Plastics: Can Life Cycle Assessment Rise to the Challenge?

How to critically assess LCA for policy making

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#break free from plastic

Report Commissioned by Break Free From Plastic

The #breakfreefromplastic Movement is a global movement envisioning a future free from plastic pollution. Since its launch in 2016, more than 8,000 organizations and individual supporters from across the world have joined the movement to demand massive reductions in single-use plastics and to push for lasting solutions to the plastic pollution crisis

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Executive Summary

This paper explores how life cycle assessment (LCA) – the de facto tool used across the world for assessing the environmental impacts of products and processes—is used today to influence policy decision making particularly in regards to single use plastic products. Understanding its process and limitations is key to determining LCA's place as a tool in decision making and the realisation that it neither can nor should provide all of the answers or justifications. As a tool it is also limited by the questions it is being used to answer; ask inappropriate, misleading, narrow or uninformed questions and the process will only provide answers in that vein. We look at the issues around how the context of the study can frequently be abandoned and that often those reading such studies are not experts in the field of LCA – be they policy makers, NGOs, journalists or even individuals—which presents a challenge for all involved to limit misunderstandings and poor decisions as a result.

Very often results of a study from a different country or a different time period (or both) are used to justify a position or policy without an assessment of how applicable the results would be when the context is different. The increased availability of 'off the shelf' life cycle inventory databases means data can easily be picked from a library and incorporated into a study without any expert knowledge of what went into developing it. The background data needs a lot of time and expertise to evaluate or the results can be misleading. It is therefore important to question unexpected results and analyse the underlying data to trace the cause and ascertain whether conclusions are still valid and robust.

When modelling the end-of-life, recycling rates are notoriously inaccurate and difficult to compare. LCAs often use nationally reported recycling rates and assume a closed-loop process. In reality there are many losses from the point of collection to the material actually becoming recycled and exported plastic is almost untraceable and can be a source of marine pollution—these aspects are often not considered in LCA studies. Conducting an LCA is always an act of simplifying a system to allow the assessment to take place, although it is often mistakenly misrepresented as an attempt to completely and accurately reflect reality. This scenario building is open to misuse and is why comparative studies from industry can be problematic, not due to their lack of methodological correctness, but that a narrow and carefully curated view can be taken that can be hard for a (non-expert) reader to unravel.

Whilst there is no recognised or agreed method for including the leakage of plastics (or other materials) into the environment, it is encouraged that this subject is at least discussed within an LCA study with respect to any conclusions that are drawn. There still remains significant challenges in determining flows, sinks and impacts in a quantifiable way and there are also many other leakage points along the value chain that would need to be incorporated into existing datasets.

Throughout this paper, guidance is provided for the reader, the practitioner and the study commissioner to help avoid or mitigate some of these issues, and the following provides a summary of the most important to consider.

Guidance for Readers

- Is the study scope representative of a general scenario which can be applicable more widely, or is it aimed at answering a very specific question?
- For comparative studies, look for a third-party critical review, particularly if the practitioner and/or the study commissioner is not independent (i.e. there may a vested interest in a particular result).
- Are future waste scenarios included? If so, are they realistic in that particular geography and how applicable would these conclusions be in other places given the variety of long term commitments that exist throughout the world.
- Has the study included some realistic future scenarios for recycling/ reuse and circularity or is it based on a current scenario.
- Increasingly, a lack of methodology is not enough for a practitioner to exclude all reference to littering impacts particularly for single-use items; this paper presents one such way that can be used as an interim measure whilst further research is conducted into creating a formalised indicator.

Guidance for Practitioners

- Understand the policy context for your study and determine whether the results are likely to be influential in systems change.
- Exercise caution around what is placed in an Executive Summary – only present the strongest and most defensible results.
- Do not exclusively rely on secondary datasets that are key to the outcome of the study – these are usually derived from industry inventories, but are not always updated regularly.
- If exports are the predominant route for the waste, it is important to question the true fate of the materials and be conservative with recycling assumptions.
- If the end of life appears to have a considerable influence on the result, perform a sensitivity analysis to determine whether realistic future changes to this would affect the result.
- Explicitly identify the forms of plastic pollution into both marine and terrestrial environments (e.g. littering, pellet loss, losses from exports) even if these cannot be quantified. This will allow decision-makers using the LCA to be aware of the limits of using this tool alone, and the need to complement it with estimates for these missing elements.

Guidance for Commissioners

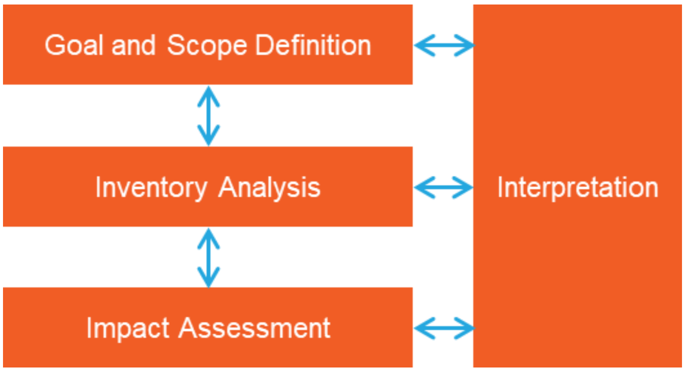
- When communicating the results, be mindful of not making claims beyond that which the study scope is valid for.
- For comparative assertions that are to be made public, it is important to reserve time and budget for the peer review process. The peer reviewers should include independent experts in the particular field with no vested interest in the result, but with a mix of perspectives and expertise that will help to strengthen the study.
- Ask the practitioner to include plastic leakage in the scope of the study and include time and budget to assess this with the appropriate amount of rigour given the state of knowledge and methods available at the time.

1.0 Introduction to LCA

Life cycle assessment (LCA) is a methodological tool for assessing environmental impacts associated with all the stages of a product's life-cycle from raw material acquisition, production, use, end-of-life treatment, recycling and final disposal (i.e. cradle-to-grave). It can be used by individuals or businesses to help make internal decisions through to creating a justification for wide ranging policy decisions. Understanding its process and limitations is key to determining its place as a tool in decision making.

Internationally, ISO 14040/44[1] is used by LCA practitioners as the guiding framework for conducting and reviewing studies. The framework is designed to be adaptable enough to be used for any type of LCA and therefore it does not provide strict criteria for assessment. Figure 1 shows the key phases of an LCA which clearly demonstrates that it is not a strictly linear process; for example, as a study progresses the scope may need to be changed to suit the availability or quality of data.

Figure 1: Phases of an LCA



Source: Adapted from EN ISO 14040:2006

Because of the flexibility of the LCA framework, results are often criticised as being biased or unhelpful for decision making. This is not necessarily the case if LCA is viewed as one of many tools (and not the only one) that can be used to help decision making. This discussion paper identifies some of the common ways in which LCA can be misinterpreted or misused and provides some of the key knowledge and questions that a reader can use to help interrogate and discuss results. It also provides some guidance for LCA practitioners that want to produce a more robust and defensible study and want to reduce instances where their study is taken out of context. Finally, we look at how LCA might develop in the future to incorporate more progressive methodologies, such as integrating circular economy principles and how we might include a simple plastic leakage impact indicator.

[1] ISO (2006) *ISO 14044 Environmental management - Life cycle assessment - Requirements and guidelines*, 2006

2.0 Interpreting LCA Results

When reading a study, it is often the results that are the first element to be examined, however these shouldn't be viewed without context. This is why understanding the goal (or aim) of the study, and how the scope has been designed to achieve that goal, should be the first step. A third-party peer reviewer will always refer back to the goal to determine whether the study achieved what it set out to do. Several cycles of tweaking the study, so that goals and results are aligned, can often follow and this iterative review process should be transparently presented alongside the report.

2.1 Maintaining Context – Specific or Generalised?

The scope description that drives an LCA is key in understanding how applicable the results are to a given situation. Generally, the narrower the scope, and more precisely defined the circumstances, the more reliable the results will be in regards to that situation. At one extreme, an LCA could answer whether a particular individual should use a glass or plastic bottle based on what might be available to them currently. The LCA could look closely at the particular situation with minimal assumptions and be confident of the result—but only for that specific person. At the other extreme, the same question could be asked, but at a global scale for every human on earth. It is clear that the latter scenario would be impossible to model and generalise with any degree of accuracy; the results would have little value. In reality, studies will sit somewhere between these two extremes and it is not always immediately obvious whether results are applicable to different situations.

An example of this in practice can be taken from a Danish EPA comparative carrier bag LCA study from 2018.[2] The study compared various grocery bag types and established an average specification from bags available at 23 stores in Denmark. Single use polyethylene (PE) bags were available at all of the stores, but the nine reusable alternatives were only available at up to three of the stores and there was only one example of an organic cotton bag. To compare these bags a *functional unit* was required – this allows the study to take into account the ability for a product to provide a certain service (the carrying of groceries) in order to make a like for like comparison. In this case it was the ability to carry 12kg and 22 litres of groceries which was the volume and weight carrying capacity of the available single use LDPE bags. Unfortunately, the only organic cotton bag that was available was one that could carry 20 litres which resulted in the author determining that two of these bags were required to fulfil the functional unit; thereby doubling all impacts for this type of bag.

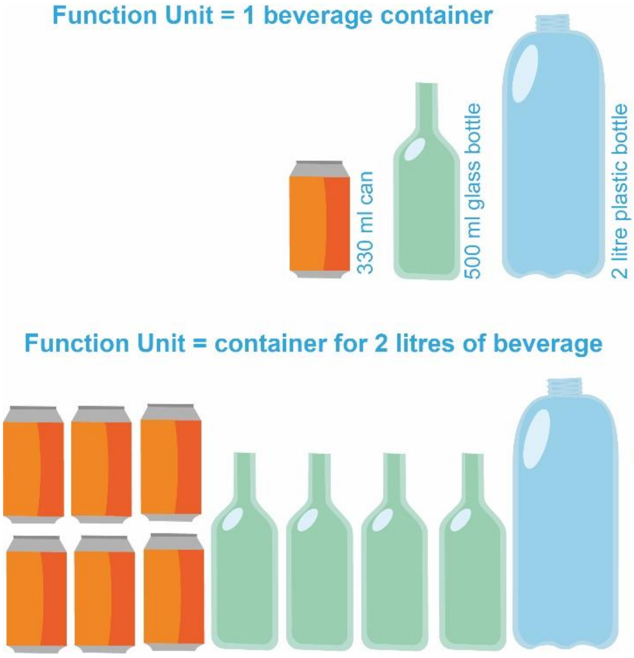
This means the study might be relevant to an individual who is presented with this particular selection of bags in Denmark (and acts strictly according to the author's assumptions), but less helpful to form the basis of national policy on the subject,

[2] Bisinella, V., Albizzati, P.F., Astrup, T.F., and Damgaard, A. (2018) *A Lifecycle Assessment of Grocery Carrier Bags*, Report for The Danish Environmental Protection Agency, February 2018

particularly for countries other than Denmark. A better, but albeit simplistic approach, might have been to assume that an organic cotton bag that was capable of carrying 22 litres requires an additional 10% mass to do so and that the design could be changed accordingly. Equally, the functional unit could be changed to a whole years' worth of grocery shopping so the difference in carrying capacity is averaged out over a year.

The study itself discusses this issue at length and includes some suggestions on design optimisation, however qualitative comments can often be overlooked by the reader in favour of the headline result. This study is a good example of how a result can be taken out of context in a way that could lead to a less optimal outcome; very often results from a study from a different country or a different time period (or both) are used to justify a position or policy without an assessment of how applicable the results would be when the context is different.

Figure 2: Functional Unit Examples



The example also highlights the importance of choosing an appropriate *functional unit*. Figure 2 shows two further examples of how different a study might be depending upon this choice. A comparison of beverage containers might literally just compare the container itself, but more usually its function of containing the beverage is the important one. In this example, we see that it would take six cans to fulfil the same function as a large plastic bottle which would give very different results compared with a single packaging type alone. However, either approach might be correct depending upon whether it is appropriate for meeting the goal and scope of the study; all decisions link back to this very beginning premise.

It is often claimed that it is not the responsibility of the LCA practitioner (or indeed, the peer reviewer) to predict how a study will be subsequently used. However, it could also be argued that if the way the study is scoped or presented allows the results to be distorted or taken out of context, then it is the practitioner's responsibility to seek to reduce this as much as possible. This is particularly important for well recognised divisive subjects such as single use packaging.

2.2 Reuse – The Missing Aspects of Consumer Behaviour and Optimum Design

When reuse scenarios are added into comparative LCAs the question of whether they are realistic is often not addressed and often based on assumptions around behaviour. If a cotton bag is required to be reused ***X*** times for the equivalent environmental impact of a PE bag, what value of ***X*** is too high? What is consumer behaviour in this regard, how does this differ culturally, and how can we influence people to use the product for that long (and longer)? Equally, is the product capable of being reused this many times, and if not, can one be designed that does so with minimal material use? LCA could be used to explore the environmental impacts associated with such scenarios which may be more instructive than being limited by a marketplace that is generally not driven by (or even considers) environmental interests.

Continuing with carrier bags as an example; PE bags have been engineered to use the least amount of material for their intended purpose. Every gram that is saved is a tremendous cost saving when a retailer might purchase these in their millions. Reusable bags have not seen the same focus, primarily because the consumer is required to pay for them, not the retailer. Producing a reusable bag that is fit for years of reuse with the minimum amount of material has not been the driving force for their design.

A different perspective for the purpose of LCA is that it can also be used to determine the specification thresholds that design needs to achieve (e.g. the maximum mass of virgin material or the optimum volume and carry capacity). It should not be seen as a tool that assesses absolutes and provides certainties. If used dynamically it can be part of the design process that can help reach optimal outcomes.

One scenario often posited in an LCA is that of the reuse of a single use item as a replacement for another product after its initial use. A common example of this is for a carrier bag to be reused as a waste bin liner. This appears sensible, but it is important to use realistic assumptions around user behaviour in this aspect – this extra use is not an implicit part of the product, but relies on individuals making this decision. If there is no reliable survey data around this behaviour it is prudent to be conservative or use a range for this reuse assumption; a series of 'what if?' scenarios. Assuming 100% reuse would clearly be unlikely for example.

Again, we often fail to look at this kind of scenario from a systems perspective, but only through the lens of existing habits. Waste systems are changing, often with the emphasis on segregating different material streams for recycling. In particular organic waste is increasingly being collected separately from a household (and will be law in the EU by 2024). Does the continued use of single use bags followed by a reuse as a bin liner represent the best possible way to transport both goods and waste or is this behaviour the result of a lack of joined up thinking between the producers of products and waste management? Often the number of reuses required to compare with the status quo is the main reported figure. This can be problematic as it is often portrayed as the responsibility of the individual to achieve these reuse numbers. Instead, it may be more productive to reframe it into a discussion around how a system can be put in place that normalises, encourages and makes this option the obvious one and not a choice between convenience and the environment.

Guidance for Study Readers

- Study results should never be presented independently of the scope as the two are inextricably linked.
- Is the study scope representative of a general scenario which can be applicable more widely, or is it aimed at answering a very specific question?
  - If the latter, will any of the assumptions significantly change if the scenario changes and how might this affect the outcome?

- If used to justify policy decisions, it is based upon an idealised scenario or on current practice?

- For comparative studies, look for a third-party critical review, particularly if the practitioner and/or the study commissioner is not independent (i.e. there may a vested interest in a particular result). Ideally this should be a panel involving two or more independent experts and the process transparently appended to the report.
- If the commissioner does not make the full study available to the public, be cautious of the results; the study itself may have been reviewed by experts, but any marketing or subsequent summaries may not have been.
- How were the comparative products chosen?
    - Are they representative of the best technological solution or simply what is available locally? – are some optimised and some not?
    - Are other more favourable alternatives missing from the analysis? e.g. for bags, fibre types other than cotton.

Guidance for Practitioners

- Think about whether the functional unit chosen allows a fair comparison, particularly if the chosen products have not been optimised for the purpose of the assessment.
- Understand the policy context for your study and determine whether the results are likely to be influential in systems change.
- Apply a cautionary approach to setting goals and scope – caveats are not always enough to mitigate misuse of the study – headline results are often shared around without context.
- Exercise caution around what is placed in an Executive Summary – only present the strongest and most defensible results.
- Be clear about which assumptions influence the results significantly so that the reader can identify whether the results of the study are applicable to other scenarios.

Guidance for Commissioners

- When communicating the results, be mindful of not making claims beyond that which the study scope is valid for.
- For comparative assertions that are to be made public, it is important to reserve time and budget for the peer review process. The peer reviewers should include *independent* experts in the particular field with no vested interest in the result, but with a mix of perspectives and expertise that will help to strengthen the study.

2.3 How We Use Data

One of the challenges with LCA is that there are many environmental indicators (e.g. climate change, ozone layer depletion, ecotoxicity), but they are not all considered equal either in terms of impact or the science they are based upon. There is no scientific consensus around which impacts are more important; hence the reason why weighting of impact categories for external communication is discouraged in ISO standards due to the additional uncertainty involved. Part of the reason for this is some impacts are global, e.g. climate change, and some manifest locally, e.g. particulates as air pollution.

The practice of LCA expects that indicators are presented equally, but some may take more precedence depending upon the situation (both physical and political). The scientific uncertainty is recognised by the European Commission's Joint Research council (JRC) who are leading the work to create a harmonised LCA methodology for products across Europe—the Product Environment Footprint (PEF) model. Out of the sixteen impact categories that are recommended to be used in all studies, only three are considered "satisfactory" —climate change, ozone depletion and particulate matter—whilst seven should "… be applied with caution."[3] The current list of these impact categories and methodologies can be found in Appendix 4.0.

Another underlying issue is the increased availability of life cycle inventory databases – these databases contain a vast amount of primary data collected over many years, from different countries and for a variety of industrial processes. They can easily be picked from a library and incorporated into the study without any expert knowledge of what went into developing the data (the Single Market for Green Products Initiative in the EU is designed to address this by verifying data source, but this is only available for a small number of product categories presently[4]). The availability of datasets allows practitioners to create studies that do not require significant amounts of primary research to conduct. This is helpful for businesses who want to assess their environmental impacts, but have neither the means nor expertise to develop their own primary datasets—it allows quick 'screening studies' that can help with internal decision making to be undertaken, with a relatively low investment.

It is therefore important to question unexpected results and analyse the underlying data to trace the cause and ascertain whether conclusions are still valid and robust – if not, other, more robust data sources should be sought or alternatively the scope of the study modified to exclude that aspect from the results. This is particularly important if the results rely on one key data source.

Taking the Danish EPA study as an example again, one of the headline results is that cotton bags are required to be reused 7,100 and organic cotton bags 20,000 times in order to be comparative with single use PE bags from the perspective of *ozone depleting emissions*. In order to model organic cotton, the authors modified an existing *global*

[3] Zampori L, and Pant R (2019) *Suggestions for updating the Product Environmental Footprint (PEF) method*, Report for Joint Research Centre (JRC), 2019
[4] https://ec.europa.eu/environment/eussd/smgp/index.htm

average[5] cotton production dataset by reducing the crop yield and removing pesticides (including all of the impacts across their lifecycle). This in itself is a fairly simplistic approach that does not recognise that organic cotton growing is a fundamentally different process. Evidently, the dataset used is older and not representative and likely to overestimate impacts when compared with recent studies[6] – for example despite the crop yield for organic cotton being slightly lower, the greenhouse gas (GHG) emissions per tonne produced are actually somewhat reduced. Indeed, the cultivation of cotton itself actually contributes rather less to the overall environmental impact (~10% for climate change) compared with the *textile production* itself, other than for eutrophication and water use impacts. This means that the large difference between the cotton types cannot be accounted for simply by differences in crop yield.[7] This is a good example of why datasets that heavily influence the results should at least be evaluated against three main data quality criteria; time, geography and technology.

1) **Time** – the data should be representative of the current year i.e. it can be older, but an investigation should be conducted to determine whether changes might have occurred that affect the results.
2) **Geography** – the data should be representative of the geographic situation defined in the scope of the study.
3) **Technology** – this concerns all other technical aspects that should be representative.

The cotton datasets in this study would likely fail to meet an assessment of data quality for at least the time and technology aspects.

It is also unclear where significant ozone depleting emissions would come from in woven cotton production as there are no direct emissions from the process and these would only be emitted in small amounts as part of electricity production. Whilst textile production uses significant amounts of energy, the latest data suggests ozone depletion is 16 times lower than the results from the dataset used in this study which reduces the reuse break-even value down to around from 7,100 to 446 uses for non-organic cotton.

It is also important to address the impact category of *ozone depletion* itself. This is the propensity for an air emission to contribute to the breaking down of atmospheric ozone into oxygen. Emissions that contribute to this are now tightly controlled so the key question would be; is increasing the emissions resulting from producing a PE bag by 446 times likely to be a significant contributor to ozone depletion?

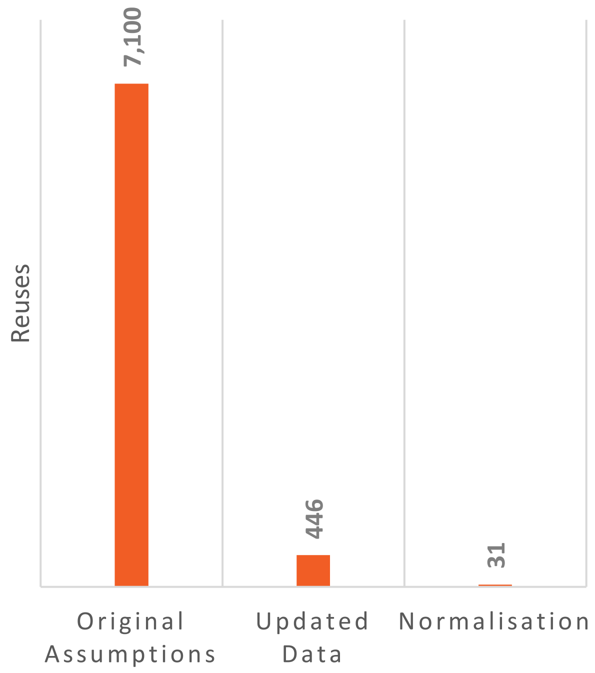
[5] The study used global averages for all materials in the study – this may or may not be appropriate for a study depending upon the value chain. One aspect global averages do not work for is where decisions can be made to improve impacts by switching supply to a different region.

[6] Thinkstep Sustainability Solutions (2018) *Life Cycle Assessment of Cotton Cultivation Systems: Better Cotton, Conventional Cotton and Organic Cotton*, Report for C&A Foundation, May 2018

[7] John Jewell (2017) *LCA UPDATE OF COTTON FIBER AND FABRIC LIFE CYCLE INVENTORY*, Report for Cotton Incorporated, March 2017

To put this into context a process called *normalisation* can be used which takes the calculated average impacts per person in a given geography i.e. the annual environmental footprint of an individual. By doing this for the EU[8] we find that over 34,000 cotton bags would have to be purchased to equal the average overall *ozone depletion* impacts associated with one person over the course of a year. However, only 2,300 would have to be purchased to equal the *global warming potential* from one person; around 14.6 times more impactful. By taking this into account and we now find that only 31 reuses would be necessary to break even with a PE single use bag as shown in Figure 3.[9]

Figure 3: Cotton Bag Reuses Required to Break Even with PE Bags



This demonstrates that whilst cotton bags might be considered hundreds of times worse than PE for *ozone depletion*, the difference between the two in reality is relatively small when looking at it in a broader context. Equally, it also demonstrates how it is relatively straight forward to produce different results and therefore different outcomes when certain data sources are used.

Generally, when conducting a study that results in very stark or unexpected results, it is important to investigate the route of that result e.g. "why does ***X*** have a significantly larger influence on an impact than ***Y***?" Results should be treated with caution where questions such as this are not addressed in a study (particularly if this appears prominently in an executive summary) as there are so many underlying data points and assumptions that form the results that inaccuracies can easily seep in. The use of standard datasets with outdated assumptions are often a chief cause of this problem, as demonstrated.

The difficulty lies in determining this without the expert knowledge of the datasets and the time to investigate assumptions fully. Therefore, it is important for LCA practitioners to be aware of these issues and to work to mitigate, and for those reading the study to be aware of the key aspects that should be questioned. Some guidance on what may be considered for both groups is proposed below.

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[8] Lorenzo Benini, Lucia Mancini, Serenella Sala, Simone Manfredi, Erwin M. Schau, and Rana Pant (2014) *Normalisation method and data for Environmental Footprints*, Report for European Commission Joint Research Centre, 2014

[9] Normalisation factors per person: Climate change = 9.22E+03, Ozone depletion 2.16E-02. This is based on inventory data from the EU 27 in 2010 with 499 million inhabitants.

Guidance for Study Readers

- Look for a data quality assessment which should ideally be quantitative and shows key data sources rated against its representativeness in time, geography and technology.

Guidance for Practitioners

- Do not exclusively rely on secondary datasets that are key to the outcome of the study – these are usually derived from industry inventories, but are not always updated regularly.

Guidance for Commissioners

- It may be possible to complete studies with small budgets and narrow timescales, but it is important to recognise that the quality, robustness and depth of the study may be questionable and not present good value for money.

2.4 Discussing the Limitations of LCA

A standard part of the LCA reporting process is to write a description of the limitations of the study. This often involves a critique of the data or a discussion around how applicable the results might be outside of the defined scope of the study.

However, there are often limitations that are implicit, but not stated, as it is assumed that the reader has a good understanding of LCA methodology more generally. This is problematic for public facing studies that may be read by non-experts.

From the perspective of plastics, the emissions resulting from the industrial processes, and to a lesser extent the end-of life are well studied and understood. This often includes an assessment of the toxicity in different environments of all the chemicals that might be emitted. However, the emissions of the material itself during production and transport from spillages of pellets before they are moulded into products (Figure 4), are not usually part of a study – this is discussed further in Section 4.0. Any toxicity (chemical exposure) associated with the use-phase of the product is also rarely included in an LCA; this is less of an oversight by the practitioner and more associated with the lack of a methodological framework to do so within LCA.

Equally, it is questionable whether it should be within the remit of LCA to address issues of chemical exposure during use. Arguably, once the science is developed enough to identify human toxicity risks during use, this is the point in which legislation is used to restrict the usage (as with the REACH[10] and Food Contact[11] Regulations in the EU). LCA is therefore not necessarily the tool to highlight and compare the inherent risk during use.

Figure 4: **Plastic Pellets Found on a UK Beach**

Source: Eunomia

Importantly, LCA does not provide the only method for decision making or as a metric for justifying choices in regard to every environmental issue. An LCA cannot determine whether a product is 'sustainable' or not. The process does not provide any thresholds or limits for impacts that would make this type of assessment possible. It also generally does not identify any social issues or the impacts associated with consumption and consumerism. LCA can help to answer certain specific environmental questions, but doesn't provide a framework for asking the 'right' questions.

Guidance for Study Readers
• Look for a description of the limitations of the study to determine whether these have been described adequately.

Guidance for Practitioners
• Think carefully about the limitations of the study from the perspective of a non-expert; conveying to the reader that the study is not likely to address all possible environmental impacts is important. Equally, do not use a description of the limitations to excuse a study that could have been scoped differently to avoid these limitations.

[10] https://eur-lex.europa.eu/legal-content/en/TXT/?uri=CELEX:02006R1907-20200428
[11] https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:02004R1935-20090807

3.0 Considering End of Life

In terms of single use packaging in particular, the assumptions for the end of life can be the most decisive aspect, regardless of the upstream impacts associated with raw materials, production and transport. This includes whether the product is reused, gets recycled (and into what) or ends up in landfill or an incinerator. These can often be somewhat idealised scenarios based on national averages and published recycling statistics, with less time spent identifying what actually takes place at the end of life. This is particularly important for plastics, given that large amounts are known to be exported to countries with more lenient regulatory frameworks where the fate is largely unknown. In light of this, the assumption that even plastic collected for recycling actually ends up recycled is increasingly being challenged as plastic exports are beginning to be identified as a source of marine plastic pollution.[12]

An end of life scenario may also include other forms of 'mismanagement' such as littering which frequently happens with many single use items and often as a result of poor waste management, particularly packaging. However, the environmental impacts of any non-controlled disposal are notoriously difficult to estimate given the uncertainties present.

3.1 The Changing Nature of Residual Waste Treatment

One of the reasons plastics often appear to perform well in LCA is that, up until recently, landfill has often been the de facto waste treatment method for much of the world. The impact of plastics in landfills are usually associated only with transport and maintaining the landfill, with no direct emissions as they remain largely inert. There is evidence to suggest that plastics may not be entirely inert, but the transition from scientific hypotheses to being embedded in life cycle inventories is not a fast process and emissions are still likely to be minimal.[13] Developing landfill GHG emissions inventories for individual materials is also notoriously difficult given that the landfill reacts differently depending upon many factors including its waste composition. Nevertheless, compared with burning plastic in an incinerator—where all its carbon is released—landfill is somewhat less impactful from the perspective of *climate change*.

Most modern incinerators also act somewhat like coal-fired power stations by generating electricity from the heat— energy-from-waste (EfW). In LCA it is typical to include the 'benefit' (or *'avoided burden'*) obtained by incinerating plastic waste as the

[12] Bishop, G., Styles, D., and Lens, P.N.L. (2020) Recycling of European plastic is a pathway for plastic debris in the ocean, *Environment International*, Vol.142, p.105893

[13] Royer, S.-J., Ferrón, S., Wilson, S.T., and Karl, D.M. (2018) Production of methane and ethylene from plastic in the environment, *PLOS ONE*, Vol.13, No.8, p.e0200574

electricity (and sometimes heat) generated means less energy needs to be produced from other sources.

In the US, landfill still dominates although there is a slow move towards EfW. A plastic packaging LCA study from 2018 used US EPA waste data from 2010 which indicated that only 18% of waste was sent to incineration, however more recent data for 2017 shows this has grown to 20%.[14] Although this difference is relatively small, it is important to use the latest waste statistics for the region of study and even develop future scenarios around what is likely to happen based on current policy commitments—this is particularly important when using LCA to justify long term decision making. It is also clear that applying the results from this study in a country with very different waste treatment infrastructure would lead to inaccurate conclusions.

In the EU at least, there has been an increasing trend towards EfW plants (from 38% in 2010 to 55% in 2017[15]) as landfill becomes marginalised. Therefore, studies based on waste data from several years ago may not reflect the current reality—and this reality may still change yet again as the realisation emerges that becoming 'locked in' to incineration actually inhibits the ability to reach high recycling rates. Equally, studies based on either the US or EU waste systems cannot be considered comparable without isolating this factor. Even within the EU care must be taken as waste systems vary enormously. In Sweden for example, landfill is banned, and the aim is to prevent plastic entering incinerators entirely, whilst in Wales the aim is to eliminate plastic from landfill, but EfW is considered acceptable. This is another reason why, both *temporal and geographical representativeness* should be transparent in such a study.

Importantly, for those countries that are relying more on EfW studies will generally show increasingly worse environmental impacts for plastics that are not recycled. This is due to the trend towards decarbonising energy systems—if energy generated from burning plastic replaces renewable technologies (instead of more polluting fossil fuels) there will come a point where this is incompatible with future decarbonisation targets. LCAs that look at future scenarios from 2030 and beyond (which a study focused on influencing policy should) will find that burning of plastics becomes increasingly untenable.

[14] https://www.epa.gov/facts-and-figures-about-materials-waste-and-recycling/national-overview-facts-and-figures-materials

[15] Eurostat

Guidance for Study Readers

- Check the assumptions used for residual waste treatment
  - Are they realistic for the given geography?
  - Is the timeframe current enough that no significant changes are likely to have taken place?
- Are future waste scenarios included? If so, are they realistic in that particular geography and how applicable would these conclusions be in other places given the variety of long term commitments that exist throughout the world.

Guidance for Practitioners

- Look for the latest data on waste treatment available, and make sure that is representative e.g. for the geography in question
- Consider performing a sensitivity analysis to determine whether a possible future scenario will have a significant effect on the outcome of the study.

Guidance for Commissioners

- Look for a practitioner who understands and has experience in waste systems and the complexities involved in modelling these.

3.2 Looping of Material – How LCA Deals with Recycling and Circularity

Recycling rates are also notoriously inaccurate and difficult to compare – LCAs often use nationally reported recycling rates and assume a closed-loop process. In reality there are many losses from the point of collection to the material actually becoming recycled. This is why the EU has recently adapted its recycling measurement method to only include material that becomes a recycled product (rather than assuming it will be recycled if it is collected for recycling).[16] This will likely have the result of radically reducing reported recycling rates in the EU particularly for plastics.

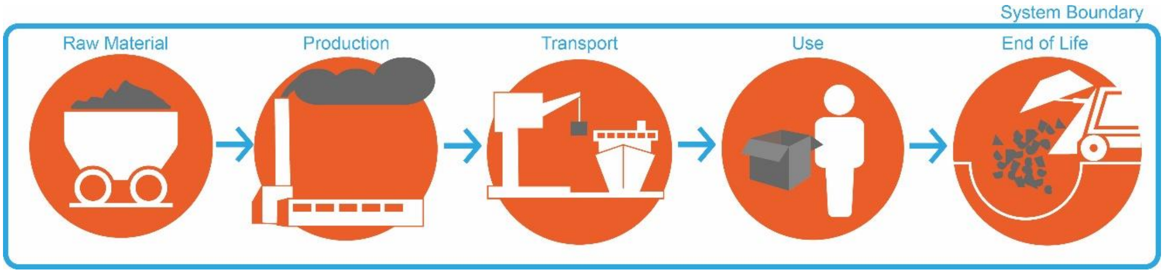
In the same way, it is important to be accurate with residual waste treatment, recycling rates also require similar attention to detail. Very often *current* reported recycling rates are used in studies where the end-of-life assumptions prove pivotal to justify *future* decisions. Again, this doesn't reflect the best outcome given future recycling targets and the drive towards more optimised systems. An example of this is a 2017 study for a composite carton manufacturer which compared various milk containers with multi-

[16] https://eur-lex.europa.eu/eli/dec_impl/2019/1004/oj

layer cartons in the Nordic countries.[17] The results clearly indicated that the end-of-life destination played a key role in determining which packaging system had the lowest overall impact and a sensitivity analysis looking at a future scenario was not performed. Without this type of scenario analysis, the study is of limited use for policy decision making due to its narrow focus and its value is therefore primarily to the manufacturer as a marketing tool (often with the results taken out of context as discussed in Section 2.0). This is generally why comparative studies from businesses can be problematic, not due to their lack of methodological correctness, but that a narrow view can be taken that can be hard for a (non-expert) reader to unravel.

Most comparative studies focus on a cradle to grave methodology where the product is assessed through its life cycle from raw material acquisition to disposal at the end of life (Figure 5). The 'end-of-life' for that cycle may be recycling or a number of reuses, but often no particular consideration is given to subsequent use of that recycled material. This is fine for LCA in the context of individual products placed on the market (for example, if a company wanted to understand the carbon footprint of one of their products) and worked reasonably well in a traditional linear system, but as the focus moves towards how the grave of one product is the cradle of another, modelling the system becomes more complex and difficult to communicate.

Figure 5: Typical Linear System Model



There will be an increased need for brand owners and producers to look at multiple life cycles as they are increasingly required to be responsible for products once they reach the end of their useful (first) life. Being made responsible for, or even accepting perpetual ownership of material streams, means that the priority may change and a way of quantifying this over multiple lifetimes will be required. This also becomes important in policy making where a simple comparison between stand-alone products is insufficient in determining impacts at a macro level.

Conceptually, this is shown in Figure 6 where a material is given a recycling rate – or in reality a type of *material use efficiency rate*, where the proportion is what ends up in another product of the same type (closed-loop). At one end of the scale, a reusable item of packaging (regardless of material) maintains 100% its material value with each subsequent reuse and is considered the epitome of circularity. On the other end of the

[17] Institut für Energie- und Umweltforschung Heidelberg (ifeu) (2017) *Comparative Life Cycle Assessment of Tetra Pak® carton packages and alternative packaging systems for liquid food on the Nordic market*, April 2017

scale, materials such as biodegradable plastics have their material value lost forever once disposed of. Other packaging materials will fit somewhere in between, with high yield materials such as aluminium maintaining a high level of circularity compared with many plastics. The graph shows that for a material that becomes a new product at a rate of 65% (higher than the EU plastic recycling target of 55% for 2030), the original virgin material only cycles enough to create the equivalent of two additional new products before it is essentially lost (i.e. only 65% of the material is retained after each loop so it quickly diminishes). In contrast, a product with a 90% recycling rate retains enough material to produce eight new products before the material is lost.

Figure 6: Material Efficiency – The Limits to Circularity

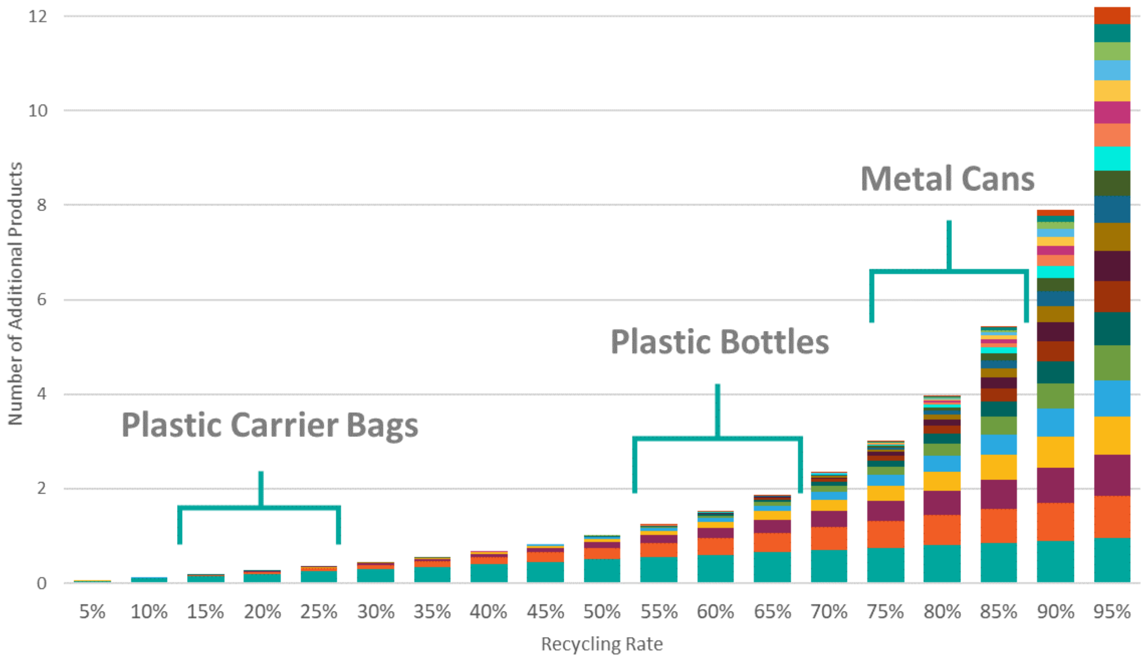
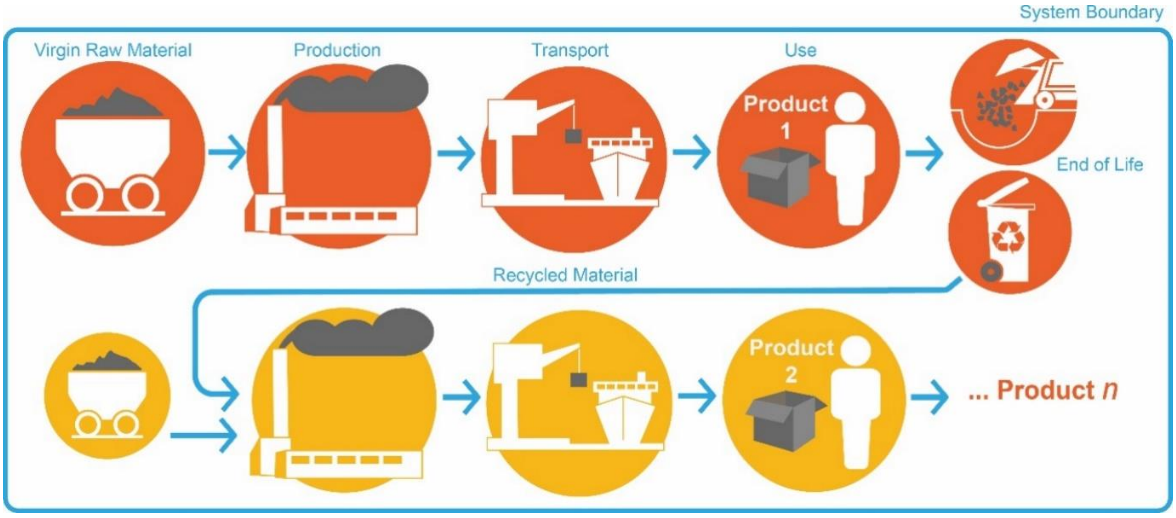


Figure 7 shows what this might look like if we expand the *system boundary* of an LCA study to include the subsequent 'lives' of the material. Less virgin raw material is required to produce Product '2' and this can continue until 100% virgin material is required to make the final product in the sequence. For reuse scenarios, no additional virgin raw material would be required and the production may be substituted by a cleaning operation. The leakage of material across the supply chain and during use could also be accounted for which would reduce the material efficiency. Where this circularity concept becomes particularly difficult to model is when material does not move around a closed loop (i.e. Product '1' is not the same as Product '2'), but cascades out into other open-loops, e.g. used PET bottles becoming polyester clothing. *Expanding the system boundary* of the LCA to include these cascades is both challenging to model and to accurately determine what is likely to happen as the material cascades further and further away from the initial product.

Figure 7: Expanding the LCA System Boundary for Material Circularity



Despite these challenges. this basic concept could be applied in LCA to show how much new (virgin) material is required for each subsequent life of a product. This could be used in several ways, including to determine:

- The best material for an application given the likely recycling rates and material yields in the current system
- Theoretically the best material for an application in an optimised system
- Whether optimised reuse systems outperform high recycling rate systems

Guidance for Study Readers

- Question whether the study results are still valid from a systems perspective.
- Has the study included some realistic future scenarios for recycling/ reuse and circularity or is it based on a current scenario.
- Has the author attempted to estimate material losses in recycling processes?

Guidance for Practitioners

- If the end of life appears to have a considerable influence on the result, perform a sensitivity analysis to determine whether realistic future changes to this would affect the result.
- Look to investigate how material circularity could be incorporated into the LCA in the form of an additional scenario to compliment the core analysis.

Guidance for Commissioners

- Ask the practitioner to build-in scenarios that increase the value of the study for third parties that will want to know that decisions taken now will still be valid in the future.

4.0 Plastic leakage – the Hidden Impact

One of the key criticisms with LCA in the context of plastic is that it is usually assumed that at the end of life the product will enter a formal waste stream and that there are no losses of the material throughout the lifecycle. It is true that conducting an LCA is always an act of simplifying a system to allow the assessment to take place, although it is often mistakenly misrepresented as an attempt to completely and accurately reflect reality, rather than an idealised scenario.

There is currently no recognised or agreed method for including the impacts of plastics leakage (or other materials) into the environment. The practitioner can assume a certain proportion will fall outside of formal waste streams—by littering, for example— but studying the flow of the product in the open environment and the consequences thereof, is beyond what is typically expected from an LCA study. This is particularly problematic for single use packaging where a certain amount of littering is almost inevitable. However, other leakage points throughout the lifecycle of all types of product—for example, the loss of the raw plastic pellets in the plastic supply chain—are not a core part of datasets for industrial processes and are therefore rarely considered. Arguably this is not something that LCA is equipped to or even should address for two reasons; firstly, the issue of pellet loss in particular is not a problem that can be generalised and therefore included in standardised datasets – it needs to be assessed and monitored throughout a supply chain and specific business activities will be highly variable. Secondly, whilst some emissions to the environment are an inevitable part of a process, this is not the case with pellet loss as it is a problem that can be mitigated and therefore not necessarily an inherent part of the plastics supply chain. This is essentially a market failure where the raw material is valued less than mitigation measures to reduce its loss. Nevertheless, this issue cannot be ignored and should form an integral part of any decision to use plastic in a particular application (alongside LCA results and other metrics). This is particularly important if the organisation specifying this material has no ability to verify or mitigate supply chain losses.

Several studies have attempted to include some form of qualitative or quantitative recognition of plastic leakage with specific regard to littering impacts. These vary from measuring the 'visual disamenity' or 'aesthetics' of littering[18,19] to the probability of escape and the resulting persistence of material in the environment.[20] Studies vary

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[18] Parker, G., and Edwards, Chris (2012) A Life Cycle Assessment of Oxo biodegradable, Compostable and Conventional Bags, *Intertek Expert Services*, p.46

[19] ExcelPlas Australia, Centre for Design at RMIT, and Nolan-ITU (2003) *The Impacts of Degradable Plastic Bags in Australia*, Report for Department of the Environment and Heritage (Australia), 2003

[20] Ecobilan (2004) *Évaluation des Impacts Environnementaux des Sacs de Caisse Carrefour*, Report for Carrefour, 2004

between whether the mass or the surface area is used as the defining factor,[21] but invariable some attempt is made to assess the length of time the material will persist in the environment. Many studies also entirely scope out littering altogether which may or may not be appropriate depending upon the goal of the study. The Danish EPA carrier bag comparison[22] determined that *"the effects of littering were considered negligible for Denmark and not considered"* and whilst there was no justification as to why littering either does not take place or that it results in no impact, the same cannot be said for all geographies. Indeed, exports of plastic waste will increase the risk of plastic leakage regardless of how developed the waste infrastructure is in the waste generating country. As previously discussed, care should be taken in presuming that assumptions made in one study are applicable to different circumstances (or even correct at all).

Progress in Developing a Plastic Leakage Indicator

Whilst there is no recognised or agreed method for including littering of plastics (or other materials), it is encouraged that this subject is at least discussed within the study with respect to any conclusions that are drawn. The Forum for Sustainability through Life Cycle Innovation have been discussing this issue for a number of years with the goal of developing a method for inventory of and life cycle impact assessment of plastic emissions into the marine environment. There still remains significant challenges in determining flows, sinks and impacts in a quantifiable way.[23] For LCAs that rely on secondary data in third party datasets (with the problems highlighted in Section 2.3) there is no clear way of integrating this; the problem is that the flow of plastic in particular, will very much depend on the type of product, and how and where it is used. The assumptions for this would be key and need to be flexible to allow for changes in design to influence the outcome. Typical waste datasets are ambivalent to the product type and focus on materials to determine impacts and often do not provide the flexibility to vary assumptions in the way that would be needed. There are also many other leakage points along the value chain that would need to be incorporated into existing datasets.

In terms of *impact* the distinction should be made around whether a *midpoint* or an *endpoint* impact category is developed. An example of a midpoint indicator is global warming potential (GWP) or climate change, measured in CO_2 equivalents. This categorises air emissions by their tendency to contribute to global warming compared with CO_2. An endpoint example would be to quantify the damage that climate change is likely to inflict on either human health or ecosystems—the consequences. Evidently,

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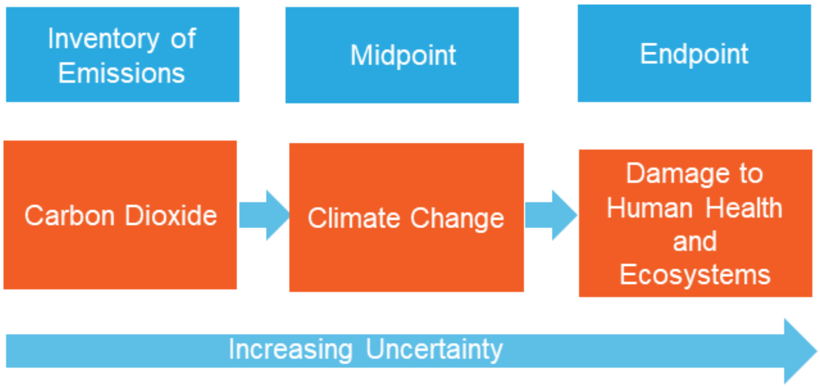
[21] ExcelPlas Australia, Centre for Design at RMIT, and Nolan-ITU (2003) *The Impacts of Degradable Plastic Bags in Australia*, Report for Department of the Environment and Heritage (Australia), 2003

[22] Bisinella, V., Albizzati, P.F., Astrup, T.F., and Damgaard, A. (2018) *A Lifecycle Assessment of Grocery Carrier Bags*, Report for The Danish Environmental Protection Agency, February 2018

[23] Philip Strothmann, Guido Sonnemann, Daniel Maga, and Nils Thonemann (2020) *Linking the Life Cycle Inventory and Impact Assessment of Marine Litter and Plastic Emission - Workshop-Report*, Report for Forum for Sustainability through Life Cycle Innovation e.V., March 2020

endpoints require more data, modelling and assumptions to produce and therefore introduce a layer of uncertainty in addition to midpoint calculations. This is why endpoints are used with caution and often not at all for public facing LCA studies; in the same way that it is difficult to predict the endpoint damage related to each kg of CO_2e, the same is true of plastic leakage into the environment.



It is also clear that regionality is likely to be even more important for plastic leakage than for many other impacts. Factors for leakage pathways will include the local waste infrastructure, wastewater treatment technology (for microplastics) and even local cultural or legislative drivers that might influence the propensity to litter. For example, a PET bottle in an EU country, after the forthcoming 90% collection target is implemented by 2029[24] , is less likely to become marine pollution than in a country where waste infrastructure is less well developed. The actual *impact* of the leakage of a bottle will also differ depending upon the proximity to water bodies and whether those water bodies contain species sensitive to (micro/nano) plastic. Furthermore, the evidence of impacts directly on human health as a result of ingesting fish that contain microplastics is limited and it appears that humans are actually far more likely to inhale plastic particles on a daily basis than ingest them through food.[25]

It is therefore likely, at least initially, that a midpoint indicator will be developed. This could take the form of a *plastic into the ocean equivalent* where materials and products are rated against this benchmark. This would rely on developing a methodology for rating plastic pollution types against each other – for example 1kg of microplastic could be 10kg of *plastic equivalents* if micro/nano plastic is found to result in a greater impact or 1kg of marine biodegradable plastic (if one could verify this in practice) might be 0.1kg of *plastic equivalent*.

Notably, as with other indicators, it will not allow a relative comparison *between* indicators e.g. does 1kg of *plastic equivalent* in the ocean have a greater impact than 1kg

[24] European Commission (2019) Directive (EU) 2019/904 on the reduction of the impact of certain plastic products on the environment

[25] Hann, S., Jamieson, O., Alice Thomson, and Sherrington, C. (2019) *Understanding Microplastics in the Scottish Environment*, Report for Scottish Environment Protection Agency (SEPA), November 2019

of *CO_2 equivalent* in the atmosphere? This will still rely on a decision being taken of the relative 'value' of an impact dictated by the goal of the study.

It is also important to recognise that by creating an impact assessment methodology for plastic, this implies that other materials have zero impact. This could lead to a situation where a non-plastic material alternative is chosen, rather than the focus being shifted towards reducing or mitigating leakage in the first place. This means that ideally, *plastic equivalents* should also be developed for some of the other common materials initially.

For incorporation into LCA, simplicity is likely to be the key. Whilst plastic can leak into the environment throughout a product life-cycle in all parts of the value chain, this is likely best addressed through one of the many tools that various organisations have developed with varying degrees of depth (and complexity).[26] Focusing on end-of-life plastic leakage should be the priority for LCA at least in the short term.

Proposal for a Simple Littering Impact Factor

In a 2016 a study by Eunomia for the EU Commission a simple method for comparing five materials commonly used to carry vegetables from a supermarket to households was tentatively proposed.[27] The study used two elements; the likelihood that a littered item will persist in both the marine and terrestrial environments (i.e. linked to biodegradability), and the mass (i.e. the amount that ends up in the environment).

More recently, an additional parameter of the likelihood of littering was proposed in a 2019 journal paper by *Civancik et al*[28], where the cost of the product is assumed to directly relate to the likelihood of littering when used in the example of carrier bags i.e. valuable items are littered less. However, the relationship is unlikely to be linear and—as the authors also acknowledge—other factors such as the effectiveness of the local waste management system will also have a bearing on the likelihood of littering. Cost may also not be appropriate for other products that the consumer places zero value on, which would include the majority of packaging. Littering rates could be a potential substitute, but this type of data is surprisingly difficult to find particularly for different countries. The important factor is to determine whether there is likely to be a significant difference in littering rate between the products being compared; if not, then the parameter can be removed altogether. This could be achieved for comparisons between single use items and reusable items, by using the likely reuse rate e.g. if 100 reuses are expected, it is 100x less likely to be littered. Nevertheless, cost is still a useful starting point as data can be obtained relatively easily and it would also work in reuse scenarios where cost to the

[26] Julien Boucher, Carole Dubois, Anna Kounina, and Philippe Puydarrieux (2019) *Review of plastic footprint methodologies*, Report for IUCN, 2019

[27] Eunomia Research & Consulting (2016) *Study to assist the Commission to carry out a life cycle impact assessment of different possibilities to reduce the consumption of very lightweight plastic carrier bags*, Report for European Commission, July 2016

[28] Civancik-Uslu, D., Puig, R., Hauschild, M., and Fullana-i-Palmer, P. (2019) Life cycle assessment of carrier bags and development of a littering indicator, *Science of The Total Environment*, Vol.685, pp.621–630

consumer would be a differentiator. Equally, other incentives such as deposit systems could also be built into the cost.

An environmental dispersion factor was also included, based on the mass of the object. This was also proposed in a French carrier bag LCA for Carrefour in 2004.[29] The mass of a product is likely to be a relatively crude indicator for this, given other influencing factors, especially whether a material is hydroscopic (water absorbing e.g. paper, PET, and some compostable plastics) or hydrophobic (water repelling e.g. conventional plastics such as PE and PP). Absorbing water would increase the mass which would reduce the risk of dispersal. Again, we can recognise that this input factor is not perfect at present, but may be improved in the future.

A further difference between the two methods is whether to focus on mass or surface area of the product as part of the calculation. Surface area is likely to be the biggest influence in the visual impact of a littered item and in its likelihood to act as an accumulator and vector for other pollutants, but the mass may have a direct effect on persistence (i.e. the item could be twice as thick with the essentially the same surface area). It is therefore recommended to choose one and perform a sensitivity analysis on the other to identify if this affects the order of results significantly.

One aspect that the Eunomia Study included, but was omitted by Civancik, were the inclusion of two persistence factors; one for terrestrial and one for marine litter. The reasoning for including both is that biodegradation is much more consistent in soil compared with the marine environment. The lack of fungi in the latter and the fact that the conditions vary considerably at depth means that uncertainty is increased and different materials can behave differently, i.e. one might biodegrade in soil, but not in sea water. However, any biodegradation factors are likely to be the most uncertain of any littering impact calculation as there is no scientific consensus for measuring this in open environments (and a strong argument for why biodegradability should not be considered as a solution to leakage). This would largely rely on the LCA practitioner to research this subject, which will most likely fall outside of their expertise to judge which scientific literature to refer to. One way to address this is to use ranges from the literature rather than a specific single figure which would go some way towards dealing with the uncertainty in this field.

Any persistence factor that uses biodegradation time also does not take into account any harm caused during or after this process. In reality, biodegradable plastics are likely to be subjected to ecotoxicity testing through existing standards (for example, EN 13432 for composting and EN 17033 for mulch films in soil) and therefore would perhaps present a lower risk in the environment compared to conventional plastics which have not. Again, accounting for this would add an additional layer of complexity that is not well understood at present.

[29] Ecobilan (2004) *Évaluation des Impacts Environnementaux des Sacs de Caisse Carrefour*, Report for Carrefour, 2004

The important aspect of both the Eunomia and Civancik indicators is that they do not calculate absolute impacts, but relative impacts between the comparative products in the study. This means the choice of products affects the outcome, and the results cannot be compared with other studies. This is an acceptable outcome for an indicator that is simple to apply whilst more sophisticated methodologies are developed; and whilst there are many complex factors that cannot be incorporated at this time, this should not be a barrier to including such an indicator with knowledge as it stands presently.

Based on the above discussion an adaptation to *Civancik et al*'s equation is proposed by including an additional persistence factor. This modified equation is shown in Figure 8. Depending upon the circumstances, *P*5 could be a marine or riverine factor, with the latter used in landlocked countries. Equally, each one of the parameters could include additional factors that are considered to be important in the particular study. A further explanation of how this equation could be applied can be found in Appendix A.2.0.

Figure 8: Littering Impact Potential Equation

$$\text{LiP} = \frac{P1}{P2 \times P3 \times P4 \times P5}$$

LiP = Littering Impact Potential
*P*1 = Quantity
*P*2 = Release Factor
*P*3 = Dispersion Factor
*P*4 = Terrestrial Persistence Factor
*P*5 = Aquatic Persistence Factor

Guidance for Study Readers

- Increasingly, a lack of methodology is not enough for a practitioner to exclude all reference to littering impacts particularly for single-use items; this paper presents one such way that can be used as an interim measure whilst further research is conducted into creating a formalised indicator.

Guidance for Practitioners

- Try to be realistic with end of life scenarios – don't assume 100% of the product will end up in formalised waste streams, and make sure to verify waste statistics for their accuracy as there is a general tendency towards overestimates for recycling.
- If exports are the predominant route for the waste, it is important to question the true fate of the materials and be conservative with recycling assumptions.
- If there is no way to quantify the impact of littering within the assessment itself, do not exclude it entirely by assuming the product enters other

formalised waste streams – at very least, reduce these accordingly (e.g. less recycling)

- Incorporate a littering indicator such as the one described in this paper to help – this could be used as a basic framework and adapted to the study goals.
- Explicitly identify the forms of plastic pollution into both marine and terrestrial environments (e.g. littering, pellet loss, losses from exports) even if these cannot be quantified. This will allow decision-makers using the LCA to be aware of the limits of using this tool alone, and the need to complement it with estimates for these missing elements.

Guidance for Commissioners

- Ask to practitioner to include plastic leakage in the scope of the study and include time and budget to assess this with the appropriate amount of rigour given the state of knowledge and methods available at the time.

APPENDICES

A.1.0 Environmental Footprint Impact Categories

EF Impact Category	Impact category Indicator	Unit	Source	Level[1]
Climate Change	Radiative forcing as global warming potential (GWP100)	kg CO2 equivalent	Baseline model of 100 years of the IPCC (based on IPCC 2013)	I
Ozone Depletion	Ozone Depletion Potential (ODP)	kg CFC-11 equivalent	Steady-state ODPs as in (WMO 2014 + integrations)	I
Ecotoxicity for aquatic fresh water	Comparative Toxic Unit for ecosystems	CTUe	USEtox model 2.1 (Fankte et al, 2017)	III
Human Toxicity - cancer effects	Comparative Toxic Unit for humans	CTUh	SEtox model 2.1 (Fankte et al, 2017)	III
Human Toxicity – non-cancer effects	Comparative Toxic Unit for humans	CTUh	SEtox model 2.1 (Fankte et al, 2017)	III
Particulate Matter/Respiratory Inorganics	Impact on human health	disease incidence	PM method recommended by UNEP (UNEP 2016)	I
Ionising Radiation – human health effects	Human Health effect model	kg U235 equivalent	Dreicer et al., 1995	II
Photochemical Ozone Formation	LOTOS-EUROS model	kg NMVOC equivalent	Van Zelm et al., 2008 as applied in ReCiPe	II
Acidification	Accumulated Exceedance model	mol H+ eq	Seppälä et al.,2006; Posch et al., 2008	II
Eutrophication – terrestrial	Accumulated Exceedance model	mol N eq	Seppälä et al.,2006; Posch et al., 2008	II
Eutrophication – aquatic	EUTREND model	fresh water: kg P equivalent marine: kg N equivalent	Struijs et al., 2009 as implemented in ReCiPe	II
Water Use	User deprivation potential (deprivation weighted water consumption)	m3 world eq	Available Water Remaining (AWARE) as recommended by UNEP, 2016	III
Resource use, fossils	Abiotic resource depletion (ADP ultimate reserves)	MJ	van Oers et al., 2002	III
Resource Use minerals and metals	Abiotic resource depletion (ADP ultimate reserves)	kg antimony (Sb) equivalent	van Oers et al., 2002	III
Land Use	• Soil quality index • Biotic production • Erosion resistance • Mechanical filtration • Groundwater replenishment	• Dimensionless (pt) • kg biotic production • kg soil • m3 water • m3 groundwater	Soil quality index based on LANCA (Beck et al. 2010 and Bos et al. 2016)	III

1. Levels: “I” (recommended and satisfactory), level “II” (recommended but in need of some improvements) or level “III” (recommended, but to be applied with caution).

A.2.0 Littering Impact Indicator Calculations

The basis calculation methodology is provided in Table 1. This builds and adapts upon *Civancik et al*[30] where a more detailed methodology can be found.

Table 1: Littering Impact Indicator Calculations

Parameter		Indicator	Calculation[1]
P1	Quantity of product to fulfil function	Surface area or Mass per functional unit	**= p / p max** p= value p max = max value of all options
P2	Probability of release into the environment	Price (€/$)	
P3	Probability of dispersion throughout the environment	Mass per item (kg)	
P4	Persistence in the marine/riverine environment	biodegradation per day[2]	
P5	Persistence in the terrestrial environment	biodegradation per day[2]	
1. Each calculated indicator is divided by the maximum result of all options to provide a relative impact; a factor of 1 is given to the highest figure and all other options are <1.			
2. Biodegradation rates of materials can often be found in the form of a threshold that is reached (typically 90%) in **X** days. Divide the biodegradation observed by the number of days the test ran for e.g. 90% in 60 days = 0.9/60 = 0.015. This approximates, but does, however, simplify what is not a linear process in reality.			

[30] Civancik-Uslu, D., Puig, R., Hauschild, M., and Fullana-i-Palmer, P. (2019) Life cycle assessment of carrier bags and development of a littering indicator, *Science of The Total Environment*, Vol.685, pp.621–630