LABPLAS

Horizon 2020
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for Research & Innovation

Land-Based Solutions for Plastics in the Sea

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D5.7. LCA Report

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2	UNIVERSIDADE DA CORUÑA	UDC	SPAIN	HES
3	Bundesanstalt fuer Gewaesserkunde	BfG	GERMANY	RTO
4	LABORATORIO IBERICO INTERNACIONAL DE NANOTECNOLOGIA	INL	PORTUGAL	RTO
5	KATHOLIEKE UNIVERSITEIT LEUVEN	KUL	BELGIUM	HES
6	HELMHOLTZ ZENTRUM FUR OZEANFORSCHUNG KIEL	GEOMAR	GERMANY	RTO
7	NATIONAL OCEANOGRAPHY CENTRE	NOC	UNITED KINGDOM	RTO
8	SORBONNE UNIVERSITE	SU	FRANCE	HES
9	OPEN UNIVERSITEIT NEDERLAND	OUNL	NETHERLANDS	HES
10	LEIBNIZ INSTITUTE FOR BALTIC SEA RESEARCH	IOW	GERMANY	RTO
11	ASSOCIACAO PARA O DESENVOLVIMENTO DO ATLANTIC INTERNATIONAL RESEARCH CENTRE	AC	PORTUGAL	RTO
12	UNIVERSIDADE FEDERAL DO SAO PAULO	UNIFESP	BRAZIL	HES
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14	TG ENVIRONMENTAL RESEARCH	ER	UNITED KINGDOM	SME
15	CONTACTICA S.L.	CTA	SPAIN	SME
16	STICHTING EGI	EGI	NETHERLANDS	Non-P
17	STICHTING RADBOUD UNIVERSITEIT	RU	NETHERLANDS	HES
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Executive summary:	This document corresponds to D5.7, the LCA report resulting from subtask 5.5.2 of the LABPLAS project. Its main aim is to use the Life Cycle Assessment (LCA) methodology to compare the environmental performance of conventional (fossil-based and non-biodegradable) plastics with biodegradable alternatives. To achieve this, two case studies were proposed, focusing on mulch film and garbage bags, to assess their respective environmental footprints from a cradle-to-grave perspective. The results indicate that biodegradable mulch film has the lowest environmental impact in the categories of photochemical ozone formation, particulate matter, resource use and toxicity-related categories. However, in the remaining categories, it shows higher impacts than conventional plastic due to the impacts associated with PBAT production. Focusing on the EoL phase, the biodegradable alternative appears to be a better option. Regarding garbage bags, paper bags show the worst environmental performance due to the energy intensity during their production. Although conventional plastic bags have a lower impact during production, their EoL phase poses a problem, as microplastics tend to accumulate in the soil. Additionally, their EoL alternatives have a higher impact than biodegradable options like composting or anaerobic digestion. For both cases, the impact of

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	microplastics has been analysed through the categories of *physical effects on biota* and *terrestrial ecotoxicity*. More efforts should be directed toward developing reliable inventory datasets for the effects of microplastics. This is crucial, as plastic microparticles have been demonstrated to significantly impact the environmental assessment of a product's life cycle.

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ABBREVIATIONS AND ACRONYMS

Abbreviation / Acronym	Description
BMB	Biomass Balance
CF	Characterisation Factor
CFF	Circular Footprint Formula
CTU	Comparative Toxic Unit
CTUe	Comparative Toxic Unit for Ecosystem
CTUh	Comparative Toxic Unit for Human
D	Deliverable
EEF	Exposure and Effect Factor
EF	Environmental Footprint
EoL	End-of-life
EU	European Union
EUBA	European Bioeconomy Alliance
EUBP	European Bioplastics
FF	Fate factor
FU	Functional Unit
GHG	Greenhouse gas emissions
iLUC	Indirect Land Use Change
JRC	Joint Research Centre
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
LDPE	Low-density polyethylene
LR	Loss rate
LUC	Land Use Change
N/A	Not applicable
PAF	Potentially affected fraction of species

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PBAT	Polybutylene adipate terephthalate
PEF	Product Environmental Footprint
PLA	Polylactic acid
$RedR_{frw\_ocean}$	Redistribution rate from freshwater to the ocean
$RedR_{terenv\_terenv}$	Redistribution rate from terrestrial environment to another terrestrial environment
$RelR_{frw}$	Release rate to freshwater
$RelR_{terenv}$	Release rate to terrestrial environment
SMNP	Small micro- and nano-plastics
WP	Work package

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1 INTRODUCTION

1.1 Aim of this document

Nowadays, it is estimated that about 5,250 billion plastic particles are floating on the surface of the seas and oceans around the world. Once in the ocean, these particles fragment into smaller ones and move with the currents and ocean gyres before washing up on the coastline. They are hardly removed from the ocean, so proactive action regarding research on plastic alternatives and strategies to prevent plastic from entering the environment should be taken promptly. Despite the research increasing, there is still a lack of suitable and validated analytical methods for the detection and quantification of small micro- and nano-plastics (SMNP) evidencing a huge obstacle for large-scale monitoring.

In this sense, LABPLAS is a 48-month project whose vision is to create capacities to evaluate rapidly and precisely the interactions of plastics with the environmental compartments and natural cycles leading to the development of effective mitigation and elimination measures, as well as making management decisions. It will assess reliable identification methods for more accurate assessment of the abundance, distribution and toxicity determination of SMNP in the environment. It will also develop practical computational tools that should facilitate the mapping of plastic-impacted hotspots and promote scientifically sound plastic governance.

This document corresponds to D5.7. LCA Report resulting from Subtask 5.5.2 of the LABPLAS project. The primary objective of this study is to use the Life Cycle Assessment (LCA) methodology to compare the environmental performance of conventional (fossil-based and non-biodegradable) plastic with biodegradable alternatives. Two case studies have been proposed, focusing on mulch film and garbage bags, to assess their respective environmental footprints from a cradle-to-grave perspective. This analysis enables the identification of the most environmentally friendly options, as well as the hotspots with significant environmental impacts. Therefore, potential recommendations for enhancing the environmental performance of both products can be proposed based on these findings.

Moreover, this report builds on the results of the LABPLAS project to address current methodological gaps in LCA and validate its applicability in the context of environmental assessment of plastics.

1.2 Context

LCA is a methodology that allows for the analysis of all inputs and outputs associated with a product, process or service, resulting in its environmental profile. This methodology is based on the systematic collection of data to establish objective criteria for decision-making in a company or production process environmental strategy. In addition, it also helps to identify critical points in the system, which are those stages of the process that have a greater environmental impact, providing a reference framework for defining improvement actions throughout the system (ISO, 2006a, 2006b).

Consequently, it enables the quantification of the potential environmental burdens associated with a product, process or activity by accounting for and analysing the materials and energy resources consumed and the pollutant emissions associated with the system under study. There are four interrelated phases in every life cycle study, which are presented in **Figure 1.**

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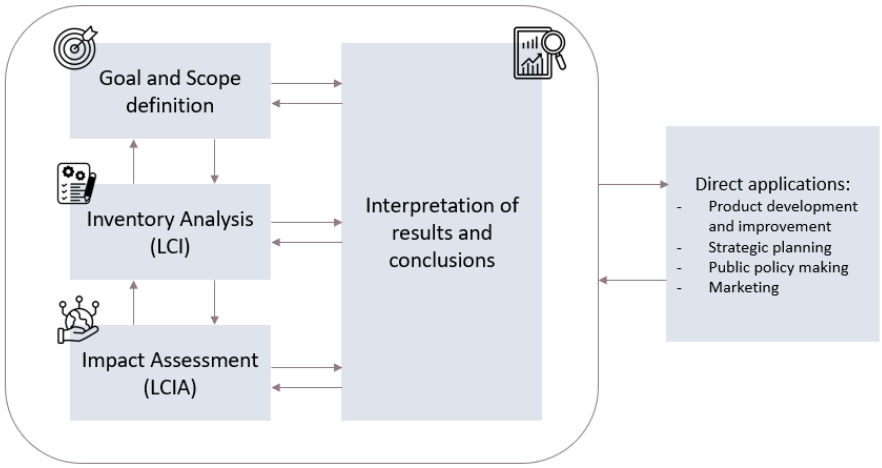

Figure 1. Steps to perform an LCA. Adapted from ISO 14040:2006 (ISO, 2006a).

Goal and Scope Definition

In this stage, the reasons for conducting the study and the intended application are established. It is essential to define the Functional Unit (FU), system boundaries and inclusion and/or exclusion criteria for inputs and outputs, which will be used for the environmental inventory and impact analysis.

Life Cycle Inventory Analysis

The inventory analysis involves a qualitative and quantitative description of the unit processes required to model the life cycle stages of the system to be evaluated. It primarily involves a material and energy balance, accounting for all inputs and outputs.

Data should be systematically collected and, if possible, standardized. This includes the use of information obtained directly from a producer (primary data), apart from the use of averages from multiple industry databases or extrapolation using current values and future projections (secondary information). Data validation is also performed by identifying data sources and handling any excluded data. Additionally, allocation rules are defined, along with their justification for application, and scenarios considered.

Life Cycle Impact Assessment

Life Cycle Impact Assessment (LCIA) interprets the inventory by analysing the produced impacts and their environmental burdens. For LCA, LCIA encompasses four elements, of which only the first two are mandatory:

- Classification: Environmental burdens of the system are categorized into different impact categories based on the type of expected environmental effect.
- Characterisation: Using Characterisation Factors (CF), environmental indicators are assigned to impact categories.
- Normalisation: Characterisation results are compared to a reference value to assess their significance.

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- Weighting: Values for different impact categories are weighted and aggregated to obtain a single environmental index.

➲ **Interpretation of Results**

Finally, an analysis of the obtained results and the quality of the data is conducted. During the interpretation phase, the selection of values, ratios, or statements to communicate is made, which varies depending on the objectives. Decisions are made based on previous results, allowing for the identification of the most impactful processes or inputs. This identification enables the proposal of mitigation measures or recommendations to facilitate a better decision-making process.

According to ISO 14040 and 14044 regulations, various institutions and organisations are focused on developing specific methodologies for the application of LCA in different contexts. Regarding plastics, a notable example is the Plastics LCA method developed by the Joint Research Centre (JRC) of the European Commission to assess the environmental impact of plastic products throughout their life cycles. The Plastics LCA method provides detailed methodological and modelling rules to conduct LCA studies of plastic products based on a common and harmonised framework, to enable as much as possible reproducible, consistent, robust and verifiable studies at the EU (European Union) level. While the focus is especially on plastic products relying on different feedstocks (fossil and bio-based), the method also applies to products with different biodegradability properties such as compostable plastics (Nessi et al., 2021).

This methodology was chosen as a reference point for the development of the LCA methodology of the LABPLAS project due to its alignment with the Product Environmental Footprint (PEF) framework but making adaptations to address concerns raised by the European Bioeconomy Alliance and the European Bioplastics (EUBA & EUBP, 2021), as described in D5.6 *LCA Methodology from Subtask 5.5.1*. All the differences and adaptations made from the Joint Research Centre (JRC) Plastics LCA method were described in D5.6, but a summary is presented in Table 1 to contextualise what is taken into consideration as the primary guide for the preparation of this report.

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Table 1. Contribution from the LABPLAS project based on the JRC Plastics LCA methodology issues, as well as EUBA and EUBP concerns. Adapted from Table 2 of D5.6.

Issue	JRC Plastics LCA methodology	Contribution from the LABPLAS project
Carbon sequestration	CFs for biogenic CO_2 uptake and emissions are set to zero in the impact assessment, so undermines the key advantage of bio-based products, which is to remove carbon from the atmosphere and sequester it into products.	N/A
Comparison of mature and immature production systems	The methodology does not provide a guide on how to compare different maturity systems to avoid biased and unfair comparisons.	Both products, non-biodegradable and biodegradable, were assessed using primary data considering the same level of maturity.
Data reporting requirements	Data quality requirements are higher for bio-based systems compared to fossil-based systems. While industrial average datasets are acceptable for fossil-based systems, much more detailed inventory data is required for bio-based products.	Primary data obtained from LABPLAS project partners was prioritized. If it was not available, secondary data came from studies and statistics with the same geographical location, scope and that were updated within the last 3 years.
Incorporation of Land Use Change (LUC)	LUC must be included for bio-based systems while for fossil-based, this shall be accounted for but are typically not captured in LCA. Furthermore, soil carbon uptake needs to be considered in calculations and not just added as additional information.	The JRC Plastics LCA guidelines were followed, which use the EF impact assessment method. Therefore, LUC was considered in the selected datasets within the climate change indicator, which comprises up to three different sub-indicators: fossil, biogenic and, land use and land use change.
Incorporation of indirect effects	Indirect LUC (iLUC) is only considered as additional information. For bio-based is recommended to be included, while for fossil-based plastics is explicitly ruled out.	Given the lack of robust data and methods to account for indirect effects, in the LABPLAS project, LCA was performed with an attributional approach without considering indirect effects (i.e., consequential approach).
Requirements for providing proof	No proof is required for negative indirect effects of bio-based products, while proof is required for positive effects.	N/A

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Table 1 (cont.). Contribution from the LABPLAS project based on the JRC Plastics LCA methodology issues, as well as EUBA and EUBP concerns. Adapted from Table 2 of D5.6.

Issue	JRC Plastics LCA methodology	Contribution from the LABPLAS project
Biodiversity impacts	Biodiversity impacts are only provided as additional information due to a lack of accepted methodology. Consequently, effects on biodiversity due to leakage of plastics into the environment are missing.	Biodiversity impact assessment methods are not currently addressed by the EF method, so additional information was provided. Therefore, a CF has been included to consider the toxicity of microplastics released into the environment. This factor is based on toxicity data of plastics in marine and terrestrial compartments collected by the UVigo ECOTOX and GEA groups as part of the WP5 LCA task.
Reflecting end-of-life (EoL) realities	LCA should be performed on waste stream level, rather than on product level to capture the real value of industrial composting in comparison to other EoL options. In JRC Plastics LCA, the EoL realities are described using the Circular Footprint Formula (CFF) parameters listed in the annex of the guide.	EoL options are proposed. This stage was assessed at a waste stream level to consider the real value of EoL options (i.e. industrial composting).
Normalisation and weighing	Normalisation and weighting are mandatory steps. However, EU is only considered and weighting factors are not updated as the reference year is 2018.	Normalisation and weighing were implemented using the factors included in the Environmental Footprint (EF) method. The comparison between products was performed at the characterisation level.
Feedstock supply data requirements	Sustainable feedstock sourcing needs to be included or at least attached to it, especially in those comparisons where the major differences are coming from the feedstocks themselves.	N/A

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In summary, the need for methodological adjustments to mitigate the effects of gaps related to disparities in data availability and quality to accurately conduct the comparative LCA has been recognised. In this regard, project-specific datasets have been used to treat the impacts of plastics as closely as possible to reality. Therefore, CFs were developed considering the toxicity and biodegradability of microplastics to assess their impacts when released into different compartments (i.e., soil and marine environment). These methodological aspects will be addressed in more detail throughout the deliverable.

1.3 Report structure

This document comprises two separate case studies. The first case study consists of a comparative LCA of mulching film, considering conventional and biodegradable plastic (**Section 2**). The second case study examines a comparative LCA of different types of garbage bags, including conventional plastic, biodegradable plastic and paper (**Section 3**). Each case study follows a systematic process, as previously mentioned, adhering to the ISO 14040/14044 standards, starting with goal definition, progressing to inventory data collection, followed by environmental impact calculation, and concluding with the interpretation of results.

2 COMPARATIVE LCA FOR MULCH FILM

2.1 Goal and Scope

2.1.1 Goal Definition

The purpose of the study was to compare the environmental impacts caused by two types of agricultural mulch film materials used for tomato cultivation, using a cradle-to-grave approach aligned with the PEF methodology. The analysis included the different biodegradability of materials and final disposal options. Specifically, it evaluated the environmental impact of soil biodegradation for a biodegradable material and compared it to open burning and littering processes for a conventional material (i.e., fossil-based and non-biodegradable). This assessment aimed to understand how the choice of final disposal options influences their potential environmental impacts.

The ultimate goal was to determine the most environmentally friendly product and identify the stages and categories that significantly contribute to environmental impacts. This understanding allows for informed decision-making and the implementation of appropriate measures to mitigate environmental impacts, optimize resource utilisation and enhance sustainability throughout the life cycle, including plastic leakage during the final phase of its life cycle.

Finally, in terms of the target audience, the conclusions regarding the applied methodology were addressed to LCA practitioners, while the conclusions regarding the comparison of the two product systems were addressed to industry stakeholders, policymakers and the general public.

2.1.2 Scope

2.1.2.1 Product system

In this case study, the potential impacts of conventional and biodegradable mulching film alternatives were quantified by modelling two separate LCA scenarios. The most common alternative currently used for large-scale agriculture is non-biodegradable low-density polyethylene (LDPE) mulch film, which was considered as

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a first scenario, assuming that polymer production is entirely based on virgin material from fossil-based feedstock.

In the second scenario, a biodegradable mulch film alternative was proposed. In this case, the raw material used for its production was polybutylene adipate-*co*-terephthalate (PBAT), which was derived from 1,4-butanediol, adipic acid and terephthalic acid, all of them obtained from fossil-based sources. Additionally, polylactic acid (PLA), which was obtained from starch, was integrated into the blend. The proportion of biogenic carbon in the final blend is 9%.

The characteristics of each product, including thickness and density, are presented below in **Table 2**. Note that from here on "conventional" refers to the non-biodegradable product, while "biodegradable" to the biodegradable alternative.

Table 2. Mulch film characteristics for each scenario.

Parameter	Unit	Conventional	Biodegradable
Material composition	-	LDPE	PBAT PLA
Source (bio/fossil)	-	Fossil	Fossil Bio-based
Thickness	µm	15	10
Density	$kg \cdot m^{-3}$	910	1,250

2.1.2.2 Functional Unit

The Functional Unit (FU) serves as the reference unit for quantifying system performance. Meaningful comparisons can only be established when products serve the same function, so this must comprehensively describe the functions of the products, both qualitatively and quantitatively.

Mulch film is a plastic film designed to protect and improve the quality of crops by regulating soil temperature (maintaining or increasing), limiting weed growth and preventing water loss by retaining moisture in the soil and roots. Therefore, the FU was defined as follows: "Mulch film needed to cover 1 hectare of agricultural land cultivated with tomatoes". The questions to be filled by the FU are presented in **Table 3**.

Table 3. FU description for mulch film.

Question	Conventional	Biodegradable
What?	Utilisation of 136.5 kg of plastic mulch in tomato fields	Utilisation of 125 kg of plastic mulch in tomato fields
How much?	1 hectare	
How well?	Protecting tomato crops and improving final quality by controlling weeds, retaining soil moisture and regulating soil temperature	
How long?	For one entire growing season	
Where?	European context (EU with 28 member countries)	

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Regarding the reference flow, it is defined as the amount of polymer required to fulfil the functional unit for each scenario. It can be calculated based on the density of the film material and the volume of de film, which depends on the thickness of the film and the net area of land to be mulched (**Table 2**).

In addition, it is worth noting that the study was conducted within a European context (EU with 28 member countries); but more specifically, the polymers and mulch film were produced in Germany and subsequently exported for utilisation in Spain. In cases where specific datasets were unavailable, a European average served as an alternative.

2.1.2.3 System boundaries

The system boundaries define which life cycle stages and processes belong to the system under analysis. In this case, the system boundaries adopted a **cradle-to-grave approach**. This means that the entire product life cycle is covered, from raw material acquisition to product manufacturing, distribution, use phase and final disposal of the product. For each life cycle stage, a thorough analysis of the results was conducted, taking into account their respective inputs and outputs, including waste generation and treatment.

Conventional and biodegradable mulch films share the same life cycle stages, differing primarily in the raw materials used and EoL treatment. The life cycle stages and processes considered are detailed below and visually represented in the system boundary diagrams presented in **Figure 2** for both products.

- **Polymer production**: Initially, this stage includes the acquisition of raw materials. In the case of fossil-based monomers, this encompasses activities such as extraction, processing, transportation and refining of crude oil and natural gas. On the other hand, bio-based materials involve the cultivation of crops, forestry and harvesting. Subsequently, the respective monomers are converted into long-chain polymers through a process known as polymerisation. In addition, the transportation of materials involved in these activities was also considered.
- **Manufacturing**: This stage covers the extrusion of polymer granulate to produce the final products. Extrusion consists of melting the polymers and forming them into a thin and flexible film. This film is then cooled and solidified to obtain the desired thickness and strength.
- **Distribution**: It refers to the transportation of products from manufacturing sites located in Germany to consumers in Spain. It was assumed that they were transported in a lorry of the size class >32 metric tons.
- **Use**: This stage is considered when it involves energy and resource consumption or emissions to the environment during the use of the products. In this regard, the consumption of water, fertilizers and other chemicals used during the cultivation of 1 hectare of tomatoes was considered.
- **End of Life (EoL)**: This stage covers activities related to the collection, transport and treatment of the disposal products. For conventional mulch film, it was assumed that half of it is open-burned, while the other half is littered in soil. In contrast, the biodegradable mulch film was assumed to naturally biodegrade in soil.

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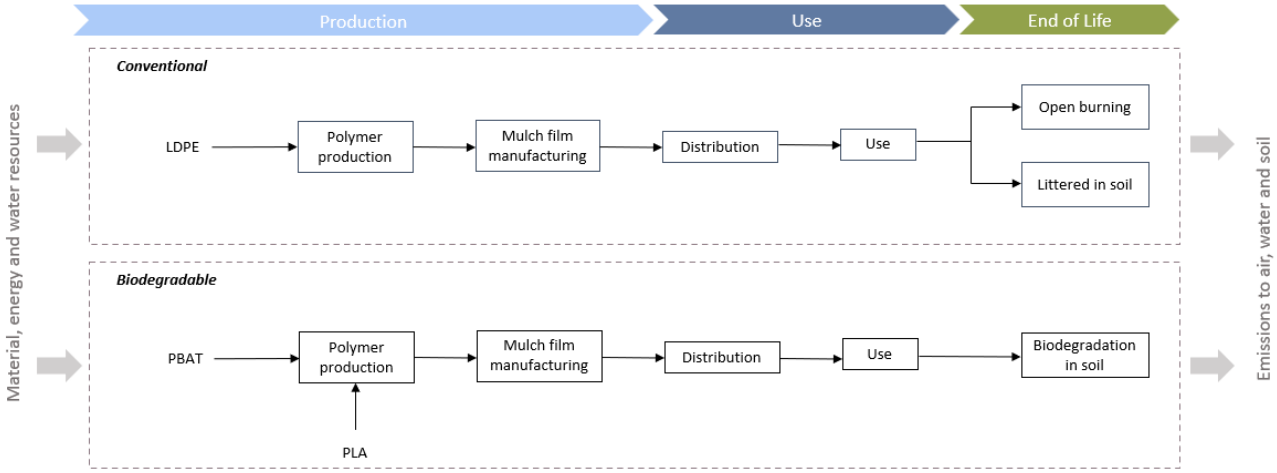

Figure 2. System boundaries of conventional and biodegradable mulch film.

The system boundaries are designed to include all processes and activities linked to these stages, excluding only those processes or activities based on a cut-off rule. Therefore, capital goods such as plant construction and the purchase of machinery and means of transport are not included.

Finally, it is important to note that plastic waste released into nature experiences fragmentation, leading to the formation of macroplastics, microplastics and nano-plastics within soil, air and water environments. Therefore, this study addressed the often-overlooked consequences of microplastic leakage during EoL phases, incorporating this aspect as supplementary information. To do so, both the biodegradability and toxicity effects of plastic have been considered to calculate their impacts on biodiversity. Through this inclusion, a more comprehensive understanding of microplastic pollution and its ecological implications was achieved.

Regarding the conventional mulch film that is left in the soil, unlike its biodegradable counterpart where residual material was assumed to remain in the soil, consideration was given to the potential displacement of microplastics by different types of transfers such as leaching, transport in freshwater bodies or wind blowing after deposition on the soil. Consequently, since primary data were unavailable, the leakage of macroplastics from conventional mulch film across environmental compartments was modelled following the Plastic Leak Project (PLP) methodological guidelines (Peano et al., 2020), as described below.

Leakage refers to the phenomenon where these plastics are lost or released into the environment through a process involving transfer and redistribution. To quantify the rate at which macroplastics enter the aquatic or terrestrial environments, **Equation 1** and **Equation 2** were used, respectively. These equations consider several variables including the loss rate (LR), initial release rate (RelR) and redistribution rates (RedR).

- LR serves as a metric for plastic that leaves a properly managed product or waste management system. It was proposed that 50% of the conventional mulch film remained uncollected, leading to littering in the soil.
- RelR measures the amount of plastic transported to an initial environmental compartment. Several parameters can influence the release rate, such as the cultural background, climatic conditions or waste characteristics, but due to limited quantitative studies and models, PLP guidance only considered waste

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residual value and size to calculate release rates. Therefore, two options were considered: release to freshwater ($RelR_{frw}$) and release to other terrestrial environments ($RelR_{terenv}$).

- RedR quantifies the plastic redistributed to a final environmental compartment. It was assumed that all plastic released into freshwater ultimately reaches the ocean ($RedR_{frw\_ocean}$), while plastics released into terrestrial environments remain there ($RedR_{terenv\_terenv}$).

$$Leak\ macro\ ocean = LR \cdot RelR_{frw} \cdot RedR_{frw\_ocean} \quad \textbf{Equation 1}$$

$$Leak\ macro\ soil = LR \cdot RelR_{terenv} \cdot RedR_{terenv\_terenv} \quad \textbf{Equation 2}$$

Bearing the abovementioned in mind, the data obtained using PLP guidance are shown in **Table 4**.

Table 4. RelR and RedR for ocean and soil compartments.

	Ocean	Soil
RelR	5%	95%
RedR	100%	100%

In summary, when mulch film is left on the ground, 95% of the resulting macroplastics remain within the soil, while the remaining 5% may be transported to a freshwater compartment before ultimately being redistributed to the ocean. Finally, the formation of microplastics is the result of both the leakage of macroplastics and the subsequent fragmentation rate.

2.1.2.4 Impact assessment method and selected impact categories

The impact assessment method selected was the EF method in its last version (3.1) included in the PEF guidelines (European Commission et al., 2018). The EF 3.1 is widely recognised and utilised for assessing the environmental performance of products and systems, and additionally, it is recommended by the JRC guidelines Plastics LCA for comparative LCA for plastics production (Nessi et al., 2021).

This method encompasses 16 environmental impact categories such as climate change, water use and acidification potential, among others, to assess the negative (or positive) effects on water, terrestrial and air environments, as well as human health. By incorporating these indicators, the method provides a comprehensive assessment of the environmental impacts associated with the life cycle of the products. Finally, Simapro (version 9.5) was used as the software tool (Prè-Product Ecology Consultants, 2020).

The impact categories in the EF 3.1 method applied in the study are presented in **Table 5.**

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Table 5. Description of impact categories of the EF 3.1 method.

Impact category	Unit	Description
Climate change[1]	kg CO_2 eq.	Increase in the average global temperature resulting from greenhouse gas emissions (GHG)
Ozone depletion	kg CFC-11 eq.	Depletion of the ozone layer protecting from ultraviolet radiation
Human toxicity – cancer, non-cancer	CTUh	Impact on human caused by absorbing hazardous substances. Divided into non-cancer and cancer related toxic substances
Particulate matter	Disease incidence	Potential incidence of human disease due to particulate matter emissions
Ionising radiation – human health	kBq U^{235} eq.	Impact of exposure to ionising radiations on human health
Photochemical ozone formation – human health	kg NMVOC eq.	Potential formation of harmful tropospheric ozone from air emissions
Acidification potential	mol H^+ eq.	Acidification mainly derived from combustion processes in electricity generation, heating and transport
Eutrophication – terrestrial	mol N eq.	Eutrophication caused by nitrogen and phosphorous emissions mainly due to fertilizers, combustion and sewage systems
Eutrophication – freshwater	kg P eq.	
Eutrophication – marine	kg N eq.	
Ecotoxicity – freshwater	CTUe	Impact on organisms due to toxic substances released into freshwater ecosystems
Land use	Dimensionless	It measures the changes in soil quality related to human activities (loss of species, organic matter, soil, filtration capacity, permeability)
Water use	m^3 world eq.	Depletion of available water for human activities and ecosystem integrity
Resource use – minerals and metals	kg Sb eq.	Depletion of non-renewable resources and deprivation for future generations
Resource use – fossils	MJ	

[1] Consists of three sub-indicators: Climate change – fossil, climate change – biogenic and climate change – land use and land use change.

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Characterisation of littered plastic emissions

As discussed in **Section 2.1.2.3**, it was assumed that the conventional plastic would eventually disperse between the ocean and soil and that the residual part of the biodegradable plastic would remain in the soil. Therefore, it has been necessary to develop CFs to assess the negative impacts of microplastics on marine and terrestrial environments.

Currently, robust models for assessing the impact of microplastics released into the environment are still lacking. This is mainly due to an incomplete understanding of the underlying mechanisms governing the fate, exposure and subsequent physical and toxicological effects of plastic products and particles on ecosystems and humans (Nessi et al., 2022). Therefore, the use of primary data from LABPLAS partners, as well as recent advances in impact methodologies, have made it possible to include the impacts of microplastic leakage in the EoL phase.

In this context, several authors have developed CFs to assess the impact of plastics released into the environment, particularly in the ocean. For instance, Maga et al. (2022) have introduced a CF that focuses on the residence time of plastics in the environment. This CF is solely determined by the fate factor, which considers the final compartment the plastic is disposed of after potential redistribution and the expected degradation rate in that compartment in relation to a reference degradation rate, which is usually set to 1 year. The resulting fate factor is expressed as kg of plastic pollution equivalent (PPe) per kg of plastic emitted.

On the other hand, Saling et al. (2020) formulated a CF for different types of polymers based on the number of particles per product entering the environment, the ecotoxicological effects on different organisms, and the time horizon these particles will persist. These findings led to the development of the midpoint-related Marine Microplastic Potential (MMP), measured in pellet equivalents. Furthermore, Salieri et al. (2021) consider USEtox as a basis for calculating a simplified CF for microplastics in the impact category of freshwater ecotoxicity. This was adjusted using ecotoxicity and material properties data from the materials under investigation.

Lastly, the MarILCA project is noteworthy in this field for its focus on assessing the impacts of microplastics in the ocean (Boulay et al., 2021), so it was used as a reference in this document. The MarILCA project focuses on developing a framework to integrate the potential environmental impacts of marine litter (specifically plastics) into LCA. To this end, it works on harmonizing the development of environmental impact pathways and CFs for marine impact assessment through the impact category of *physical effects on biota*. The impact category *physical effects on biota* aims at capturing the physical impacts of plastic litter on organisms, both through internal (ingestion) and external (entanglement, smothering) pathways (Corella-Puertas et al., 2022).

Conversely, there is currently a lack of established methods or scientific literature covering the assessment of the impacts of microplastics on soil. To address the toxicity and ecotoxicity impacts in LCA, the current consensus is to use the USEtox model. This was developed by an expert working group from the United Nations Environment Programme (UNEP) and the Society of Environmental Toxicology and Chemistry (SETAC) Life Cycle Initiative (USEtox, 2024). This model was originally designed for organic compounds and does not account for microplastics. Additionally, CFs for terrestrial ecotoxicity are absent in USEtox, which focuses primarily on freshwater ecotoxicity.

Given these limitations, it has been necessary to adapt the method following studies that search for the impacts of various compounds, such as metals or chemicals, on soil to address the specific scenario of microplastics.

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In this regard, the impacts of metals on soil have been evaluated using the *terrestrial ecotoxicity* impact category, as demonstrated by Aziz et al. (2018), Owsianiak et al. (2013) and Santos et al. (2018).

A common structure has been followed to obtain both CFs (i.e., *physical effects on biota* and *terrestrial ecotoxicity*). Therefore, the CFs were calculated by combining the fate, exposure and effect factors (Golsteijn, 2014).

- Fate factor (FF): Represents the distribution of the substance in the environment. It depends on the environmental compartment where this substance is found (i.e. air, soil or water) at a steady state (Owsianiak et al., 2023).
- Exposure and effect factor (EEF): Based on the work developed by Lavoie et al. (2022), exposure constitutes the bioavailable mass fraction in the compartment of exposed ecosystems, while the effect factor is the intrinsic toxicity potential.

The CFs are explained in more detail in ***Annex I Physical effects on biota*** and ***Annex II Terrestrial ecotoxicity***.

2.1.2.5 Assumptions and limitations

In this section, the limitations found during the definition of the systems, as well as the assumptions made to address them, are listed.

If information on additives used, such as plasticisers, compatibilizers and pigments is available as primary data, the effects of their production are included in the study. In contrast, the effect of additive leakage was not included in the assessment due to the lack of comprehensive and representative data on additive use and its environmental impacts.

For the biodegradable mulch film, no credits are assigned to its degradation in soil as a substitute for soil conditioners, since the mulch decomposes primarily into carbon dioxide and methane. Regarding the biodegradation rate of biodegradable plastic in soil, the material complies with European standards, specifically EN-17033:2019 for mulch film. Consequently, the biodegradation rate was determined to be 90%, in alignment with the minimum biodegradability (in 2 years at ambient temperature) required by this standard, which reflects the portion of carbon mineralised into CO_2. The residual part that does not biodegrade was assumed to remain in the soil in the form of biomass.

Lastly, knowledge on the fragmentation of macroplastics into microplastics is insufficient, so three fragmentation states are proposed as following the work done by Corella-Puertas et al. (2022): 10% (slow), 50% (medium) and 100% (fast) fragmentation of macroplastics into microplastics. Fragmentation is understood as the breaking of plastics into smaller plastic particles via mechanical pathways.

2.2 Life Cycle Inventory

Data collection involves both direct sources (primary data) and indirect sources (secondary data). Primary data are gathered through an LCI template filled out by BASF partners. This template compiles detailed information about product characteristics, including thickness, material composition and weight, as well as the energy and chemicals required for polymer production. On the other hand, secondary data, which includes information on the production of fuels and chemicals, as well as processes like extrusion for film manufacture, is sourced from the Ecoinvent database (Wernet et al., 2016). Information related to de production of polymers is not presented in the report due to its confidential nature. The LCI is shown in **Table 6**.

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Table 6. Life cycle inventory of conventional and biodegradable mulch film. Note that the values refer to 136.5 kg of conventional mulch film and 125 kg of biodegradable mulch film.

Conventional			Biodegradable		
Transport					
Input	**Value**	**Unit**	**Input**	**Value**	**Unit**
Distance	$2.06 \cdot 10^{3}$	km	Distance	$2.06 \cdot 10^{3}$	km
Usage					
Input	**Value**	**Unit**	**Input**	**Value**	**Unit**
Water	$9.50 \cdot 10^{2}$	L	Water	$9.50 \cdot 10^{2}$	L
Fertilizer	$1.23 \cdot 10^{2}$	kg N	Fertilizer	$1.18 \cdot 10^{-2}$	kg N
Fertilizer	$1.58 \cdot 10^{-2}$	kg P_2O_5	Fertilizer	$1.51 \cdot 10^{-2}$	kg P_2O_5
Fertilizer	$8.61 \cdot 10^{-3}$	kg K_2O	Fertilizer	$8.24 \cdot 10^{-3}$	kg K_2O
Insecticides	$2.41 \cdot 10^{-5}$	kg	Insecticides	$2.30 \cdot 10^{-5}$	kg
Fungicides	$1.93 \cdot 10^{-4}$	kg	Fungicides	$1.85 \cdot 10^{-4}$	kg
Herbicides	$1.50 \cdot 10^{-5}$	kg	Herbicides	$1.44 \cdot 10^{-5}$	kg
Output	**Value**	**Unit**	**Output**	**Value**	**Unit**
Crop yield	$2.41 \cdot 10^{4}$	kg	Crop yield	$2.51 \cdot 10^{4}$	kg
EoL phase					
Input	**Value**	**Unit**	**Input**	**Value**	**Unit**
Open burning			**Biodegradation in soil**		
Waste film	$6.83 \cdot 10^{1}$	kg	Waste film	$1.25 \cdot 10^{2}$	kg
Microplastics formation in soil					
Microplastics (10% fragmentation)	6.48	kg			
Microplastics (50% fragmentation)	$3.24 \cdot 10^{1}$	kg			
Microplastics (100% fragmentation)	$6.48 \cdot 10^{1}$	kg			
Microplastics distribution to ocean					
Microplastics (10% fragmentation)	$3.41 \cdot 10^{-1}$	kg			
Microplastics (50% fragmentation)	1.71	kg			
Microplastics (100% fragmentation)	3.41	kg			

2.3 Life Cycle Impact Assessment

Table 7 shows the characterisation results for the EF impact categories as well as the single scores for each scenario. Bearing in mind these results, it can be observed that most of the characterization results by impact category between conventional and biodegradable materials are of the same order of magnitude. However, notable differences in mulch film production profiles are appreciated depending on the polymer used.

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Consequently, conventional mulch film shows a higher environmental impact compared to biodegradable mulch film in the impact categories of HTN, HTC, PM, POF, EC and RUF.

The impact in the toxicity-related categories (i.e., HTN, HTC and EC) and PM can be explained by the fact that the high toxicity emissions generated during the incineration process at de EoL of the product. Concerning RUF and POF, the production phase of LDPE granules plays a significant role due to the consumption of fossil resources and energy.

Conversely, the biodegradable mulch film has a higher environmental impact in all other categories compared to the conventional one. In these categories, the impact is primarily attributed to the production of the PBAT polymer, caused by high energy and material consumption in the manufacturing stage, especially the synthesis processes of PBAT and its raw materials, since raw materials such as 1,4-butanediol, adipic acid and terephthalic acid cannot be obtained from renewable resources.

Additionally, it is important to note that current LCIA methodologies are not capable of fully accounting for the effects of microplastics formation and its biodegradability. Therefore, to fully understand these results, it is of paramount importance to consider the impacts of microplastics, as discussed in **Section 2.3.1**.

Lastly, to justify the environmental results of the different profiles assessed, a discussion of the individual contribution of each impact category to the total environmental impact after the optional weighting and normalisation steps was done. Moreover, an analysis of the contribution by stage and by scenario in each of the impact categories studied was also carried out in **Section 2.4.1.**

Table 7. Characterisation results per FU of conventional and biodegradable mulch film and single score. Climate change (CC), ozone depletion (ODP), non-cancer and cancer human toxicity (HTN and HTC, respectively), particulate matter (PM), ionizing radiation (IR), photochemical ozone formation (POF), acidification (AC), terrestrial, freshwater and marine eutrophication (TEU, FEU and MEU, respectively), freshwater ecotoxicity (EC), land use (LU), water use (WU), minerals and metals and fossil resource use (RUM and RUF, respectively).

Impact Category	Unit	Conventional	Biodegradable
CC	kg CO_2 eq.	$5.73 \cdot 10^{2}$	$6.27 \cdot 10^{2}$
ODP	kg CFC-11 eq.	$6.31 \cdot 10^{-6}$	$5.39 \cdot 10^{-4}$
HTN	CTUh	$9.93 \cdot 10^{-7}$	$3.88 \cdot 10^{-7}$
HTC	CTUh	$4.11 \cdot 10^{-6}$	$3.91 \cdot 10^{-6}$
PM	disease inc.	$3.15 \cdot 10^{-5}$	$1.78 \cdot 10^{-5}$
IR	kBq U^{235} eq.	9.32	$1.99 \cdot 10^{1}$
POF	kg NMVOC eq.	1.61	1.59
AC	mol H^{+} eq.	1.24	1.97
TEU	mol N eq.	3.20	4.08
FEU	kg P eq.	$9.97 \cdot 10^{-3}$	$2.14 \cdot 10^{-2}$
MEU	kg N eq.	$2.89 \cdot 10^{-1}$	$3.56 \cdot 10^{-1}$
EC	CTUe	$2.05 \cdot 10^{3}$	$1.66 \cdot 10^{3}$
LU	Pt	$1.46 \cdot 10^{3}$	$2.46 \cdot 10^{3}$
WU	m^3 depriv.	$2.95 \cdot 10^{2}$	$7.04 \cdot 10^{2}$
RUM	kg Sb eq.	$2.93 \cdot 10^{-4}$	$2.31 \cdot 10^{-3}$
RUF	MJ	$1.22 \cdot 10^{4}$	$9.38 \cdot 10^{3}$
EF single score	mPt	$4.61 \cdot 10^{1}$	$4.87 \cdot 10^{1}$

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2.3.1 Impact of littered plastics

Firstly, the impacts of littered microplastics in the ocean are presented through the impact category of *Physical effects on biota* for LDPE microplastics. The results depend on the sedimentation scenarios proposed (i.e., slow, medium and fast) and the fragmentation rate, as shown in **Table 8**.

It can be observed that the highest impacts occur when 100% of macroplastics are converted into microplastics. Regarding the sedimentation scenario, the slow sedimentation scenario has the most significant impacts. In this scenario, sedimentation occurs slowly, leading to a longer residence time in the ocean, which in turn increases the influence of degradation. Therefore, the worst-case scenario is slow sedimentation combined with a 100% fragmentation rate.

Table 8. Physical effects on biota impacts for littered microplastics of conventional mulch film.

Fragmentation	Sedimentation scenario	Physical effects on biota (CTUe)
10%	Slow	3.91
	Medium	3.90
	Fast	3.87
50%	Slow	$1.96 \cdot 10^{1}$
	Medium	$1.95 \cdot 10^{1}$
	Fast	$1.94 \cdot 10^{1}$
100%	Slow	$3.91 \cdot 10^{1}$
	Medium	$3.90 \cdot 10^{1}$
	Fast	$3.87 \cdot 10^{1}$

The assessment of the impacts of microplastics generated during the EoL phase of mulch film that remain in the soil is conducted through the impact category of *terrestrial ecotoxicity*, as shown in **Table 9**. The bottleneck, once again, is the 100% fragmentation rate, which results in more microplastics being released into the environment. The contribution to *terrestrial ecotoxicity* by LDPE microplastics is dominated by the fate factor. Due to the non-biodegradability of these plastics, they will remain in the soil for a long time, causing prolonged effects.

Table 9. Terrestrial ecotoxicity impact for littered microplastics of conventional mulch film.

Fragmentation	Terrestrial ecotoxicity (CTUe)
10%	$1.16 \cdot 10^{1}$
50%	$5.79 \cdot 10^{1}$
100%	$1.16 \cdot 10^{2}$

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2.4 Interpretation of results

2.4.1 Hotspot analysis

2.4.1.1 Identification of the most relevant impact categories

The single score results indicate the absolute potential impact of a product. To obtain this value, the characterisation results were normalised to the average global per capita (for an average European citizen) emissions and then multiplied by the set of weighting factors to obtain a final single score expressed in milli-points (mPt). The results were disaggregated per impact category for the two scenarios, as shown in **Figure 3.**

For the conventional mulch film, three impact categories are identified as most relevant: CC, RUF and PM, with relative contributions of 35% for CC, 34% for RUF, and 10% for PM. The remaining impact categories have negligible contributions (less than 5% of the total impact). Similarly, with the biodegradable mulch film, the three main impact categories are CC, RUF and WU, with relative contributions of 36%, 25% and 11% respectively, while the other impact categories contribute less than 5%.

The predominance of these impact categories can be attributed to the fossil origin of both products and their associated consumption of fuels and energy, which are predominantly fossil-based.

It is important to note that this single score depends on a set of weighting factors, which reflect the perceived relative importance of the considered impact categories based on the subjective consideration of a group of experts. In the EF methodology, priority is given to the CC category (21%), while the categories with the lowest weighting factor are the toxicity ones (approximately 2% for each). Therefore, these results should be interpreted with caution and attention should be given to each category according to the specific interest of the study.

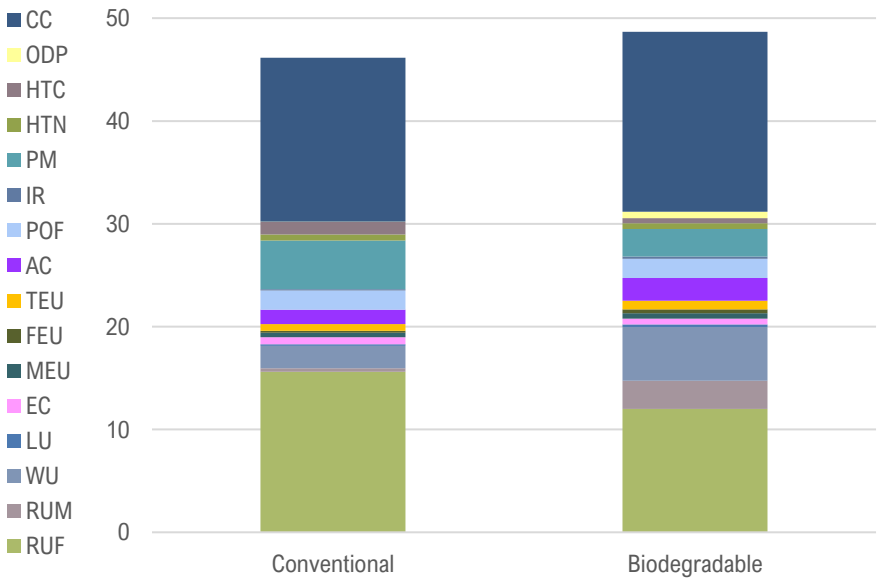

Figure 3. Contribution of impact categories to the total normalised and weighted scenario impact for mulch film. Climate change (CC), ozone depletion (ODP), non-cancer and cancer human toxicity (HTN and HTC, respectively), particulate matter (PM), ionizing radiation (IR), photochemical ozone formation (POF), acidification (AC), terrestrial, freshwater and marine eutrophication (TEU, FEU and MEU, respectively), freshwater ecotoxicity (EC), land use (LU), water use (WU), minerals and metals and fossil resource use (RUM and RUF, respectively).

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2.4.1.2 Identification of the most relevant life-cycle stages

Contribution analyses of conventional and biodegradable mulch film are shown in **Figure 4** and **Figure 5**, respectively.

Given **Figure 4**, the production of LDPE pellets is the main source of impact for most categories, notably RUF (89%), ODP (74%), AC (63%) and FEU (62%). This is mainly due to the consumption of non-renewable resources such as the ethylene used in its production or the consumption of electricity.

The EoL stage significantly contributes to several impact categories (e.g., HTC (92%), EC (73%), HTN (70%) and PM (59%)) due to the emissions released during the combustion of the film. The contribution to the impact of mulch film manufacturing is also notable in categories like IR (69%), LU (63%), RUM (44%), and WU (40%), which is because of the high electricity consumption during film extrusion.

Additionally, transport has a relevant impact contribution in the categories LU (21%) and RUM (27%) from fuel production and combustion. On the other hand, during the usage stage, the WU category (26%) is the only one with a remarkable contribution based on the water consumed for irrigation during cultivation.

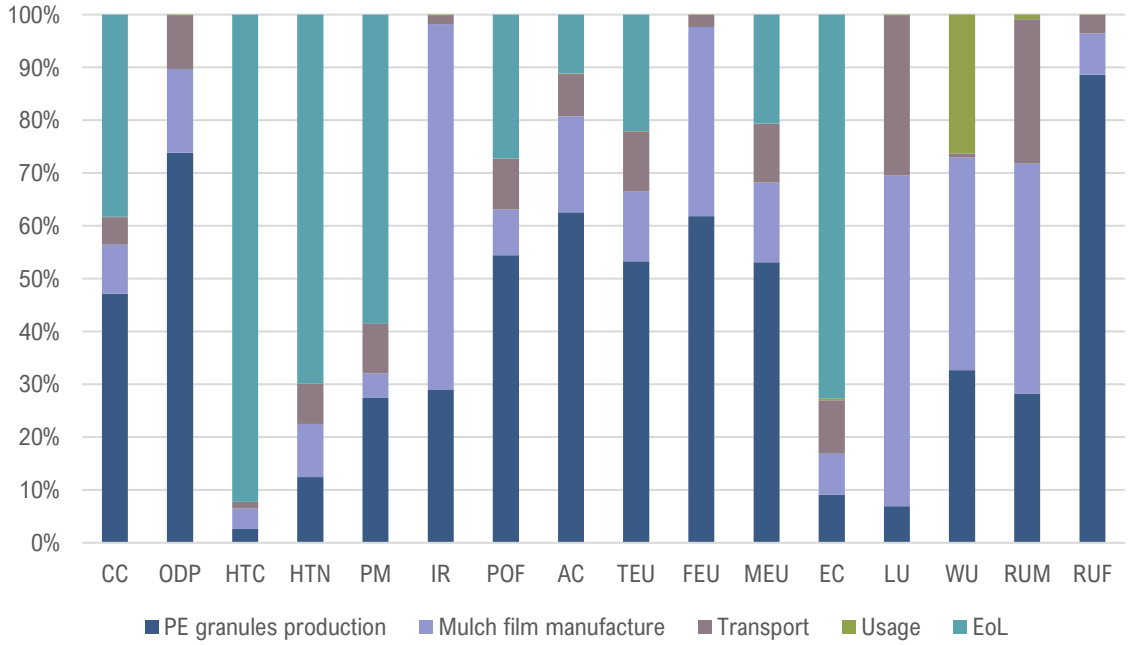

Figure 4. Relative contribution to life cycle impact of conventional mulch film. Climate change (CC), ozone depletion (ODP), non-cancer and cancer human toxicity (HTN and HTC, respectively), particulate matter (PM), ionizing radiation (IR), photochemical ozone formation (POF), acidification (AC), terrestrial, freshwater and marine eutrophication (TEU, FEU and MEU, respectively), freshwater ecotoxicity (EC), land use (LU), water use (WU), minerals and metals and fossil resource use (RUM and RUF, respectively).

Figure 5 shows the impact contribution in the biodegradable scenario for the different life-cycle stages considered. The polymer production phase is the most impactful in all impact categories, with a relative contribution ranging from 49% to nearly 100% for the LU and ODP impact categories, respectively. The primary contributors to these impacts are adipic acid and 1,4-butanediol used in PBAT production, both derived from fossil-based materials. Therefore, the impact of these substances is mainly due to the consumption of non-renewable raw materials such as cyclohexane for adipic acid and acetylene for 1,4-butanediol, as well as the consumption of stationary fossil fuels (e.g. natural gas) during their production processes. It is important to

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mention that there are now more sustainable options, such as certified biomass-balanced (BMB) PBAT. This involves replacing the fossil raw materials commonly used in the production process with renewable raw materials at the beginning of the value chain (Biernat, 2024). The impact reduction associated with this approach will be discussed in section 2.5 below.

The manufacture of mulch film is the second-largest contributor to the overall impact. Notably, it has a relative contribution of 33% for LU, 32% for IR, 16% for FEU, and 15% for WU. These figures, as in the previous scenario, are mainly due to the electricity consumption from the grid and their associated emissions.

Transport contributes significantly to categories such as LU (16%), PM (15%) and EC (13%), mainly due to fuel consumption and GHG emissions during transport. Additionally, the EoL stage has only a significant relative impact on CC (10%). The above can be explained by the fact that fossil carbon is emitted as CO_2 during the biodegradation process. Lastly, the use stage shows a negligible impact contribution except for WU (11%) because of the use of water for crop irrigation.

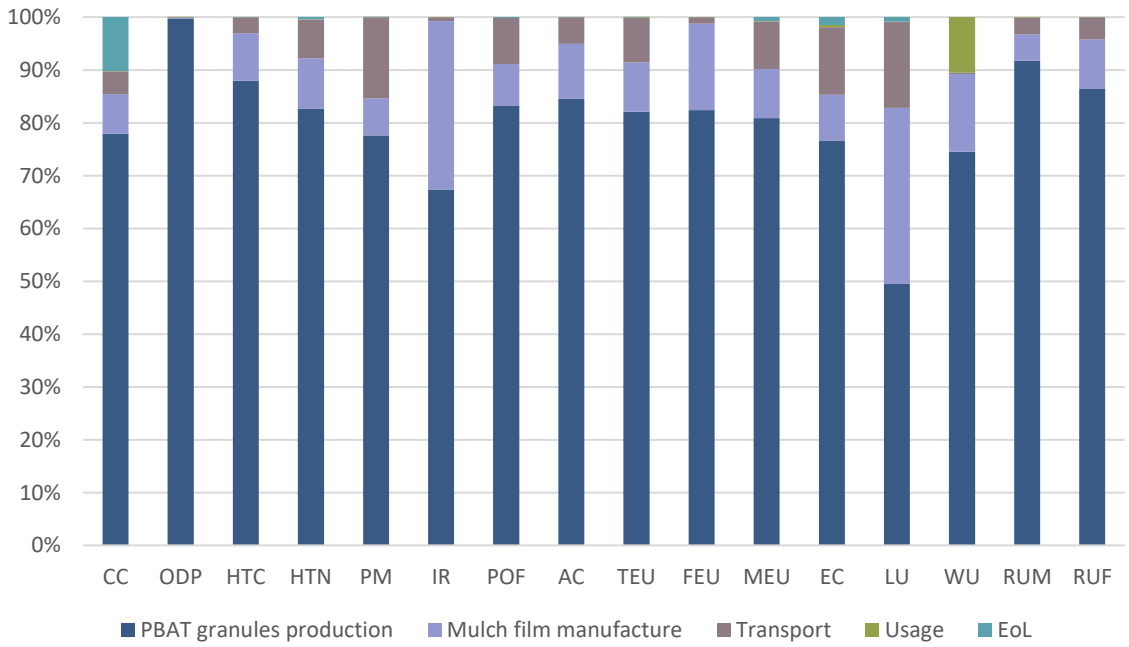

Figure 5. Relative contribution to life cycle impact of biodegradable mulch film. Climate change (CC), ozone depletion (ODP), non-cancer and cancer human toxicity (HTN and HTC, respectively), particulate matter (PM), ionizing radiation (IR), photochemical ozone formation (POF), acidification (AC), terrestrial, freshwater and marine eutrophication (TEU, FEU and MEU, respectively), freshwater ecotoxicity (EC), land use (LU), water use (WU), minerals and metals and fossil resource use (RUM and RUF, respectively).

2.4.2 Sensitivity analysis

A sensitivity analysis was conducted considering alternative EoL scenarios, as shown in **Figure 6**. This analysis proposes different EoL alternatives for conventional LDPE mulch film and includes two scenarios with different thicknesses for biodegradable mulch film (10 and 18 µm), as the purpose of biodegradable film scenario already includes only in-situ biodegradation. The proposed case for conventional mulch film, which assumes 50% open burning and 50% soil dumping, is considered the base case. Additionally, for conventional mulch film, mechanical recycling, incineration and landfilling were considered individually to better understand the effects

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of these options. Therefore, the main goal of this sensitivity analysis is to evaluate how the potential impacts of mulch film alternatives are affected by different EoL scenarios.

For the recycling scenario, it must be noted that mulch film presents low reported recycling rates due to barriers during the collection and recycling processes, as well as the generally poor quality of pellets produced. This study intended to consider all the limitations, but it is important to incentivize future research to make this alternative a feasible option.

Firstly, mulch film cannot be completely removed from the soil without tearing, resulting in fragments remaining in the soil. Therefore, a higher thickness (20 µm) for this scenario is considered, along with an estimated loss rate of 34% plastic losses during collection. Another problem is the high soil contamination rates, since when collecting mulch film, a portion of soil and organic matter is also collected, affecting the recycling process and increasing transport costs. A contamination rate of 45% for mulch films is expected, and consequently, due to this soil contamination, an intensive washing step must be included before processing (Hann et al., 2021). Once the film is collected, the recycling process includes sorting at specific facilities, followed by recycling through grinding, washing, and extrusion into pellets. In this regard, recycled polymer granulate was assumed to replace virgin granulate, so the primary production burdens of virgin granulate were credited to the system. To account for the lower overall quality of recycled polymer compared to virgin material, a substitution ratio of 0.75 was used for LDPE, following the recommendations of Nessi et al. (2022).

On the other hand, it was assumed that waste incineration occurred in a plant equipped with flue gas cleaning. For landfilling, an inventory for a sanitary landfill for municipal solid waste, including gas and leachate collection, was considered. Finally, in all of these alternatives where the collection of the film is necessary, it is important to mention that the removal of topsoil may affect soil fertility, which is critical for agricultural land. The removal of topsoil increases soil erosion and causes changes in soil texture, soil moisture, infiltration rate, and the vegetation community within the disturbed area (DeVilleneuve et al., 2023). Due to a lack of data, this aspect could not be included in the study, but it should be considered when selecting these options.

In the light of the results obtained, no single scenario exhibited higher environmental performance throughout all impact categories, although some conclusions can be drawn. The recycling scenario shows a beneficial impact on categories such as RUF, WU, AC, POF and CC, since the impact associated with new PE production is considered to be avoided. Conversely, recycling shows the worst environmental performance for ODP, IR, FEU, LU and RUM compared to other alternatives due to the high consumption of electricity and fuel during the recycling process, as well as the poor efficiency of the collection process.

The biodegradable mulch film demonstrated the lowest impact on most impact categories, such as HTC, HTN, IR and EC, and showed better environmental performance than the base case, landfilling, and incineration in all categories except for CC and LU. When compared to recycling, without considering the credits associated with the avoidance of producing new PE polymer, biodegradation appears to be a better option in all impact categories. The impact of the biodegradation process is due to the mineralization and conversion of carbon to CO_2 and CH_4. Consequently, the thicker mulch film has a slightly higher impact than the thinner one due to its greater weight.

On the other hand, the base case shows the worst results in terms of HTC, HTCN, PM, POF, AC, TEU, MEU and EC, primarily because of the release of gases during open burning without any precautions. Additionally, landfilling and incineration do not appear to be the preferred options in any impact category, as they tend to

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have significant environmental impacts across various categories, particularly those related to toxicity and resource use. This is because landfills emit CH_4 and CO_2 due to the decomposition of waste, while incineration releases CO_2 and other toxic pollutants from the burning of waste, and also consumes energy and materials in the process.

In conclusion, landfilling, open burning and incineration present significant environmental burdens across several impact categories, and although the recycling process has credits for avoiding the production of new material, it is highly energy-intensive and faces several barriers during collection such as plastic fragmentation and the collection of topsoil. Therefore, biodegradation emerges as the preferable disposal alternative from an environmentally friendly perspective.

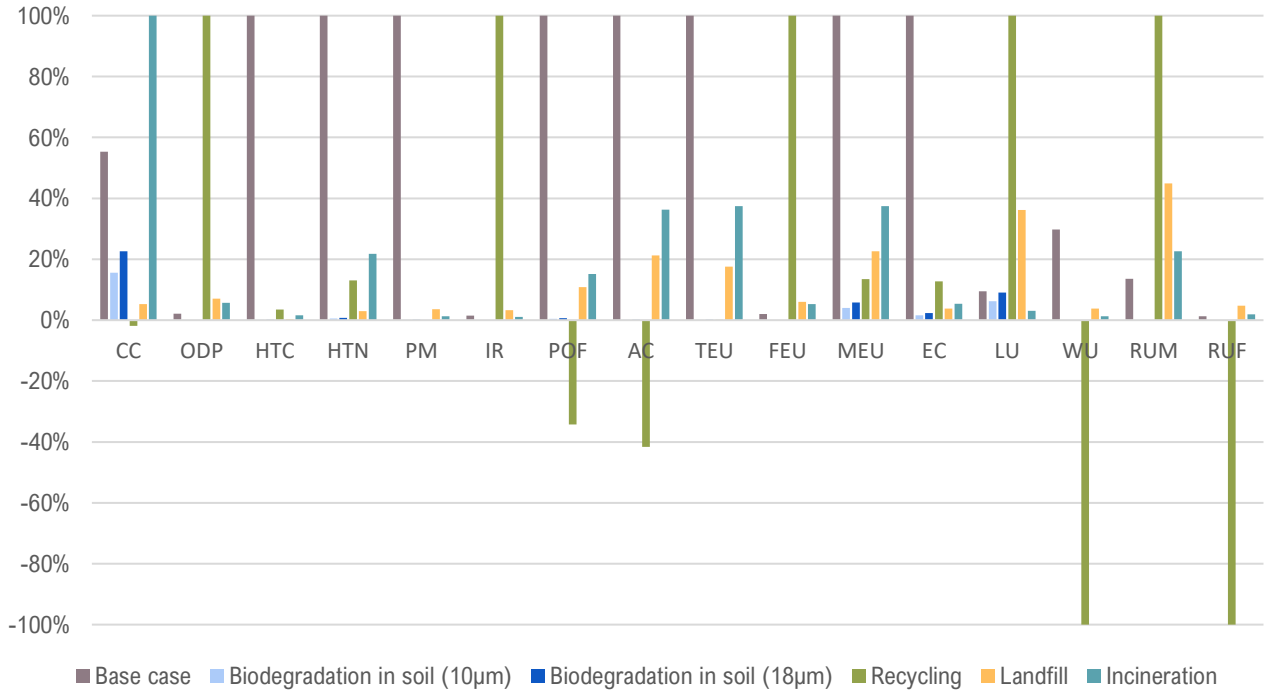

Figure 6. Comparative analysis of different EoL options for mulch film. Climate change (CC), ozone depletion (ODP), non-cancer and cancer human toxicity (HTN and HTC, respectively), particulate matter (PM), ionizing radiation (IR), photochemical ozone formation (POF), acidification (AC), terrestrial, freshwater and marine eutrophication (TEU, FEU and MEU, respectively), freshwater ecotoxicity (EC), land use (LU), water use (WU), minerals and metals and fossil resource use (RUM and RUF, respectively).

2.5 Conclusions and recommendations

A range of different environmental impact categories were assessed in this study, including climate change potential, air pollution, aquatic eutrophication, and human and freshwater toxicity, among others. For the biodegradable mulch film, the high impact values of categories like ODP, RUM or CC are mainly caused by high energy requirements and material consumption during the manufacturing stage of PBAT, especially the consumption of raw materials such as 1,4-butanediol, adipic acid and terephthalic acid, which are obtained from fossil-based sources. One option to decrease the impacts associated with its production could be increasing the biobased proportion. In the production of PBAT, it is currently not possible to completely avoid the use of fossil resources, but the biomass balance approach is a promising option to increase its biobased carbon content (Biernat, 2024).

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PBAT BMB replaces the fossil raw materials commonly used in the production process with renewable raw materials at the beginning of the value chain. The renewable raw material comes from biomass waste and residues and is attributed to the PBAT grade through a certified mass balance approach following the methodology described by Jeswani et al. (2019). Comparing the impact of using PBAT BMB for mulch film in our case study, it would achieve a single score 26% lower than conventional mulch film. Additionally, it shows a significant reduction in numerous impact categories compared to the original, such as PM (-69%), RUF (-56%), POF (-39%), and CC (-37%). Therefore, by reducing dependence on non-renewable resources and utilizing biodegradable materials this alternative presents a sustainable option to conventional mulch films.

In this regard, it is crucial to pay attention to material choices, as production is a critical stage throughout the life cycle. In addition to product composition, recent research on the composition of compostable plastics reveals the presence of additives such as talc, esters and phthalates can influence both the degradation rate of plastics and the subsequent release of microplastics into the environment (Mendes et al., 2024).

In impact categories such as EC, HTC and HTN, the biodegradable film presents relatively lower values compared to conventional film due to the absence of collection and waste management activities for this product. If we focus solely on the EoL stage, the biodegradable mulch film, assumed to naturally biodegrade in soil, shows better environmental performance in all evaluated impact categories compared to conventional film (assumed that half is burned outdoors and the other half is discarded in soil).

When considering alternative EoL options for conventional recycling, although recycling offers environmental improvements by avoiding the production of virgin LDPE, the effectiveness of collecting mulch film is critical. The thickness of the mulch film can lead to breakage during collection or due to ultraviolet exposure, and contamination with soil further complicates its retrieval for recycling.

Another intriguing aspect of this study is the analysis of the impact of microplastics through the categories of *physical effects on biota* and *terrestrial ecotoxicity*, which is an emerging problem receiving significant attention nowadays but is not usually assessed by current LCA methodologies. Biodegradable materials can ultimately be mineralized and converted to CO_2 over time, whereas PE is not biodegradable, resulting in the continual accumulation of microplastics in the soil. This demonstrates the importance of characteristics such as degradation at the end of a product's useful life.

Based on the above, it is important to continue researching this topic with more initiatives that gather direct data on microplastics, thus reducing the uncertainty that is currently faced. The goal must be to integrate the impact of microplastics into LCA studies in the future, providing a comprehensive and integrated understanding. This study demonstrates the importance of accounting for all impact categories throughout the life cycle of a product. Typically, the toxicity of microplastics is not assessed in LCA, yet this study shows that the impacts are significant. To have a comprehensive understanding that allows for correct discussion of results and informed decision-making, it should be considered.

In conclusion, this study analyses the environmental impacts of the entire life cycle of conventional and biodegradable mulch films, highlighting that although biodegradable films have significant impacts during production, they play an important role in the EoL stage.

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3 COMPARATIVE LCA FOR GARBAGE BAGS

3.1 Goal and Scope

3.1.1 Goal definition

The purpose of the second part of this report is based on the comparison of the environmental impacts associated with garbage bags for kitchen waste made from three different materials: LDPE, PBAT, and kraft paper, considering industrial composting as the only EoL scenario. Therefore, the biodegradability and microplastics formation of these materials is a key point for the study.

The analysis encompasses additional life cycle stages, such as the manufacture of primary materials, bag production, usage, and inter-stage transportation. This LCA also follows a cradle-to-grave approach and selects the impact categories from the PEF method. Using this evaluation is expected to help to understand the influence of the selection of different materials and their potential environmental impacts.

Similar to the study on mulch film, the conclusions of this study will inform more environmentally aware decisions regarding the most sustainable material. On the other hand, it will also identify the processes with the greatest environmental impact and the most relevant impact categories for kitchen waste bags.

3.1.2 Scope

3.1.2.1 Product system

Three case studies were selected for this analysis to assess the environmental impacts of various kitchen waste bags. For the first scenario, LPDE waste bags were selected as the non-biodegradable alternative, these bags are commonly used for compostable waste due to their durability and ability to prevent leaks in kitchen bins. The second alternative is compostable PBAT waste bags. The last scenario is biodegradable kraft paper bags used as kitchen waste bags. Kraft paper bags are commonly used in Germany for kitchen waste. Therefore, Germany was chosen as the location for this study.

The features of each bag are presented in **Table 10**.

Table 10. Waste bags characteristics of each scenario.

Parameter	Unit	Conventional	Biodegradable	Kraft paper
Material composition	-	LDPE	PBAT PLA	Paper pulp
Source (bio/fossil)	-	Fossil	Fossil Bio-based	Bio-based
Capacity (volume)	L	6	6	7
Weight	g	7.2	10	20.32

3.1.2.2 Functional Unit

The FU for the kitchen waste bags study was defined as the "waste bags required for the disposal of 1.4 kg of kitchen waste". The following table (**Table 11**) contains additional requirements for the study.

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Table 11. FU description for kitchen waste bags.

Question	Conventional	Biodegradable	Kraft paper
What?	Use of 1 garbage bag of 7.2 g	Use of 1 garbage bag of 10 g	Use of 1 garbage bag of 20.32 g
How much?	1.4 of kitchen waste		
How well?	Retain kitchen waste by preventing leaks into the kitchen bin or kitchen floor.		
How long?	One use		
Where?	Germany		

Regarding the reference flow for the study, it is defined as the amount of bag composition material necessary to meet the functional unit for each scenario (**Table 11**). As mentioned previously, the study is located in Germany, where the production and processing of raw materials, as well as their use, occur. When specific data were not available for Germany, the European average was used to fill this gap.

3.1.2.3 System boundaries

The system was analysed from a **cradle-to-grave** perspective, covering the extraction of raw materials up to their final disposal, as shown in **Figure 7**. For each stage, inputs and outputs, including emissions, were taken into consideration. The stages for each type of bag are the same, although they are based on different raw materials, so the inputs in their manufacturing processes may differ.

- **Raw Material Production:** This step includes the extraction, processing, transportation, and refining of crude oil and natural gas for conventional bags. For biodegradable bags, it involves the cultivation, forestry, and harvesting of raw materials, and then converting them from monomers to polymers. In the case of paper bags, it includes cultivation, forestry, collection of wood, transformation into pulp, and then pressing. The transportation of materials involved in these activities was also considered.
- **Manufacturing:** This stage covers the extrusion of polymer granules to produce the final products. Extrusion involves melting polymers and shaping them into a thin, flexible film, which is then cooled and solidified to obtain the desired thickness and strength. For kraft paper, the process includes cutting, folding, and glueing.
- **Distribution:** Refers to the transportation of products from manufacturing centres to consumers. The German medium distance between manufacturing centres and main cities was used, assuming transportation in trucks of size class >32 tonnes.
- **Use:** Once the bags are filled with kitchen waste, consumers dispose of them in specialized containers for organic waste in Germany, typically found on the street. Paper bags begin first to degrade in these containers, causing liquid spills, which necessitates regular washing to prevent smells and pest proliferation (Olaosebikan Oluwatosin et al., 2014; Puyuelo et al., 2013). Therefore, the water consumption required to wash these containers is also considered in this stage.
- **End of Life (EoL):** Collection, transportation, and treatment of disposal products. For all three scenarios with different types of bags, industrial composting is taken into account. Biodegradable and paper bags degrade, while conventional bags fragment and remain on the ground.

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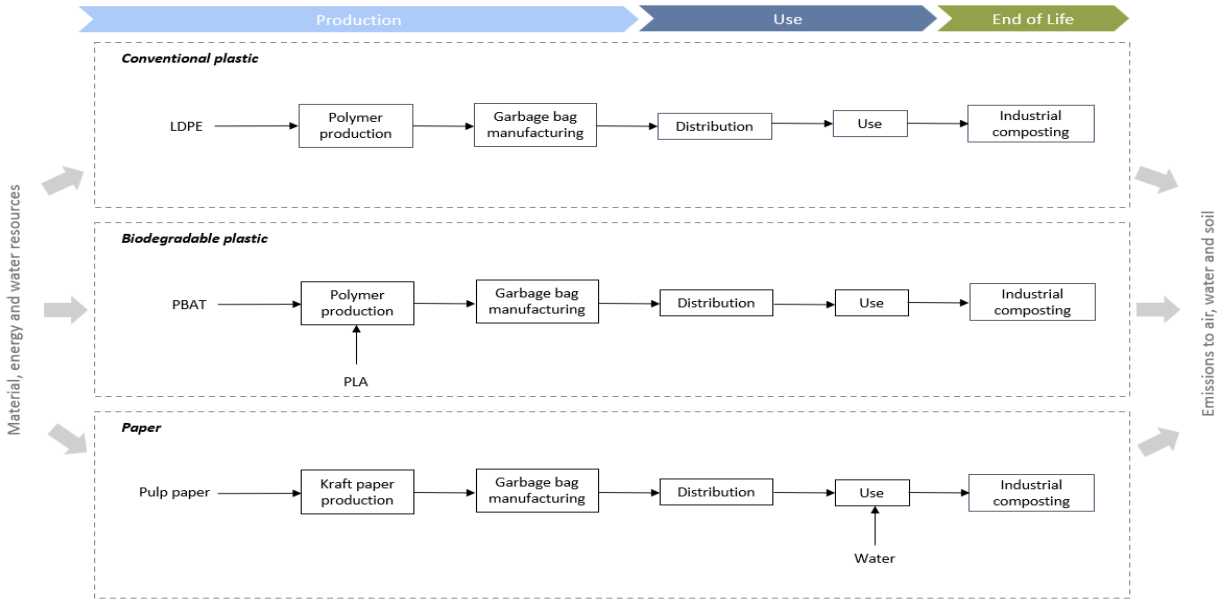

Figure 7. System boundaries of conventional, biodegradable and paper waste bags.

3.1.2.4 Impact assessment method and selected impact categories

The impact categories and different methods are the same as shown in **Section 2.1.2.4** (Impact assessment method and selected impact categories). The EF method version 3.1 was used to assess up to 16 environmental impact categories.

Characterisation of littered plastics emissions

According to the bags, it was considered that they are only disposed of on soil. Therefore, the methodology established in **Section 0** was implemented, but only to develop the CF of microplastics in the terrestrial environment, specifically focusing on terrestrial ecotoxicity. The detailed calculations can be found in **Annex II *Terrestrial ecotoxicity***.

3.1.2.5 Assumptions and limitations

The assumptions and limitations for the mulch film study were also considered for the waste bags. These are detailed in **Section 2.1.2.5**. Additionally, no credits were given for using compost as a substitute for fertilizer, since it is out of the scope of this study, which is the assessment of the use of different materials of plastic bags.

3.2 Life Cycle Inventory

Primary data regarding conventional and biodegradable bags were acquired from BASF's inventory. This data covers the bag specifications, material composition, and the required inputs for polymer production. The information related to polymer production is not shown due to their confidential character. The manufacturing modelling processes for these bags were obtained from the Ecoinvent database. Similarly, inventory modelling for paper bags, including production processes, electricity consumption, and material usage, were also derived

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from the Ecoinvent database (Wernet et al., 2016). The inventory data collected is shown in the following **Table 12.**

Table 12. Life cycle inventory of conventional, biodegradable and paper waste bags film.

						Paper		
						Paper production		
						Input	Value	Unit
						Pulpwood	$3.00 \cdot 10^{-2}$	m^3
Conventional			Biodegradable			Electricity	$5.00 \cdot 10^{-1}$	kWh
Transport								
Input	Value	Unit	Input	Value	Unit	Input	Value	Unit
Distance	$4.43 \cdot 10^{2}$	km	Distance	$4.43 \cdot 10^{2}$	km	Distance	$4.43 \cdot 10^{2}$	km
Usage								
Input	Value	Unit	Input	Value	Unit	Input	Value	Unit
						Water	$1.51 \cdot 10^{1}$	l
EoL phase								
Input	Value	Unit	Input	Value	Unit	Input	Value	Unit
Kitchen waste	1.40	kg	Kitchen waste	1.40	kg	Kitchen waste	1.40	kg
Bag waste	$7.20 \cdot 10^{-3}$	kg	Bag waste	$1.00 \cdot 10^{-2}$	kg	Bag waste	$2.03 \cdot 10^{-2}$	kg
Microplastics (10% fragmentation)	$7.20 \cdot 10^{-4}$	kg						
Microplastics (50% fragmentation)	$3.60 \cdot 10^{-3}$	kg						
Microplastics (100% fragmentation)	$7.20 \cdot 10^{-3}$	kg						

3.3 Life Cycle Impact Assessment

Table 13 presents the characterisation results for EF impact categories, as well as the EF single score for each scenario. Based on them, it is evident that paper bags exhibit a higher environmental impact than plastic bags in nearly all categories, except for RUF. For the categories of PM, AC, TEU and EC, although the impact for paper bags is slightly higher, the results are quite similar. This similarity is because the composting stage contributes the most (over 90%) and is a common process across all scenarios. In the categories of HTC, HTN, IR, POF, FEU and MEU the impact is distributed in bag manufacturing (due to electricity and fuel consumption), usage (due to pumping water) and composting. For the LU impact category, pulp production is the main contributor to impact. Finally, in the WU category, the impact comes from the consumption of water for cleaning the containers, due to the poor resistance to moisture and the rapid degradation of paper compared to plastic bags.

The biodegradable bag occupies a middle position in most categories, yet they demonstrate the poorest performance in the categories of CC, ODP, RUM and RUF. The high ODP impact is primarily due to the

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production of PBAT, while the impacts in CC, RUM, and RUF are influenced by both PBAT production and composting.

Conventional plastic bags demonstrate better environmental performance overall, coinciding with the fact that the weight of plastic bags plays an important role in their environmental impact, with the conventional plastic bag being the lightest. Additionally, it is important to note that conventional plastics are being used for composting in this study, which is not their typical use. Current LCIA methodologies are not capable of fully accounting for the effects of non-biodegradability over the product's lifecycle. Therefore, to fully understand these results, it is crucial to consider the impacts of microplastics, as discussed in **Section 3.3.1**.

As explained in the previous case study, the detailed results are addressed in **Section 3.4.1**.

Table 13. Characterisation results per FU for biodegradable plastic, conventional plastic and kraft paper bags, including the single score. Climate change (CC), ozone depletion (ODP), non-cancer and cancer human toxicity (HTN and HTC, respectively), particulate matter (PM), ionizing radiation (IR), photochemical ozone formation (POF), acidification (AC), terrestrial, freshwater and marine eutrophication (TEU, FEU and MEU, respectively), freshwater ecotoxicity (EC), land use (LU), water use (WU), minerals and metals and fossil resource use (RUM and RUF, respectively).

Impact category	Unit	Conventional	Biodegradable	Paper
CC	kg CO_2 eq.	$1.03 \cdot 10^{-1}$	$1.28 \cdot 10^{-1}$	$1.09 \cdot 10^{-1}$
ODP	kg CFC-11 eq.	$8.42 \cdot 10^{-10}$	$4.36 \cdot 10^{-8}$	$1.17 \cdot 10^{-9}$
HTC	CTUh	$5.03 \cdot 10^{-11}$	$7.73 \cdot 10^{-11}$	$8.00 \cdot 10^{-11}$
HTN	CTUh	$6.55 \cdot 10^{-10}$	$8.97 \cdot 10^{-10}$	$1.06 \cdot 10^{-9}$
PM	disease inc.	$1.70 \cdot 10^{-8}$	$1.79 \cdot 10^{-8}$	$1.80 \cdot 10^{-8}$
IR	kBq U^{235} eq.	$3.85 \cdot 10^{-3}$	$7.16 \cdot 10^{-3}$	$7.18 \cdot 10^{-3}$
POF	kg NMVOC eq.	$2.49 \cdot 10^{-4}$	$3.13 \cdot 10^{-4}$	$3.24 \cdot 10^{-4}$
AC	mol H^+ eq.	$3.19 \cdot 10^{-3}$	$3.30 \cdot 10^{-3}$	$3.31 \cdot 10^{-3}$
TEU	mol N eq.	$1.38 \cdot 10^{-2}$	$1.41 \cdot 10^{-2}$	$1.43 \cdot 10^{-2}$
FEU	kg P eq.	$1.45 \cdot 10^{-5}$	$2.26 \cdot 10^{-5}$	$4.99 \cdot 10^{-5}$
MEU	kg N eq.	$1.56 \cdot 10^{-4}$	$1.71 \cdot 10^{-4}$	$1.93 \cdot 10^{-4}$
EC	CTUe	$1.18 \cdot 10^{1}$	$1.20 \cdot 10^{1}$	$1.21 \cdot 10^{1}$
LU	Pt	$7.91 \cdot 10^{-1}$	$9.06 \cdot 10^{-1}$	4.62
WU	m^3 depriv.	$-4.60 \cdot 10^{-3}$	$3.47 \cdot 10^{-2}$	$6.32 \cdot 10^{-1}$
RUM	kg Sb eq.	$1.52 \cdot 10^{-7}$	$3.21 \cdot 10^{-7}$	$2.20 \cdot 10^{-7}$
RUF	MJ	1.16	1.25	$9.06 \cdot 10^{-1}$
EF single score	µPt	$1.86 \cdot 10^{1}$	$2.07 \cdot 10^{1}$	$2.50 \cdot 10^{1}$

3.3.1 Impact of littered plastic emissions

Table 14 shows the assessment results of the impacts of microplastics generated during the EoL phase of conventional garbage bags through the impact category of *terrestrial ecotoxicity*. The generation of microplastics depends on fragmentation, with a higher rate leading to a greater impact. Conventional plastic bags, made from LDPE, tend to break down into smaller fragments over time without fully mineralizing, resulting in the persistent accumulation of microplastics in the environment.

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The characterisation factors are dominated by the fate factor, which has a high contribution due to the non-biodegradability of the material, which has the potential to accumulate and affect terrestrial micro-organisms over a long period of time.

Table 14. Terrestrial ecotoxicity impact for littered microplastics of conventional garbage bags.

Fragmentation	*Terrestrial ecotoxicity* (CTUe)
10%	$1.29 \cdot 10^{-3}$
50%	$6.43 \cdot 10^{-3}$
100%	$1.29 \cdot 10^{-2}$

3.4 Interpretation of results

3.4.1 Hotspot analysis

3.4.1.1 Identification of the most relevant impact categories

The single score results, expressed in micro-points (μPt), were disaggregated per impact category for the three scenarios, as shown in **Figure 8**.

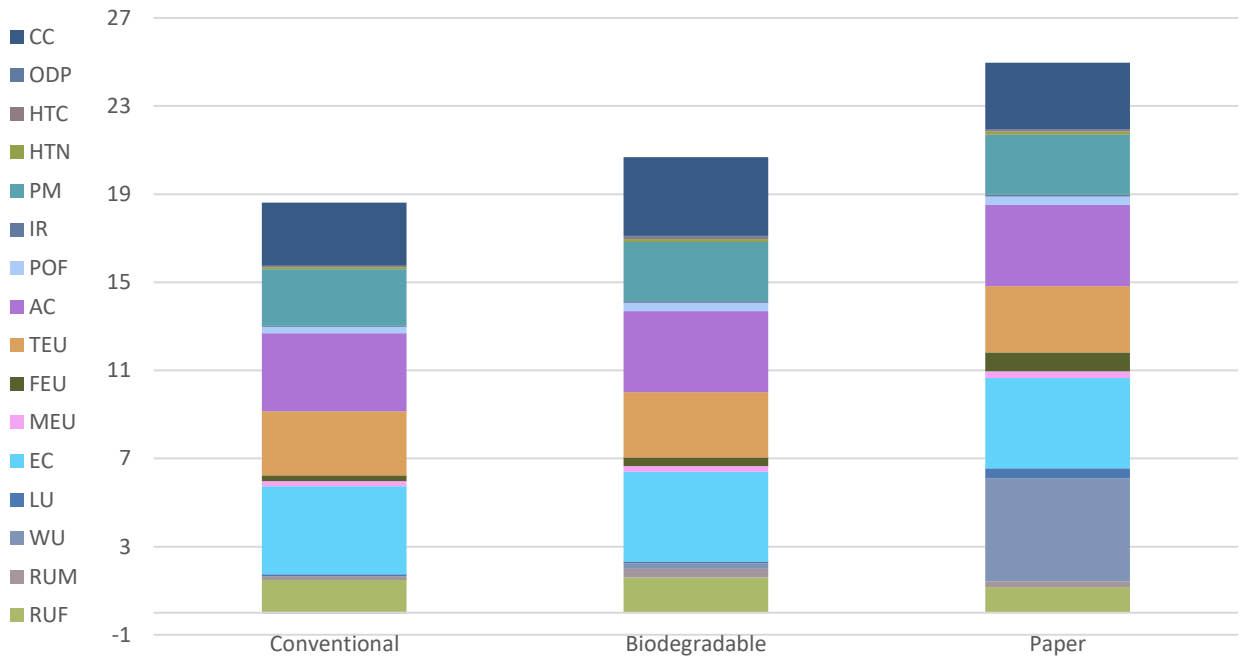

Figure 8. Contribution of impact categories to the total normalised and weighted scenario impact for waste bags. Climate change (CC), ozone depletion (ODP), non-cancer and cancer human toxicity (HTN and HTC, respectively), particulate matter (PM), ionizing radiation (IR), photochemical ozone formation (POF), acidification (AC), terrestrial, freshwater and marine eutrophication (TEU, FEU and MEU, respectively), freshwater ecotoxicity (EC), land use (LU), water use (WU), minerals and metals and fossil resource use (RUM and RUF, respectively).

There are several impact categories identified as significant across the three scenarios: CC, PM, AC, TEU and EC. For conventional plastics, the most relevant category is EC (22%) followed by AC (19%), TEU (16), CC (15%) and PM (14%). In the case of biodegradable plastic, the critical impact categories are also EC (20%)

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closely followed by AC (18%), CC (17%), TEU (14%) and PM (13%). Finally, for paper, in addition to the previously mentioned categories, the most relevant is WU (19%), followed by EC (16%), AC (15%), CC (12%), TEU (12%) and PM (11%). This aligns with the fact that paper bags require additional water for bin cleaning.

The commonality of these relevant impact categories across scenarios suggests a shared critical point, likely related to the composting of kitchen waste. Composting processes can influence categories like eutrophication and acidification due to the release of nutrients and chemicals during decomposition. This explains why these impact categories are consistently significant across different material scenarios.

3.4.1.2 Identification of the most relevant life-cycle stages

Contribution analyses of conventional plastic, biodegradable plastic and kraft paper bags are shown in **Figure 9**, **Figure 10** and **Figure 11**, respectively.

Firstly, for the conventional plastic scenario, the composting process is the main source of impact for all the categories, ranging from 59% to nearly 100% for WU and EC, respectively. The impact of composting in categories such as PM, RUM, HTC, HTN, LU and CC is primarily due to electricity and fuel emissions, along with the extraction of non-renewable materials. For categories such as EC, AC and eutrophication, the impact depends on the gas emissions of the process (NH_3, N_2O and H_2S). Notably, composting has a positive impact in the WU category because the wastewater and leachate generated during the process are treated, and the recovered water is assumed to replace the use of new water.

The production of LDPE granules significantly contributes to the impact categories of RUF (49%), ODP (29%) and TEU (20%) due to the consumption of ethylene and electricity during its production. The manufacturing of the bag, including film extrusion, has a relevant impact on IR (30%), WU (23%) and (12%) due to the production of the electricity used and its associated emissions.

Finally, transport has a negligible impact, contributing less than 2% in all impact categories.

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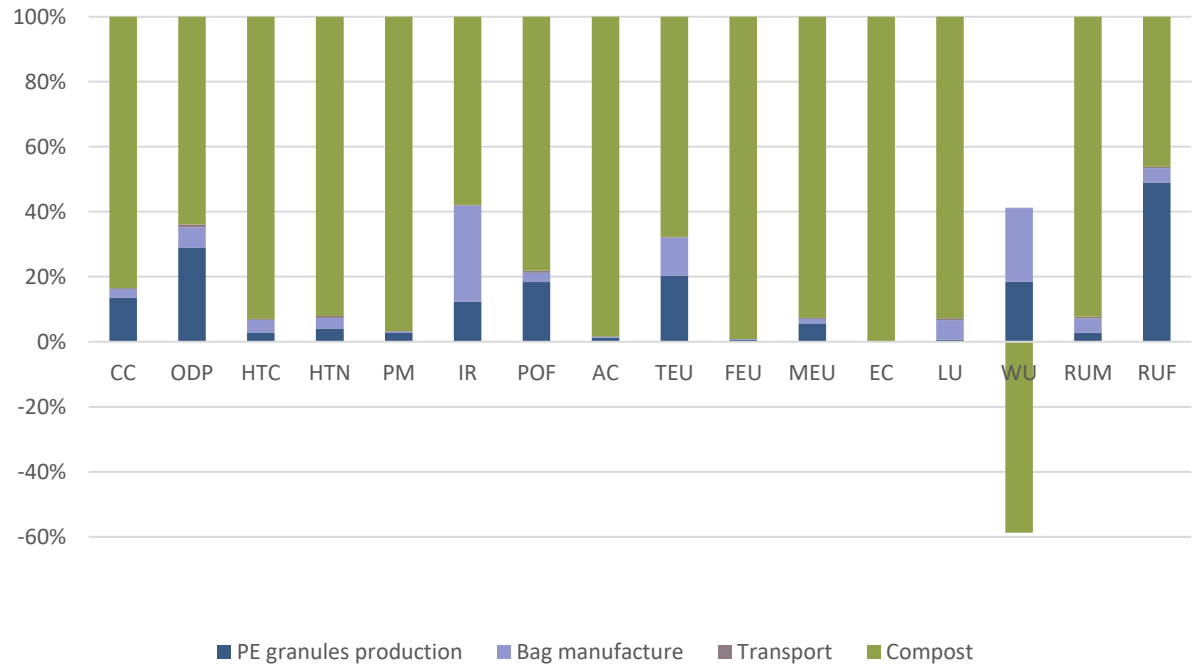

Figure 9. Relative contribution to life cycle impact of conventional garbage bags. Climate change (CC), ozone depletion (ODP), non-cancer and cancer human toxicity (HTN and HTC, respectively), particulate matter (PM), ionizing radiation (IR), photochemical ozone formation (POF), acidification (AC), terrestrial, freshwater and marine eutrophication (TEU, FEU and MEU, respectively), freshwater ecotoxicity (EC), land use (LU), water use (WU), minerals and metals and fossil resource use (RUM and RUF, respectively).

Figure 10 shows the impact contribution in the biodegradable plastic scenario for the different life-cycle stages considered. Again, the composting process is the main contributor in most of the impact categories, notably in EC (99%), TEU (98%) and AC (95%), caused by chemicals emitted during the process as well as the electricity and fuel consumption.

The second-largest contributor to impact is the production of polymer granules, which has a significant relative contribution to ODP (99%), WU (64%) and RUM (53%), due to the use of non-renewable raw material and the consumption of fossil fuels during its production.

The manufacture of mulch film has relevant contributions of 22% for IR, 13% for WU and 11% for FEU. These impacts, as in the previous scenario, are primarily due to the electricity consumption from the grid and its associated emissions. Finally, transport has a negligible impact in all impact categories, contributing less than 2%.

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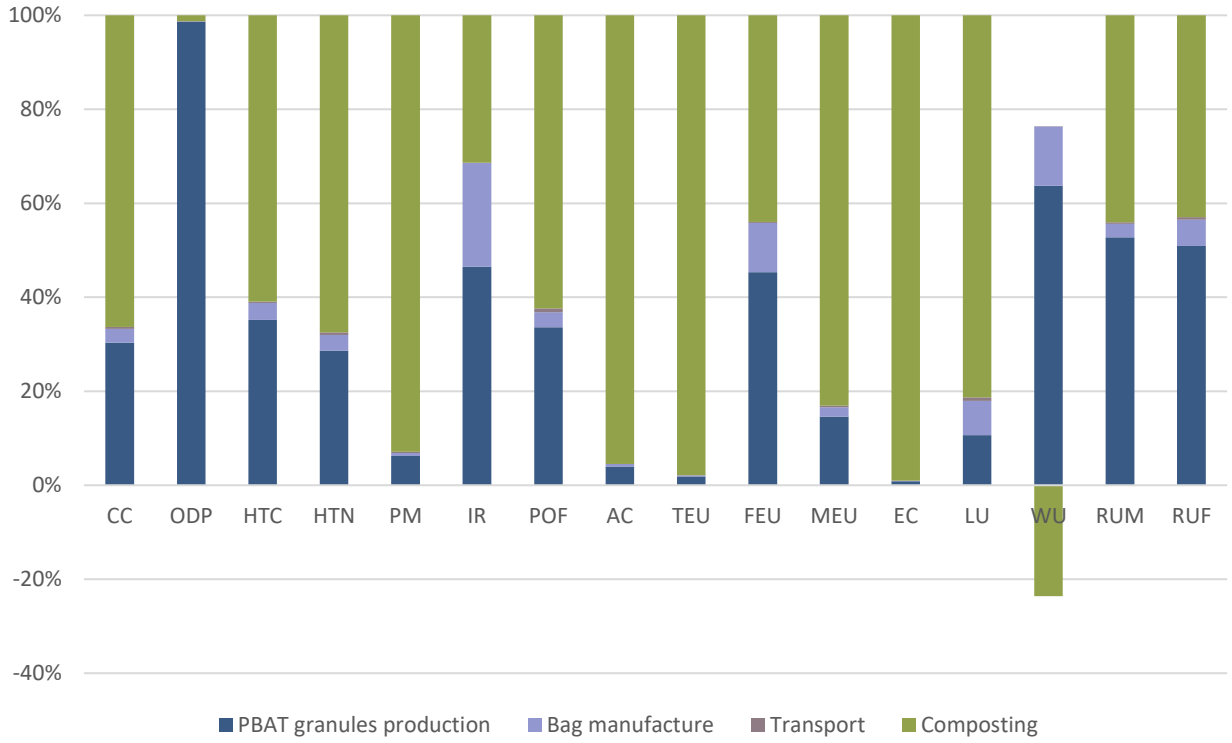

Figure 10. Relative contribution to life cycle impact of biodegradable garbage bags. Climate change (CC), ozone depletion (ODP), non-cancer and cancer human toxicity (HTN and HTC, respectively), particulate matter (PM), ionizing radiation (IR), photochemical ozone formation (POF), acidification (AC), terrestrial, freshwater and marine eutrophication (TEU, FEU and MEU, respectively), freshwater ecotoxicity (EC), land use (LU), water use (WU), minerals and metals and fossil resource use (RUM and RUF, respectively).

Lastly, **Figure 11** presents the impact contribution in the kraft paper scenario considering the different life-cycle stages. As in the previous cases, the composting process emerges as the main contributor in most of the impact categories, specifically in EC (99%), TEU (97%) and AC (96%), due to the emission of chemicals during degradation and the electricity and fuel consumption involved the process.

The manufacturing of the bag also has significant contribution to various impact categories, such as FEU (73%), IR (43%) and ODP (38%), due to the consumption of electricity and fuels, as well as all emissions at the mill, both from the cooking and recovering processes and combustion of fuels at different points at the plant. As mentioned earlier, paper bags degrade faster than plastic bags, accelerated by the humidity of the waste, necessitating bin cleaning. This is represented in the usage phase, which has a substantial impact on WU (96%), HTC (26%) and HTN (25%) due to the pumping of raw surface water. Pulp production has the highest contribution to LU at 73%, derived from silviculture and forestry activities. Regarding transportation, this has a negligible impact in all categories.

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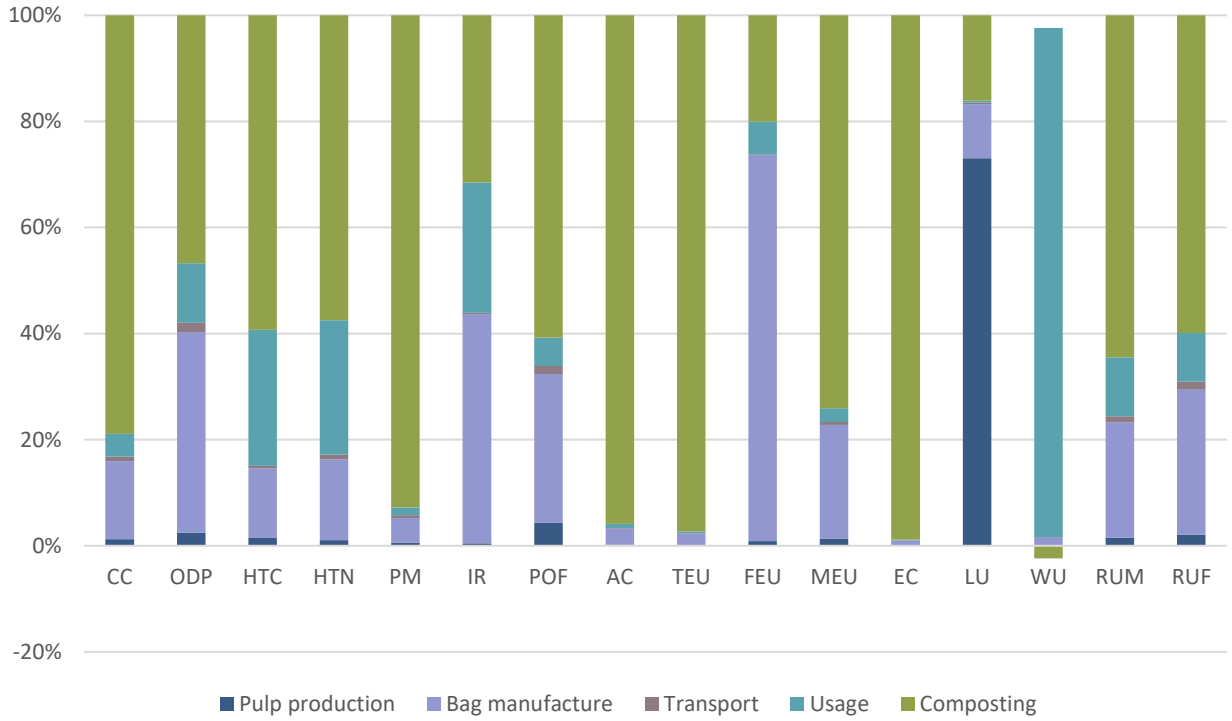

Figure 11. Relative contribution to life cycle impact of paper garbage bags. Climate change (CC), ozone depletion (ODP), non-cancer and cancer human toxicity (HTN and HTC, respectively), particulate matter (PM), ionizing radiation (IR), photochemical ozone formation (POF), acidification (AC), terrestrial, freshwater and marine eutrophication (TEU, FEU and MEU, respectively), freshwater ecotoxicity (EC), land use (LU), water use (WU), minerals and metals and fossil resource use (RUM and RUF, respectively).

3.4.2 Sensitivity analysis

A sensitivity analysis was conducted considering alternative EoL scenarios, as shown in **Figure 12**. In this study, sensitivity analysis focused on biodegradable waste bags, as correct usage dictates avoiding conventional bags for composting due to their non-biodegradable nature. To correctly understand their impacts, the ecotoxicity of resulting microplastics from these bags has already been assessed. Therefore, the primary objective of this sensitivity analysis was to evaluate how the environmental impacts of disposing of 1.4 kg of kitchen waste in biodegradable plastic bags are influenced by various EoL scenarios, including industrial composting (base case), home composting, anaerobic digestion, landfill and incineration.

Home composting has the lowest impact in all categories except HTC, POF and TEU. In fact, for POF, it is the option with the highest impact. The generally lower impact of home composting compared to industrial composting is due to reduced electricity and fuel consumption. However, home composting faces challenges such as odour, pests and incomplete decomposition.

Anaerobic digestion has a lower impact than the base case in categories such as AC, TEU, PM and IR. Conversely, it performs worse in CC, POF and FEU, due to emissions of gases and compounds other than methane, which is utilized for energy production.

A landfill is the worst option in the categories of CC, FEU, MEU and WU, while incineration is the worst for HTC and HTN. So, while these options perform better in some areas than the base case, they have significant environmental drawbacks, particularly in categories related to toxicity and resource use.

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Therefore, the most environmentally beneficial options would be composting and anaerobic digestion. Although not explicitly considered in this analysis, these options also offer the added benefit of avoiding the consumption of new resources. Composting reduces the need for synthetic fertilizers, while anaerobic digestion decreases the reliance on natural gas by producing biogas.

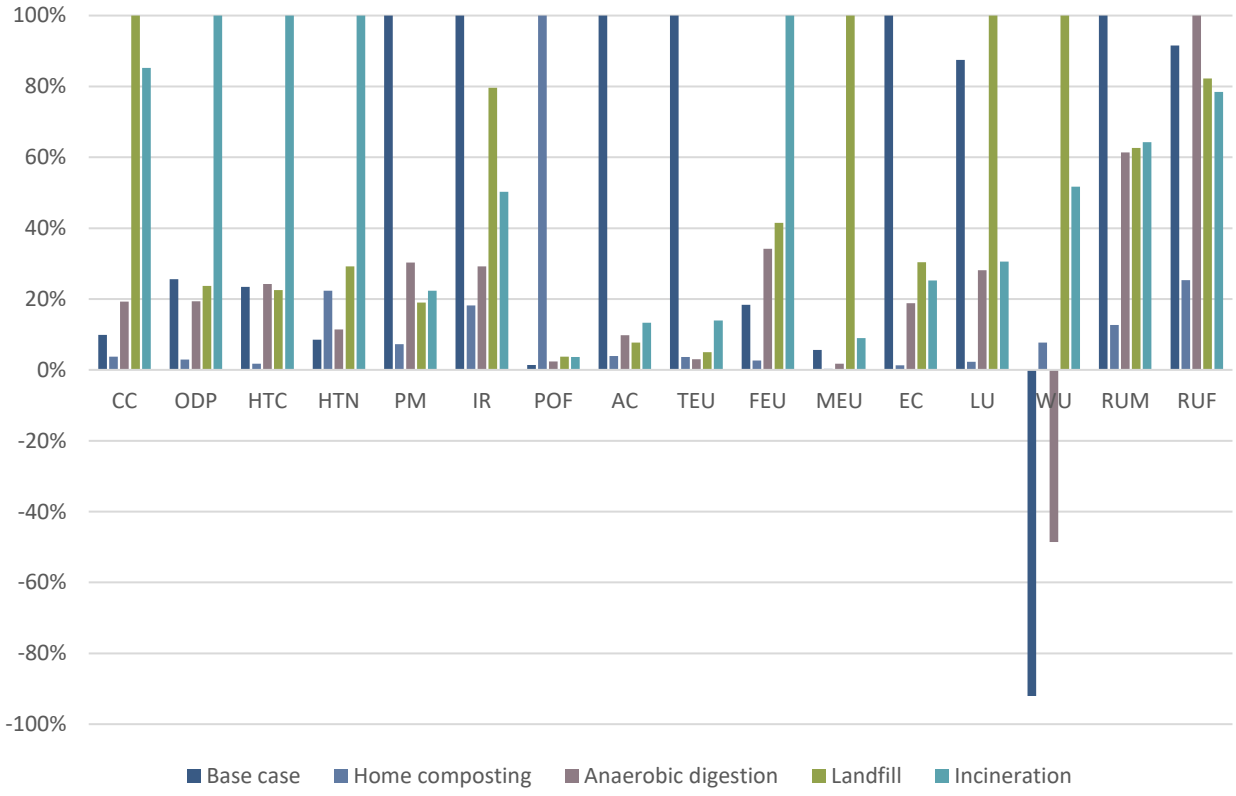

Figure 12. Comparative analysis of different EoL options for garbage bags per FU. Climate change (CC), ozone depletion (ODP), non-cancer and cancer human toxicity (HTN and HTC, respectively), particulate matter (PM), ionizing radiation (IR), photochemical ozone formation (POF), acidification (AC), terrestrial, freshwater and marine eutrophication (TEU, FEU and MEU, respectively), freshwater ecotoxicity (EC), land use (LU), water use (WU), minerals and metals and fossil resource use (RUM and RUF, respectively).

3.5 Conclusions and recommendations

In this case, study analysing the environmental impacts of using different compostable garbage bags, several clear conclusions can be drawn. When comparing the weight of materials needed to achieve a bag of the same capacity, paper bags weigh more than plastic bags, leading to higher resource consumption in the production of the bag. This results in a significantly higher impact during manufacturing due to high energy consumption and chemical treatment required for pulp paper. Additionally, in their use as compostable garbage bags, another critical point is the susceptibility of the material to moisture, which may cause the bag to break prematurely, failing to meet its intended purpose and potentially necessitating container washing, thereby affecting water consumption more significantly.

In the case of compostable garbage bags, the high impact values are attributed to the PBAT manufacturing phase, which was similar to the observations made for biodegradable mulch films. As demonstrated in the

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previous case, increasing the percentage of biogenic carbon through a biomass balance approach can reduce environmental impacts significantly, making the product a more sustainable option.

When assessing the impacts of microplastics, it was demonstrated that the use of non-biodegradable materials for composting has an impact on terrestrial ecosystems, as the accumulation of persistent microplastics increases over time. Additionally, sensitivity analysis confirms that composting and anaerobic digestion techniques are more beneficial options compared to incineration and landfilling toward a more sustainable world. This underscores the importance of biodegradable materials that can undergo a second lifecycle, reducing environmental impacts and contributing to a circular economy.

As in the previous case study, it is necessary to continue researching the formation of microplastics and their potential impacts to obtain more primary data and integrate them into the other impact categories in LCA studies. This is crucial for having a comprehensive view that allows for complete decision-making without overlooking any data. Therefore, it would be interesting to continue evaluating their impacts considering more materials, their composition (i.e. additives), and the environment in which they are released. For now, this study has demonstrated that the impacts of microplastics should not be overlooked, as they are significant. Additionally, although environmental impact assessments of a product usually focus on the category of climate change, it is important to cover other categories, such as the mentioned toxicity, to have a full understanding of the results.

In conclusion, this study provides an examination of the environmental impacts throughout the life cycles of conventional plastic, biodegradable plastic, and paper trash bags. It reveals that paper turns out to be the least favourable option due to its higher production impacts, while plastic bags show lower production impacts. When considering end-of-life scenarios, biodegradable options demonstrate a reduced footprint on ecosystems, highlighting their potential environmental benefits.

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ANNEX I *PHYSICAL EFFECTS ON BIOTA*

To determine the CF of physical effect on biota, it was necessary to develop specific fate, exposure and effect factors for the LDPE.

The CF has units of $\frac{PAF \cdot m^3 \cdot day}{kg_{emitted}} = \frac{CTU_e}{kg_{emitted}}$, which is the same as the ecotoxicity impact category in the PEF method. Note that PAF stands for the *Potentially affected fraction of species* and CTU_e represents the *Comparative toxic unit for Ecosystems*.

1. Fate factor

The work conducted by Corella-Puertas et al. (2023) served as a reference for developing the FF. In their study, the FFs vary based on the type of polymer, shape (sphere, cylinder or film fragments) and size (1, 10, 100, 1000, 5000 µm). For film fragments, the size specifically refers to their thickness.

The FF comprise the main microplastic fate mechanisms of degradation and sedimentation, which take place in a single sub-compartment of the marine environment, including both surface water and the water column. Following the USEtox framework, it was assumed that the marine sub-compartment is a homogeneous box at steady-state. In this case, FF can be calculated by inverting the rate constant matrix K (which was renamed as $k_{water,total}$ to adapt it to our case study), as shown in **Equation 3.** The unit is $kg_{in\ compartment} \cdot day \cdot kg_{emitted}^{-1}$.

$$FF = -K^{-1} = -k_{water,total}^{-1} \qquad \textbf{Equation 3}$$

The rate constant encompasses both degradation and sedimentation mechanisms, as detailed in **Equation 4**, where the degradation rate constant is expressed as $k_{degradation}$ and the sedimentation rate constant as $k_{sedimentation}$, both in units of $kg_{mass\ loss} \cdot (kg_{in\ compartment} \cdot day)^{-1}$.

$$k_{water,total} = k_{degradation} + k_{sedimentation} \qquad \textbf{Equation 4}$$

Both rates are further described below. It is important to note that the degradation rate depends on polymer type, size and shape, while the sedimentation rate depends solely on the polymer type.

1.1. Degradation rate

To develop this rate, Corella-Puertas et al. (2023) assumed that the degradation occurred perpendicular to the surface of the particle. Note that degradation is defined as the loss of total mass of plastic litter over time because of the deterioration of plastic properties (for example due to UV irradiation or heat which results in oxidation, chain scission or cross-linking, and ultimately mineralization). Based on a common structure of degradation rate (**Equation 5**), the authors adapted it for an LCIA study.

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$$-\frac{dm(t)}{dt} = SA(t) \cdot k_{SSRD} \cdot \rho \cdot f_C \qquad \textbf{Equation 5}$$

Where:

- $\frac{dm(t)}{dt}$ is the differential mass loss per unit of time.
- SA is the surface area (cm^2)
- k_{SSRD} is the specific surface degradation rate (µm/year)
- ρ is the polymer density ($g \cdot cm^{-3}$)
- f_C surface area correction factor, which allows the consideration of the polymer surface roughness and porosity. It is calculated as $f_C = \frac{SA_{actual}}{SA_{theoretical}}$.

To adapt the **Equation 5**, several assumptions were made. It was assumed that, in a steady-state condition, there is a constant influx of micro-particles entering the system, each with a maximum radius r_{max}. Although individual micro-particles degrade and lose mass over time, new micro-particles continuously enter the system to replace the degraded ones, ensuring that the total mass of the system remains constant. Over time, the degradation process reduces the size of these micro-particles. It is assumed that the degradation occurs perpendicular to the surface of the particles at a constant rate. This means that the radius of each particle decreases steadily and uniformly at a constant rate denoted as k_{SSDR}. As a result, the micro-particles gradually reduce to nanoparticles and eventually degrade completely.

The mathematical equations are developed in the Multimedia component 1 of Corella-Puertas et al. (2023). The equation obtained (**Equation 6**) considering the previous assumptions is presented below:

$$k_{degradation|film} = -\frac{2 \cdot k_{SSDR} \cdot f_c}{r_{max}} \qquad \textbf{Equation 6}$$

The k_{SSDR} was obtained using the rate constants proposed by Maga et al. (2022). The factor f_C, which accounts for polymer surface roughness and porosity, is typically set to 1. The r_{max} corresponds to half the initial thickness of microplastic film fragments, using the thickness of the mulch film. The summarized data is presented in the following table.

Table 15. Data used to calculate the degradation rate.

Parameter	Value	Unit
$k_{degradation\|film}$	$-8.04 \cdot 10^2$	$kg_{mass\ loss} \cdot (kg_{in\ compartment} \cdot day)^{-1}$
k_{SSDR}	8.26	$\mu m \cdot year^{-1}$
f_C	1	
r_{max}	7.5	µm

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1.2. Sedimentation rate

The sedimentation rates were calculated assuming first-order kinetics, as shown in **Equation 7**. Due to the uncertainty in sedimentation behaviours, three scenarios—slow, medium, and fast sedimentation—were proposed. These scenarios were defined according to the density of the polymer and its probability of sedimentation (Corella-Puertas et al., 2023). LDPE is considered a medium-density polymer (with a density close to seawater, 0.8-1.1 g/cm³), so sedimentation occurs more slowly than for high-density polymers (Jordan et al., 2016). Therefore, in the fast sedimentation scenario, even large microparticles have sufficient time to undergo significant degradation before being removed from the water sub-compartment by sedimentation.

$$k_{sedimentation} = \frac{-(\ln(A) - \ln(A_0))}{t} \qquad \textbf{Equation 7}$$

Where:

- A_0 is the initial proportion of polymers, set to 100 %.
- A is the proportion of the polymer at time t, calculated as *A - Sedimentation percentage* (%).
- t is the residence time.

The sedimentation rate for the different proposed scenarios is presented in **Table 16.**

Table 16. The sedimentation rate depending on the different scenarios.

Scenario	$k_{sedimentation}$ ($kg_{mass\ loss} \cdot (kg_{in\ compartment} \cdot day)^{-1}$)
Slow	$-1.90 \cdot 10^{-5}$
Medium	$-4.89 \cdot 10^{-5}$
Fast	$-1.26 \cdot 10^{-4}$

2. Exposure factor and effect factor

Following the work done by Lavoie et al. (2022), the exposure and the effect factor were combined into a single factor. On one hand, the exposure factor (XF) is defined as ratio of plastic available for exposure to organisms related to the total quantity of plastic reaching the aquatic compartment, as expressed in **Equation 8**.

$$XF = \frac{kg_{bioavailable\ plastic}}{kg_{plastic\ in\ compartment}} \qquad \textbf{Equation 8}$$

A conservative approach was taken, setting this variable to 1, assuming that the entire quantity of plastic in the compartment is available for exposure to organisms.

On the other hand, the effect factor, which represents the ecotoxicity effects of the polymer, was calculated from $HC20_{EC10}^{eq}$ data based on the recommendations provided by the USEtox group (Owsianiak et al., 2023). Specifically, this effect factor is derived as the 20th percentile of a species sensitivity distribution (SSD), constructed using measured EC10 values.

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$$EF = \frac{0.2}{HC20_{EC10^{eq}}}$$ **Equation 9**

The $HC20_{EC10^{eq}}$ was derived from the arithmetic mean and standard deviation of log-normally distributed $EC10^{eq}$ for the considered species ($SSD\mu_{logEC10^{eq}}$ and $SSD\sigma_{logEC10^{eq}}$, respectively), and the z-value of this distribution corresponding to the 20th percentile, as is shown below (**Equation 10**, **Equation 11** and **Equation 12**):

$$\log HC20_{EC10^{eq}} = SSD\mu_{logEC10^{eq}} + z_{0,2} * SSD\sigma_{logEC10^{eq}}$$ **Equation 10**

$$SSD\mu_{logEC10^{eq}} = \frac{1}{n} * \sum_{i=1}^{n} (\log EC10_i^{eq})$$ **Equation 11**

$$SSD\sigma_{logEC10^{eq}} = \sqrt{\frac{1}{n}\sum_{i=1}^{n}\left(logEC10_i^{eq} - SSD\mu_{logEC10^{eq}}\right)^2}$$ **Equation 12**

The EC10 data were obtained through experimental activities carried out by the UVigo ECOTOX and GEA groups as part of the WP5 LCA task. This data is not presented here due to its confidential nature. Note that, as no primary data were available for mulch film, experimental toxicity data for bags were used. In both cases, plastic films of similar thickness were used, so primary data from the same source were prioritized over secondary data.

To sum up, the EF data obtained for LDPE was 1.25E·10^{-1} PAF·m^3·kg$_{\text{in compartment}}^{-1}$.

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ANNEX II *TERRESTRIAL ECOTOXICITY*

Fate and exposure factors are below developed adapting the USEtox method for *terrestrial ecotoxicity*. This CF also has units of $\frac{PAF \cdot m^3 \cdot day}{kg_{emitted}}$, the same as the ecotoxicity impact category of the EF method.

1. Fate factor

The fate module of USEtox accounts for the removal processes and intermediate transport processes of substances in the environment. Examples of removal processes are biodegradation by micro-organisms, burial into the sediment, leaching to the groundwater and escape to the stratosphere (Fantke et al., 2015). To run the USEtox fate module for organic or inorganic (currently limited to cationic metals) substances, a set of substance-specific input parameters must be provided, as shown in **Table 17**.

Table 17. Input parameters to calculate the fate factor in USEtox.

Parameter	Symbol	Unit
Molar mass	MW	g·mol
Acid dissociation constant	pKa	
Partitioning coefficient between n-octanol and water	K_{ow}	$L \cdot L^{-1}$
Henry's law constant	K_{H25C}	$Pa \cdot (m^3 \cdot mol)^{-1}$
Vapor pressure	P_{vap25}	Pa
Solubility	Sol_{25}	$mg \cdot L^{-1}$
Rate constant degradation in soil	k_{degsl}	$1 \cdot s^{-1}$

To model the fate factor in soil, the assumptions made by Salieri et al. (2021) were followed. For a hypothetical worst-case scenario of a high persistence of microplastics in soil, USEtox was used with a high molecular weight to confer high impact resistance, an extremely low partitioning coefficient between octanol and water (K_{OW}) to represent the non-solubility of plastics, an extremely low Henry's law coefficient (K_{H25C}) to represent that microplastics have no potential of partitioning between air and water, and an extremely low vapor pressure ($Pvap_{25}$), as well as assuming no solubility (Sol_{25}) due their hydrophobic nature.

The degradation rates constant in soil for both LDPE were obtained from Maga et al. (2022). Finally, the FFs obtained was $2.03 \cdot 10^{1}$ $kg_{incompartment} \cdot day \cdot kg_{emitted}^{-1}$.

2. Exposure factor and effect factor

As in section 2, the exposure factor was set to 1 assuming that the entire quantity of plastic in the compartment is available for exposure to organisms. Similarly, the effect factor, which represents the ecotoxicity effects of the polymer, was calculated from $HC20_{EC10}^{eq}$ data based on the recommendations provided by the USEtox group (Owsianiak et al., 2023).

The EC10 data were provided by partners UVI-ECOTOX and UVI-GEA research groups for different terrestrial species. To sum up, the EF data obtained was $8.80 \cdot 10^{-2}$ $PAF \cdot m^3 \cdot kg_{in\ compartment}^{-1}$ for LDPE.

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