparing PET, glass and aluminium as bottling materials exist in the literature. For example, Romero-Hernández et al., (2009) have looked into this as part of their environmental implications and market analysis of soft drink packaging systems in Mexico using a waste management approach. However, their study was at a national level with little spatial granularity. In addition, the end of life options they considered included recycling and landfill but not incineration. Other studies investigated specific applications of glass containers including, e.g., a comparison between compared glass jars and plastic pots for baby food packaging (Humbert et al., 2009), an analysis of the impacts of glass and PET for extra virgin olive oil packaging (Accorsi et al., 2015) and a report on the carbon impact of bottling Australian wine in the UK using PET and glass bottles (Best Foot Forward Ltd for Wrap, 2008). The most recent and comprehensive study was carried out by Simon et al (Simon et al., 2016) who assessed the life cycle impacts of different beverage packaging materials and focused on the collection of post-consumer bottles. They examined five different packaging materials during their whole lifecycle and six bottle collection systems such as kerbside bin, kerbside bag, deposit-refund, combinations with thermal compression of plastic bottles and refill-bottles. However, their study was based on a generic hypothetical case study that did not reflect actual amounts of different types of waste generated and the actual amounts of waste treated in different ways. Overall, these existing studies tend to neglect local context in terms of volumes and types of waste, management practices and infrastructure and consumer recycling behaviour. This study aims to assess the climate change impact resulting from the potential substitution of PET by glass as the packaging material for drinks using high resolution data on consumer waste disposal behaviour, waste infrastructure and current waste management practices. Life Cycle Assessment (LCA) is used to calculate a wide range of environmental impacts including Global Warming Potential (GWP), an indicator for climate change impact. The English county Cornwall is used as the case region given that it hosts many plastic-free initiatives (including the first plastic-free town in the UK) and mitigating climate change is a top priority in its environmental agenda (Cornwall Council, 2019). Our study will be crucial in informing sustainable material substitution in the rising plastic-free movements as consumer waste disposal behaviour and waste infrastructure and management can vary regionally and locally and waste contracts can last for many years or even decades.

2 Materials and methods

The model developed allows users to perform comparative LCA to assess the potential impacts of substituting PET bottles by equivalent glass ones to meet the same level of demand for drinks packaging in the domestic sector in Cornwall. The main interface of the model has been developed in Excel so that users who are not experts in LCA or do not have access to specialist LCA software can modify key input parameters and investigate alternative scenarios.

Figure 1 is a flowchart that presents the overall methodology and describes the sources of data for the model developed in this study. In the next subsection, the specific system under study is presented, followed by the subsections with the analysis on the LCA stages for the two main scenarios investigated. The LCA stages according to the International Standard Organization, (2006a, 2006b) include the goal and scope definition, compilation of the Life Cycle Inventory

(LCI), the Life Cycle Impact Assessment (LCIA) and interpretation of the results. Although these activities are analysed in detail in the next subsections, this flowchart highlights that a significant part of this study has been dedicated to quantifying the material flows of the glass and PET bottle waste streams. These have been used to inform the LCI stage of the LCA and combined with the LCIA results on the specific processes that comprise the production, transport and end of life.

2.1 System description

The sociotechnical boundary of the system under study is the PET bottles used by households in the Cornwall County in the South West of England. Cornwall Council is the local authority responsible for the collection of household waste from the 213 smaller administrative units called civil parishes. The household waste can be categorised into two main types: recyclables and residual. Residual waste is collected weekly at the kerbside and transported to the Cornwall Energy Recovery Centre (CERC), the only waste-to-energy facility in Cornwall that started operation in 2017 (Cornwall Council, 2018a). The recyclables are separated by the residents and placed in four different containers provided by the Council, including a black plastic box for textiles and glass bottles and jars, a red sack for metal and plastics, a blue sack for paper and an orange sack for cardboard. The recyclables are collected every fortnight at the kerbside and transported to one of the two Material Recovery Facilities (MRF) situated in the towns of Bodmin and Pool.

At the MRFs, the plastic bottles (including the PET bottles) are consolidated, tied with wire in bales and transported by lorry out of the county to one of the three reprocessing facilities in Rochdale, Leicester and Bedford to be recycled according to Cornwall Council (Cornwall Council, 2018b). The recyclable glass bottles and jars are sent to Portugal for recycling, so we assume that they are transported first by lorry from the MRFs to Falmouth Harbour where they are loaded onto ships. Figure B1 in Appendix B shows the journey PET and glass bottles follow before they are processed further (either incinerated or recycled).

The residents can also take household waste to the 13 Household Waste Recycling Centres (HWRCs) and 66 bring banks located in different parts of the county. As this is a more complex operation and data on the exact types, amounts and origins of these wastes is not readily available, they have been excluded in this study. This should not affect our study significantly as the majority of wastes taken to the HWRCs are wastes that cannot be collected at kerbside (e.g., bulky waste such as furniture and electrical and electronic devices) and the amount of recyclable plastics collected at the HWRCs and bring banks are small in comparison to kerbside collection.

In this specific illustrative case study, data for the 2017-18 financial year from the Cornwall Council is used. Total household waste collected was approximately 160,576 tonnes, including 128,805 tonnes (~80%) of residual waste and 31,770 tonnes (~20%) of waste intended for recycling. 1.47% of the residual waste and 6.48% of the waste intended for recycling were estimated to be plastic bottles in the most recent waste compositional analysis for Cornwall conducted in 2017 (Waste and Resources Action Programme (WRAP), 2013; Wells, 2017). According to WRAP, 61.5% of plastic bottles in the UK are made of PET (Waste and Resources Action Programme (WRAP), 2013; Wells, 2017). This means that PET bottles collected at the kerbside was 2,468 tonnes, of which 1,164 tonnes (47.17%) was in the residue waste and sent for incineration while 1,304 tonnes (52.78%) was recycled. This recycling rate is lower than that for glass bottles, 84% and 16% of which are in the waste intended for recycling and the residual waste, respectively [6].

2.2 LCA goal and scope

The goal of this LCA is to evaluate the potential environmental consequences of substituting all the PET bottles used by households in Cornwall with glass ones under the existing waste infrastructure and manage practices. The functional unit of this case study is therefore the liquids packaging service provided by 2,468 tonnes of PET bottles to households in Cornwall. In order to estimate the equivalent amount of glass bottles that would be required to substitute these PET bottles, the users can specify the glass/PET mass ratio for the bottling of the same quantity of liquid content. This ratio depends on many factors such as the types and sizes of plastic and glass bottles used in different applications and is not available from the existing waste data. A range of 12.78-13.09 was reported in the literature (Accorsi et al., 2015; Simon et al., 2016) and the minimum value (12.78) is used in our study. This was chosen because there is an effort to reduce the weight of the glass bottles (British Glass, 2018) and it would provide a conservative estimate of the impacts of glass bottles. As a result, 31,542 tonnes of glass bottles would be needed to substitute the 2,468 tonnes of PET bottles.

Ideally, a complete cradle to grave LCA would have covered all the key stages over the life cycle of bottles, including production, transportation to drinks manufacturers, distribution to retailers, transportation to households, collection at kerbside and transportation to the MRFs, CERC and recycling facilities, incineration, recycling and transportation back to the bottle producers. Figure 2 illustrates these life cycle stages and highlights the system boundary of the LCA used in our model with the red dotted line.

Our system boundary includes the following life cycle stages: bottle production, collection at kerbside and transportation to the MRFs, CERC, recycling companies, incineration and recycling. This is because detailed data is available for these stages, assuming that the processes of bottle production, recycling and incineration are similar within Europe and that the datasets for these processes available in the LCA databases are representative for Europe. Transportation to drinks manufacturers, retailers, households and back to bottle producers are excluded because there is insufficient information about the locations of the bottle producers, drinks manufacturers and retailers. Therefore, our study is not a complete cradle-to-grave LCA, but the system boundary could be expanded in the future to include the rest of the transportation stages when more data become available.

2.3 Life Cycle Inventory

The model uses data from external sources and allows the user to define the preferred values and based on these it calculates the inputs that are necessary for the LCI. There are three main sets of data for each waste flow type that the model feeds into the LCI: i) the mass of the bottles that need to be produced, ii) the distances travelled and iii) the mass of the bottles that are incinerated and recycled. It should be noted that the actual amount of waste bottles generated by households was bigger than the amount collected at kerbside as there were other flows, including, e.g., a small amount of bottles taken to the HWRCs and bring banks, bottles discarded outside of homes and potential leakages to the environment (e.g., plastic bottles might be swept away by wind at kerbside before being collected). This will be further investigated in a future version of the model when data on other flows are available.

2.3.1 Production

The number of bottles that is produced is based on the estimation of the mass of the bottles that are collected at the kerbside. For the production of PET bottles, the LCI dataset for the stretch blow moulding process in the Ecoinvent 3.5 database is used and the raw material is assumed to be 35% of recycled bottle grade PET granulates and 65% of virgin PET granulates (Shen et al., 2011). The model allows users to specify the shares of green, brown and white glass bottles in the kerbside collected waste streams. However, for simplicity and lack of detailed data 100%

share of white glass with a 58% cullet content is assumed in this case study and the LCI dataset for white packaging glass production in Ecoinvent 3.5 is used.

2.3.2 Transportation

The transportation requirements are calculated based on two main parameters: the mass and distances of bottles transported. The model allows users to specify the total amounts of residual and recyclable waste collected per parish, which were available from Cornwall Council. It then computes the amounts of PET bottles in the residual and recyclable waste per parish based on the percentages mentioned in Section 2.1. The equivalent amount of glass bottles per parish is then estimated based on the glass/PET mass ratio (12.78).

The transportation distances are estimated based on the routes shown in Figure 2, with Google Maps (Google Maps, n.d.) used to calculate the distances for road transportation and the online model seadistances.org ("SEA-DISTANCES.ORG - Distances," n.d.) used to calculate the distances between seaports. Table 1 presents the distances of the transportation routes for the waste bottles to reach the respective PET or glass recyclers.

The distances between the CERC and MRFs and the parishes depend on their geographical locations. Figure B2 in Appendix B is a map produced to illustrate the average distances between the kerbside collection in all parishes and their closest MRF calculated based on the Ordnance Survey base maps for Cornwall and its parishes (Ordnance Survey (GB), 2019). Similarly, the distances between kerbside collection and the CERC are calculated for all parishes.

The transportation distances of PET bottles from the MRFs to the PET recyclers are assumed to be the average values of the distances between the two MRFs and the three PET recyclers in Rochdale, Leicester and Bedford. The locations of the three Portuguese harbours and two glass recyclers at Vidrologic Gralda and Parque Industrial del Gala are used to estimate the average transportation distance between the Portuguese harbours and glass recyclers. The average transportation distances are used as default inputs in the model, but all the distances shown in Table 1 are also provided as predefined values so that the users can choose specific facilities (e.g. UK PET recycler in Rochdale). The predefined values can also be overwritten to allow investigation of different PET recycling facilities in England and glass recycling facilities in Portugal.

Based on these transportation distances and the masses of the PET and glass bottles the model calculates the transportation requirements in thousand tonne-kilometres (kt-km) for each route. The total road transportation requirement for all the parishes amounts to 706 kt-km for PET bottles and 8,925 kt-km for glass bottles. The glass bottles require an additional 36,430 kt-km of sea transportation, which is approximately 4 times their road transportation requirement. The Ecoinvent LCI datasets for 16-32 metric ton lorry freight transportation and transoceanic ship sea freight transportation are used for the road and sea transportation, respectively.

2.3.3 End-of-Life

For the end-of-life stage, the waste bottles are either recycled or incinerated depending on whether they are in the recycling or residual waste streams. As mentioned in Section 2.1, out of the 2,468 tonnes of PET bottles collected at the kerbside, 1,164 tonnes (47%) were in the residue waste and sent to be incinerated while 1,304 tonnes (53%) were in the recyclable waste and sent to recycling. Out of the equivalent glass bottles (31,542 tonnes), 5,047 and 26,495 tonnes would be incinerated and recycled, respectively, based on the shares of glass bottles in the residue waste (16%) and recyclable waste (84%) (Wells, 2017).

For the incineration process, the Ecoinvent 3.5 LCI dataset for ‘treatment of waste polyethylene terephthalate at municipal incineration with fly ash extraction’ is used for the PET bottles while the dataset for ‘treatment of waste glass, municipal incineration with fly ash extraction’ is used for the glass bottles. During the PET bottle incineration process, 0.825 kWh of electricity is generated and used internally for every 1 kg of PET incinerated. As this substitutes electricity that would otherwise have to be provided from the grid, the same amount of grid electricity is assumed to be avoided. The Ecoinvent 3.5 LCI dataset for 'electricity, high voltage, production mix' for Great Britain is used for the avoided grid electricity production.

For the recycling processes, the Ecoinvent 3.5 LCI dataset for 'Europe without Switzerland: treatment of waste polyethylene terephthalate, for recycling, unsorted, sorting' is used for the PET bottles and 'treatment of waste glass from unsorted public collection, sorting' for the glass bottles. Based on the Ecoinvent data we have estimated that the losses during the recycling process are approximately 25% for PET and 8% for glass. Users can accept these values already predefined in the model or define their own. In order to credit the system for the avoided impacts we follow the 'net scrap' avoided burden approach which means that we take into account both the amount of the material (e.g. PET) that can be recycled at the end of life and reduce it by the amount of recycled material (e.g. PET) that is already included in the production of the bottles. For the PET bottles, the avoided virgin PET production would be the percentage that is sent for recycling (53%) minus the losses during the recycling processes (25% x 53% = 13% of the initial PET bottles) and the percentage of the recycled content that was used for the production of the initial PET bottles (35%), equal to 5% of the initial PET bottles. For the glass bottles, the avoided virgin glass (without cullet) production would be the percentage that is sent for recycling (84%) minus the losses during the recycling processes (8% x 84% = 6% of the initial glass bottles) and the percentage of the cullet that was already used for the production of the initial bottles (58%) (Wernet et al., 2016), equal to 20% of the initial glass bottles. In both cases, the model calculates these deductions automatically and when the percentage of the bottles that are recycled is less than or equal to the percentage of recycled content in the original bottles then the credit the model allows is zero. The model also considers the losses incurred during the recycling activities, so the quantities of PET and glass bottles collected are not equal to the quantities that are actually recycled.

2.4 Life Cycle Impact Assessment

The life cycle environmental impact results for the transportation of 1 tonne of waste for 1 km by lorry and by ship were calculated and included in the model along with the impacts associated with the processes included for the production, incineration, recycling of the bottles and their credits from the avoided impacts.

Using the GaBi software (GaBi, 2018), the Ecoinvent database (Wernet et al., 2016) and CML 2001 impact assessment method (Guinée, 2002) the model can calculate the impacts from the preferred PET bottle elimination scenario defined. The CML 2001 LCIA method is one of the most widely used and it was chosen because other studies found in the literature used the same method. In addition, this method provides transparency by keeping the results for 11 life cycle environmental impact categories disaggregated without weighting. The 11 impact categories include Abiotic Depletion Potential – elements (ADP elements), Abiotic Depletion Potential – fossil (ADP fossil), Acidification Potential (AP), Eutrophication Potential (EP), Freshwater Aquatic Ecotoxicity Potential (FAETP), Global Warming Potential (GWP), Human Toxicity Potential (HTP), Marine Aquatic Ecotoxicity Potential (MAETP), Ozone Layer Depletion Potential (ODP), Photochemical Oxidant Creation Potential (POCP) and Terrestrial Ecotoxicity Potential (TETP).

3 Results

The model was used to investigate the life cycle environmental impacts resulting from the hypothetical substitution of PET bottles consumed by households in Cornwall with glass ones. Firstly, a comparison of the environmental impacts of PET and equivalent glass for the 11 impact categories is given and then the results for a more detailed analysis that focuses on the GWP are presented. Finally, as a sensitivity analysis we investigated the glass/PET mass ratios needed to equalise the life cycle environmental impacts of the two types of materials for bottles as well as the changes in the losses during the recycling and the recycled content of the bottles.

3.1 Life cycle environmental impacts

Based on the lifecycle stages we described in the Figure 2 that shows system boundaries, we considered three main groups of activities for the waste bottles: i) transportation, ii) production and iii) end of life (recycling and incineration). The absolute values of the results for PET and the equivalent glass bottles for these three stages are presented in Table A1 and more detailed results with the exact impact values for PET and glass can be found in Table A2 Appendix A. For PET, the main contributor for the majority of the impacts is the production stage and only for the FAETP and MAETP the main contributor is the end of life stage. The PET transportation stage contributes to less than 1.3% of the total life cycle impacts for almost all categories except for ODP (3.2%). For PET, the end of life stage creates net benefits only in ADP fossil and AP (-3.8% and -1%, respectively) and for glass in ADP elements, GWP and MAETP (-14.4%, -1.7% and -0.3%, respectively). The greatest net burden for glass is in TETP (9%) while for PET the TETP is 22% and the greatest burden is for FAETP (57%). This can be explained by the fact that only 53% of PET is sent for recycling, avoiding 4% of virgin PET to be produced while 84% of glass is sent for recycling, avoiding 20% of virgin glass to be produced. The production stage contributes the most to all categories for glass, followed by transportation except for HTP and TETP to which the end of life stage contributes more. Nevertheless, the transportation stage is not negligible for glass as it can contribute between 0.9% (MAETP) and 8.4% (ODP) of total life cycle impacts, due to glass being heavier and transported further than PET.

Figure 3 presents a comparison of the life cycle environmental impacts for PET and the equivalent glass required for the bottling of the same quantity of liquids for the specific case study on Cornwall.

It is clear that substituting PET with glass would lead to an increase in 10 of the 11 environmental impacts considered: ADP elements, ADP fossil, AP, EP, GWP, HTP, MAETP, ODP, POCP, TETP. The greatest difference is for ODP and AP where the impact for PET is 17% of that of the equivalent glass and the smallest difference is for MAETP and EP where the impact for PET is 73% and 54% of that of the equivalent glass.

The only impact category for which glass performs better is FAETP, where the impact for glass is 65% of that of PET. The higher FAETP impact for PET can be attributed to the end-of-life stages as both the recycling and incineration processes have net impacts despite the credits for the electricity generated and the avoided production of virgin PET material.

3.2 Global Warming Potential comparison

Figure 4 shows a more detailed comparison of the GWP results for PET and the equivalent glass expressed in thousand tonnes of CO_2-equivalent (kt CO_2e) by main life cycle stage. The net GWP for PET is 11.3 kt CO_2e, about 38% that of the equivalent glass (29.9 kt CO_2e). The main reasons for this are the production stage, which contributes most to GWP for both PET and glass and the impact from production, which is much larger for glass (28.5 kt CO_2e) than for PET (8.9 kt CO_2e). This is also the reason why the credit from avoided virgin material through recycling is much more significant for glass (6.52 kt CO_2e) than for PET (0.35 kt CO_2e).

The impacts of the transportation stages are much less significant than those of other life cycle stages. For example, GWP of transportation is 0.12 kt CO_2e (or 1% of the life cycle net GWP) for PET and 1.9 kt CO_2e (or 6.3% of the life cycle net GWP) for glass. GWP of transportation is much less for PET than glass mainly because the means of land transportation are the same for both types of bottles (lorry) but PET bottles weigh less than 8% of the equivalent glass ones. This suggests that even if the transportation routes change (e.g., different PET recycling facilities in the UK are used or the glass bottles are recycled locally instead of in Portugal), the overall GWP impacts would not change significantly. This also shows that excluding the life cycle stages of bottle transportation and distribution via the retailers and the customers and back to the bottle producers is not expected to affect the results significantly. That is because these additional transportation requirements are not expected to be greater than the ones we have included, which prove to be less significant than the other life cycle stages in terms of impacts. It is worth noting that international transportation to Portugal (the longest transportation distance) is not the main contributor to GWP over the glass life cycle, as it accounts only for 0.41 kt CO_2e (or less than 22% of total transportation GWP and less than 2% of the total net GWP). The reason is that sea transportation has considerably lower carbon footprint per tonne-km than that of the road transportation.

For the end-of-life stage of glass, the GWP impacts from incineration are rather low (less than 1.5% of the credit received from the recycling) even though there is no credit for electricity generation. This is because incineration of glass does not emit CO_2 (unlike PET) and most of the glass (84%) is recycled, resulting in a considerable credit to the system. The end-of-life results suggest that PET has higher impacts from the recycling and incineration than that of glass despite credit for electricity generation from PET incineration.

The end-of-life performance highlights the sustainability of glass as a recyclable material and in a more comprehensive model where washing and reusing the bottles are also considered as an option this could potentially improve the GWP results for glass. According to some sources (British Glass, 2018) glass is an infinitely recyclable material as opposed to PET and other plastics, which can be recycled only for a limited number of times due to the breakdown of the polymer chain and the deterioration of their quality.

3.3 Data quality and sensitivity analysis

In order to investigate the robustness of the results and the significance of alternative modelling choices, we first evaluate the data quality and in the next paragraphs we perform a sensitivity analysis for the glass/PET mass ratio, for the losses during recycling and for the recycled content (cullet).

3.3.1 Data quality analysis

Using a pedigree matrix with 6 quality indicators (Weidema and Wesnæs, 1996) we performed a data quality analysis taking into account the sources used, date, geographical scope, technology covered, reliability, completeness and uncertainty. Table 2 presents the results from this data quality analysis with scores of 1-5, where 1 is best and 5 is worst with colour coding accordingly. The total scores for each one of the factors examined are given at the last column of the table. As far as uncertainty is concerned, the following factors were highlighted: total mass of equivalent glass (score: 17/30) and losses during recycling (score: 19/30). The uncertainty in the total mass of equivalent glass is the most important because of the high uncertainty in the estimation of the glass/PET mass ratio use due to lack of specific field data. Uncertainty is also high for losses during recycling as these are based on the available Ecoinvent data only, also due to lack of specific field data. Issues with the temporal coverage are present in the impact inputs but their uncertainty is low because of the quality of the Ecoinvent database which was used as a source.

3.3.2 Sensitivity analysis on the glass/PET mass ratio

In this subsection, we present the investigation on the glass/PET mass ratio that could equalise the life cycle environmental impacts of the two types of bottles. The focus initially is the GWP because avoiding the potential adverse climate change impact has been the impetus of this work. Making glass more lightweight to reduce the glass/PET mass ratio to 4.85 could equalise the GWP, bringing both types of bottles to approximately 11.3 kt CO_2e, and improve the performance of glass compared with PET in other impact categories (see figure 4).

In order to for glass to perform better than PET in all impact categories, the ratio needs to reduce to 2.12 (see Figure 5). This comparison highlights the two impact categories that glass perform relatively poorly: AP and ODP. These two impact categories have very low impacts for PET compared to the equivalent glass bottles for all stages and this is mainly due to the high weight of the glass bottles. More specifically, the key process that contributes the most to AP and ODP is production (40.35 tonnes SO_2-eq. and 0.59 kg R11-eq., respectively).

Achieving these mass ratios would require important technological improvements and there is no evidence to suggest that this can be feasible in the foreseeable future. Potentially significant reductions in the impacts of glass are possible when a combination of strong interventions such as the development of a glass recycling facility in the county and the introduction of very lightweight glass bottles take place. However, these interventions depend on a wide range of factors and caution is needed when such scenarios are investigated to support policy making.

3.3.3 Sensitivity analysis on the losses during the recycling

We assumed that 25% of the collected PET and 8% of the collected glass bottles will be lost in the recycling process therefore not recycled in the base case. In this subsection we investigate how much the results change when we vary these values by ±10% (i.e., 25%±2.5% for PET and 8%±0.8% for glass). We also test two sets of extreme values for both types of bottles: i) zero

(0%) so that we can assess the case of an ideal systems where all the PET and glass bottles collected become recycled PET and glass cullet respectively, and ii) 50% where half of the collected material becomes recycled material available for reuse. The results of this analysis are given in full in Table A3 in Appendix A and show that although the absolute values change, impacts of PET are still lower than glass for all categories except for FAETP.

The 10% variation around the base case value results in a change of ±0.88% for the PET GWP results while the changes in the other categories range from -1.82% to 1.62%. For glass, the 10% variation around the base case value results in a change of up to -0.33% for GWP and the other categories range from -0.58% to 0.46%.

In the ideal scenario where losses are zero, the PET GWP would decrease by -7.14% and the changes in the other categories would be between -14.47% and 1.28%. For glass, the ideal scenario results in impact reductions ranging from -5.10% to 0.74% (-1.01% for GWP). When the losses become 50% the changes in the PET impacts increase from 0.30% to 3.50% (1.75% for GWP). On the contrary, in the 50% loss case the glass impacts changes range from -2.74% to 15.83% (3.34% for GWP). This differences in the effects of the recycling losses on PET and glass impacts are due to the high recycling rate for glass (84%) compared with PET (52.8%).

3.3.4 Sensitivity analysis on the recycled cullet content

Our model is based on the results extracted from using the Ecoinvent database and that implies that we also accepted the assumptions they have made for the recycled content. In order to perform a sensitivity analysis on the recycled content used we would need to change the amount of cullet in the respective process, but this is not straightforward. For example, if lower levels of cullet are used in the production of glass, the reduction in cullet contents needs to be compensated by increases in the use of other materials. However, the glass production process uses a range of materials such as silica sand and dolomite and it would be difficult to estimate the levels of increase needed for each of these materials without actual data. In addition, changes in the levels of different materials used would change the amount of processing energy required. Therefore, it is considered to be unrealistic to change the amount of recycled contents only. Nevertheless, the Ecoinvent database includes datasets that represent a case of no cullet being used for the production of glass (i.e., 0% recycled content) and a case of 80% cullet content. Although these datasets do not provide an equal increase and decrease in the recycled content around the 58% figure used in the base case, they can still serve as a sensitivity analysis on changes in the recycled content.

The results of this analysis are given in full in Table A4 in Appendix A and show that although the absolute values change, impacts of PET are still lower than glass for all categories except for FAETP. Reduction in recycled content increases the GWP for both PET and glass (by 1.21% and 27.65%, respectively) and increase in recycled content leads to decrease of the GWP (by -18.88% and -3.31%, respectively). The GWP increase is greater for glass while the GWP decrease is greater for PET. This can be attributed to the fact that in the base case scenario the recycled content is 35% for PET and 58% for glass. Reducing recycled content to 0% would lead to changes in other impacts ranging from -22.56% (for TETP) to 7.28% (for ODP) for PET and from -40.77% (for FAETP) to 33.21% (for ADP elements) for glass. Increasing recycled content to 80% would lead to changes in other impacts ranging from -40.54% (for ADP elements) to 43.52% (for MAETP) for PET and from -5.01% (for ADP elements) to 16.10% (for TETP) for glass.

4 Discussion

In this section we discuss the results and we focus on two parts: the limitations of our study and the comparison with results from other relevant studies.

4.1 Limitations

Although considerable efforts were made to cover the majority of the factors that can affect the collection and recycling/incineration of the PET and equivalent glass bottles, there are some limitations in our analysis. These limitations are mainly associated with data availability and collection and might introduce uncertainties to the results. For example, the estimation of the amount of the PET bottles collected was based on data about the share of PET in all plastic bottles at a national level and the estimation of the equivalent glass bottles was based on a glass/PET mass ratio found in the literature. These values can therefore be refined when better data become available. In the future, it would be useful for the stakeholders who are responsible for the collection to measure or estimate these values via a survey on the shares of the desired wastes (PET and glass bottles in this case).

We acknowledge that the activities of production, incineration and recycling are influenced by many factors that cannot be controlled by decision makers at the local level and that adds further uncertainties to our study. For example, the ratio of virgin/recycled PET granules used in the production of the PET bottles or cullet used in the glass bottles is up to the individual manufacturers. For simplicity and a lack of more detailed data we also assumed as a base case that all the glass bottles are made of white glass with a 58% cullet content while in reality these bottles can be of different colour with the composition depending on the intended use. Likewise, we excluded the caps and labels which can be made of a wide variety of materials (plastic, metal, cork etc.).

4.2 Comparison with results from relevant studies

Although the base case in our study reflect the hypothetical scenario where PET bottles consumed in the domestic sector in Cornwall are replaced with glass ones under the current recycling behaviour and waste management infrastructure and practices, the sensitivity analysis extend the range of results that are potentially comparable with other relevant studies. For example, our results are in agreement with the finding in Accorsi et al (2015) that the recycled PET scenario has the lowest GWP for all end-of-life strategies and the finding in Humbert et al (2009) that plastic pots lead to 28-31% lower GWP than glass jars. The WRAP report (2008) on the carbon impact of bottling Australian wine found a lower footprint for the 54g PET bottle with 0% recycled PET content (446g of CO_2) than the equivalent 496g glass bottle with 81% recycled content (476-550g of CO_2). This is in agreement with our study which suggests a PET bottle with 0% recycled content has a lower carbon footprint than an equivalent glass bottle with 80% recycled content. The importance of lightweighting glass bottles that we highlighted with our sensitivity analysis is also mentioned in the WRAP report (2008), which showed that glass can become better than PET when its weight is reduced by more than 23% and its recycled content exceeds 90%. Using the values of Simon et al (2016) for the 0.5l PET and 0.5l glass bottles for the production, distribution, waste collection, incineration and recycling including the potential credit, the carbon footprint of the PET bottle is also lower than the glass one. All of the above results are specific to different circumstances, but they all highlight that replacing PET bottles by glass ones can potentially result in an increase in climate impacts.

5 Conclusion

Our study aims to investigate whether eliminating PET bottles entirely under existing waste infrastructure and management practices could potentially have an adverse effect on climate change mitigation. An analysis on the life cycle environmental impacts from the hypothetical substitution of PET with glass as the material for bottling liquids in the domestic sector in Cornwall, England is used as a case study.

The results suggest that without changing the current waste infrastructure and management practices, the substitution of PET bottles consumed by households in Cornwall with glass ones could lead to significant increases in GWP and hinder efforts to tackle climate change. It seems that in this specific case PET bottles help to lower GWP thanks to their lightweight, but the development of more favourable conditions for the glass bottles does not exclude the overturn of this finding.

Potential improvements might be achieved by making glass bottles lighter. For example, lowering the glass/PET mass ratio to 4.85 could equalise the GWP of PET and glass while a reduction to 2.12 could make glass perform better than PET in all impact categories. Less significant improvements might be achieved by keeping the recycling activities within the county's geographic boundary and avoiding any transportation out of the county. This would lead to less than 1% reductions in the impacts for PET and less than 6% reductions in impacts for glass. Future versions of the model could include more stages of the life cycle as well as more detailed LCI of the bottles and their materials based on a solid market analysis.

Switching from PET to glass could increase AP and ODP by approximately 500%, POCP by 337%, HTP by 182%, ADP elements by 181%, GWP by 164%, ADP fossil by 160%, TETP by 110%, EP by 86% and MAETP by 36%. The only impact that would be decreased is FAETP (-35%). These results suggest that a wide range of impacts need to be considered in addition to GWP when making decisions on replacement of plastics.

It is important to note that these conclusions apply only locally and cannot be generalised as waste management may vary across regions and countries. In order to extend these conclusions to replacing plastics more widely, future research is needed to evaluate other plastics forms and possible replacements scenarios.

Overall, our study suggests that it is necessary and crucial to consider the specific waste infrastructure and management practices in place and use science-based models that incorporate life cycle thinking to evaluate any solutions to plastic pollution in order to avoid problem shifting like the case study presented in this work.

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