SWEDISH LIFE CYCLE CENTER

Incentives for recycling and incineration in LCA: Polymers in Product Environmental Footprints

Report number: 2021:02
January 2021 — Gothenburg, Sweden

January 2021
Gothenburg, Sweden
Swedish Life Cycle Center, Chalmers University of Technology
Report no (Swedish Life Cycle Center’s report series): 2021:02
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Summary

For material recycling to occur, waste material from a product life cycle must be made available for recycling and then used in the production of a new product. When recycling is beneficial for the environment, the LCA results should give incentives to collection for recycling and also to the use of recycled material in new products. However, most established methods for modelling recycling in LCA risk giving little or even wrong incentives. Many methods, such as the Circular Footprint Formula (CFF) in a Product Environmental Footprint (PEF), assign some of the environmental benefits of recycling to the product that uses recycled materials. This means that the incentive to send used products for recycling will be lower. If energy recovery also provides an environmental benefit, because the energy recovered substitutes energy supplied with a greater environmental impact, the LCA results may indicate that the waste should instead be sent to incineration – even when recycling is the environmentally preferable option for the society.

This study aims to increase the knowledge on the extent to which PEF results, and LCA results in general, risk giving incorrect incentives for energy recovery from plastic waste. Our calculations focus on the climate impact of the recycling and incineration of LDPE waste generated in Sweden. Since this is a pilot study, we use easily available input data only. We estimate the net climate benefit through simple substitution, where recycled material is assumed to replace virgin material and where energy recovered from LDPE waste is assumed to replace average Swedish district heat and electricity. We then apply the CFF to find whether a PEF would give the same indications. Our results show no risk of a PEF or LCA giving incorrect climate incentives for incineration of fossil LDPE. However, an LCA can wrongly indicate that renewable LDPE should be incinerated rather than recycled. Our results indicate this can happen in a PEF when the heat and electricity substituted by incineration has 40-200% more climate impact than the Swedish average district heat and electricity.

Our study also aims to increase knowledge about the extent to which correct incentives can be obtained through a more thorough analysis of incineration with energy recovery – specifically, through:

1. a deeper understanding of Factor B, which in the CFF can be used to assign part of the burdens and benefits of energy recovery to the energy instead of the product investigated, but which in the PEF guidelines by default is set to 0, or
2. a broader systems perspective that accounts for the effects of energy recovery on waste imports and thus waste management in other countries.

We estimate Factor B based on the observation that waste incineration can be described as a process with multiple jointly determining functions. Waste treatment and energy recovery both contribute to driving investments in incineration. This, in turn, defines the volume of waste incinerated, the quantity of energy recovered, and the quantity of energy substituted. We propose that expected revenues from gate fees and energy are an appropriate basis for calculating Factor B. Up-to-date estimates of the expected revenues in the relevant region should ideally be used for the calculations. Lacking such data, we suggest the value B=0.6 can be used in the CFF when modelling waste incineration in Sweden. Our PEF calculations with Factor B=0.6 indicate such a PEF will identify the environmentally best option for plastic waste management in almost all cases. However, this is at least in part luck: Factor B will vary over time and between locations, and other parts of the CFF varies between materials.

To account for the broader systems perspective, we develop two scenarios based on different assumptions on whether change in Swedish waste imports affects the incineration or landfilling in other European countries. The scenarios bring a large uncertainty into the results. This uncertainty is real in the sense that it is difficult to know how a change in Swedish waste imports in the end will affect waste management in other countries. The uncertainty still

makes it difficult to draw conclusions on whether renewable LDPE should be recycled or incinerated.

Our suggestions for Factor B and European scenarios both make the CFF more balanced and consistent: it now recognizes that not only recycling but also energy recovery depends on more than the flow of waste from the life cycle investigated. However, neither Factor B nor the broader systems perspective amends the fact that LCA tends to focus on one product at a time. This might not be enough to guide a development that requires coordinated or concerted actions between actors in different life cycles – such as increased recycling or energy recovery. Assessing decisions in one product life cycle at a time might in this context be compared to independently assessing the action of clapping one hand. This will most probably not result in an applaud.

Besides a more thorough assessment of energy recovery, we also discuss the option to give correct incentives for recycling from LCA by assigning the full environmental benefit of recycling to the product that generates waste for recycling but also to the product where the recycled material is used. We find that this 100/100 approach can give negative LCA results for products produced from recycled material and recycled to a high degree after recycling, because the benefits of recycling are counted twice. The LCA results would indicate that you save material resources by producing and recycling such products without ever using them. The 100/100 approach also lacks additivity, does not model foreseeable consequences, and does not assign a well-defined environmental value to the recovered secondary material.

To guide concerted actions, like recycling or energy recovery, it seems systems analysis should ideally assess the necessary actions in combination. Many situations require the environmental impacts to be estimated for a specific product or a specific action. In some cases, however, the LCA results can be calculated and presented with, for example, the following introduction:

“When the material is sent to recycling, you will, together with the recycler and the actor using the recycled material, jointly achieve this net environmental benefit: ...”

Such joint assessment of supply and demand for secondary materials means the allocation problem is avoided. It is also consistent with the recommendation in the old SETAC “Code of Practice” to assess life cycles with recycling by studying the inputs and outputs from the total linked system.

Project organizations

SWEDISH LIFE CYCLE CENTER

CHALMERS

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TERRA

Contents

Project information

Project title

Incitament för energiutvinning i livscykelanalys av plast / Incentives for energy recovery in LCA for plastics

Funded by

Swedish Environmental Protection Agency

Aim

This pilot project aims to increase knowledge on the extent to which results from a life cycle assessment provide incorrect incentives for energy recovery from plastic waste. It also aims to increase knowledge about the extent to which this problem can be addressed through 1) use of Factor B in the methodology for Product Environmental Footprints, and 2) a broader system perspective that includes the effects of energy recovery on waste imports and thus waste management in other countries. We do this by combining case study calculations for plastic waste with the knowledge and insights from previous projects.

Project leader

Maria Rydberg, project manager at Swedish Life Cycle Center

Coordination of the project

Maria Rydberg, project manager at Swedish Life Cycle Center

Project team

This project was carried through within the competence center Swedish Life Cycle Center. The research was conducted by docent Tomas Ekvall at TERRA (Tomas Ekvall Research, Review & Assessment AB) in cooperation with Tomas Rydberg, Marie Gottfridsson, Johan Nilsson, and Maja Nellström at IVL Swedish Environmental Research Institute.

Time period

From November 2020 to January 2021

Acknowledgements

The project was funded by Swedish Environmental Protection Agency as part of their task to contribute to a sustainable use of polymers in Sweden. We are grateful to Pernilla Cederstrand at Essity for making us aware of the risk for incorrect incentives for incineration. We are also grateful to Magdalena Svanström, Chalmers University of Technology, for pointing at the 100/100 approach as a possible solution.

About Swedish Life Cycle Center

Swedish Life Cycle Center is a national competence center for credible and applied life cycle thinking in industry and society. More information on www.lifecyclecenter.se.

Abbreviations

ALCA	Attributional LCA, i.e., an LCA that aims to identify the share of the global activities and their environmental burdens that belong to a product system.
CFF	Circular Footprint Formula
CLCA	Consequential LCA, i.e., an LCA that aims to estimate how the global environmental burdens are affected by the production and use of the product investigated.
EC	European Commission
EPD	Environmental Product Declaration
EU	European Union
ISO	International Organization of Standardization
LCA	Life cycle assessment
LDPE	Low-density polyethylene
LHV	Lower heating value
PEF	Product Environmental Footprint

1. Introduction

Background

Life cycle assessment

A life cycle assessment (LCA) is typically an environmental assessment of a single product. Here we distinguish between attributional LCA (ALCA) and consequential LCA (CLCA). The former aims to identify what share of the global environmental impacts that belongs to the product (see Figure 1). The latter aims to quantify how the production and use of the product affects the global environmental impacts.

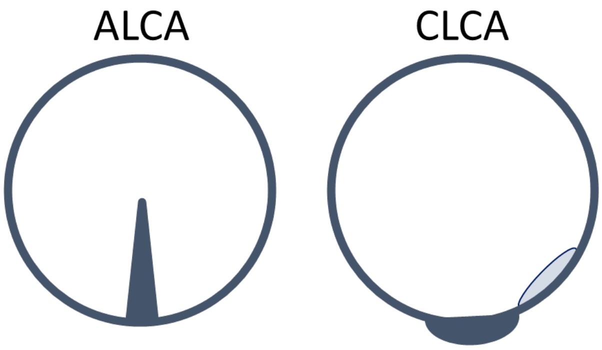

Figure 1 – Illustration of attributional and consequential LCA, were the circle symbolizes the global environmental impacts. Source: Weidema 2003.

Modelling recycling

When material is recycled from one product into another (Figure 2), an allocation problem arises. In an ALCA the challenge is to decide what share of the environmental burdens of virgin production (E_V), recycling (E_R) and final disposal (E_D) of the material belongs to the product investigated.

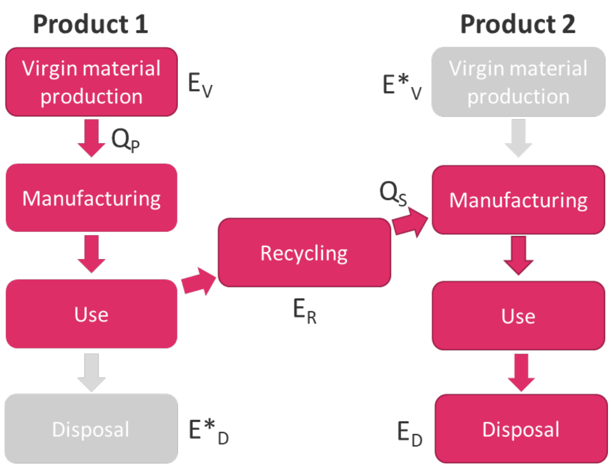

*Figure 2 – Illustration of recycling. EV, ER and ED are the environmental burdens of virgin material production, recycling, and final disposal, respectively. E*D and E*V are the environmental burdens of the disposal and virgin production avoided through recycling. QP and QS are the quality of primary and secondary material, respectively.*

In a CLCA the focus is instead on the consequences of using recycled material and of recycling the product after use. The recycling process has an environmental impact (E_R). On the other hand, recycling means impacts of virgin material production (E^*_V) and final disposal (E^*_D) are both avoided. This typically results in a net environmental benefit: $E^*_V+E^*_D-E_R>0$. The challenge is then to decide how much of this benefit is due to waste being sent to recycling, and how much is due to recycled material being used in new products.

Many different methods have been developed and applied to model recycling in LCA. The results of attributional approaches should be additive in the sense that the environmental burdens allocated to the products where the material is used (Products 1 and 2 in Figure 2) should add up to the total environmental burdens of virgin material production, recycling and final disposal (E_V+E_R+E_D).

Consequential approaches to the allocation problem often also generate additive results in the sense that the net environmental benefit assigned to the product recycled after use (Product 1) plus the benefit assigned to the product that use the recycled material instead of virgin material (Product 2) adds up to the total net environmental benefit of recycling (E^*_V+E^*_D-E_R). This happens as soon as the credit obtained for sending a material to recycling is the same as the burden assigned to the recyclable material when used as input in a new product. This credit or burden can be called the environmental value of the recyclable material (Tillman et al. 1994; Ekvall 2000).

A common method to model recycling is to assume the recycled material to replace virgin material. The entire environmental benefit of recycling is then attributed to the product that is recycled after use. The entire responsibility for virgin material production is attributed to the product that is not recycled, regardless of whether it was produced from virgin or recycled material. This approach is known under many names, for example the end-of-life approach.

The cut-off approach is also common. It is the main approach in, for example, Environmental Product Declarations (EPDs). With a cut-off approach, most of the environmental benefit is attributed to the product produced from recycled materials. This is because the product that consists of virgin material bear full responsibility for that material production.

Several methods for modelling recycling and energy recovery have been proposed within the EU initiative to develop a common LCA methodology called Product Environmental Footprint (PEF). An early version of the PEF Guide (JRC 2012, Annex V) includes a formula for calculating the so-called Resource Use and Emissions Profile (RUaEP). This formula assigned no burden to recyclable material used as input to the product; however, it assigned a credit for sending a material to recycling corresponding to (using notation from Figure 2) $Q_S/Q_P \times E^*_V - E_R$.

The RUaEP formula was soon replaced by the-so-called End-of-Life (EoL) formula (EC 2013, Annex V). Despite similar names, the EoL formula is quite different from the end-of-life approach above. The EoL formula models recycling with a quality-adjusted 50/50 method scientifically published by Allacker et al. (2017). This approach assigns an environmental value to recyclable inputs as well as outputs. The environmental value of recyclable input material is $0.5 \times (E_V - E_R)$. The environmental value assigned to recyclable output material is quality-adjusted: $0.5 \times (Q_S/Q_P \times E^*_V - E_R)$.

The current PEF methodology models recycling and energy recovery through an equation called the Circular Footprint Formula (CFF; EC 2018a). This formula divides the environmental benefits of recycling between Product 1 and Product 2. The shares are determined by a factor A, which varies depending on the material recycled, and by quality losses in the recycling. In plastic recycling, the default values are A=0.5 and Q_S/Q_P=0.9. This means that the product that is recycled by default gets credited with almost half of the difference between virgin production and recycling: $0.5 \times 0.9 \times (E^*_V - E_R)$ to be precise. The rest is assigned to the product that is produced from recycled material.

The incentives

In a recently completed project within the Swedish Life Cycle Center, Ekvall et al. (2020) studied these end-of-life, cut-off, quality-adjusted 50/50, CFF, and eight other approaches for modelling material recycling in LCA. Ekvall et al. (2020) found that all methods for modelling recycling risk giving incorrect incentives. The end-of-life approach gives, as stated above, the entire environmental benefit of the recycling to the product that is recycled after use. This means that producers do not get an incentive to use recycled material in their production.

An EPD, PEF, and most other methods attribute at least some of the environmental benefits to the product that uses recycled materials. This means that the incentive to send used products for recycling will be lower.

Waste incineration with energy recovery yields an environmental benefit when emissions from the incineration are less than the emissions from substituted energy supply and avoided disposal ($E^*_E+E^*_D-E_{ER}>0$ in Figure 3). A climate benefit can arise, for example, at energy recovery from bio-based plastic waste. When system expansion with substitution is applied in the LCA, the full net benefit of energy recovery is typically assigned to the product generating the incinerated waste. This holds for the RUaEP and EoL formulas and, by default, also for CFF.

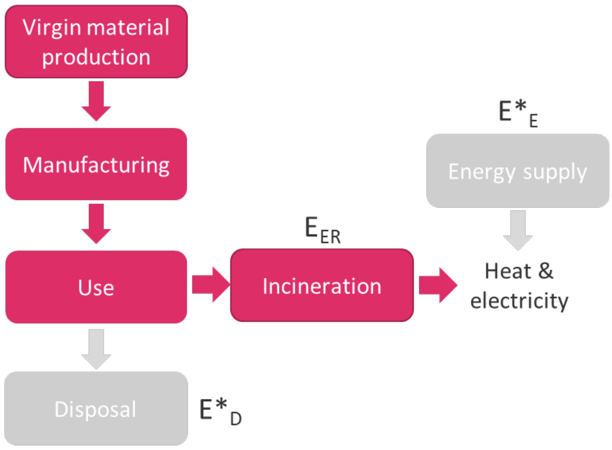

Figure 3 – Illustration of waste incineration with energy recovery from Product 1. E_{ER} is the environmental burdens of the incineration. E^*_E is the environmental burdens of the energy supply substituted by electricity and heat from the incineration.

The environmental benefits are often greater for recycling, compared to incineration. However, if only part of the recycling benefit is assigned to products recycled after use, the benefit of incineration can still be greater than this part. Mengarelli et al. (2017) argue, for example, that the EoL formula is biased towards incineration because it accounts for the full benefit of energy recovery but only half the benefits of recycling.

Figure 4 illustrates a hypothetical case where the PEF methodology is applied to compare different options for managing a polymer product after use. In this case, recycling and incineration with energy recovery both generate a net environmental benefit. The environmental benefit of recycling is greater than the net benefit of energy recovery ($E^*_V-E_R > E^*_E-E_{ER}$). To obtain the greater benefit of recycling, PEF results should give an incentive to use recycled plastics as well as an incentive to send plastic waste to recycling. However, since default PEF calculations credit less than half the benefit of recycling to a polymer product sent to recycling, it gets a larger credit if sent to incineration. This means the PEF results give incorrect incentives in this case: they indicate that the product should be sent to incineration, even though recycling is the better option for the environment.

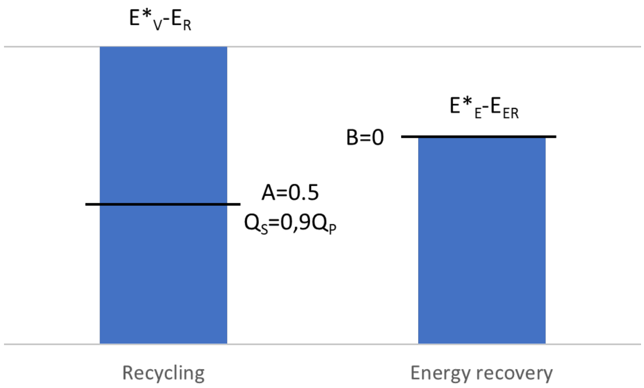

Figure 4 – The net environmental benefits of recycling and energy recovery in a hypothetical case of managing post-consumer polymer waste. Default PEF calculations (where Factor A=0.5, secondary material has 90% of primary material quality, and Factor B=0) in this case indicate that the waste should be incinerated even though recycling is better for the environment.

Alternative modelling of incineration

The CFF includes a Factor B, which can be used for allocating away part of the burdens and benefits of energy recovery from the product investigated (EC 2018a). Such a factor may be relevant because the environmental benefits of energy recovery do not arise simply because combustible waste is made available for incineration. There must also be capacity in the incinerators and the demand for the energy extracted. However, the default value for factor B is 0 (see
Figure 4), which means that the environmental benefit of energy recovery is in practice rarely written down at all. If factor B was given a higher value, the LCA results would provide less incentives for energy recovery, and thus more often provide incentives for material recycling.

The results in Figure 4 are based on the assumption that incineration of plastic waste means more energy is recovered at the waste incinerator. Some LCAs instead assume that a product that is sent for waste incineration in Sweden does not increase energy recovery. This is based on the observation that waste incineration plants tend to be utilized to the maximum - by importing waste when there is spare capacity. If we send more waste to incineration, imports decrease, which affects waste management in other countries. The effect in the countries of origin has been studied previously (Fråne et al. 2016). The ultimate impact on the waste management and energy systems in other countries is highly uncertain, but scenarios have been developed to deal with this uncertainty (Hagberg et al. 2017).

Purpose

This pilot project serves to increase the knowledge on the extent to which results from a life cycle assessment provide incorrect incentives for energy recovery from plastic waste. It also aims to increase knowledge about the extent to which this problem can be addressed through 1) use of Factor B in the PEF methodology, and 2) a broader systems perspective that includes the effects of energy recovery on waste imports and thus waste management in other countries. We do this by combining case study calculations for plastic waste with the knowledge and insights from the above-mentioned projects.

The focus is on the PEF methodology and CFF because it includes factor B, because the PEF methodology is relatively new and untested, and because it can be of great importance to policy makers, producers and consumers both in Sweden and in the rest of the EU.

We have the following objectives:

- Combine qualitative insights from previous projects on modelling of material recycling and energy recovery with environmental data for plastics.
- Discuss the basis for factor B.

- Model waste treatment of low-density polyethylene (LDPE) produced with fossil and renewable raw materials, respectively.
- Disseminate results and conclusions to key actors, which primarily include staff at the Swedish Environmental Protection Agency and the group of authorities within Swedish Life Cycle Center, but also to researchers and other actors in the LCA world, for example within the European Commission and the EU Joint Research Center.

Method

We discuss Factor B partly on the basis of available knowledge about PEF and LCA in general, and partly on the basis of available knowledge about the availability of combustible material, capacity for waste incineration and the demand for heat and electricity.

Our calculations focus on the climate impact of the recycling and incineration of LDPE waste generated in Sweden. Since this is a pilot study, calculations for the treatment of LDPE waste are carried out using environmental data from easily available databases.

We estimate the net climate benefit of mechanical recycling (E^*_{Vm}-E_{Rm}), chemical recycling (E^*_{Vc}-E_{Rc}) and incineration (E^*_{E}-E_{ER}) through simple substitution: recycled material is assumed to replace virgin material and energy recovered from LDPE waste is assumed to replace average Swedish district heat and electricity. The latter introduces a significant error in the estimate: heat and energy from waste incineration do not replace average energy but marginal heat and electricity. We account for this error in the discussion of the results.

We apply the CFF with default values (A=0.5; Q_S/Q_P=0.9; B=0) to find whether a PEF would give the same indications as a simple substitution. Again, we use Swedish average data to model the substituted heat and electricity. A PEF should ideally include the residual average data for electricity. In Sweden, this residual is based on the Nordic electricity production mix (EI 2021) and has a much higher climate impact than Swedish average electricity. However, since only a small share of the recovered energy is electricity, the impact on the PEF results is limited.

We then modify CFF by applying a revised Factor B in the CFF to investigate whether this brings PEF results more in line with the estimated net climate benefit of recycling and incineration with energy recovery.

Finally, we modify the CFF with two scenarios (based on Hagberg et al. 2017) on how a change in the incineration of LDPE waste would affect waste important and, hence, the waste management in other European countries.

The results are used as basis for a discussion aiming to draw conclusions about the extent to which PEF, and LCA in general, risk giving incorrect incentives for energy recovery, and about the extent to which there is a good basis for methods to alleviate this problem.

2. Factor B

Interpreting Factor B

The PEF methodology models recycling as well as energy recovery with substitution (EC 2018a). For energy recovery, the CFF is calculated according to Equation 1.

$$(1-B)R_3 \times (E_{ER} - LHV \times X_{ER,heat} \times E_{SE,heat} - LHV \times X_{ER,elec} \times E_{SE,elec}) \qquad (1)$$

where R_3 is the share of material going to energy recovery, E_{ER} is the environmental impact of the energy-recovery process, and the avoided burdens are defined by the lower heating value of the material (*LHV*), the heat and electricity efficiency of the recovery ($X_{ER,heat}$ and $X_{ER,elec}$), and the impacts of the alternative production of heat and electricity ($E_{SE,heat}$ and $E_{SE,elec}$).

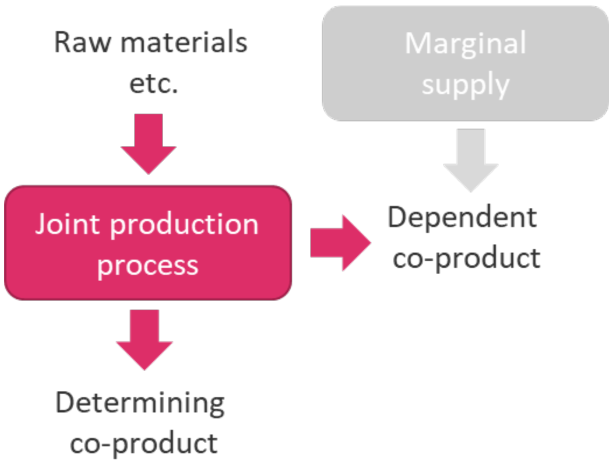

Figure 5 – Distinction between determining and dependent co-products, simplified from Weidema (2001). Demand for the determining co-product affects both the joint production and the marginal supply of dependent co-products. Demand for a dependent co-product.

When applying substitution at a joint production process, Weidema (2001) distinguishes between the co-product that determine the production volume of the process, and dependent co-products that are produced in volumes decided by the demand for the determining co-product. A consequential LCA of a determining co-product will include the joint production process and a credit for the avoided marginal supply of competing products substituted by dependent co-products (see Figure 5). A CLCA of a dependent co-product will not include the joint production since it is not affected by the demand for the product. Instead, it will include the affected, marginal supply of the competing product. This means the use of the dependent co-product in an CLCA is assigned the burdens of the marginal supply.

The PEF methodology is not consequential in the sense that it includes marginal data. However, PEF models of recycling and energy recovery both account for substitution of material and energy displaced by the material and energy recovered from waste. Using the default value B=0 in the energy substitution is equivalent to assuming that the waste treatment service is the determining function of energy recovery, i.e., that the volume of waste incinerated is determined by the quantity of combustible waste (cf. Figure 6).

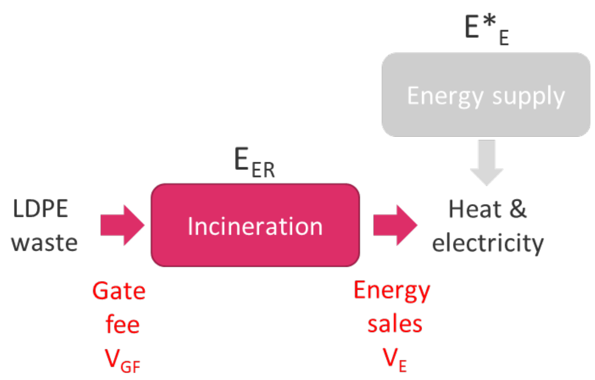

Figure 6 – Waste incineration has two functions: waste treatment and energy recovery. Is waste treatment the determining function? VGF and VE are the expected economic value of gate fees and energy, respectively.

The assumption above is often wrong. In many countries, combustible waste is deposited at landfills and the volume incinerated is much less than the volume of combustible waste.

In Sweden, on the other hand, waste incinerators are constructed even though the existing capacity is more than enough to treat the domestic combustible waste that is not recycled (Waste Sweden 2012a; Profu 2013). Of the nearly 6 million tonnes of waste combusted in Swedish waste incinerators in 2018, 1.5 million tonnes were waste from other European countries (Waste Sweden, 2019). This is called waste import, even though the facilities charge a gate fee when receiving the combustible waste. It might be less misleading to call it export of waste-treatment services.

The expansion of waste incineration in Sweden is driven by good economic conditions for incinerators in the country. The energy in the waste is used more efficiently in Sweden, compared to many other countries, because the heat can be used in district-heating networks. The Swedish taxes on fossil fuels also make waste a more competitive fuel.

Waste incineration has high fixed costs: investment costs are higher than for other fuels, since waste incinerators require advanced technology for combustion as well as flue gas treatment. On the other hand, the variable cost of energy recovery from waste is very low, if not negative, because of the gate fee. Therefore, waste incineration plants are base-load plants that are used as much as possible. The owners of incineration plants import waste (or export waste-treatment services) to utilize the incineration capacity as much as possible. Some incineration plants are fully used even when all heat recovered cannot be utilized. Hence, the volume of waste incinerated, and the corresponding energy recovery is mainly determined by the waste-incineration capacity.

Approach for calculating B

The concept of determining functions provides a basis for calculating a value of B that differs from zero. From the above we observe that, in Sweden, at least:

- the quantity of energy recovered from waste is determined mainly by the waste incinerator capacity, and
- increases in waste incinerator capacity are determined by the expected profitability.

Heat, electricity, and gate fees paid to deliver the waste all contribute to this profitability (see Figure 6).

Hence, waste incineration can be described as a process with multiple determining functions: waste treatment and energy recovery contribute to driving the process in proportion to their economic value. We propose that expected revenues from gate fees and energy are an appropriate basis for calculating Factor B:

$$B = V_E / (V_E + V_{GF}) \qquad (2)$$

where V_E and V_{GF} in Equation 2 are the expected economic value of energy and gate fees, respectively.

Factor B in Sweden

The national trade organization Waste Sweden (2014) recommends economic allocation of emissions from waste incineration with 58.7% allocated to the energy and the remaining 41.3% to the waste treatment. Using the same values for calculating B we get B=0.587. Using three digits will indicate a precision that does not exist, however. The economic revenues will vary with time and between locations. In Gothenburg, for example, the energy generates only 30% of the revenues in a waste-management system dominated by incineration (Renova 2020). Factor B should ideally be calculated based on updated data on expected revenues in the relevant region. However, when such data are lacking, we propose that a default value B=0.6 can be used for Sweden, based on rounded figures from Waste Sweden.

3. A wider systems perspective

Impacts in Sweden

As clear from the previous chapter, waste incinerators typically operate at full capacity. This capacity is often given by limitations in the energy flow is the incinerator. When we send polymer waste to incineration, this means we do not necessarily get more energy from waste. Instead, the polymer waste is likely to replace other waste flows in the incinerators.

Landfilling of combustible waste is not allowed in Sweden. Hence, an increase in incineration of waste polymers will not lead to an increase in landfilling of other waste flows in Sweden. Instead, it is primarily the import of combustible waste that is affected. Several previous reports indicate that imported waste is the marginal fuel in Swedish waste incineration plants (Waste Sweden 2012b; Gode et al. 2013; cf. Figure 7).

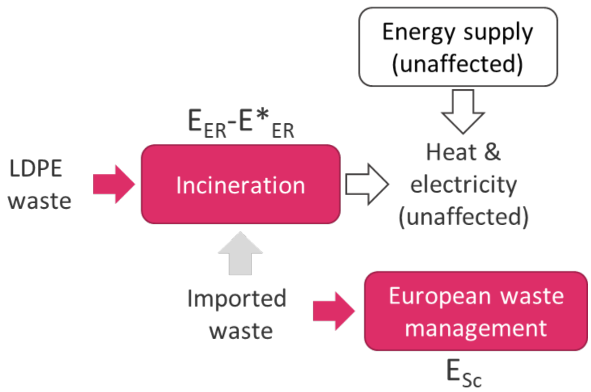

Figure 7 – A wider systems perspective: LDPE incineration affects waste imports, which in turn affects waste management elsewhere in Europe. E_{Sc} represents the net environmental burdens of the affected European waste treatment.

Impacts in other countries

Identifying how a change in waste imports to Sweden affects the waste management in other countries is a challenge. Sweden imports large amounts of combustible waste from Norway and the United Kingdom. Smaller quantities are imported from Ireland, the Netherlands, Finland, and other countries. Fråne et al. (2016) examined how this trade in waste affects waste systems in Sweden, Norway, the United Kingdom and Ireland. This study is the basis for our analysis.

Exports of waste from Norway increased sharply when they introduced a ban on landfilling of biological waste in 2009. The possibility of exporting this waste to Sweden has in at least some cases affected the expansion of infrastructure for biological treatment of food waste. It may also have affected the expansion of incineration plants in Norway. Norway can in addition export waste for incineration to other countries.

The waste exported from the UK is largely residues from sorting facilities (so-called Material Recovery Facilities; MRF) or from Mechanical Biological Treatment (MBT). Most of this is sent to the Netherlands for energy recovery, and only a small part is exported to Sweden. It is possible to deposit the waste in UK landfills, but this is expensive due to high landfill taxes. The option of sending residues for energy recovery in Sweden helps keeping down the costs of MRF and MBT facilities. However, the Swedish gate fees are set just low enough to compete with other options for treating the residues. Hence, the export to Sweden probably has a minor impact on the economy of the MRF and MBT facilities and hardly affects the investments or operation of such facilities (Fråne et al. 2016). A change in the export to Sweden therefore primarily affect how much is deposited in the United Kingdom and how much is exported to other countries for incineration there.

In Europe as a whole, the capacity for incineration is significantly smaller than the supply of combustible waste. Large amounts of combustible waste are still landfilled, even if an increasing share of the deposited waste first goes through MBT or other pre-treatment processes. If a reduction in Swedish waste imports means that the countries of origin instead export the waste for incineration in other European countries, it will probably, in the end, lead to more waste having to be landfilled somewhere in Europe.

In general, Fråne et al. (2016) argue that Swedish waste imports contribute to keeping costs down for waste treatment outside Sweden's borders. In theory, this makes it more difficult for material recycling and other waste treatment to compete. However, since the Swedish facilities negotiate gate fees that are just below the cost of other waste treatment, this effect will be very small.

The EU aims to significantly reduce landfill, and the capacity for incineration and other waste treatment is therefore being greatly expanded in Europe. This could in the long run lead to a European overcapacity for incineration. If so, there will be competition for the combustible waste, resulting in lower gate fees. In such a situation, a reduced import of waste to Sweden can contribute to keeping the gate fees up. This can lead to increased material recycling, for example through an increased degree of sorting in MRF facilities or through increased investments in infrastructure for source sorting. However, increased reception fees can also lead to increased investments in energy recovery and/or biological treatment, or to increased landfilling in the countries where it is permitted.

Our scenarios

As stated above, an increased flow of waste plastics to Swedish incinerators can reduce the flow of imported waste. This can have a range of impacts on the waste management in other countries and, thereby, affect the environment in many different ways. The actual consequences are likely to be a combination of, for example:

- increased disposal of untreated residual waste in the exporting countries or elsewhere in Europe,

- increased disposal of MBT and MRF residues in the countries of origin or elsewhere in Europe,
- increased incineration in other countries,
- increased biological treatment in other countries, and/or
- (in the long run) increased material recycling.

The uncertainty is great in what effects will dominate the mix of consequences. This uncertainty can be illustrated and managed with scenarios. The scenarios should be widely separate without being unreasonable. They should also be simple enough to communicate and understand. Therefore, we have chosen two scenarios based on Hagberg et al. (2017) with only one kind of effect in each:

1. European incineration: increased incineration of plastic waste in Sweden leads to reduced waste imports and to more waste being incinerated with electricity production in another European country.

2. European landfill: increased incineration of plastic waste in Sweden leads to reduced waste imports, which in this scenario leads to an increase in the disposal of untreated residual waste in another European country.

We can calculate the potential environmental impacts of these scenarios with WAMPS (WAste Management Planning System), an LCA model for waste management (https://wamps.ivl.se/prod/) developed by IVL. The results can be used to illustrate the uncertainty in the environmental burdens and benefits of Swedish waste incineration in a European-wide systems perspective (see Chapter 4).

4. Calculations and results

We carry through a simple case study to investigate and illustrate the risk of LCA providing incorrect incentives for energy recovery from plastic waste, and to test and illustrate to what extent a modified Factor B or a broader systems perspective solves this problem. For this purpose, we calculate the climate impacts of mechanical recycling, chemical recycling, and incineration of 1 tonne waste LDPE in Sweden. The polymer is produced either from 100% fossil or from 100% renewable raw materials.

The three waste-management options are compared with four different methodological approaches:

- simple substitution, a common application of the end-of-life approach where the full benefit of recycled material substituting virgin material is assigned to the LDPE sent to recycling, and where energy from incineration substitutes heat and electricity modelled with average data,
- the default PEF approach with B=0,
- PEF calculations with B=0.6, and
- adjusted PEF calculations with a European systems perspective on Swedish incineration.

Models for mechanical recycling, chemical recycling and incineration with energy recovery are created in GaBi Software with databases from thinkstep/Sphera and EcoInvent. As mentioned in Chapter 3, the scenarios with a European systems perspective are calculated with WAMPS.

Options for waste management

Various routes for management of waste polymers exist. In this study we compare three options:

Mechanical recycling

The mechanical recycling route used for this study include the following steps: the collected LDPE waste is sorted and baled, after which washing occurs, followed by melting, and granulation. The output product of the mechanical recycling route is 1 tonne recycled LDPE per 1 tonne LDPE waste. The recycled polymer is assumed to replace 1 tonne virgin LDPE.

The circular footprint (CF) for mechanical recycling was calculated according to Equation 3:

$$CF = (1 - A)R_2 \times \left(E_{recyclingEoL} - E_V^* \times \frac{Q_{Sout}}{Q_P}\right) \quad (3)$$

where A is the allocation factor of burdens and benefits between supplier and user of the recycled material, and R_2 is the proportion of the material in the product that will be recycled in the following system. $E_{recycledEoL}$ is the specific emissions from the recycling process at end-of-life (EoL), E^*_V is the specific emissions from the acquisition and pre-processing of virgin material assumed to be substituted by the recycled material, Q_{Sout} is the quality of the outgoing secondary material, i.e., the quality of the recycled material at the point of substitution and Q_P is the quality of the primary (i.e., virgin) material (Wolf, 2020).

The variable values used in Equation 3 are presented in Table 1 in Annex 1.

Chemical recycling through pyrolysis

As for mechanical recycling, the chemical recycling route through pyrolysis involves sorting and baling, followed by washing and melting. After these initial steps the pyrolysis occurs where the products coke, syngas and pyrolysis oil are formed. We assume that the coke and syngas are incinerated, and that the pyrolysis oil replaces naphtha (crude oil). The output product from the chemical recycling process is 720 kg pyrolysis oil per tonne recycled LDPE, based on mass balance calculations for pyrolysis.

The CF for chemical recycling though pyrolysis is calculated according to Equation 3. The variable values and results for Equation 3 are presented in Table 1 in Annex 1.

Incineration with energy recovery

The route for incineration with energy recovery includes incineration of the plastic waste where the generated energy is used for electricity and district heating according to the distribution of electricity and heat in Sweden.

The CF for incineration with energy recovery is calculated according to Equation 1. The variable values and results from Equation 1 are presented in Table 2 in Annex 1.

A modified CFF formula was used for calculating the climate impacts of incineration with the two European scenarios in Chapter 3. The modified formula is presented in Equation 4.

$$CF = (1 - B)R_3 \times (E_{ER} - E_{ER}^* + E_{Sc}) \qquad (4)$$

where E^*_{ER} is the specific emissions from the avoided energy recovery process of imported waste and E_{sc} is the scenario-dependent specific emissions from the alternative treatment of the imported waste in a European country other than Sweden.

The variable values and results from the CFF calculations using Equation 4 are presented in Table 3 in Annex 1.

Key assumptions and limitations

The recycling and incineration routes are all assumed to be located in Sweden. Hence, datasets relevant for Swedish conditions are chosen as far as possible.

Domestic transportation was excluded from the study since the distances are assumed to be relatively short and equal in the simulation of all waste-management routes. The collection of plastic waste is also excluded from the study since it mainly consists of domestic transportation. International transports of waste are accounted for in the European scenarios (see below).

Emissions of biogenic CO_2 are assumed to be climate-neutral and, hence, do not affect the results.

Mechanical recycling

For mechanical recycling, a production process of LDPE granules from virgin crude oil is substituted. The dataset for LDPE production is based on European conditions since no dataset for Sweden was available. An average Swedish electricity mix is used for the recycling process steps.

Chemical recycling through pyrolysis

In the model for chemical recycling the pyrolysis oil is assumed to substitute crude oil production for the Swedish market. CO_2-emissions from the pyrolysis process are based on calculations from an assumed yield of a thermal pyrolysis process. CO_2-emissions from renewable LDPE are assumed to be climate neutral, which means they are excluded from our climate calculations. When applying the CFF to chemical recycling, variables A and Q_s/Q_p are assumed to be 0.5 and 1, respectively. Factor A is approximated with a CFF default value for exhausted olive oil presented by the European Commission (EC 2018b). The assumption is made that no quality degradation occurs, thus Q_s/Q_p is set to 1.

Incineration with energy recovery

The lower heating value (LHV) of LDPE is estimated to be 42.83 MJ/kg (Phyllis 2020). Efficiencies for heat and electricity of 0.85 and 0.3 are based on Seyed et al. (2020).

Our calculation results on incineration are sensitive to assumptions about what energy sources are substituted by the heat and electricity generated in Swedish waste incineration. We model the substituted energy with input data representing the average district heat and electricity used in Sweden. This does not fully correspond to the method developed in this project. In a PEF, the substituted electricity should ideally be modelled with data representing average national residual electricity. In Sweden, the residual electricity mix is defined based on the Nordic electricity system (EI 2021), which makes the climate impact much higher for the residual mix, compared to the Swedish average consumption mix.

Incineration with European scenarios

For the scenario calculations in WAMPS, we assume the affected European waste flow to be a residual waste mix which includes 11% plastic waste. The amount residual waste corresponding to the energy content of 1 tonne LDPE was calculated based on heating values. The LHV for LDPE is 42.83 MJ/kg (Phyllis 2020) and the LHV of residual waste is by default set in WAMPS to 10,6 MJ/kg. This means each tonne of LDPE incinerated affects the treatment of 4.04 tonne residual waste.

The calculations in WAMPS do not include waste collection or other local transport. International transports are assumed to take place by truck (approx. 3 * 30 m^3 per crew) which travels 3000 km per single journey.

In the scenario European incineration, the electricity efficiency is assumed to be 30%. The results are sensitive to assumptions about which energy supply is substituted by the waste incineration. We assume that the electricity from European incinerators replaces electricity produced from natural gas in a modern combined cycle power plants with 58% efficiency. The same assumptions were made in the incineration scenario of Hagberg et al. (2017), except that they assumed the substituted electricity to be produced from natural gas in a less efficient (hence, older) plant.

In the scenario European landfilling, the results are highly sensitive to assumptions of how much of the methane generated in the landfill is captured. The results are also affected by assumptions regarding how the extracted gas is used and what it replaces. We assume the landfill to be a modern, well-designed landfill, where 70% of the methane formed during a hundred-year period is utilized as landfill gas. The extracted landfill gas is assumed to be used to produce electricity (25%) and heat (75%) with 80% efficiency. 10 % of the methane gas that is not extracted, i.e., 3% of the generated methane, is assumed to oxidize in the landfill coverage and does not affect the climate. These assumptions are all the same as in Hagberg et al. (2017).

Results

The climate impact results (in kg CO_2-equivalents/tonne LDPE) obtained from our calculations are presented in the charts below (Figures 8 through 15). The numerical results are presented in Annex 1, Table 4.

Results from simple substitution for fossil and renewable LDPE are presented in Figures 8 and 9, respectively. Simple substitution is equivalent to CFF where variables are set to A=0 and Q_s/Q_p=1 for both mechanical and chemical recycling.

Climate impacts calculated with the default PEF approach (with A=0.5, Q_s/Q_p=0.9, and B=0) are shown in Figures 10 and 11.

For the PEF case with factor B=0.6, climate impact for fossil LDPE and renewable LDPE can be seen in Figure 12 and Figure 13, respectively.

Climate impact for the two different PEF scenarios (European incineration and European landfill) for energy recovery with factor B=0 is presented below. Climate impact for fossil LDPE is shown in Figure 14 whereas results for renewable LDPE can be seen in Figure 15.

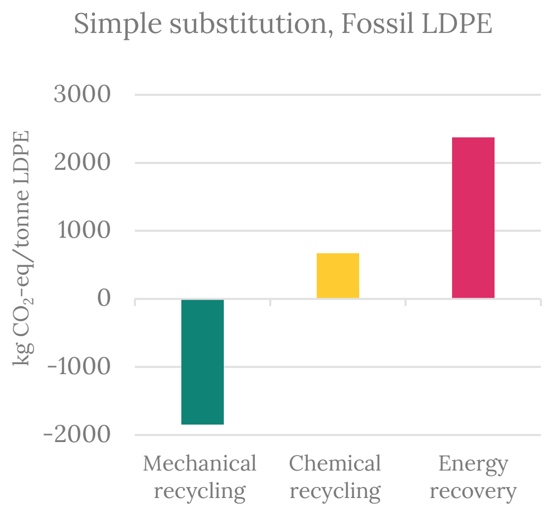

Figure 8 - Climate impact for waste treatment of fossil LDPE with simple substitution including mechanical recycling, chemical recycling and energy recovery where CFF variables are as follows: A=0, B=0 and Q_s/Q_p=1.

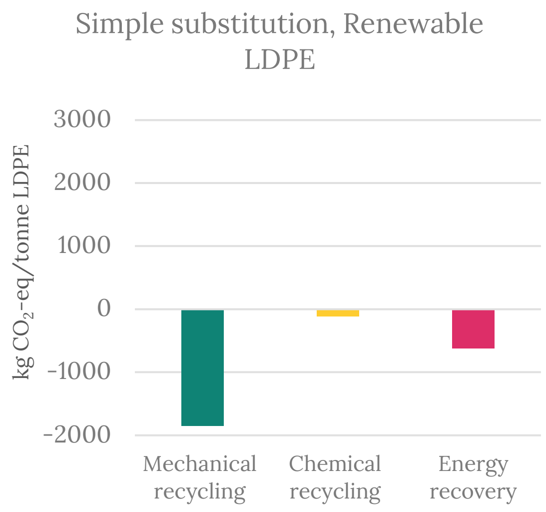

Figure 9 - Climate impact for waste treatment of renewable LDPE with simple substitution including mechanical recycling, chemical recycling and energy recovery where CFF variables are as follows: A=0, B=0 and Q_s/Q_p=1.

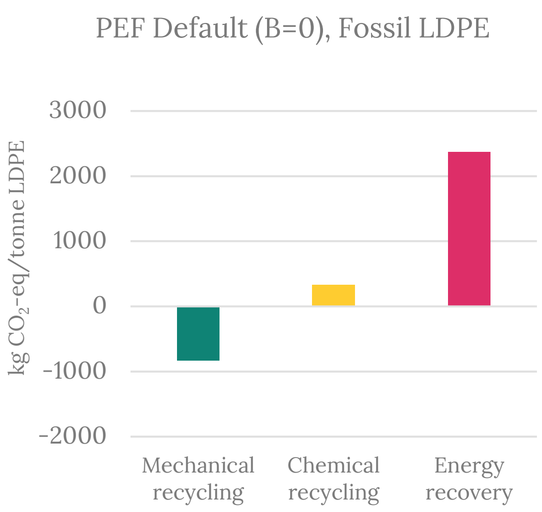

Figure 10 – Climate impact for waste treatment of fossil LDPE in the PEF default case including mechanical recycling, chemical recycling and energy recovery where factor B=0.

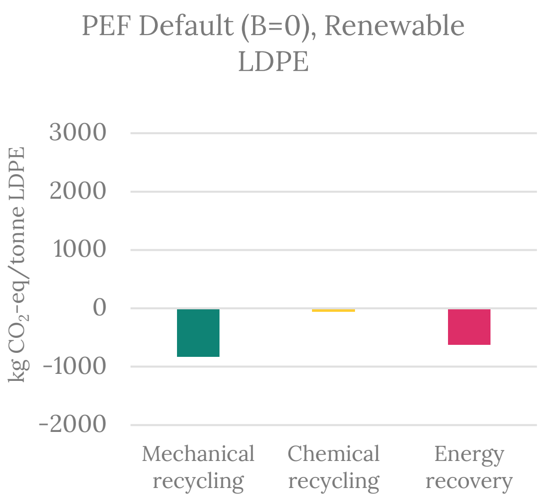

Figure 11 – Climate impact for waste treatment of renewable LDPE in the PEF default case including mechanical recycling, chemical recycling and energy recovery where factor B=0.

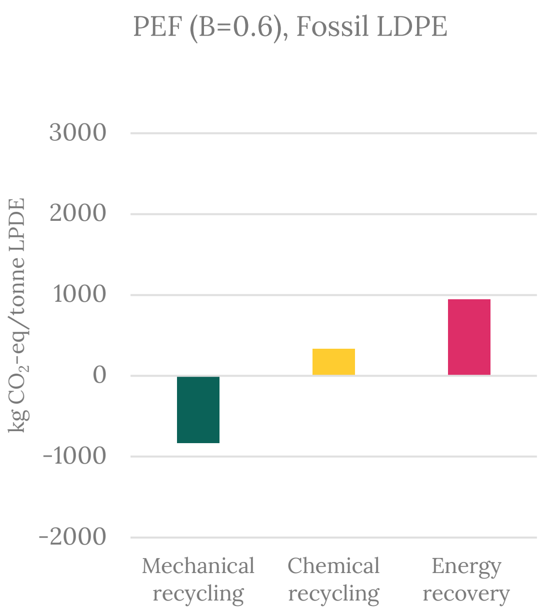

Figure 12 - Climate impact for waste treatment of fossil LDPE in the PEF case with B=0.6 including mechanical recycling, chemical recycling and energy recovery.

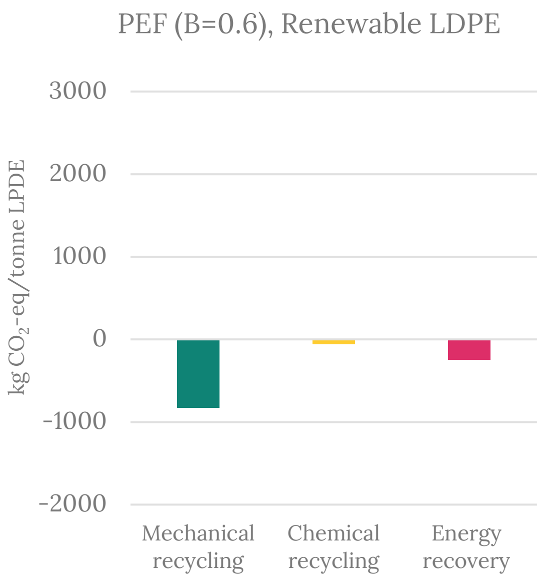

Figure 13 - Climate impact for waste treatment of renewable LDPE in the PEF case with B=0.6 including mechanical recycling, chemical recycling and energy recovery.

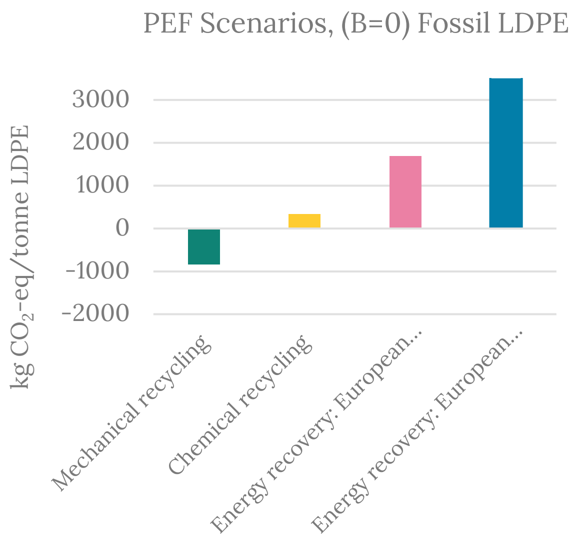

Figure 14- Climate impact for waste treatment of fossil LDPE scenario analysis including mechanical recycling, chemical recycling and scenarios for energy recovery with European incineration and European landfill.

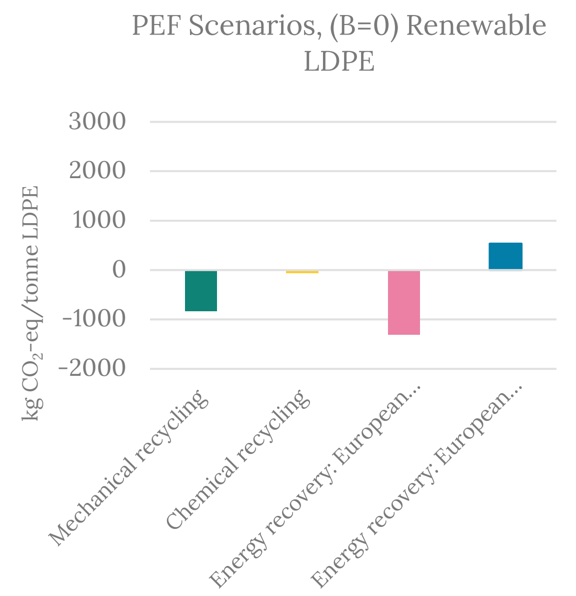

Figure 15 – Climate impact for waste treatment of renewable LDPE scenario analysis including mechanical recycling, chemical recycling and scenarios for energy recovery with European incineration and European landfill.

5. Discussion

Interpretation of results

Incineration of fossil LDPE

The results indicate that LCA does not give any incentive for Swedish energy recovery from waste plastics with a fossil origin. On the contrary, incineration of fossil plastics has a large net impact on the climate in our default calculations (see Figure 8). This is because the incineration of fossil plastics generates much more greenhouse gas emissions than the heat and electricity production avoided through the energy recovery.

The difference is large because the default calculations are based on average data on the production of district-heat and electricity used in Sweden. The average district-heat and electricity production both have a very low impact on the climate, which means there is little climate benefit from substituting this production. If we instead modelled the substituted electricity with average residual electricity, as stipulated for PEF, the difference between energy recovery and recycling in Figure 8 would be a bit smaller. The effect on the results might be significant, since Swedish residual electricity is defined by data representing Nordic residual electricity.

If we used marginal data for both electricity and district heat, the net climate impact of waste incineration could have been much smaller. There would probably still be a net impact, however, because the combustion of plastics generates more CO_2 than most fossil fuels.

The results indicate that waste incineration in Sweden is worse for the climate than waste incineration elsewhere in Europe. This is evident from the fact that incineration has lower net impact on the climate in the scenario with European incineration (Figure 14), compared to the default calculations (Figure 8). This is because the net impacts of incineration elsewhere in Europe (E_{Sc} in Equation 4, modelled with assumptions from the scenario European incineration) is less than the net impact of incineration of the same waste in Sweden (E^*_{ER}). Incineration elsewhere is better in our calculations, even though the Swedish district-heating system and associated heat demand makes the total energy efficiency in Swedish incineration plants much greater than in countries where waste incineration generates electricity only.

Again, this difference is sensitive to the energy supply substituted by energy from the waste. In European incineration, we assume the electricity produced to replace electricity from natural gas produced with a high efficiency (58%) and correspondingly low emissions. However, the energy supply substituted by incineration in Sweden has still much less emissions. As indicated above, this is because average data were used in the modelling of Swedish district-heat and electricity production. With this approach, LCA results are likely to indicate that all waste should be incinerated in countries where the average electricity supply is largely based on coal, for example Poland.

The use of average data neglects the fact that the energy replaced is not the average energy supply but the marginal supply. The use of national data neglects the significance of electricity trade across borders. If waste is incinerated in Sweden, the electricity generated can be exported to, e.g., Poland and substitute electricity production there.

A consequential LCA (CLCA) with marginal data and a sufficiently broad systems perspective might be able to account for these systems effects and give correct indications on where residual waste should be incinerated. However, the large uncertainty in marginal energy data might hamper CLCA in this application: the indications of the CLCA results might be unclear. Note that this uncertainty is not inherent in CLCA; instead, it reflects the fact that actual consequences are difficult to foresee in a system as complex as the international electricity system.

A PEF where the substituted electricity is modelled with data representing Nordic residual electricity mix, might also indicate that it is better for the climate to incinerate waste in Sweden compared to Poland. This will depend on the difference in climate impact between electricity in the Nordic countries and Poland, and on the climate impact of the heat substituted in Sweden.

Incineration of renewable LDPE

Most results indicate a relatively small net benefit from incineration with energy recovery from waste plastics with renewable origin. Figure 15 illustrates that, in a European perspective, the consequence can be anything from a large climate benefit to a negative climate impact, when the renewable polymer replaces imported waste in Swedish incinerators.

The climate benefit is small in the default calculations (Figures 9 and 11), even though we assume CO_2 emissions from incineration of the renewable material to be climate neutral. This is because the energy from incineration substitutes average Swedish district heat and electricity, which have a very low climate impact. The benefit of energy recovery from renewable plastics would probably be much greater if the calculations were instead based on Nordic residual electricity or accounted for marginal impacts in the district-heat and electricity systems.

Chemical recycling of fossil LDPE

The results indicate that chemical recycling through pyrolysis of fossil LDPE has a negative impact on the climate (Figure 8). The emissions from the pyrolysis etc. are greater than the emissions avoided when the pyrolysis oil substitutes naphtha produced from crude oil.

The net climate impact of chemical recycling is in our calculations much lower than the net impact of incineration with energy recovery. This indicates that chemical recycling of fossil LDPE is a good option for the climate, if the alternative is incineration. However, as stated above, the large net impact of energy recovery from fossil LDPE in our calculations occurs at least partly because we do not account for the Nordic residual electricity or for marginal impacts in the district-heat and electricity systems.

Chemical recycling, renewable LDPE

Our results indicate that chemical recycling of renewable LDPE brings a small climate benefit. We assume emissions of bio-based CO_2 from the pyrolysis to be climate-neutral. On the other hand, the climate benefit of substituting naphtha is small.

The benefit of chemical recycling is in the default calculations (Figures 9 and 11) less than the climate benefit of energy recovery. This difference would be even greater if the calculations accounted for the Nordic residual electricity or for marginal impacts in the district-heat and electricity systems.

On the other hand, the calculations also do not account for climate benefits at the end-of-life of subsequent life cycles. When oil from pyrolysis of renewable plastics is used to produce new plastics, the new product will be biobased rather than produced from crude oil. Incineration or pyrolysis at the end-of-life of this product will emit biogenic rather than fossil CO_2. This impact is not accounted for in our calculations. Hence, our calculations do not provide a basis for deciding whether chemical recycling or incineration of renewable LDPE is better for the climate.

Mechanical recycling

The results indicate a large net benefit of mechanical LDPE recycling, independent of whether the polymer is produced from renewable or fossil raw materials. Mechanical recycling appears to be a significantly better option than chemical recycling. This is in part because of lower emissions from the recycling, and in part because the mechanically recycled polymer substitutes a virgin polymer rather than naphtha.

Mechanical recycling is also a better option than energy recovery in all default calculations (Figures 8-11). In the default PEF calculations, the difference between mechanical recycling and energy recovery is small, however (Figure 11). Default PEF calculations can indicate a greater net climate benefit for energy recovery from renewable LDPE, compared to mechanical recycling, if the substituted energy mix includes a larger share of fossil fuel.

The risk of incorrect incentive

The correct incentive

Simple substitution assigns the full environmental benefit of replacing virgin material to the product sent to recycling (E^*_V-E_R in Figure 4). Figures 8-9 indicate that the climate benefit of mechanical recycling is much greater than the benefit of chemical recycling.

Simple substitution also assigns the full environmental benefit of energy recovery to the incinerated product (E^*_E-E_{ER} in Figure 4). There is no net benefit from incineration of fossil LDPE (Figure 8). Energy recovery from renewable LDPE results in a climate benefit, the size of which depends fully on what energy is substituted. If the substituted energy is average Swedish district heat and electricity, our calculations indicate that the benefit of energy recovery is just a third of the benefit of mechanical recycling (Figure 9). If the climate impact of the substituted energy is three times higher, the benefit of energy recovery and mechanical recycling would be approximately the same.

From this we conclude that recycling to replace virgin LDPE is better for the climate than energy recovery, as long as the energy substituted has less than three times the climate impact of average Swedish district heat and electricity. In these cases, LCA results should indicate that products should be sent to mechanical recycling and new products should be produced from recycled materials.

If the energy actually substituted by waste incineration has a climate impact that is more than three times the average Swedish energy, fossil LDPE should still be recycled; however, renewable LDPE should in this case be incinerated with energy recovery to increase the renewable share of the energy supply.

All this, of course, assuming that other input data etc. in our calculations are correct.

PEF Incentives

The results in Figures 10 and 11 are calculated with the default PEF approach. This means that less than half the environmental benefit of mechanical recycling is assigned to the recycled waste LPDE. The same goes for the environmental burdens of recycling. Compared to the simple substitution, the net benefits and net burdens are both reduced by 55%, based on the default values for polymers in the CFF:

$$A \times Q_S/Q_P = 0.5 \times 0.9 = 0.45 \qquad (5)$$

The results still indicate that mechanical recycling is a better option for the climate, compared to energy recovery. In other words, they give a correct incentive to send waste LDPE to mechanical recycling.

However, for renewable LDPE, the difference between mechanical recycling and substitution is small. It is also sensitive to the climate impact of the energy substituted by incineration. If the substituted energy has 40% higher climate impact than the average Swedish district heat and electricity, the PEF results would indicate that renewable LDPE should be incinerated. This could be bad for the climate: as stated above, recycling is the better option as long as the energy substituted has less than three times the climate impact of average Swedish energy.

This means there is a risk of incorrect incentive for incineration only for renewable LDPE. Our PEF calculations will give an incorrect incentive when the energy substituted by incineration has 40-200% more climate impact than the average Swedish energy. Again, this assumes other input data etc. are correct in our calculations.

Generalizations

Recycling is often, but not always, a better option than incineration for the climate and/or other environmental aspects. In some cases when waste should be sent to recycling, PEF results will still indicate that the waste should be incinerated with energy recovery. This is because the CFF by default assigns the full incineration benefit to the product going to incineration, but only part of the recycling benefit to the product generating the recyclable waste.

The same holds for various other methods to model recycling. These includes some attributional approaches, for example price-based allocation (ISO 2018) and 50/50 allocation (Lindfors et al. 1995). They also include consequential methods such as 50/50 substitution (Ekvall 2000), the price elasticity approach (also Ekvall 2000), and price-based substitution (Schrijvers et al. 2016).

Note that even ambitious methods that aim to model the foreseeable consequences of sending a material to recycling can fail to give correct information on when a material should be sent to recycling. Ekvall (2000) and Schrijvers et al. (2018) observe that supply and demand for recyclable material are both needed for recycling to occur. This means sending a material to recycling is not enough: it can affect not just the use of recycled material but also the collection for recycling somewhere else in the world. When part of the benefit of sending a material to recycling is off-set by reduced collection for recycling elsewhere, an accurate and comprehensive modelling of the consequences should account also for this consequence and assign only part of the recycling benefits to the product that supplies recyclable material. As concluded above, this can make the LCA results indicate that waste should be incinerated, even when the climate or environment benefits more from recycling.

Accurate modelling the consequences of sending a material to recycling is clearly not enough to always give correct indication regarding how to treat the waste. This might be because recycling requires a concerted action between actors in different product life cycles, and LCA focusses on one product at a time: ALCA focusses on an individual product life cycle, and CLCA focusses on decisions made in an individual product life cycle. This might not be enough to guide a development that requires concerted actions between actors in different life cycles.

When actions are needed in different life cycles, the focus on individual life cycles might be an important limitation with LCA. This is the case with recycling, but probably also in many other situations that require actors in different life cycles to be coordinated.

The limited relevance of separately assessing individual actions goes far beyond LCA. To clarify with a drastic example: consider assessing the foreseeable consequences of clapping your right hand. The hand is most likely to hit air only. It might also hit an object or person in front of you. The chances that you are going to hurt yourself or someone else is much greater that the likelihood that your right hand is going to produce a clapping sound. In conclusion, you should refrain from clapping your right hand. The same goes for the left hand, of course. Hence, there will be no applaud.

Factor B as a solution

Factor B is used when modelling energy recovery in CFF. A change in Factor B affects the results for energy recovery only. When B is changed from 0 to 0.6 (Figures 12 and 13), the environmental burdens and benefits of incineration are reduced by 60%, compared to the default PEF results (Figures 10 and 11) and also compared to the simple substitution (Figures 8and 9).

While simple substitution accounts for the full benefits and burdens of recycling and incineration, PEF results with B=0.6 reduce the net results of recycling by 55% and the net results of incinerations by 60%. In other words, the relationship between recycling and incineration are almost the same as if calculated with simple substitution. If the input data etc. are correct, and the simple substitution correctly indicates whether the LDPE waste should be incinerated or recycled, PEF with B=0.6 will point in the same direction as simple substitution in almost all cases. The risk that PEF with B=0.6 will give incorrect incentives for incineration or recycling is very small.

Our method for estimating B is based on the observations that, at least in Sweden:

- the quantity of energy recovered from waste is determined mainly by the waste incinerator capacity,
- increases in waste incinerator capacity are determined by the expected profitability, and
- gate fees and energy recovery both contribute to the expected profitability.

These arguments are relevant in an LCA aiming to assess the long-term consequences of sending waste to incineration. They are only relevant in the long-term perspective, because waste incinerator capacity is in the short term fixed and hence cannot be affected by a change in the flow of plastic waste.

Our approach for estimating Factor B accounts for the fact that incineration with energy recovery does not increase simply because a waste flow is sent to the incinerator. This is in parallel to Factor A, which accounts for the fact that recycling does not increase simply because a waste flow is sent to recycling. In this sense, our approach makes the CFF more consistent and balanced.

If CFF becomes more balanced, though, it still has the limitation of accounting for consequences of actions in a single product life cycle at a time. It still does not assess in combinations the concerted actions in the life cycle generating waste and the life cycle utilizing the recycled material or recovered energy.

Note that the accuracy in the incentives given by PEF with B=0.6 in our calculations is a bit of a coincidence. Factor A in CFF varies between materials. PEF results for polymers with B=0.6 will provide accurate indication regarding the waste management of most polymers (A=0.5); however, it might give incorrect incentives for incineration of textiles (A=0.8), and incorrect incentives for recycling of many paper grades (A=0.2). In addition, Factor B as estimated with our method will vary over time and between locations, depending on to what extent the recovered energy contributes to the expected profitability of the incinerator.

The European perspective

Scenarios with a European perspective are particularly relevant in the short-term perspective, because

- waste incinerator capacity is typically fully utilized,
- incinerator capacity is in the short term unaffected by a change in the incinerated flow of plastic waste, but instead
- the flow of plastic waste primarily affects the import of waste from other European countries.

The scenarios European incineration and European landfill illustrates the uncertainty in the climate impact of a change in waste imports (Figures 14 and 15). The conclusion that fossil LDPE should be recycled and not incinerated seems robust even with respect to this uncertainty (Figure 14). However, for renewable LDPE the large uncertainty means that the difference between recycling and incineration is no longer significant.

Just like Factor B above, the expanded systems perspective makes the CFF more consistent and balanced: it accounts for the fact that not only recycling but also energy recovery depends on more than the flow of waste from the life cycle investigated. However, the large uncertainty in the European impacts means the PEF results gives little guidance on whether renewable LDPE should be incinerated to recycled. In this sense, it removes the incentives for recycling as well as energy recovery.

In the choice between the two solutions discussed so far, it is worth noting that Factor B is relevant when assessing long-term effects of plastic waste incineration while the waste imports are most clearly affected in the short term. An LCA, just like other environmental assessments, aims at increased long-term sustainability. In this perspective long-term effects, such as investment in new incinerator capacity, are likely to be more important than short-term fluctuations in waste flows and their treatment. This indicates that it is more relevant to estimate Factor B than to estimate the impacts of a change in waste imports.

However, since B>0, a change in the flow of plastic waste to incinerators will not fully be met by an increase in incinerator capacity. Part of the long-term effects will also include crowding out of imported waste. Comprehensive modelling of the foreseeable long-term consequences of an increased plastic waste flow will require a combined approach: an estimate of Factor B to decide to what extent the increase is met by incinerator investments, and scenarios to model the European impacts of the change in the waste imports.

100/100 modelling of recycling

We have stated above that simple substitution gives the correct incentive by assigning the full net benefit of recycling and energy recovery to the product delivering waste to recycling and incineration, respectively. Instead of a deeper analysis of waste incineration, might not simple substitution be a much easier solution?

Simple substitution is part of the end-of-life approach to modelling recycling (see Background): the recycled material is assumed to replace virgin material and credited for the full benefit of this substitution. This approach will give an incentive to recycle when the full benefit of recycling is greater than the full benefit of energy recovery. However, the approach will not give any incentive to use recycled material, because scrap material will be assigned the same burdens as virgin material. This is an important limitation in an LCA of plastics or textiles, where the demand for recyclable material needs to be stimulated.

A possible solution can perhaps be to assign the full benefit of recycling to the product that generates recyclable waste – and also to the product where the recycled material is used. This approach to modelling recycling will always give a correct incentive to recycle and also give an incentive to use recycled material. The EoL formula suggested for PEF at an early stage (JRC 2012) is a version of this approach.

An objection to such a method is that the benefits of recycling are accounted for twice, and that the LCA results for the two products are not additive. Additive results are typically expected from ALCAs, because ALCA aims to identify the share of environmental burdens that belongs to the product. The Guide to PAS 2050 (BSI 201x2011), for example, states that the environmental benefits of recycling must be allocated to the input or output of recyclable material, but not to both.

A CLCA does not require additivity. In a CLCA, a relevant objection to the 100/100 approach is instead that it does not reflect the foreseeable consequences of generating and using recyclable waste material in any consistent scenario: assigning all benefits of recycling to the product generating the recyclable waste is accurate only if the supply of recyclable material is the bottleneck that decides how much material is recycled; assigning all benefits to the use of recycled material is accurate only if the demand for recycled material is the bottleneck that decides how much material is recycled.

Regardless of whether the LCA is an ALCA or CLCA, the 100/100 approach is inconsistent with the tradition or rule that the credit given at recycling should correspond to the burden carried by the use of the same recycled material. This credit and burden can be called the environmental value of the recyclable material (Tillman et al. 1994, p.25; Ekvall 2000, p.96 & p.103).

Perhaps most importantly, a product that is produced from recycled material and also recycled after use will be fully credited for the recycling at both ends of the life cycle. The double counting of the recycling benefit might make the total LCA results negative for such products. In other words, the results might wrongly indicate that the product reduces the environmental impact of the world. These results would give an incentive to produce the product even when it is not needed or wanted.

The last problem can be alleviated if the recycling is modelled as a closed loop to the extent that the inflow and outflow of recycled materials match each other. A net inflow or a net outflow or recycled material would still be assigned the full benefit of recycling. However, this solution would completely remove the incentive to recycle a product that is produced from 100% recycled materials: such a product would be assigned the full benefit of recycling whether it is recycled after use or not.

Assessing the concerted action

The focus on individual life cycles is apparently a limitation with LCA. None of the solutions discussed here solves this problem completely. To guide coordinated or concerted action, it seems systems analysis should assess the necessary actions in combination. There is a solid basis for stating that the net environmental benefit of recycling is the joint result of at least two actions: 1. sending material to recycling, and 2. using recycled material instead of virgin material.

The net environmental benefit of energy recovery is also the joint result of at least two actions: sending waste to energy recovery and using the energy recovered from waste instead of energy from other sources.

Many situations require the environmental impacts to be estimated for a specific product or a specific action. In some cases, however, the LCA results can be calculated and presented with, for example, the following introduction:
“When the material is sent to recycling, you will, together with the recycler and the actor using the recycled material, jointly achieve this net environmental benefit: …”

When such statements are sufficient, the allocation problems of recycling and waste incineration can be avoided. There is no longer a need to subjectively choose a method to divide the environmental benefits of recycling or energy recovery between the life cycle generating the waste and the life cycle where the recovered material or energy is used. This solution is in line with the recommendation in the old SETAC “Code of Practice” to assess life cycles with recycling by studying the inputs and outputs from the total linked system (SETAC 1993, p.21).

6. Conclusions and further research

From the perspective of climate change, PEF and LCA in general give no incentive for energy recovery from waste plastics with a fossil origin; however, PEF might give an incorrect incentive to incinerate renewable plastics when biogenic CO_2 emissions are counted as climate-neutral. To make an accurate estimate of this risk, the calculations in our pilot study would have to be refined. In particular, the electricity data representing average Swedish consumption should be replaced by data representing the average residual mix to be in line with the PEF guidelines.

The risk of incorrect incentives for incineration with energy recovery can be reduced through an analysis of this option. Energy recovery from waste is not driven only (or even primarily) by the availability of combustible residual waste. Instead, it is mainly determined by the capacity of waste incinerators.

In the short term, the incineration capacity is fixed. Sending more waste plastics to Swedish incinerators is likely to reduce the import of waste, more than affect the quantity of energy recovered. This will have an impact on the waste management elsewhere in Europe. The impact on the European waste treatment is highly uncertain. The impact on the environment even more so. Further analysis is required to investigate to what extent this uncertainty can be reduced. Until then, at least, scenario analysis is an approach to demonstrate the uncertainty and assess the robustness in any conclusions from the LCA.

In the long term, incineration capacity can be affected by a flow of waste plastics. However, investments in waste incineration capacity also depends on revenues for the energy recovered. Factor B can be defined to reflect to what extent the expected revenues, profitability and, hence, investments depend on the electricity and heat generated in the waste incineration. This is relevant in an LCA with a long-term perspective. Factor B will vary with time and between locations, but it can be defined with a higher precision than the short-term effects on European waste management.

If waste incinerators are used at maximum capacity, Factor B can be interpreted as an estimate of to what extent a flow of combustible waste from the life cycle investigated crowds out imported waste from the incinerators. The consequences of such displacement of other waste can be modelled with scenarios. For consistency, these scenarios should have a long-term perspective.

Our scenarios for impacts in the European waste management and our approach for estimating Factor B both make CFF more consistent and balanced. It now accounts for the fact that not only recycling but also energy recovery depends on more than the flow of waste from the life cycle investigated.

Combining Factor B with long-term European scenarios is probably a more constructive approach than only using scenarios for the impact on European waste management. A combination of Factor B and scenarios reflects long-term impacts, which are likely to dominate in the long run. The long-term perspective, in turn is often considered more important from a sustainability perspective. In addition, Factor B has a higher precision, compared to the scenarios.

Neither Factor B nor the European scenarios will always give a correct indication on whether renewable LDPE waste should be recycled or incinerated. This is probably related to the fact that LCA tends to focus on one product at a time. The limited perspective seems to be an obstacle when used to assess recycling, energy recovery and other developments that require concerted actions of actors in different product life cycles. Further research might shed light on the extent and significance of this limitation.

The net environmental benefit of recycling is achieved by the concerted action to recover material for recycling and to use the recycled material instead of virgin material. When possible, the actions to recover material for recycling and to use the recycled material should be jointly assessed. Similarly, the actions to recover energy and to use the recovered energy should be jointly assessed. This will not only reduce or eliminate the risk of incorrect incentives for incineration or recycling. It will also avoid the subjective choice of method for dealing with the allocation problems.

Besides the risk of an incorrect incentive for incineration in general, LCA results can indicate that incineration should be located to countries with an environmentally poor energy system, rather than in incineration plants with a high total energy efficiency. This risk might be alleviated through consequential LCA. Further research is needed to investigate to what extent this is a problem, and to what extent consequential LCA can solve it.

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Annex 1: Input variables and calculation results

Table 1 – The variable values used in Equation 3 used for mechanical and chemical recycling of fossil and renewable LPDE, respectively (Wolf, 2020).

		Mechanical recycling				Chemical recycling			
Description of variable	**Vari-able**	**Fossil LDPE**	**Fossil LDPE**	**Renew-able LDPE**	**Renew-able LDPE**	**Fossil LDPE**	**Fossil LDPE**	**Renew-able LDPE**	**Renew-able LDPE**
Allocation factor of burdens and benefits between supplier and user of recycled materials. [-]	A	0.0	0.5	0.0	0.5	0.0	0.5	0.0	0.5
Proportion of the material in the product that will be recycled (or reused) in a subsequent system. [-]	R_2	1	1	1	1	1	1	1	1
Specific emissions arising from the recycling process at EoL, including collection, sorting and transportation process. [kg CO_2-eq./tonne LDPE]	$E_{recycledEoL}$	51	51	51	51	838	838	51	51
Specific emissions arising from the acquisition and pre-processing of virgin material assumed to be substituted by recyclable materials. [kg CO_2-eq./tonne LDPE]	E^*_v	1900	1900	1900	1900	165	165	165	165
Qs=Quality of the outgoing secondary material, i.e. the quality of the recycled material at the point of substitution. Qp=Quality of the primary material, i.e. quality of the virgin material. [-]	Q_s/Q_p	1.0	0.9	1.0	0.9	1	1	1.0	1
Circular footprint. [kg CO_2-eq./tonne LDPE]	CF	-1849	-829	-1849	-829	673	337	-114	-57

Table 2 - The variable values used in Equation 1 used for incineration with energy recovery of fossil and renewable LPDE, respectively (Wolf 2020; Phyllis 2020; Seyed et al. 2020).

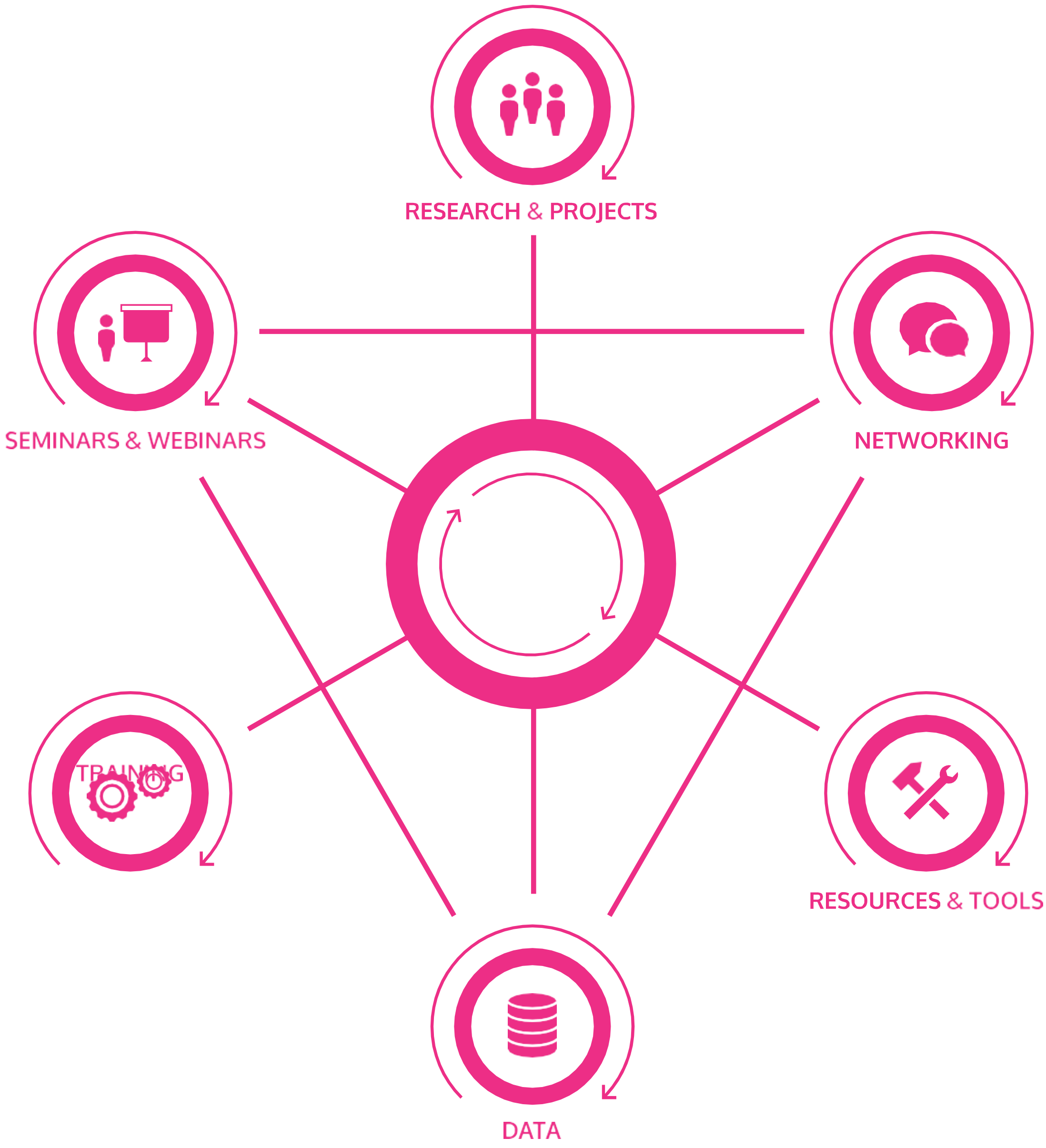
	Energy recovery				
Description of variable	**Variable**	**Fossil LDPE**	**Fossil LDPE**	**Renewable LDPE**	**Renewable LDPE**
Allocation factor of energy recovery processes. It applies both to burdens and benefits [-]	B	0	0.6	0	0.6
Proportion of the material in the product that is used for energy recovery at EoL. [-]	R_3	1	1	1	1
Specific emissions arising from the energy recovery process. [kg CO_2-eq./tonne LDPE]	E_{ER}	3000	3000	4	4
Lower heating value of the material in the product that is used for energy recovery. [MJ/tonne LDPE]	LHV	42830	42830	42830	42830
The efficiency of the energy recovery process for heat. [-]	$X_{ER,heat}$	0.85	0.85	0.85	0.85
Specific emissions that would have arisen from the specific substituted heat. [kg CO_2-eq./MJ]	$E_{SE,heat}$	0.01	0.01	0.01	0.01
The efficiency of the energy recovery process for electricity. [-]	$X_{ER,elec}$	0.3	0.3	0.3	0.3
Specific emissions that would have arisen from the specific substituted electricity. [kg CO_2-eq./MJ]	$E_{SE,elec}$	0.01	0.01	0.01	0.01
Circular footprint [kg CO_2-eq./tonne LDPE]	CF	2372	949	-623	-249

Table 3 - The variable values used in Equation 4 used for calculating climate impacts of incineration with energy recovery that affects treatment of residual waste elsewhere in Europe (European incineration and European landfill) for the treatment of affected (Wolf 2020; Phyllis 2020).

		Energy recovery							
Scenarios		**European incineration**				**European landfill**			
Description of variable	**Vari-able**	**Fossil LDPE**	**Fossil LDPE**	**Renew-able LDPE**	**Renew-able LDPE**	**Fossil LDPE**	**Fossil LDPE**	**Renew-able LDPE**	**Renew-able LDPE**
Allocation factor of energy recovery processes. It applies both to burdens and benefits. [-]	B	0	0.6	0	0.6	0.0	0.6	0.0	0.6
Proportion of the material in the product that is used for energy recovery at EoL [-]	R_3	1	1	1	1	1	1	1	1
Specific emissions arising from the energy recovery process. [kg CO_2-eq./tonne LDPE]	E_{ER}	3000	3000	4.27	4.27	3000	3000	4	4
Specific emissions arising from the avoided energy recovery process of imported waste. [kg CO_2-eq./tonne LDPE]	E^*_{ER}	2274	2274	2274	2274	2274	2274	2274	2274
Specific emissions arising from landfill with energy recovery in a European county other than Sweden [kg CO_2-eq./tonne LDPE]	E_{Sc}	958	958	958	958	2805	2805	2805	2805
Lower heating value of the material in the product that is used for energy recovery [MJ/tonne LDPE]	LHV	42830	42830	42830	42830	42830	42830	42830	42830
Circular footprint [kg CO_2-eq./tonne LDPE]	CF	1684	674	-1311	-525	3531	1412	535	214

Table 4 - The variable values used for the waste options in the four cases presented in Figures 8 to 15.

Case	**Waste option**	**A**	**Q_s/Q_p**	**B**	**Circular footprint [kg CO_2-equ./tonne fossil LDPE]**	**Circular footprint [kg CO_2-equ./tonne renewable LDPE]**
Simple substitution	Mechanical recycling	0	1		-1849	-1849
	Chemical recycling	0	1		673	-114
	Energy recovery			0	2372	-623
PEF (default)	Mechanical recycling	0.5	0.9		-829	-829
	Chemical recycling	0.5	1		337	-57
	Energy recovery			0	2372	-623
PEF (B=0.6)	Mechanical recycling	0.5	0.9		-829	-829
	Chemical recycling	0.5	1		337	-57
	Energy recovery			0.6	949	-249
PEF (scenarios)	Mechanical recycling	0.5	0.9		-829	-829
	Chemical recycling	0.5	1		337	-57
	Energy recovery: European incineration			0	1684	-1311
	Energy recovery: European landfilling			0	3531	535



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