Resources, Conservation & Recycling 209 (2024) 107807

Contents lists available at ScienceDirect

Resources, Conservation & Recycling

journal homepage: www.sciencedirect.com/journal/resources-conservation-and-recycling

Life cycle assessment of mechanical recycling of low-density polyethylene into film products – towards the need for life cycle thinking in product design

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ARTICLE INFO

Keywords:
Life Cycle Assessment
Recycling
Environmental break-even
Material testing
LDPE
Design for recycling

ABSTRACT

In this study, we conducted LCA on the environmental impacts of recycling LDPE films within two system boundaries. System boundary 1 analysed the operations of a recycling company producing recycled LDPE granules. The results were comparable to the literature, yielding 0.44 kg CO_2-eq./kg LDPE. Polymer testing revealed that recycled LDPE foil products had a higher mass per product than virgin materials to compensate for inferior material properties. System boundary 2 was modelled to analyse existing film products, incorporating both recycled and virgin LDPE and using data from system boundary 1, to fulfil the same function. Due to the incineration at the products' end of life, some recycled products showed higher climate change footprint, due to the additional mass. Accordingly, the idea of a threshold called the environmental break-even was introduced, indicating the maximum surplus of low-quality recyclate usable to achieve the same climate change impact than a purely virgin competitor.

1. Introduction

Packaging plastic accounts for 42 % of all non-fibre plastic ever produced (Geyer et al., 2017). Moreover, it was estimated that the mean life cycle of plastic packaging is shorter than a year, leading to a considerable amount of plastic waste that needs disposal. In 2020, each citizen in the European Union was accountable for 34.6 kg of plastic packaging waste, with only 13.0 kg being recycled (Eurostat, 2022). In Austria, low-density polyethylene (LDPE) is the primary polymer used in packaging, with a recycling rate of approximately 26 %, in line with European statistics (Van Eygen et al., 2018). Unfortunately, recycling often equals downcycling because of phenomena such as the breaking and crosslinking of carbon chains, which ultimately result in materials with inferior properties (Schyns and Shaver, 2021). Polyethylene (PE) is no exception to this rule, as was recently confirmed by Felgel-Farnholz et al. (2023). Surprisingly, numerous life cycle assessment (LCA) studies (Chen et al., 2019; Civancik-Uslu et al., 2021) on plastics recycling assume that recyclates can substitute virgin material to a high extent, ranging from 90 to 100 %. In a recent review, Pellengahr et al. collated LCA studies on PET recycling a material that has high recyclability, mostly due to separate collection. Substitutability values between 71 and 85 % were found for recycled PET when compared to virgin material (Pellengahr et al., 2023)

Rajendran et al. postulated a break-even point, suggesting that it is difficult for recyclates to completely substitute virgin materials 1:1. Therefore, more recycled material is needed, resulting in higher emissions during both the production and disposal phases. They concluded that a 70 % substitution ratio would be the break-even point at which HDPE recycling becomes sustainable (Rajendran et al., 2013). In a recent study, Holly et al. interviewed key players in the Austrian recycling landscape to identify barriers for the implementation of a circular economy. Representatives from numerous companies stated that regulatory interventions, such as implementing a distinct policy focused on recycling and providing subsidies, should be regarded as a key solution. In our opinion, if subsidies are to be granted, sustainability needs to be quantified by exact rules, and the circular plastic value chain needs to be unquestionably preferable to the status quo (Holly et al., 2023).

To address this issue, this study aimed to analyse two system

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https://doi.org/10.1016/j.resconrec.2024.107807
Received 2 April 2024; Received in revised form 7 June 2024; Accepted 5 July 2024
Available online 16 July 2024

boundaries (SB) via LCA. SB1 addressed the production of 1 kg of recycled LDPE pellets (*r*-LDPE) using primary data from a recycling company operating in the DACH (Germany, Austria, and Switzerland) region of Europe. These results were subsequently transferred to SB2, in which four distinct commercially available *r*-LDPE film products were benchmarked against virgin LDPE (*v*-LDPE) products from competitors. Using these hands-on products, we were able to circumvent methodological issues related to substitutability, thereby obtaining a fair comparison of recycled and virgin products. Thus, the research question, whether recyclate-containing consumer products exist which contradict the waste hierarchy by causing higher emissions than products made from virgin material, was addressed and possible break-even points were identified.

2. Materials and methods

2.1. LCA

The LCA conducted followed the four phases outlined in ISO 14040/14044.

2.1.1. Definition of goal & scope

This study aimed to assess the environmental impact of *r*-LDPE. SB1 employed a cradle-to-gate SB and a functional unit (FU) of 1 kg *r*-LDPE produced. SB2 expanded upon the results obtained in SB1 to simulate the production of hands-on consumer products that are available in stores in Austria. The following four product classes underwent material testing and LCA in SB2:

- A 60-L waste bag for solid waste
- A 30-L waste bag for solid waste
- Transparent packaging for 10 rolls of toilet paper
- Transparent packaging for 8 rolls of kitchen paper

2.1.2. Life cycle inventory

The life cycle inventory (LCI) was obtained from a plastic recycling plant operating in the DACH region. For confidentiality reasons, the inventory could not be presented in this study. To compensate, the process was described as comprehensively as possible in Section 3.1. The results were benchmarked using a similar process design based upon background data from Sphera's professional database (Sphera, 2023). As the electricity input used is a decisive factor in LCA on plastics recycling, four distinct electricity inputs were investigated, which were the Austrian (AT) and European (EU) as well as fictitious Austrian mixes for 2030 (AT2030, Vilbergsson, 2021) and 2040 (AT2040, Austrian Ministry for Sustainability and Tourism, 2019), also based on background data from the professional database by Sphera. The respective mixes were described by governmental bodies and present the grid on its way to decarbonisation. Details are presented in Supporting Table S1.

2.1.3. Life cycle impact assessment

Environmental Footprint 3.1 was used for the life cycle impact assessment (LCIA) as it represents the official impact assessment method in the European legal landscape and is therefore of great regulatory interest. Authors already speculate that the method will become mandatory for certain industries in the future (Mordaschew and Tackenberg, 2024). Calculations were performed using Sphera's "LCA for Experts" software (formerly GaBi) version 10.7, using the professional database by Sphera, as well as ecoinvent v3.8 (Sphera, 2023; Wernet et al., 2016).

2.1.4. Interpretation

The interpretation phase's results are outlined in Chapter 3, Results and Discussion.

2.2. Materials and material testing

All investigated films were sourced as commercially available LDPE film products. Four different product types were purchased, as described in Section 2.1.1. As a rule, a 100 % virgin product was purchased, along with at least one benchmark product containing *r*-LDPE, with the exact percentage specified on the product. To conduct a mass-based comparison of different products, the influence of product design was eliminated. Therefore, three rectangles with dimensions of 140 mm × 30 mm were punched per material and subsequently weighed. The obtained weights were averaged and then extrapolated to represent the equivalent of 1 m^2 of film.

2.2.1. Puncture resistance

Puncture resistance was determined according to the DIN EN 14477 standard using a Zwick/Roell zwickiLine Z2.5 universal testing machine (ZwickRoell, Ulm, Germany). The test speed was set at 10 mm/min, and the maximum force prior to failure was recorded. Ten measurements were performed for each material. According to the standard, the films were conditioned at 23 °C and 50 % relative humidity for at least 48 h before testing.

2.2.2. Tensile strength

Tensile strength was measured via tensile tests conducted in accordance with the ISO 527–1 standard using a Zwick/Roell zwickiLine Z2.5 universal testing machine (ZwickRoell, Ulm, Germany). A total of 10 Type 5 specimens (ISO 527–3), which are commonly used to determine the tensile properties of films and sheets, were punched per product to obtain meaningful averages and standard deviations. As specified by ISO 527–1 and ISO 527–3, the traverse speed was set to 50 mm/min. Prior to testing, the specimens were conditioned at 23 °C and 50 % relative humidity for at least three days.

2.2.3. Microscopy

Consumer product samples were inspected via optical microscopy using a Keyence VHX-7000 digital microscope (Keyence Corporation, Osaka, Japan).

3. Results and discussion

3.1. Life cycle inventory: system boundary 1

As mentioned in Section 2.1, the detailed inventory pertaining to the recycling plant cannot be presented because of confidentiality agreements. Details of the recycling operation are listed below.

- The operation specialises in the recycling of LDPE films.
- Plastic reject is estimated to represent approximately 15 % of the total input, whereas the remainder comprises different waste fractions that are treated accordingly (e.g., cardboard waste, wire, or solid waste).
- PE waste input streams of diverse qualities are treated, ranging from high-quality inputs (e.g., 98 % pure LDPE) to challenging inputs (e.g., DSD-310) (van Rossem, 2023).
- Sorting lines operate automatically and manually.
- The process steps include shredding, sorting, washing, and extrusion, whereas the data also included auxiliary processes, such as wastewater treatment, heating provision for company facilities, warehouse operation, and transportation.
- The electricity consumption for the entire process is approximately 1 kWh/kg output.

For the LCA study, the assumption was made that the company was situated in Austria and, therefore, used the Austrian grid and utilities within that regionalisation. The main reasons for this were that the authors are from Austria, which provides them with a strong

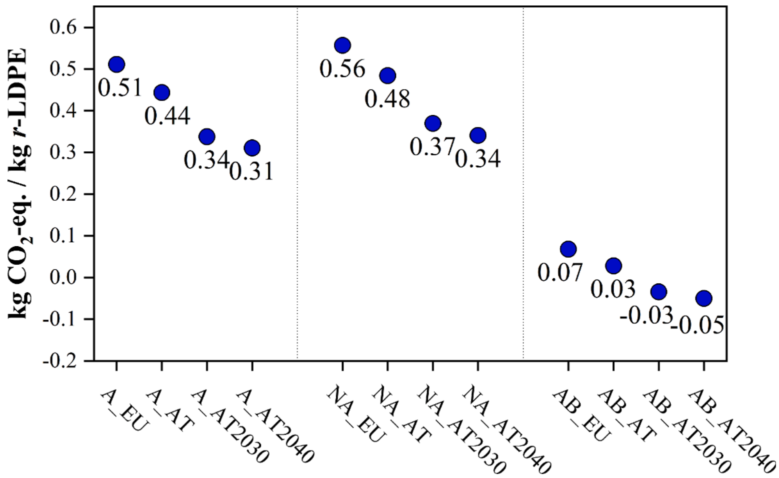

Fig. 1. Midpoint results for the climate change total impact category. Please note that the first letters prior to the underscore denote the type of multifunctionality solution: "A" for allocation, "NA" for no allocation, and "AB" for the avoided-burden approach. The letters following the underscore indicate the electricity mix used for calculating the results.

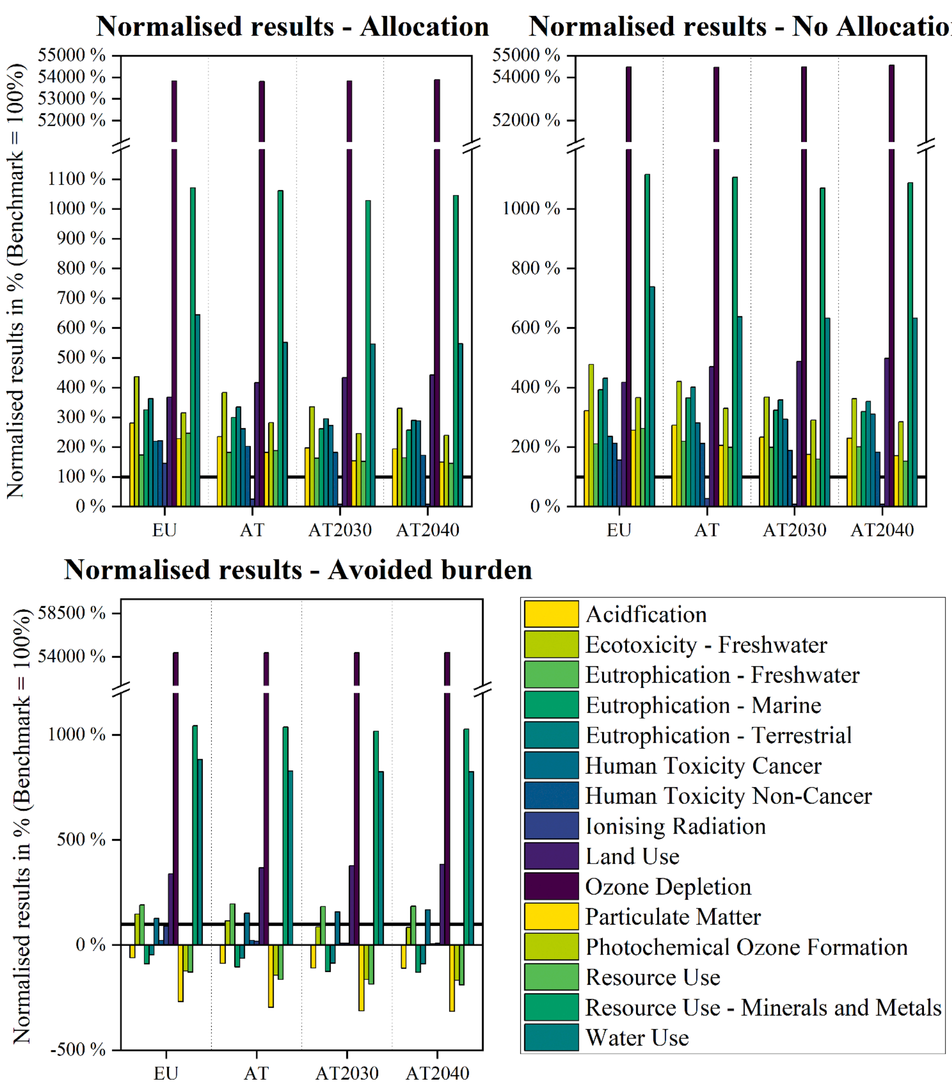

Fig. 2. The results of impact categories other than climate change, normalised to the benchmark results. The electricity mix used is shown on the x-axis.

Table 1
Material data for the four commercial case studies. The benchmark (BM) is always the product made from 100 % virgin material.

Case Study	*r*-LDPE / %	*v*-LDPE / %	Article weight / g	Mass ratio of product to BM (AW)	Weight of 1 m^2 article / g	Mass ratio of product to BM (1 m^2)	Average puncture force / N	Average tensile strength / MPa
Waste bag 60-L Black	100	0	35.02	**3.67**	33.49	2.96	1.27	7.98
Waste bag 60-L grey	80	20	29.55	**3.10**	30.56	2.70	0.95	10.18
Waste bag 60-L white – BM	0	100	9.53	**1.00**	11.31	1.00	0.74	12.30
Waste bag 30-L grey	80	20	18.48	**2.36**	27.62	2.44	1.05	11.04
Waste bag 30-L white – BM	0	100	7.83	**1.00**	11.31	1.00	0.77	12.68
Toilet paper bag 10 Pieces	50	50	13.99	**0.96**	27.38	0.91	1.6	17.82
Toilet paper bag 10 pieces - BM	0	100	14.52	**1.00**	30.16	1.00	0.95	19.68
Kitchen roll bag 8 pieces	60	40	16.97	**0.93**	27.94	0.93	0.89	16.95
Kitchen roll bag 8 pieces - BM	0	100	18.25	**1.00**	30.08	1.00	0.8	16.95

understanding of recycling practices in the country, and the fact that the consumer products were purchased at an Austrian retailer.

3.2. Solving multifunctionality in system boundary 1

15 % of the total plastic input ends up as reject. Although this flow can easily be regarded as waste, there are alternative ways of modelling. The Austrian cement industry uses a high percentage of refuse-derived fuels (RDFs) for clinker production, with plastics historically being the most important RDF (Mauschitz, 2023). Therefore, three solutions for solving multifunctionality were analysed.

1.) Allocation (A): The process' burdens were allocated by mass between the relevant RDF and *r*-LDPE.
2.) No allocation (NA): All burdens were assigned to the main product, *r*-LDPE.
3.) Avoided-burden approach (AB): It was assumed that the RDF was of an additional nature and substituted hard coal in Austrian cement kilns.

The system expansion was performed based on the assumption that the RDF had a lower heating value of 37.72 MJ/kg RDF, as specified by Istrate et al. (2021). This value was assumed to substitute the same quantity of thermal energy as from hard coal, the most common fossil fuel in Austrian kilns. (Istrate et al., 2021). The resulting emissions from incinerating the RDF were accounted for using the approach outlined by Jeswani et al. (2021).

3.3. LCIA results for system boundary 1: climate change

The results of the previously described scenarios for solving multifunctionality with different electricity mixes are shown in Fig. 1. The allocation or no allocation approaches did not result in a qualitatively different decision and lower emission levels were expected in the allocation approach due to the distribution of the burdens. The results for the avoided-burden approach were also qualitatively similar; however, the values were much lower, reaching negative values for the AT2030 and AT2040 cases. This was due to the large amount of credit acquired from displacing hard coal in cement kilns. These negative results lead to an adverse outcome, which shows that methodological issues and choices heavily influence the results. Similar findings were made by Istrate et al. who used the circular footprint formula to analyse the *r*-HDPE production of pipe grade material, as they demonstrated that the allocation approach had a minimal impact on the ranking, whereas substitution had a major impact (Istrate et al., 2021). Comparing studies on plastic recycling is challenging because of the wide range of polymer types, methodological choices that strongly alter results, and process steps being cut off. However, Martín-Lara et al. featured a very similar design to that of our study and reported a value of 0.416 kg CO_2-eq per 1 kg of *r*-LDPE pellets (Martín-Lara et al., 2022). Khoo reported 0.4 kg CO_2-eq./kg recycled plastic, which translated to 0.456 kg/kg *r*-PE, assuming the 87.7 % yield for PE presented in this study (Khoo, 2019). Finally, Civancik-Uslu et al. reported -1.162 kg CO_2-eq./kg recycled PE film, assuming a substitution potential ratio of 1:1 (2021). This can easily be converted to 0.73 kg CO_2- eq./kg *r*-LDPE, taking the substitution credit, 1.890 kg CO_2-eq./kg *v*-LDPE, (Jeswani et al., 2021) into account, which aligns with our reported values. Those partly higher emission values do not indicate inferior performance, as the quality of the obtained regranulates could not be compared despite the similarities in the process steps to the industrial case at hand.

The results for the other impact categories are presented in Fig. 2. The values obtained were compared to a benchmark, produced from both databases, mirroring the recycling operation, utilizing averaged European data. Evidently, the midpoints for the allocation and non-allocation cases were considerably higher than those of the benchmark, except for ionising radiation in the Austrian cases (due to the absence of nuclear power in Austria). Most categories were between 100 and 500 % higher than the benchmark, except for ozone depletion, which was approximately 47,500 % higher. Upon examining single contributions (without disclosing any specific numbers or figures for confidentiality reasons), we discovered that waste water treatment was the decisive contributor, with over 95 % of the total impact for ozone depletion. The main flow responsible was the production of a chemical, which is commonly used in waste water treatment, which is also included in the benchmark scenario. The exact reason for the shown discrepancy is hard to determine, yet the values presented in this work, which are similar in all 12 scenarios and lie around 2×10^{-9} kg CFC-eq., compare well to other literature values such as 7.26×10^{-9} kg CFC-eq. in the work of Martín-Lara et al. (2022). Some midpoints, such as acidification, terrestrial eutrophication, and particulate matter formation, had negative values because of the crediting of hard coal avoided in cement kilns. Heijungs and Guinée (2007) (Waste Management) have already explored the problems with this approach and the results obtained, labelling the challenges with the avoided-burden approach as "insurmountable". They especially criticised the introduction of more "what-ifs" into LCA. Moreover, Ekvall and Weidema (2004) noted that consequential LCA requires further investigation into market mechanisms or marginal technologies, raising questions such as "will hard coal be the first fuel to be replaced?" or "is there a financial incentive for using reject instead of hard coal?". In a more recent work, Ekvall et al. (2021) also questioned the approach to infer implications (credits) for the recycling/waste management system based on a single product flow, similar to our study. Therefore, a decision was made to use the most conservative approach, the no allocation approach, for SB2.

3.4. *Polymer testing results for the LDPE products used in system boundary 2*

The products' material properties are listed in Table 1.

Tensile strength was strongly correlated with the percentage of virgin material used. Comparing the 100 % *v*-LDPE waste bags with the 100 % *r*-LDPE products, the value was more than 50 % higher. Interestingly, the weight per square metre showed an inverse correlation, with virgin product being much lighter than the products containing *r*-LDPE. The difference in weight reached factors between 2.36 and 3.67 times for the waste bags, depending on the comparison methodology. Table 1 displays the two different modes. The first mode is more straightforward, which translates to weighing the purchased article. The second method involved a comparison of the weight of 1 m^2 of the product. Therefore, the weight of the normalised specimen was recorded. The reason for the

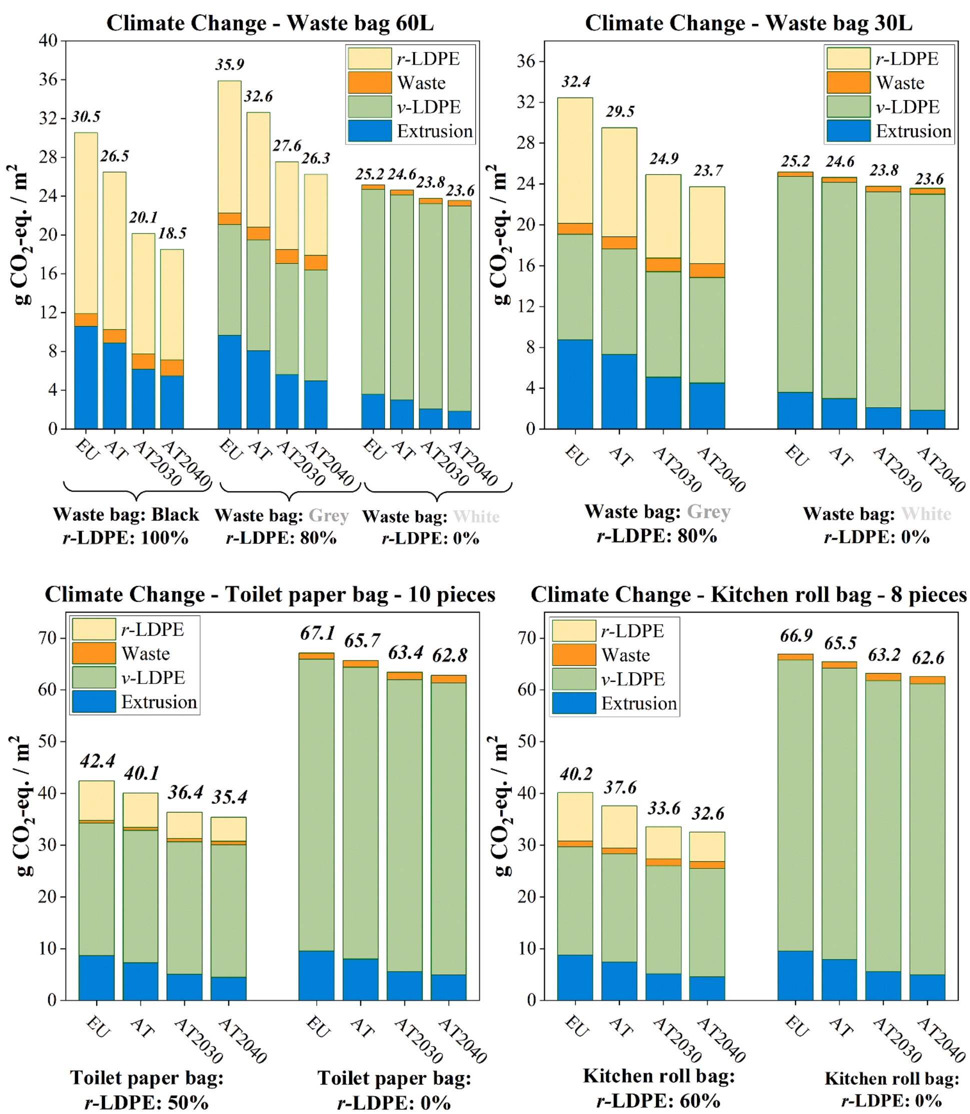

Fig. 3. The results for the impact category of climate change for the different case studies. The electricity mix used is shown on the x-axis, and the total result is presented at the top of the columns.

second mode was the elimination of product design, as some waste bags had drawstrings whereas others did not. Ultimately, products with larger weights, exhibited a much higher puncture force tolerance, likely attributed to their thicker walls and higher number of impurities. The optical microscopy results showing evidence of contamination are presented in the Supporting Information.

3.5. Modelling of system boundary 2 and sensitivity analysis

The definition of goal and scope differed for SB2. In SB2, the functional unit was the production of a sufficient amount of film to safely transport the contained product from the factory gate to the retail establishment, the consumer's residence, and then store it until consumed. The system boundary was defined as "cradle-to-gate". For the mass balance, the two approaches described in Section 3.4 were investigated. Ultimately, especially for the toilet paper bag and kitchen roll bag, the difference was very small. This indicates a good recycling material input. For the waste bags, the total-mass approach delivered similar ratios. As discussed before, we decided to use the weight of 1 m^2 film product to eliminate additional design features and focus solely on the LDPE film. The amounts of *r*-LDPE material required were modelled using the same process as the findings presented in Fig. 1 and Fig. 2. The required *v*-LDPE was analysed using a value of 1.87 kg CO_2-eq./kg *v*-LDPE, as reported by PlasticsEurope (2014) in the respective Eco-profile. The final required step was extrusion, which was sourced from the ecoinvent database v3.8 and the "*RER: extrusion, plastic film*" dataset (Wernet et al., 2016). Electricity grid mixes were again altered as in SB1. A small fraction of waste plastic was produced during extrusion, and it was assumed to be thermally recovered.

3.6. LCIA results: system boundary 2

A graphical representation of the impacts of the products on climate change in the cradle-to-gate system boundary is shown in Fig. 3. For the 60-L waste bag, the 100 % recycled product shows the lowest value for CO_2-eq., aligning with the concept of recycling as the environmentally friendly option, if a green electricity source can be provided. Interestingly, the Austrian grid is already insufficient to make the *v*-LDPE bag preferable in terms of climate change mitigation, with results worsening for the EU-grid. The grey bag, made from 80 % *r*-LDPE, was an interesting case, as the 100 % *v*-LDPE benchmark always showed better results, irrelevant of the used electricity source. The smaller 30-L bag behaved similarly as a clean grid showed CO_2-eq. on a comparable level to the 100 % *v*-LDPE benchmark, albeit a little higher than the virgin benchmark. The two waste bags clearly confirm that the avoided CO_2-eq. emissions from using recycled material instead of virgin input, are not always sufficient to provide a product with better CO_2-eq. balance, as the *v*-LDPE products needed much less raw material. These results indicate that there is a threshold beyond which using recyclates is no longer preferable, as discussed in the next chapter. The kitchen roll and toilet paper bags were very different as the product containing recyclate was lighter than the virgin benchmark. This means the utilisation of *r*-LDPE resulted in much lower carbon dioxide equivalent emissions, with the reduction approaching a factor of 50 % in the AT2040 case. The comparison between waste bags and other products offers strong evidence for a previously discussed point: quality is of the utmost importance when improving the sustainability of recycling. Only upon approaching low substitution ratios around 1, a considerable reduction of CO_2-eq. emissions can be expected. Thus, the case studies make a compelling argument for striving for the best quality outputs in

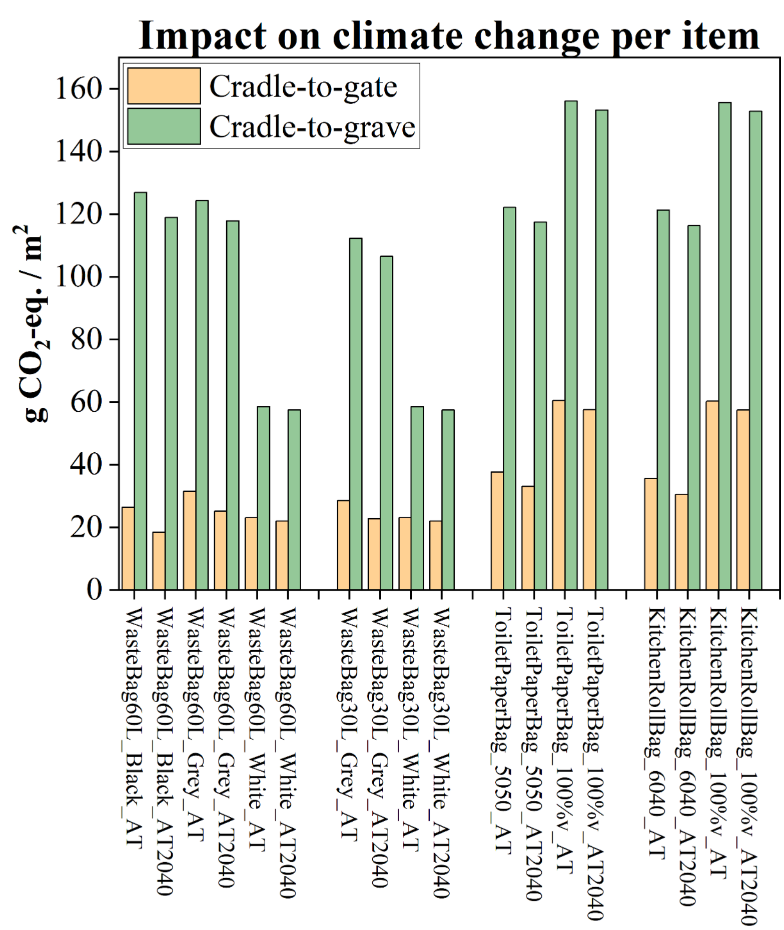

Fig. 4. Impact on climate change of four case study product systems using different electricity grids and system boundaries.

mechanical recycling to achieve the maximum emission reductions.

3.7. LCIA results of system boundary 2: end of life, the recycled content threshold and the environmental break-even

A complete LCA should always aim at describing the whole life cycle of a product. Therefore, the end of life (EoL) should also be discussed. Previously, we cited the work of Heijungs and Guinée (2007), who identified the problems within the substitution approach as “insurmountable”. They argue that there is no way of knowing or estimating whether the products would be entering one or more additional life cycles. Fortunately, this did not pose a significant problem for our approach, as we did not include any credits to begin with. To keep this assumption valid, we assumed that products underwent incineration at the end of their lifetime and do not enter a third life-cycle, presenting a worst-case scenario. For the calculation, a literature value of 3 kg CO_2-eq./kg LDPE incinerated was used (d’Ambrières, 2019; Edwards and Parker, 2012). The results for the two different SBs, cradle-to-gate and cradle-to-grave, are shown in Fig. 4, respectively. For brevity, we included only the Austrian (AT) and AT2040 mixes.

When the contributions of thermal valorisation are factored into the results, the figures increase considerably. The most drastic increase is found for the waste bags, as much more mass is required per m^2 when using *r*-LDPE. The 60-L black bag (100 % *r*-LDPE) had a greater impact on climate change in SB2, but not necessarily in SB1. Upon comparing the higher-quality products with the waste bags it becomes obvious that the surplus of CO_2-eq. emitted in SB2 is not as high for the high-quality products, due to the masses of the pure virgin products and those containing the recyclate were very similar. Therefore, the conclusion is that a cradle-to-grave LCA can unveil the adverse effects of excess low-quality recyclate. Naturally these findings lead to the question of a possible environmental break-even point: “*How much more low-quality input material can one use to reach the same impact on climate change as high-quality virgin input*?” The emissions per gram in both SBs were calculated, and the environmental break-even was determined by computing the multiplication factor required to match the CO_2-eq. emissions. The results were plotted and fitted with an exponential function and presented in Fig. 5. The data points were derived from two waste bag products. Because the LCA design was the same for both product categories (*v*-LDPE, *r*-LDPE, extrusion, and EoL), the emission factors were scaled by mass/m^2. Dividing by mass leads to the same

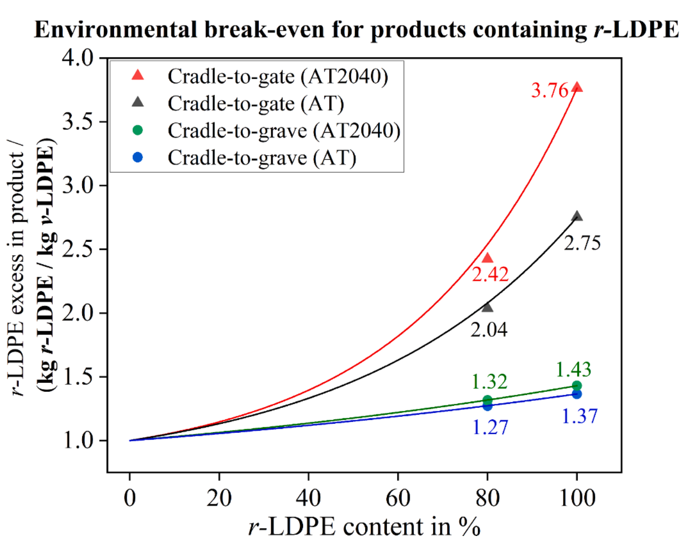

Fig. 5. The curves for environmental break-even when using more low-quality recyclates as a benchmark for virgin material. The plots were derived from figures obtained from the 60-L waste bags, as they were the only case studies featuring three data points ranging from 0 to 100 % *r*-LDPE.

ratios for both cases, the 30-L and 60-L waste bag, as they are made from the same material and manufacturer.

The curves in Fig. 5 can be used to quickly assess whether a conceptual process design is environmentally beneficial. If the product is under the curves for a given SB and electricity mix, it should feature a lower climate change impact. For example, if 60 % low-quality *r*-LDPE is used in a product and the resulting product has three times the mass of the virgin competitor, it is highly probable that the product is not a sustainable alternative in terms of climate change mitigation. The fitting was performed using an exponential function. Details can be found in the Supporting Information. However, for the cradle-to-grave products, a linear fit also reaches satisfying values of R^2 over 0.99. The 30-L bags were not included, because a 100 % *r*-LDPE product could not be obtained. The idea of environmental break-even points has been applied before, e.g. in finding exact numbers such as the mileage required for electric cars to break-even with conventional ones (Dillman et al., 2020), or the number of reuses a reusable cup needs to see before breaking-even with a single-use drinking cup (Cottafava et al., 2021). Our approach however, shows curves based upon primary data, which allows producers to qualitatively assess whether an excess use of low-quality material leads to higher or lower kg CO_2-eq. emissions when compared to a virgin-based product design which is more frugal material-wise, at a quick glance. While environmental claims should be thoroughly examined, this method eliminates inefficient designs right at the drawing table and can help to save resources.

The approach presented is only feasible for fossil-based polymers for now. Although the literature suggests that biobased PE can lower CO_2-eq. emissions in comparison to fossil PE (Benavides et al., 2020; Tsiropoulos et al., 2015), these savings are often accompanied by increased impacts in other areas such as acidification, eutrophication or land use – a common finding for different biobased products and their respective fossil benchmarks. (Moretti et al., 2021; Mousavi-Avval et al., 2023). Biobased PE is described as a future growth market, albeit global plastic production remains almost exclusively on a petrochemical basis, as only 1 % of plastics worldwide is estimated to be of biobased origin (Soo et al., 2024). Therefore, no biobased PE was discussed in the break-even curve, as it represents a negligible fraction for the moment. Contrarily, the approach is suitable for advanced LCA methods which also include former life cycles or multiple recycling loops, as only a result for a climate change impact is needed, not taking the calculation method into account. The CO_2-eq. produced for certain pathways and the differences thereof are of great interest for designing policy mechanisms to drive clean development mechanisms, such as emission trading schemes (Bing et al., 2015). This is especially relevant for plastics, as the literature suggests that plastics recycling is rather cost-ineffective in comparison to other “green” technologies (Gradus et al., 2017). In their work on net-zero chemicals and plastics (Zibunas et al., 2022) also postulate a level playing field by including incineration in carbon pricing, as well as precise sustainability monitoring, which could both be met with the approach presented.

3.8. Limitations and weaknesses of the study

Naturally, an undisclosed LCI is a significant weakness in any LCA study. We aimed to demonstrate the integrity of the study by providing detailed insights into the process and presenting comparable results from the literature. Should those facts not sufficiently convince the reader, it is worth noting that only climate change was investigated for SB2. Therefore, the companies’ system should no longer be relevant, as literature values are available and could have been used instead. Unfortunately, the fitting results did not contain many data points; however, we remain optimistic that other researchers will adopt our approach and report its viability and accuracy. Moreover, we did not include a generic formula to produce curves such as in Fig. 5 which would allow for a generic input of the carbon intensity of the electricity used. The results remain highly dependant on the chosen data for *v*-

LDPE production, *r*-LDPE production, extrusion, and EoL. We aimed to establish a framework and contribute to qualified discussions rather than provide a tried-and-tested tool. From a material science perspective, it cannot be assumed with absolute certainty that the material from the recycling plant in question (SB1) would be usable for the products in SB2

Additionally, the primary company data used as well as material testing performed make replication cumbersome, as considerable efforts are needed to arrive at a similar LCI, showcasing the need for more published and trustworthy data on plastics recycling. As recently confirmed by Pellengahr et al. (2023) hands-on data is indispensable for performing high-level LCA on plastics. Another weakness and possible future research question is found in the fact, that only one polymer type was examined in this work.

4. Conclusion

Our study presents LCA results based on primary data for producing *r*-LDPE at an existing company in SB1. These results were benchmarked by using processes available in a commercial LCA database to create a similar system, which showed comparable results for climate change and highly different results for other midpoint categories. This poses an issue as detailed primary data is hardly available in the literature.

In SB2, we discovered that using a higher mass of low-quality recyclates may not always be preferable over a virgin competitor. Although *r*-LDPE features a much lower climate change impact (approximately 0.3–0.5 kg CO_2 eq./kg) than that of *v*-LDPE (1.87 kg CO_2 eq./kg), an environmental break-even point is reached at a certain threshold, when using high quantities of low-quality material. By changing the SB from a cradle-to-gate to a cradle-to-grave approach and assuming the incineration of low-quality *r*-LDPE, this break-even point decreases considerably due to the rather high emissions of incinerated LDPE. For the higher-quality recyclates used for transporting toilet and kitchen paper, these findings do not apply because their masses are almost equivalent to the 100 % *v*-LDPE benchmark. In these cases, the impact on climate change can be considerably reduced by recycling because the reduced CO_2-eq. emissions of *r*-LDPE are not negated by the increase in material mass used.

This study proves that fostering recycling by simply increasing the percentage of plastics recycled without considering the recyclates' quality and overall recycling efficiency may not be the most effective approach for mitigating climate change. As assuming rather high substitution rations in LCAs on plastics recycling is still commonly found (Alhazmi et al., 2021), authors should verify those claims by material testing or case studies to provide robust results. Cullen (2017) elegantly explained that as we approach perfect circularity, the efforts required for additional improvements grow exponentially, based on the principles of thermodynamics. In addition, Lase et al. (2022) performed extensive testing on an improved mechanical recycling process and discovered that yields could not be substantially increased for high-quality recycling, but better qualities could be obtained.

CRediT authorship contribution statement

Lukas Zeilerbauer: Writing – original draft, Visualization, Project administration, Methodology, Investigation, Formal analysis, Conceptualization. **Jörg Fischer:** Writing – review & editing, Supervision, Project administration, Funding acquisition, Conceptualization. **Karin Fazeni-Fraisl:** Writing – review & editing, Supervision, Resources, Project administration, Funding acquisition, Conceptualization. **Moritz Mager:** Visualization, Resources, Investigation, Data curation. **Johannes Lindorfer:** Writing – review & editing, Validation. **Christian Paulik:** Writing – review & editing, Supervision.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

The data that has been used is confidential.

Acknowledgements

This work received funding from the Federal Ministry for Climate Action, Environment, Energy, Mobility, Innovation and Technology Austria (BMK) within the lead project "circPLAST-mr" in the "1. Ausschreibung Initiative Kreislaufwirtschaft," which is handled by the Austrian Research Promotion Agency (FFG).

Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.resconrec.2024.107807.

References

Alhazmi, H., Almansour, F.H., Aldhafeeri, Z., 2021. Plastic waste management: a review of existing life cycle assessment studies. Sustainability 13 (10). https://doi.org/10.3390/su13105340. Article 10.

Austrian ministry for sustainability and tourism, 2019. Integrated National Energy and Climate Plan For Austria. https://ec.europa.eu/info/energy-climate-change-environment/implementation-eu-countries/energy-and-climate-governance-and-reporting/national-energy-and-climate-plans_en.

Benavides, P.T., Lee, U., Zarè-Mehrjerdi, O., 2020. Life cycle greenhouse gas emissions and energy use of polylactic acid, bio-derived polyethylene, and fossil-derived polyethylene. J. Clean. Prod. 277, 124010 https://doi.org/10.1016/j.jclepro.2020.124010.

Bing, X., Bloemhof-Ruwaard, J., Chaabane, A., van der Vorst, J, 2015. Global reverse supply chain redesign for household plastic waste under the emission trading scheme. J. Clean. Prod. 103, 28–39. https://doi.org/10.1016/j.jclepro.2015.02.019.

Chen, Y., Cui, Z., Cui, X., Liu, W., Wang, X., Li, X., Li, S., 2019. Life cycle assessment of end-of-life treatments of waste plastics in China. Resour. Conserv. Recycl. 146, 348–357. https://doi.org/10.1016/j.resconrec.2019.03.011.

Civancik-Uslu, D., Nhu, T.T., Van Gorp, B., Kresovic, U., Larrain, M., Billen, P., Ragaert, K., De Meester, S., Dewulf, J., Huysveld, S., 2021. Moving from linear to circular household plastic packaging in Belgium: prospective life cycle assessment of mechanical and thermochemical recycling. Resour. Conserv. Recycl. 171, 105633 https://doi.org/10.1016/j.resconrec.2021.105633.

Cottafava, D., Costamagna, M., Baricco, M., Corazza, L., Miceli, D., Riccardo, L.E., 2021. Assessment of the environmental break-even point for deposit return systems through an LCA analysis of single-use and reusable cups. Sustainable Production and Consumption 27, 228–241. https://doi.org/10.1016/j.spc.2020.11.002.

Cullen, J.M., 2017. Circular economy: theoretical benchmark or perpetual motion machine? J. Ind. Ecol. 21 (3), 483–486. https://doi.org/10.1111/jiec.12599.

d'Ambrières, W., 2019. Plastics recycling worldwide: current overview and desirable changes. Field Actions Sci. Reports 19, 12–21.

Dillman, K.J., Árnadóttir, Á., Heinonen, J., Czepkiewicz, M., Davíðsdóttir, B., 2020. Review and meta-analysis of EVs: embodied emissions and environmental breakeven. Sustainability 2 (22), 12. https://doi.org/10.3390/su12229390. Article 2.

Edwards, C., Parker, G., 2012. A Life Cycle Assessment of Oxo-biodegradable, Compostable and Conventional Bags. https://www.biodeg.org/wp-content/uploads/2021/05/intertek-final-report-15.5.121.pdf.

Ekvall, T., Gottfridsson, M., Nellström, M., Nilsson, J., Rydberg, M., Rydberg, T., 2021. Modelling incineration for more accurate comparisons to recycling in PEF and LCA. Waste Manage. (Oxford) 136, 153–161. https://doi.org/10.1016/j.wasman.2021.09.036.

Ekvall, T., Weidema, B.P., 2004. System boundaries and input data in consequential life cycle inventory analysis. Int. J. Life Cycle Assess. 9 (3), 161–171. https://doi.org/10.1007/BF02994190.

Eurostat, 2022. Plastic Packaging waste: 38% Recycled in 2020. https://ec.europa.eu/eurostat/web/products-eurostat-news/-/ddn-20221020-1.

Felgel-Farnholz, A., Schweighuber, A., Klampfl, C.W., Fischer, J., 2023. Comparative study on the degradation of HDPE, LLDPE and LDPE during multiple extrusions. Polym. Degrad. Stab. 216, 110486 https://doi.org/10.1016/j.polymdegradstab.2023.110486.

Geyer, R., Jambeck, J.R., Law, K.L., 2017. Production, use, and fate of all plastics ever made. Sci. Adv. 3 (7), e1700782 https://doi.org/10.1126/sciadv.1700782.

Gradus, R.H.J.M., Nillesen, P.H.L., Dijkgraaf, E., van Koppen, R.J., 2017. A cost-effectiveness analysis for incineration or recycling of dutch household plastic waste. Ecol. Econ. 135, 22–28. https://doi.org/10.1016/j.ecolecon.2016.12.021.

Heijungs, R., Guinée, J.B., 2007. Allocation and 'what-if' scenarios in life cycle assessment of waste management systems. Waste Manage. (Oxford) 27 (8), 997–1005. https://doi.org/10.1016/j.wasman.2007.02.013.

Holly, F., Kolar, G., Berger, M., Fink, S., Ogonowski, P., Schlund, S., 2023. Challenges on the way to a circular economy from the perspective of the Austrian manufacturing industry. Front. Sustainabil. 4, 1243374 https://doi.org/10.3389/frsus.2023.1243374.

Istrate, I.-R., Juan, R., Martin-Gamboa, M., Domínguez, C., García-Muñoz, R.A., Dufour, J., 2021. Environmental life cycle assessment of the incorporation of recycled high-density polyethylene to polyethylene pipe grade resins. J. Clean. Prod. 319, 128580 https://doi.org/10.1016/j.jclepro.2021.128580.

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

Khoo, H.H., 2019. LCA of plastic waste recovery into recycled materials, energy and fuels in Singapore. Resour. Conserv. Recycl. 145, 67–77. https://doi.org/10.1016/j.resconrec.2019.02.010.

Lase, I.S., Bashirgonbadi, A., van Rhijn, F., Dewulf, J., Ragaert, K., Delva, L., Roosen, M., Brandsma, M., Langen, M., De Meester, S., 2022. Material flow analysis and recycling performance of an improved mechanical recycling process for post-consumer flexible plastics. Waste Manage. (Oxford) 153, 249–263. https://doi.org/10.1016/j.wasman.2022.09.002.

Martín-Lara, M.A., Moreno, J.A., Garcia-Garcia, G., Arjandas, S., Calero, M., 2022. Life cycle assessment of mechanical recycling of post-consumer polyethylene flexible films based on a real case in Spain. J Clean Prod 365, 132625. https://doi.org/10.1016/j.jclepro.2022.132625.

Mauschitz, G., 2023. Emissionen Aus Anlagen der österreichischen Zementindustrie Berichtsjahr 2022. https://www.zement.at/downloads/downloads_2023/Emissionen_aus_Anlagen_der_%C3%B6sterreichischen_Zementindustrie_Bilanzjahr_2022.pdf.

Mordaschew, V., Tackenberg, S., 2024. The product environmental footprint – A critical review. Procedia Comput Sci 232, 493–503. https://doi.org/10.1016/j.procs.2024.01.049.

Moretti, C., Hamelin, L., Jakobsen, L.G., Junginger, M.H., Steingrimsdottir, M.M., Høibye, L., Shen, L., 2021. Cradle-to-grave life cycle assessment of single-use cups made from PLA, PP and PET. Resour. Conserv. Recycl. 169, 105508 https://doi.org/10.1016/j.resconrec.2021.105508.

Mousavi-Avval, S.H., Sahoo, K., Nepal, P., Runge, T., Bergman, R., 2023. Environmental impacts and techno-economic assessments of biobased products: a review. Renewable Sustainable Energy Rev. 180, 113302 https://doi.org/10.1016/j.rser.2023.113302.

Pellengahr, F., Ghannadzadeh, A., van der Meer, Y., 2023. How accurate is plastic end-of-life modeling in LCA? Investigating the main assumptions and deviations for the end-of-life management of plastic packaging. Sustainable Production and Consumption 42, 170–182. https://doi.org/10.1016/j.spc.2023.09.014.

PlasticsEurope, 2014. Eco-profiles Set • Plastics Europe. Plastics Europe. https://plasticseurope.org/sustainability/circularity/life-cycle-thinking/eco-profiles-set/.

Rajendran, S., Hodzic, A., Scelsi, L., Hayes, S., Soutis, C., AlMa'adeed, M., Kahraman, R., 2013. Plastics recycling: insights into life cycle impact assessment methods. Plast. Rubber Compos. 42 (1), 1–10. https://doi.org/10.1179/1743289812Y.0000000002.

Schyns, Z.O.G., Shaver, M.P., 2021. Mechanical recycling of packaging plastics: a review. Macromol. Rapid Commun. 42 (3), 2000415 https://doi.org/10.1002/marc.202000415.

Soo, X.Y.D., Muiruri, J.K., Wu, W.-Y., Yeo, J.C.C., Wang, S., Tomczak, N., Thitsartarn, W., Tan, B.H., Wang, P., Wei, F., Suwardi, A., Xu, J., Loh, X.J., Yan, Q., Zhu, Q., 2024. Bio-polyethylene and polyethylene biocomposites: an alternative toward a sustainable future. Macromol. Rapid Commun., 2400064 https://doi.org/10.1002/marc.202400064. *n/a*(n/a).

Sphera, 2023. Life Cycle Assessment Software—Sphera LCA For Experts. Sphera (GaBi). Sphera. https://sphera.com/life-cycle-assessment-lca-software/.

Tsiropoulos, I., Faaij, A.P.C., Lundquist, L., Schenker, U., Briois, J.F., Patel, M.K., 2015. Life cycle impact assessment of bio-based plastics from sugarcane ethanol. J Clean Prod 90, 114–127. https://doi.org/10.1016/j.jclepro.2014.11.071.

Van Eygen, E., Laner, D., Fellner, J., 2018. Circular economy of plastic packaging: current practice and perspectives in Austria. Waste Manage. (Oxford) 72, 55–64. https://doi.org/10.1016/j.wasman.2017.11.040.

van Rossem, C., 2023. State of Recycling Technologies For Flexible Plastic Packaging In Europe. EPRD Office for Economic Policy and Regional Development Ltd. https://www.eeas.europa.eu/sites/default/files/documents/2023/EU%20Project%20in%20Canada%20-%20Technologies%20for%20Flexible%20Plastic%20Packaging%20in%20Europe%20January%202023_0.pdf.

Vilbergsson, K.V., 2021. Comparative Life Cycle Assessment of Hydrogen Production Via Electrolysis in Austria and Iceland. University of Reykjavik.

Wernet, G., Bauer, C., Steubing, B., Reinhard, J., Moreno-Ruiz, E., Weidema, B., 2016. The ecoinvent database version 3 (part I): overview and methodology. Int. J. Life Cycle Assess. 21 (9), 1218–1230. https://doi.org/10.1007/s11367-016-1087-8.

Zibunas, C., Meys, R., Kätelhön, A., Bardow, A., 2022. Cost-optimal pathways towards net-zero chemicals and plastics based on a circular carbon economy. Comput. Chem. Eng. 162, 107798 https://doi.org/10.1016/j.compchemeng.2022.107798.