13

Life Cycle Assessment of Bio-Based Plastics: Concepts, Findings, and Pitfalls

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13.1 Introduction and Chapter Learning Objectives

The aim of this chapter is to introduce the environmental impacts of bio-based plastics. This chapter begins by a brief recap of the different groups of bio-based and biodegradable plastics (Section 13.2), followed by the principle of LCA (life cycle assessment), its history, international standards and norms commonly practiced, and the role of LCA in various decision-making contexts (Section 13.3). Next, seven LCA case studies of bio-based products are described in Section 13.4. These bio-based products are made from novel bio-based and/or biodegradable polymers. Their environmental impacts are then compared with the petrochemical competitors. The case studies provide the statue-of-the-art understanding of the environmental impacts of bio-based plastics as of 2019–2020. Once we learn about the case studies, I invite the readers to jointly reflect: are the bio-based plastics mentioned in the case studies a game changer?

After completing this chapter, you should understand the following concepts:

– *Recap*. Bio-based and biodegradable
– The initial knowledge of LCA with the focus on the assessment of bio-based innovation.
– The environmental impacts of bio-based plastics: the “knowns” and the “unknowns.”

13.2 “Bioplastics” Is a Confusing Term

There are two dimensions when “bioplastics” are discussed: *biodegradability* and *bio-based content*. These two properties should not be confused. Bio-based polymers are not necessarily biodegradable; vice versa, biodegradable polymers are not necessarily made from biological resources. Yes, this is still confusing. Let us use these two properties to categorize polymers.

Biodegradable Polymers in the Circular Plastics Economy, First Edition.
Edited by Michiel Dusselier and Jean-Paul Lange.

In the first dimension, we use the narrow definition of "biodegradability" by the European Standard EN 13432:2000, i.e. if the polymer has passed the biodegradability test according to the standard, we call it "biodegradable" in this context. If a polymer fails to pass the test, we categorize it as "non-biodegradable." However, it needs to be kept in mind that in reality, biodegradability is not a simple "yes" or "no" property but rather a property depending on the duration of degrading, the shape and size of the material, and the ambient conditions.

In the second dimension, based on whether and to which extent the carbon in the polymer is originated from biological resources, we categorize polymers into three groups: fully bio-based, partially bio-based, and fully fossil fuel-based.

- *Fully bio-based polymers* in these polymers, all carbon (100%) in the polymer chain comes from biomass. For example, both PLA (polylactic acid or polylactide) and PHA (polyhydroxyalkanoates) are made from biochemical conversion of sugar, starch, or vegetable oils. For instance, bio-based PE (polyethylene) is commercially produced via dehydration of bio-ethanol [1]; bio-based PP (polypropylene) can be made from cracking bio-based naphtha-like oil made from hydrotreatment of used cooking oil [2]. In this category, we can further distinguish these polymers based on their biodegradability property into two groups: (i) *fully bio-based and biodegradable* polymers: for example, PLA, PHA, and starch plastics, and (ii) *fully bio-based and non-biodegradable* polymers: for example, bio-based PE and bio-based PP; they are identical polymers with petrochemical PE and PP and offer the same functionalities and applications. The only difference is the origin of carbon.
- For *partially bio-based polymers*, the origin of carbon is a mixture of bio-based and fossil fuel-based. Also, some of these polymers are biodegradable and some are not. This group can again be further categorized into two subgroups:
  (1) *Partially bio-based and fully biodegradable polymers*. For example, thermoplastic starch is often blended with biodegradable polyesters (PBAT, polybutylene adipate terephthalate, or PBS, polybutylene succinate) to create desired strength, which offers wide applications such as film extrusion and injection molding. Such a blend consists of partial bio-based carbon and is fully biodegradable.
  (2) *Partially bio-based but non-biodegradable polymers*. For example, partially bio-based PET (polyethylene terephthalate), widely used to make beverage bottles, is made from bio-based ethylene glycol co-polymerized with petrochemical terephthalic acid. About 30% of the mass in the polymer originated from biological resources and the remaining is fossil fuel-based. Partially, bio-based PET is an identical polymer as petrochemical PET; they have the same chemical and physical properties and offer the same applications.
- *Fossil fuel-based but biodegradable polymers*. These polymers are mostly polyesters such as PBAT and PBS. They are made from fossil fuels but are designed to biodegrade; the carbon has 100% fossil fuel origin and the polymers are able to degrade via biological processes.

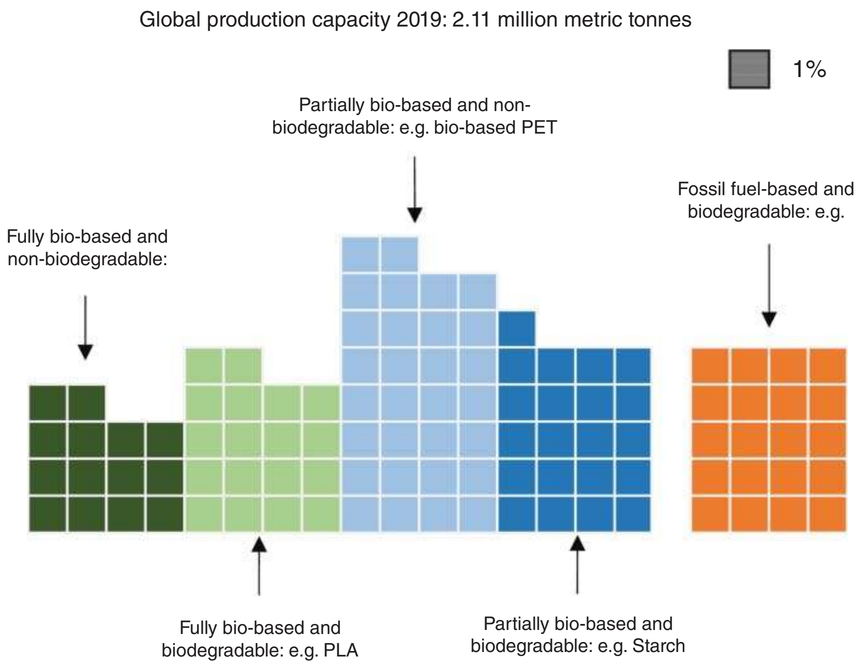

Figure 13.1 Share of global production capacity of bio-based and biodegradable plastics in 2019. Source: Compiled based on the data published by European Bioplastics [3].

In short, “bioplastics” refer to a mixed group of plastics made from biodegradable polymers *or* bio-based polymers, *or* sometimes both (see Figure 13.1). In 2019, the installed production capacity for these emerging groups of plastics was about 2.1 million metric tons (Mt) [3]. They fulfilled less than 1% of total polymers demanded presently in view of the annual global polymer production of 300–380 Mt. The share looks small, but it results from a very fast growing sector. In the past decades, the production volume has been increased drastically from 0.35 Mt in 2007 [4] to 2.1 Mt in 2019. This represented a 17% annual growth rate, whereas the global polymer market grew approximately 4% p.a. in the same period. The market demand is still growing strongly, primarily driven by two major factors: public concerns of poorly managed non-degradable plastic waste and the pressure from decarbonization of the material sector to combat climate change.

Many of these innovative bio-based plastics have the potential to offer GHG (greenhouse gas) emission savings, but they could also have other environmental trade-offs, and the savings can also depend on various factors, e.g. agricultural land management, energy and material efficiencies of production, responsible consumers, and effective waste management. To understand these potential trade-offs and to identify the key factors before making a decision, are more and more important for policy makers, company strategists and even for individuals. For this reason, the environmental impacts of the novel polymers need to be evaluated in a systematic way. LCA is a tool to carry out such evaluations.

Now, we know that "bioplastics" is not one type of plastic but has a myriad of polymers. When the environmental impacts are discussed, it is important to distinguish the type of polymer, their bio-based origin, the manufacturing processes, the logistic chains involved, and their end-of-life (EoL) options. These will be discussed in depth in Section 13.4 of this chapter. Before we go in depth into the case studies, we need to first understand the basic principles of LCA. In the following section, the method LCA is briefly introduced.

13.3 LCA in a Nutshell

LCA is a tool to assess the environmental impact of a product or a service through the entire life cycle of that product or service. The method is standardized by the international standardization organization (ISO). As of 2020, the valid standard is ISO 14040 series published in 2006 [5, 6]. LCA is probably the most applied tool to understand the environmental sustainability of a product.

13.3.1 Concept and a Brief History

The concept of LCA is straightforward. In LCA, the environmental impacts of a product are accounted for including all steps in the life cycle, i.e. raw material and resource extraction, chemical conversions, manufacturing steps, all logistic services in-between, storage, retailing, use phase, and EoL phase. In every step, energy, materials, and utilities are required, and emissions and waste are generated. In an LCA model, these data and information are gathered for all the steps through the life cycle. The resources were extracted from the environment and sent to the technosphere, and emissions released from the technosphere to the environment are quantified and then translated into environmental impacts (see Figure 13.2).

The earliest LCA-like research was probably the "Resource and Environmental Profile Analysis" carried out by the Midwest Research Institute for the Coca Cola company in 1969–1972 [7]. The energy crisis in the early 1970s strongly stimulated the research on energy balances and promoted system thinking, especially for the energy-intensive sectors. In 1977, the Swiss "Ecoscarcity" method was published; this was the first life cycle impact assessment method. In 1992, the first scientific guidance to conduct LCA, the Centrum voor milieuwetenschappen Leiden, in English: Institute of Environmental Sciences (CML) guide, was published by the University of Leiden. The world did not wait too long before the first international standards about environmental system management were made available. The ISO published the first 14 000 series in 1997. The current standards in use (ISO14040/44) were published in 2006 and are still valid to date. An important milestone in LCA was probably the International Reference Life Cycle Data System (ILCD) Handbook, published by the European Commission Joint Research Center in 2010 [8], where very detailed guide for conducting, reporting, and reviewing LCA is provided. This document is often considered by many practitioners as the "cook book."

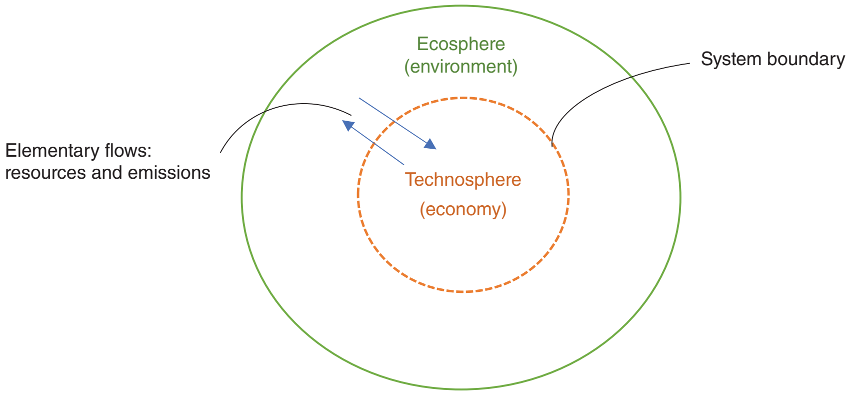

Figure 13.2 LCA is a tool to assess the environmental impact of the life cycle of a product by understanding the environmental interventions between the technosphere and the eco-sphere.

In 2013, the European Commission published the Product Environmental Footprint (PEF) method and the Organizational Environmental Footprint (OEF) method as one of the important actions of the EU initiative of Single Market for Green Products [9]. This document provides a common and consistent LCA methodological guidance for companies across different EU member states to market their products as environmentally friendly. For different markets and sectors, specific rules are made using the EF (environmental footprint) methods as the basis. These rules are called "Product Environmental Footprint Category Rules" (PEFCR) [10]. This is an important step forward to enable companies to use LCA to manage the supply chains and to appropriately market their products. In Section 13.4 of this chapter, we will introduce seven LCA case studies following some of the recommendations of the latest PEFCR.

In the past decades, LCA has been widely applied in many areas including but not limited to cooperate strategy, purchase decisions for individuals, corporates as well as public sectors, long-term policy recommendation, product and process eco-design, and supply chain management. The European Commission considers LCA as the "best framework for assessing the potential environmental impacts of products" [11].

13.3.2 Procedure, Jargons, and Sciences Behind

There are four steps to conduct an LCA, i.e. (i) *goal and scope definition*, (ii) *inventory analysis*, (iii) *impact assessment*, and (iv) *interpretation* [5]. In this section, the basic concepts and the most used terms are explained. It is important to understand some of those "jargons" because these are the terms used by professional practitioners and decision-makers all over the world to communicate LCA. It is also important to know what should be expected during each of these steps when we read, review, and interpret an LCA.

13.3.2.1 Goal and Scope Definition

Like any scientific research, the first step involves the research motivation. The ISO standard defines four requirements for the goal definition in an LCA (see Box 13.1). The motivation of the research is reflected by elements 2 and 3 (in Box 13.1). A well-reflected motivation is crucial for an LCA: the motivation of LCA determines, among many decisions during the research, the LCA decision context, the analytical scope, the types of inventory model, the choice of allocation method, tolerated data uncertainties, and the coverage of impact categories.

Box 13.1 The four elements in the goal definition of an LCA

1. *What* is the LCA about?
2. *Why* is this LCA needed?
3. *Who* are the audience of this LCA?
4. *By whom* is the study conducted?

An important concept in the goal and scope definition is *function unit*. A functional unit is the "quantified performance of a product system for use as a reference unit" [6]. It is a reference unit to be used to compare various options.

After the functional unit is defined, the scope of the LCA should be defined and described. This includes the identification of product systems, defining the geographical scope, temporal scope, technological scope, and the environmental impact categories (see also Section 13.3.2.3) [6].

13.3.2.2 Life Cycle Inventory Analysis (LCI)

In the second step of the LCA, data are gathered and verified and the inventory model is established based on the scope defined in goal and scope definition. The step of life cycle inventory (LCI) often represents one of the most time-consuming step for practitioners.

This step starts with the understanding how things are made. A flow sheet is often drawn in the first step to reflect the understanding. Figure 13.3 shows a simplified example of a flow sheet. A flow sheet consists of many processes in the life cycle of a product. The smallest process is called *unit process*. Unit processes are connected into a system that delivers the quantified function (i.e. functional unit) defined earlier. Such a system is called *product system* in LCA. *System boundary* is drawn to separate the economic activities (inside of the system boundary) from the environment (outside of the system boundary). The system boundaries can be adjusted to the question to answer (see Box 13.2). Substances that cross the system boundary (e.g. extraction of natural resources as arrows toward inside or emissions to the atmosphere as arrows toward outside) make *environmental intervention*. These substances crossing the system boundary are also called "*elementary flows*." A list of elementary flows that corresponding to the quantity defined by the functional unit forms the *life cycle inventory table*. This table is the aim of LCI. The inventory analysis has a massive

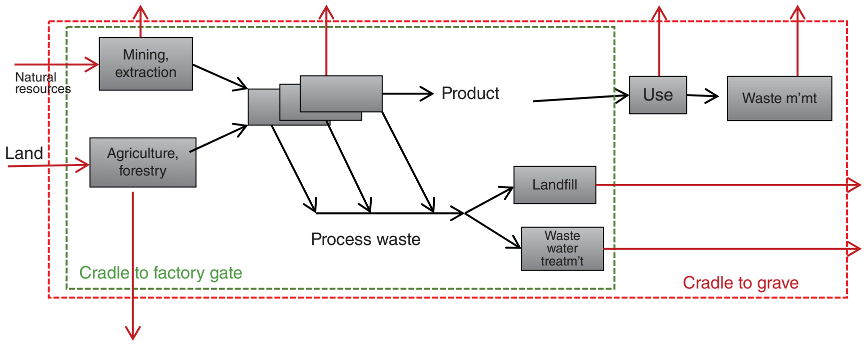

Figure 13.3 An example flow diagram in LCA with different system boundaries.

data processing requirement. The data processing is often carried out by database tools and LCA software.

Box 13.2 Cradle-to-factory gate and cradle-to-grave

A complete LCA should always be a cradle-to-grave LCA including the entire life cycle of the manufacturing, use, and post-consumer disposal phases (see Figure 13.3). However, in reality, many LCAs are conducted with the goal of assisting purchase decision in order to support supply chain management or to identify the significant environmental issues for an intermediate product instead of an end product. In those cases, a "cradle-to-factory gate" or a "cradle-to-user" systems is often analyzed, where the use phase and the EoL phase are not directly relevant for the goals of those LCAs.

13.3.2.3 Life Cycle Impact Assessment (LCIA)

The third step in LCA is life cycle impact assessment (LCIA). In this step, we move from the physical flow analysis in the technosphere to the environmental impact assessment. The aim of LCIA is to translate the environmental interventions (i.e. the result of inventory analysis) into environmental impacts. This is done in two steps: *classification* and *characterization*.

In the *classification* step, the inventory results are categorized into different environmental impact categories. For example, CO_2, N_2O, and CH_4 are GHGs and are classified into "Climate Change"; SO_2 and NO_x make the environment more acidic and are classified into "Acidification." In Appendix 13.A, the commonly used 16 impact categories are listed as recommended by the European Commission's EF's method [9]. Different *impact assessment methods* define different impact categories and their assessment frameworks. Table 13.1 shows a list of selected impact assessment methods commonly used by LCA practitioners.

In the characterization step, the classified inventory results are then quantified into environmental impacts. This step is determined by characterization models. The characterization models are cause–effect models. The outcome of these models is the so-called characterization factors, namely, a set of conversion factors to translate

Table 13.1 Commonly used impact assessment models and their references.

Name of the method	Developed by	Number of impact categories	Web links[a)]
EF	Joint Research Center, European Commission [12]	16 mid-point	https://eplca.jrc.ec.europa.eu/EFVersioning.html
ReCiPé	NL National Institute of Public Health and the Environment [13]	18 mid-point and 3 end point	http://www.rivm.nl/en/life-cycle-assessment-lca/recipe
CML	University of Leiden [14]	11 (baseline) mid-point	http://www.universiteitleiden.nl/en/research/research-output/science/cml-ia-characterisation-factors
TRACI	US EPA [15]	8 mid-point	http://www.epa.gov/chemical-research/tool-reduction-and-assessment-chemicals-and-other-environmental-impacts-traci
LIME	LCA Society of Japan [16]	7 end point	https://lca-forum.org/english/lime

a) Many LCIA methods constantly update characterization factors based on the latest knowledge in environmental impact assessment. Always check the latest available version when practicing LCA.

the inventory results into environmental impacts. In detail, an environmental impact is calculated by multiplying the inventory flows (emissions and resources) with the corresponding characterization factors of that impact category. The total environmental impact of that category is the sum of the impacts of all inventory flows that are classified to that impact category.

13.3.2.4 Interpretation

Like many scientific methods, LCA requires iteration. This means that after new information is obtained, the previous step(s) need to be revisited, and assumptions and data quality are reflected and adjusted wherever needed. The interpretation could occur in any steps in LCA. Most importantly, the interpretation of LCA results requires to answer the research questions and to evaluate and reflect the certainty and the quality of the answers. In this step, the consistency and completeness of the study are checked, and the analyses of the results are expected. Finally, like any scientific research, based on the results and interpretation, conclusions are drawn, limitations are pointed out, and recommendations are made.

13.4 LCA Case Studies of Seven Single-Use Plastic Items Made from Bio-Based Resources: Highlights and Lessons Learned

13.4.1 Background, Aim, and Scope of the BIO-SPRI Study

In this section, we will introduce seven LCA case studies of single-use plastic items made from bio-based resources and compare them with the petrochemical counterparts. The seven cases are summarized in Table 13.2. The seven cases cover three major commercialized innovative bio-based polymers, namely, bio-based PET (partially bio-based but not biodegradable), PLA (bio-based and biodegradable), and starch plastics (partially bio-based and biodegradable).

Table 13.2 Overview of the seven case studies, functional units, and product systems.

Case studies	Functional units	Bio-based product systems	Reference systems: petrochemical counterparts
Beverage bottles	Packaging of water in 100 bottles each 0.5 l providing a shelf life of at least 9 mo	Partially bio-based PET	PET
Single-use drinking cups	1000 single-use drinking vessels each for 200 ml of cold beverage	PLA from corn and sugarcane	PET and PP
Single-use cutlery	1000 sets of disposable cutleries each consisting of a knife, a fork, and a soup spoon		PS
Food packaging films	100 m^2 of transparent film packaging for fresh vegetables for 1 wk		PP
Horticultural clips	45 000 single-use clips used for horticultural purposes for 1 ha of land	Starch plastics[a)]	PP
Agricultural mulch films	Providing field mulching for 1 ha of land for 6 mo		LDPE
Single-use carrier bags	One single-use all-purpose lightweight plastic carrier bag with a volume of 20 l and 10 kg weight holding capacity		LDPE

a) Starch plastics is a generic term for a family of blends. For the three cases based on starch plastics, the compositions of these starch plastics are different.

The study was asked by the EU policy makers from the Commission's Directorate-General Research and Innovation (DG-RTD). The full project "Support to Research and Innovation Policy for Bio-based Products" (abbreviated BIO-SPRI) was conducted in 2017–2018. The policy makers asked for science-based evidence: are there environmental benefits if bio-based plastics are promoted? What are the critical environmental trade-offs? In the BIO-SPRI project, the seven cases were selected out of 20 candidate cases based on five criteria: market potential, promise for deployment, available LCA data, innovation, and potential sustainability benefits.

The aim of BIO-SPRI is to provide facts and evidence to support future policies about bio-based plastics, particularly in the context of supporting the implementation of the EU Plastic Strategy [17]. The goal of LCA was then further specified in twofolds: (i) to identify the key environmental hot spots of innovative bio-based plastics and (ii) to compare the environmental impacts with the petrochemical counterparts.

The research project, in a 745-page report, was published in February 2019 [18]. In this section, we will give a summary of this study and highlight the most important data and assumptions, results, and interpretation. The limitations of the study, the reflection of the methodology, and the lessons learned are summarized in Section 13.5.

The ISO standards [5, 6] were followed in general. Specifically, the study followed as much as possible the PEFCR guidance [10]. It needs to be kept in mind, even with the help of methodological guidance and protocols, in real LCA practice, there are often situations that are more complexed and require programmatic solutions. Some of these situations are highlighted in this section.

All cases were analyzed with the scope "**cradle to grave**" (see Figure 13.3) to align with the goal of LCA, i.e. EU-level policy support. The **geographical scope** includes the products sold, consumed, and disposed of in Europe. The supply chain of the product however, reflected the status quo reality, which is often global. For **technological scope**, only commercialized technologies that have been deployed by the industrial scale production were analyzed. Technologies under development were excluded from the scope. The **temporal scope** reflected the status quo production, use, and waste management, with a short-term vision into 2020–2025.

The LCAs were modeled in two software: SimaPro version 8 (simapro.com) from cradle to user and EASETECH (easetech.dk) for the EoL waste management. The background data were taken from the state-of-the-art LCI databases as of 2018, such as Ecoinvent (version 3.4), Agrifootprint (version 3), Gabi (version 2017), PlasticsEurope's Eco-profiles (2012–2017), and EASETECH. All detailed data and assumptions can be found in the report of the BIO-SPRI study [18].

The inventory models were built on a modular basis. For bio-based product systems, five modules were modeled separately for (i) biomass feedstock acquisition, (ii) polymer and material production from biomass, (iii) plastic conversion into end products, (iv) distribution to end user, and (v) the EoL waste management. The impact of the use phase of the packaging products was considered negligible.

Sixteen *environmental impact categories* recommended by the PEFCR (see Appendix 13.A) were analyzed. The study also included the *normalization* and *weighting* steps (see Box 13.3) in order to come to a single score of environmental impacts to simplify and to facilitate the communication. The normalization and weighting factors applied in this LCA can be found in Appendix 13.B.

Box 13.3 Normalization and weighting in LCA

Normalization and *weighting* are optional steps in LCA, according to the ISO [6]. Normalization is a step to normalize the LCA results (e.g. impacts at the mid-point) to the total environmental impacts of that impact category in a defined region and time. Appendix 13.B shows an example of normalization factors for per capita EU citizen using the EF impact assessment method developed by the European Commission in 2020. Normalization step can be done by dividing the mid-point LCA results by the normalization factors. The normalized results show the relative contribution of the environmental burden associated with the functional unit investigated to the overall environmental burden in that region. Normalization does not involve subjective opinions.

Weighting, on the contrary, involves subjective opinions about the relative importance among different impact categories. Sometimes in policy making and communication, it is difficult to come up with an overall conclusion if the environmental impact is higher in some categories and lower in other categories. To determine the weighting factors, multi-criteria approaches with stakeholder engagements are often used. The weighting factors are influenced by public concerns as well as the concerns from specific stakeholders (i.e. it depends on from whom the opinions were collected). In Appendix 13.B, an example of the weighting factors based on per EU citizen is given using the EF impact assessment method [19]. In that example, the EU citizen considered climate change a more urgent issue than human toxicity.

13.4.2 Key Findings

In this section, we first discuss the key findings of the seven bio-based products. The comparison with their fossil fuel counterparts can be found in Section 13.4.3.

Figure 13.4 shows the *normalized* and *weighted* LCA results for seven bio-based products, breaking down into major life cycle stages. For all seven cases, the "polymer and/or material manufacturing" phases play a dominant role. The plastic conversion and distribution also have significant contributions to the overall impact. The contribution from the biomass feedstock acquisition is high for beverage bottles (partially bio-based PET) but less significant for the other six products. The impacts from the EoL waste management stage depend strongly on the type of waste management assumed, the type of application, and the properties of the polymers, which is further interpreted in Section 13.4.2.4.

For all seven bio-based products, the most important environmental impact categories identified are climate change, resources depletion (fossil fuels), and human

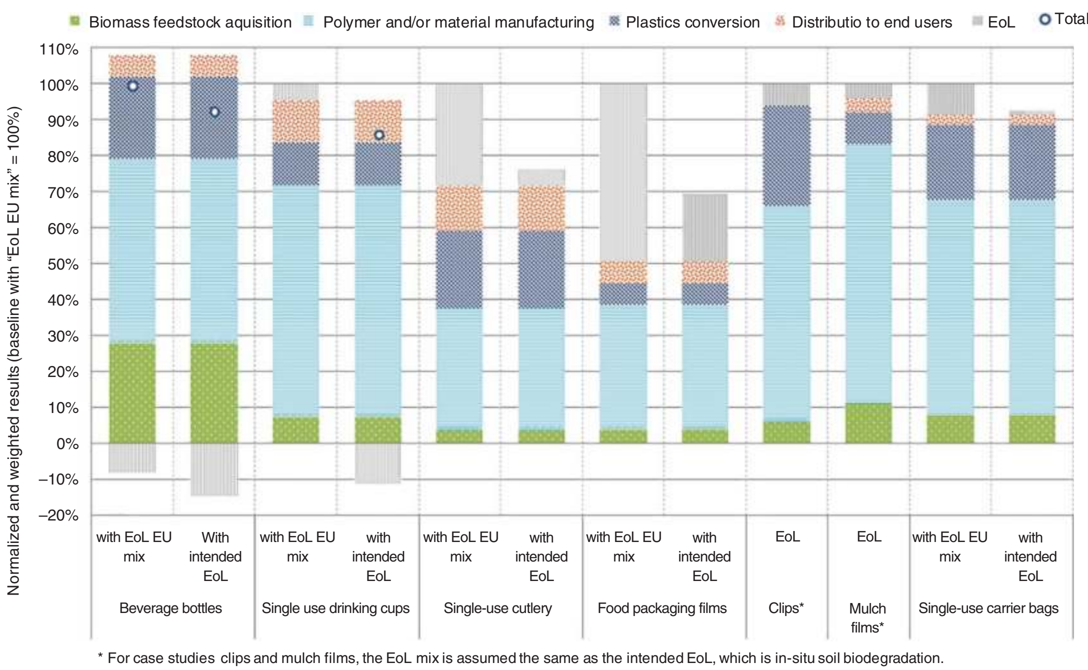

Figure 13.4 Cradle-to-grave environmental impacts of seven bio-based products, weighted results based on 16 PEFCR impact categories, breaking down into major life cycle stages. For each case, the baseline results (with EoL EU mix, see the detailed assumptions in Section 13.4.2.4) are then compared with the results based on intended end-of-life scenarios (intended EoL) to show the effects of different EoL waste managements. LCA results here are without taking into account the effect of direct or indirect land use changes. Source: COWI, DG RTD, and Utrecht University [18] / European Union / CC BY 4.0.

toxicity (see Figure 13.5). Together, these three impact categories account for approximately 30–60% of the total cradle-to-grave environmental impacts of the bio-based products. These three impact categories are highly associated with (predominantly fossil fuel) energy use in the manufacturing and distribution phases, as well as the direct air emissions from the EoL phase. We should also be aware that climate change is weighed the most urgent environmental issue based on the views provided by the EF method (see Appendix 13.B). In the following sections, the detailed impact breakdown per life cycle stages and the key data and assumptions are explained further.

13.4.2.1 Biomass Feedstock Acquisition

Sugarcane and corn are the two most important crops for the seven bio-based products studied (see Table 13.3). The LCA results presented in Figure 13.4 show that, for six out of seven bio-based products, the biomass production phase does not represent a major impact for the bio-based plastics in this study; in general, it accounts for less than 10% of the overall cradle-to-grave environmental impacts. The exception is the beverage bottles made from (partially) bio-based PET, where the high impact from biomass production (c. 28% of total cradle-to-grave impacts) is primarily contributed by the particulate matter, caused by the open-air cane residue burning in sugarcane harvesting in Brazil. It should be noted while this study was conducted in 2017–2018, Brazil is in the meantime phasing out the burning of cane trash by legislation. By 2031, the burning of cane trash should be fully phased out in the country (São Paulo State Law n. 11241/02) [20]. Thus, the impact of particulate matter is expected to be substantially reduced in the near future for sugarcane ethanol from Brazil.

Apart from the particular matter impact of sugarcane from Brazil, generally sugarcane is considered a more efficient feedstock compared to corn when fermentable sugar is required as the feedstock. For sugarcane, sucrose can be directly obtained by simple mechanical pressing and extraction, with a substantial energy benefit of using bagasse to fuel the milling and fermentation processes. For starch crops, glucose or dextrose needs to be first liberated from starch by wet or dry milling, which involves multiple-step conversions and separations, accompanied by many co- and by-products (e.g. proteins, fibers, and oils). Nevertheless, for the

Table 13.3 Biomass feedstocks used for the seven bio-based products.

Bio-based or partially bio-based polymers/materials	For the bio-based content: biomass feedstock and origin
Partially bio-based PET	Sugarcane from Brazil
PLA	Corn from the USA (67%) and sugarcane from Thailand (33%)[a)]
Starch plastics	Corn from the EU

a) The share represents the production capacities of the two major producers of PLA as of 2018.

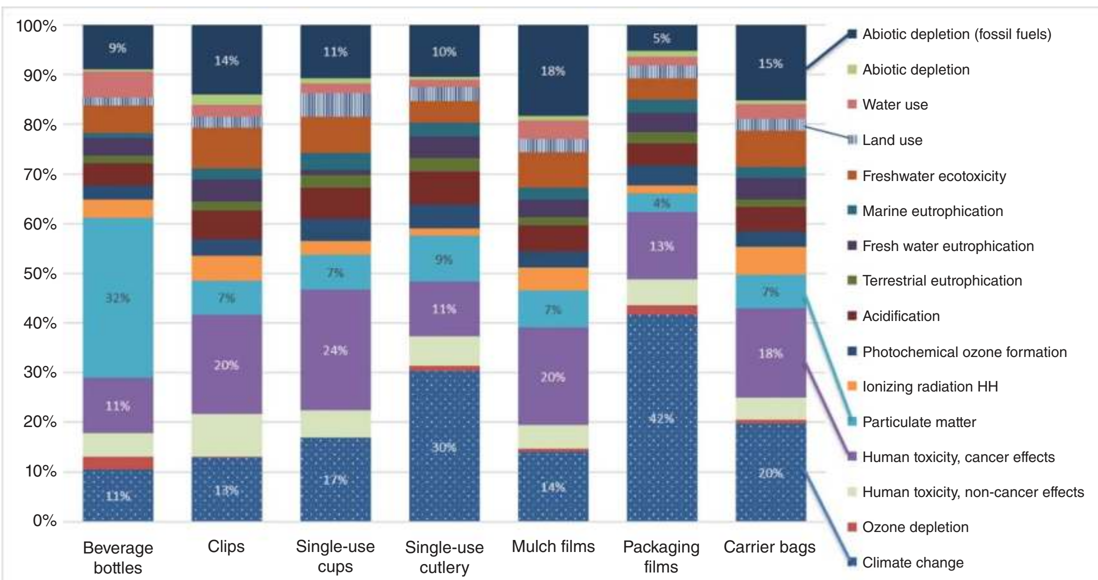

Figure 13.5 Cradle-to-grave environmental impacts of seven bio-based products, breaking down into impact categories. Normalized and weighted results assuming EoL EU mix. LCA results here are without taking into account the effect of direct or indirect land use changes. Source: COWI, DG RTD, and Utrecht University [18] / European Union / CC BY 4.0.

PLA products analyzed in the study, we observed rather negligible differences between sugarcane and corn, especially when the comparison is made from the entire life cycle perspective – i.e. the environmental impact of PLA is dominated by the monomer and polymer production and the material manufacturing phase, not by the impact from biomass acquisition (see Figure 13.4 for single-use drinking cups, single-use cutlery, and food packaging films).

Similarly, the impacts from the biomass acquisition phase for the three starch plastic products, namely, clips, mulch films, and carrier bags, also represent a small share of the overall impacts. For these starch plastic products, starch is utilized in its polysaccharide form – no sugar is required to be liberated. This leads to a relatively low impact from the biomass feedstock acquisition.

For both sugarcane and corn, the geographical differences do not represent as an influential factor because the industries source their feedstock based on the highest available yields in the regions, although bio-based products typically require (more) land than the extraction of fossil fuels. Land use and the associated impacts are often specifically investigated in the literature. In this study, we found that the impacts from land use and water use are rather limited compared to the same impacts from the manufacturing phase (Figure 13.5). This is partly explained by the high yield of the crops, partly explained by the dominance of the monomer/polymer and material production for these bio-based products.

For the impact category "Climate Change," it is important to note that the *biogenic carbon* removals were accounted for in the cradle-to-factory gate models (see Box 13.4), and it assigns a credit to the climate change impact (embedded carbon in the product is taken as a carbon sink: CO_2 is removed from the atmosphere by photosynthesis). At the EoL phase, the embedded biogenic carbon (in CO_2 equiv) was then modeled as biogenic CO_2 emission if fully oxidized or partially oxidized and form CO and/or CH_4 or stored in the compost (no emissions). This choice also results in a relatively low climate change impact for the biomass acquisition phases and a relatively high climate change impact for the EoL phases for all the bio-based products analyzed.

Box 13.4 Biogenic carbon in LCA

For bio-based products, the carbon embedded in the products is originally sequestered by biomass via photosynthesis in the "cradle" phase. This is different from the carbon embedded in the fossil fuel products, which was buried millions of years ago and intensified via geological formations. These two types of carbon need to be distinguished in the inventory analysis in LCA because the climate change impacts caused are different. The current ISO standards do not provide specific rules how to deal with biogenic carbon. Here, we follow the terms defined by a widely used method to calculate the carbon footprint of a product – PAS 2050 (Public Available Specification) (Figure 13.6) [21].

(Continued)

Box 13.4 (Continued)

When the biomass absorbs CO_2 from the atmosphere for photosynthesis, this flow of CO_2 enters the system boundary as an elementary flow (see Figure 13.5). This inbound flow is called biogenic *carbon removal*. When the product is burned at the end of its life and let us assume that all carbon is fully oxidized, CO_2 is released into the atmosphere. This outbound elementary flow is called biogenic *carbon emission*. If a part of the biogenic carbon stayed for a longer period of time in the product, e.g. biogenic carbon in a wooden table, this carbon is called biogenic *carbon storage*.

In inventory analysis, the principle to deal with biogenic carbon flows is to distinguish and report the flows separately [8–10, 21, 22]. The elementary flows of *carbon removals* and *carbon emissions* should be both assessed in the impact assessment for climate change. Note that *carbon storage* is not an elementary flow in LCA but it helps to create the correct overview of the carbon balance.

There are time-related issues about carbon removals (e.g. it takes time for forest to absorb CO_2) and carbon storage (e.g. delayed emissions could lead to reduced global warming effects). The time-related issues are relevant for assessing the climate change impact because climate change is often evaluated within a defined term, e.g. 100-year time frame. We will not expand the issue here. The current LCA norms, such as PAS 2050 and PEFCR, do not support the time-related assessment because the relevant research is still ongoing. The methodology and time-related dynamic databases that can be used for LCA are under development.

In addition, the impacts associated with *direct and indirect land use changes* (see concepts explained in Box 13.5) were investigated in this study. The study modeled the emissions from direct land use change (dLUC) as defined by PAS 2050 as well as the PEFCR guidance [10]. Indirect land use change (iLUC) was modeled using a deterministic model based on the statistical evidence observed in the past 10 years in the world for arable land converted via deforestation (expansion), e.g. using the Food and Agriculture Organization of the United Nations (FAO) statistics and literature knowledge, and to estimate the emissions due to the agricultural intensification, e.g. using the historical fertilizer statistics published by IFA (International Fertilizer Association). iLUC is not a compulsory step currently in LCA (see Box 13.5). It was modeled as an optional component in the study.

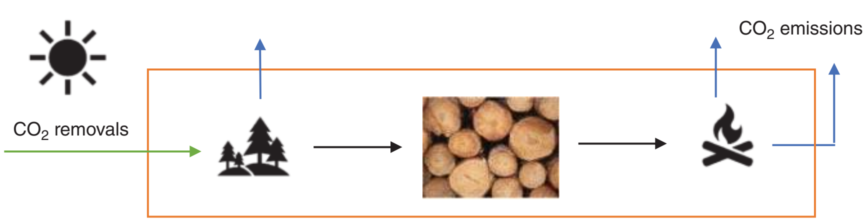

Figure 13.6 The biogenic carbon cycle: removals, emissions, and storage.

Box 13.5 Direct and indirect land use changes

Land use change is another important issue for bio-based production. Soil contains large amounts of organic carbon. When land use changes, the soil organic carbon content will also change. For example, if primary forest, which is organic carbon rich, is converted into arable land for crop production, the soil organic carbon content will decrease. The lost carbon is mostly emitted as CO_2. In LCA, this is called GHG arising from dLUC. The GHG emissions associated with land use changes was about 7% of global GHG emissions in 2017 [23].

Again, the current ISO standards do not provide specific rules about how to deal with emissions from land use changes. Both PAS 2050 and PEFCR recommend to account for GHG emissions related to *dLUC* if the land use change occurred on or after 1 January 1990. For the existing agricultural systems that are older than 30 years, the changes in soil organic carbon can be treated negligible [10, 21].

The concept of *iLUC* reflects the consequential thinking. The increase in demand of bio-based plastics creates pressure to the agricultural system, and some of the agricultural crops currently used for food or feed will be used for polymer production. In that case, the arable land is still an arable land; there is not dLUC attributed to bio-based plastics. However, the demand of food and feed remains the same. To meet the additional demand of bio-based plastics, additional land needs to be converted from primary forest. This is called *indirect land use change*. The carbon emitted from iLUC should be attributed to the bio-based plastic, which drives the additional demand of land. Alternatively, this additional demand could be met by intensifying agricultural outputs by, e.g. increasing yields. This is rather a limited response because yields cannot be increased infinitely. According to FAO statistics [24] from 2000 to 2010, about 85% of the additional demand of land was met by expansion, which leads to land use changes, and 15% of the additional demand was met by agricultural intensification.

In LCA, assessment of iLUC is not required by the current norms and specifications because it requires complexed global demand and supply models and databases. This is an important research area. Many high-resolution databases and methods are currently under development.

For the seven bio-based cases, an average iLUC factor of 4.0 t CO_2 equiv/ha/yr (with a range of 1.2–5.2 t CO_2 equiv/ha/yr) was found. The main impacts affected by the inclusion of iLUC were climate change and photochemical ozone formation. When the effects of land use changes are accounted for, the impacts of biomass production do increase. However, while the share of biomass production in the overall impacts was not dominant in the first place, the additional land use changes share to the overall cradle-to-grave impacts is relatively marginal: on average, 14% (4–17%) for climate change, 10% (4–30%) for photochemical ozone formation, and between 0.01% and 2.4% for all other impact categories.

13.4.2.2 Manufacturing Phase: From Biomass to Polymers, Materials, and End Products

The impacts of the manufacturing phase consist of the impacts from three sub-stages: the production of bio-based polymers (i.e. converting biomass into polymers), the production of (often) fossil fuel-based copolymers, and the plastic conversion step when the final products are formed. The first two sub-stages together are shown as "polymer and/or material manufacturing" in Figure 13.4.

The foreground data, especially for the processes involved in the conversion from biomass to platform chemicals and materials, compounding and plastic conversions, were the primary data collected from the key producers who have a dominant share of that product in the European market. The inventory data reflected the status quo as of 2018. If a product had more than one key producer, the weight average was formulated based on installed production capacities.

Figure 13.4 shows that the manufacturing phase has the highest contribution of all life cycle stage: it accounts for approximately 50% of the cradle-to-grave impacts of nearly all seven bio-based products studied. The contributions from the three sub-stages vary depending on the type of polymers or materials studied and summarized as follows.

- *Bio-based PET* is chemically identical with petrochemical PET. PET is a polyester made from polymerization of monoethylene glycol (MEG) and purified terephthalic acid (PTA). For the current bio-based PET available in the market, MEG is made from bio-based ethylene derived from sugarcane-ethanol, and PTA remains petrochemical based. The bio-based content is approximately 30% (by weight) in PET, representing the fraction of bio-MEG in the polymer. For (partially) bio-based PET, the manufacturing phase accounts for nearly three-quarter of the total impacts (see Figure 13.4), with the highest contributor of polymer production, which includes bio-MEG production, PTA production, and the polymerization process. These three production processes contribute to over 50% of the total cradle-to-grave impacts. The production of PTA, with a fossil fuel origin, has a dominant role. The second most important life cycle stage is the plastic conversion step (bottle blown molding), contributing to 23% of the total impacts. These impacts from the manufacturing phase are dominated by the process energy (heat and electricity) requirements.
- *PLA* (100% bio-based) is produced via polymerization of lactic acid or lactide. The monomers are products of sugar fermentation. For PLA, the conversion from sugarcane or corn into polymer PLA has the dominant impact, accounting for 40–60% of the total cradle-to-grave impacts of the three PLA case studies (i.e. single-use cups, single-use cutlery, and food packaging films). Process energy (electricity and heat) and chemical used in the lactic acid and PLA production are responsible for the major parts of the environmental impacts. The energy consumption is highly associated with the process water and its removal that needs to be dealt with in the biochemical conversion (fermentation of lactic acid). In the studied cases, a lion's share of the process energy (especially heat supply) is fossil fuel based, which contributes to the impacts of climate change and abiotic depletion (fossil fuels). The second most important contributor in the manufacturing phase of PLA is the

plastic conversion steps (e.g. thermal forming of cups, injection molding of cutlery, and film extrusion). Electricity is the key input of these conversion processes. Depending on the fuel mix of the electricity where the conversion step is located, the step could contribute to about 10% of the total cradle-to-grave impacts (e.g. single-use cups thermoformed in Europe) or up to 25% of the total impacts (e.g. cutlery molded in China where the electricity is more carbon intensive than the EU average).

- **Starch plastics** (partially bio-based) are starch thermoplastics (bio-based and biodegradable) blended with biodegradable petrochemical polymer(s). The bio-based content varies considerably depending on the formulation of the starch plastics for specific applications. For the three starch plastic applications studied, the bio-based contents range from 33% to 68%. The environmental impacts of studied starch plastics are dominated by the material manufacturing phase (c. 80% of the total cradle-to-grave impact), of which, the impacts from the fossil fuel-based co-copolymers are dominant (c. 50–60% of the total cradle-to-grave impacts). The impacts from plastic conversions (e.g. injection molding and film extrusion) account for 20–30% of the total impacts.

13.4.2.3 Distribution to End User: Impacts from Transportation

The impacts from transportation are in general not substantial (approximately 3–6% of the cradle-to-grave impact, see Figure 13.4). The exceptions are when the global supply chain is very long. For instance, based on the current supply chain of biomass, polymers, and end products, the bio-based carbon in the PLA cutlery travels nearly three quarters of the globe to come to the demand center – Europe. In this case, the impact of transportation is no longer negligible: about 13% of total impacts of the PLA cutlery are contributed by the transportation and distribution of the intermediates and the final product.

From cradle to user, the *energy requirements* along the chain, from process heat and electricity to transportation fuels, together play an important role. Energy consumption directly leads to the impact of fossil fuel depletion and climate change caused by CO_2 emitted from the combustion of fossil fuels (e.g. for heat and electricity). Combusting fossil fuels also lead to the impact of photochemical ozone formation (caused by nitrogen oxides emitted during fuel combustion) and particulate matter (dust emitted from coal combustion).

13.4.2.4 End-of-Life (EoL) Post-consumer Waste Management Scenarios

Modeling the EoL of niche products was challenging. The selected products are commercialized and have been produced on a large scale. However, compared to the petrochemical counterparts, the market share of these bio-based alternatives are still very small. European Bioplastics Association estimated about 1% of all plastics produced are bio-based (see Section 13.2). In the waste management stream, this amount has no statistical significance. Therefore, there was no so-called “status quo” primary data to model the EoL waste management of these bio-based products. The bio-based plastics waste could end up in the regular waste where they are intended for (e.g. recycling or composting) or where they are not expected (e.g. landfill or waste incineration). There is little empirical or statistical evidence to support either of these assumptions. In the BIO-SPRI project, to tackle this challenge, two EoL scenarios were distinguished:

(1) "EoL EU mix," where the plastics waste was assumed to end up in an average (mixture of) treatment that is most likely to happen to that packaging product. For example, for beverage bottles, both the bio-based and the petrochemical ones, the used bottles are likely to be disposed of for recycling, in a landfill or being incinerated in Europe (different member states have different waste collection schemes). The shares of the different waste treatments were determined using our exiting knowledge (from e.g. EuroStat) of the waste treatments of PET bottles in the EU.
(2) "Intended EoL," where the waste was assumed to end up in the ideal and desired waste treatments. For example, ideally, all PLA applications should all be either composted or recycled. Note that this is an extremely favorable ideal situation.

The reality is hard to predict. It highly depends on the waste management schemes in different member states as well as how well the consumer behaviors are steered. It is anticipated that the reality is probably somewhere in between the two scenarios.

An overview of the two EoL scenarios for the seven case studies are summarized in Table 13.4. The most foreground data were taken from the EU or country-level statistics and state-of-the-art literature data. The background data for the EoL modeling were taken from EASETECH databases and the modified Ecoinvent database. For the reference systems with petrochemical counterparts, in this book chapter, we only present the results of the average "EoL EU mix" as the benchmark (see Section 13.4.3).

It needs to be highlighted that for the two cases, "single-use cutleries" and "food packaging films" food residues were also assumed to be included in the product systems. Several studies [26–28] have reported that food leftovers often follow the packaging products and hence were often end up in the same EoL treatments as the packaging products. The impact of treating food residues was accordingly included in the system boundary.

The impacts of the *EoL* of the bio-based products studied are less dependent on the energy requirements of waste treatments but highly dependent on the type of waste treatment and the type of the plastic products. Except for PET bottles, the other six bio-based cases are all for biodegradable applications for which the materials are the newcomers for the existing waste management systems. For the six case studies where biodegradable materials are involved, the uncertainty of the EoL assessment is considered high.

- For *beverage bottles*, because PET is a currently highly recycled polymer in Europe (more than 60% collected for recycling in 2016 in Europe), the EoL waste treatment of PET bottles receives a net credit (see Figure 13.4) because they avoid the production of a virgin PET polymer. Based on the weighted LCA results for all 16 impact categories, the most favorable EoL technology of PET is recycling. By improving

Table 13.4 Key assumptions of the two scenarios of EoL waste management[a].

Case studies (bio-based vs. reference)	Bio-based product systems		Petrochemical references, "EoL EU mix" as the benchmark EoL
	EoL EU mix	**Intended EoL**	
Beverage bottles (bio-based vs. Pchem. PET)	Recycling 60% MSWI 20% Landfilling 20%	Recycling 100%	Recycling 60% MSWI 20% Landfilling 20%
Single-use drinking cups (PLA vs. PET, PP)	MSWI 39% Landfilling 31% Recycling 15% Industrial composting 15%	Industrial composting 50% Recycling 50%	Recycling 30% MSWI 39% Landfilling 31%
Single-use cutlery (PLA vs. PS)		Industrial composting 100%	
Food packaging films (PLA vs. PP)		Industrial composting 100%	
Horticultural clips (starch plastics vs. PP)	*In situ* soil biodegradation 100%	*In situ* soil biodegradation 100%	MSWI 56% Landfilling 44%
Agricultural mulch films (starch plastics vs. LDPE)			Recycling 5% MSWI 53% Landfilling 42%
Single-use carrier bags (starch plastics vs. LDPE)	Industrial composting 30% MSWI 39% Landfilling 31%	Industrial composting 100%	Recycling 30% MSWI 39% Landfilling 31%

a) Data presented in this table were mostly based on European statistics [25]. In 2017, the average treatment for plastic waste within the EU member states was reported to be 39% MSWI (municipal solid waste incineration with and without energy recovery), 31% landfilling, and the remaining 30% was collected for mechanical recycling. We used these average shares to simplify the EoL modeling. The individual EoL technologies were modeled as an average European technology, e.g. indicating that incineration includes both with and without energy recovery and anaerobic digestion includes all combinations of dry/wet, thermophilic/mesophilic, and with/without post-maturation. It should be noted that there are differences between polymer types in terms of the actual recycling rates, which depend on the economic values of the recycled polymer and the recycling technology available for the polymer.

the 60% collection rate for recycling of today, to an idealistic 100% collection in the future, the overall impacts of the bio-based PET bottle could be further reduced by 7%. It should be noted that the difference between (partially) bio-based PET and petrochemical PET is the source of MEG. Both PET bottles do not differ in terms of EoL waste management.

- For *single-use cutleries and food packaging films*, where the reference flow includes food leftovers, the mixed EoL has a significant or even dominant contribution: this

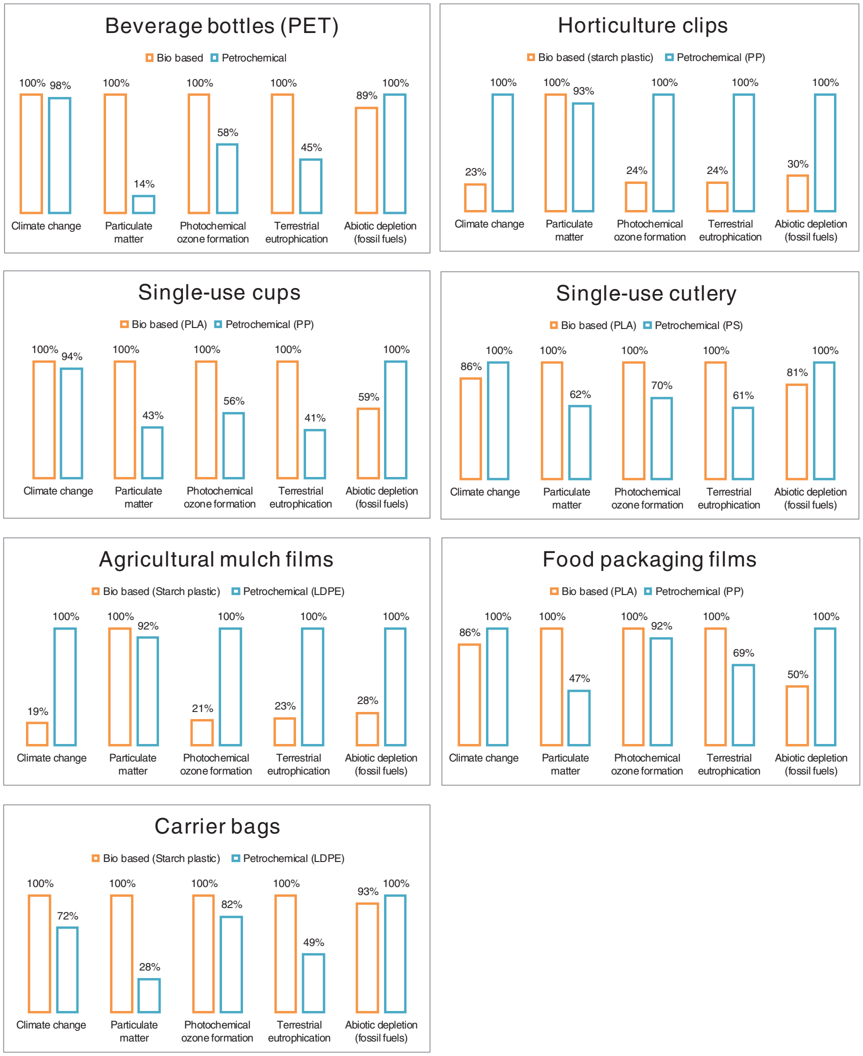

Figure 13.7 Comparing cradle-to-grave environmental impacts of the seven bio-based products with their petrochemical counterparts for five selected impact categories, results based on EoL EU mix only and without taking into account the effect of land use changes. The highest impact is normalized to 100%.

phase accounts for 30% and 50% of the cradle-to-grave environmental impacts for the two products, respectively. From Figure 13.7, it can be seen that if the "intended" EoL (industrial composting) is implemented, the overall impacts of PLA cutlery and food packaging films could decrease by 25% and 30%, respectively, largely because of the avoidance of methane emissions from landfilling. Therefore, the results of these two cases are highly sensitive to the assumed mix of waste management technologies as well as to the assumed organic waste ended up in the plastic waste streams.

- For *PLA cups and single-use carrier bags*, the differences between the "EoL EU mix" and "intended EoL" are less significant (see Figure 13.4). For starch plastics used for *mulch films* and *clips*, the impacts from EoL only contribute to 1% of the total cradle-to-grave impacts. Their EoL results are sensitive to the amount of soil assumed to be collected together with used plastics.

Overall, it can be concluded that recycling and industrial composting are more favorable than anaerobic digestion, incineration, and landfilling for the seven bio-based products studied.

13.4.3 Comparisons with Petrochemical Plastics

The types of petrochemical polymers were determined by industrial consultation and verified by market statistics. For petrochemical polymers, the average EU polymer production was assumed. The LCI data were obtained from the Eco-profiles published by PlasticsEurope [29]. The plastic conversion and final product manufacturing were modeled based on the status quo (as of 2018) plastic conversion technologies. Like the bio-based plastics, the use and disposal of the petrochemical plastics were assumed to take place in Europe. Note that because of the limited data availability, it is not possible to calculate the environmental impacts for all 16 impact categories for the comparison [18]. The Eco-profiles were compiled with industrial average data and were presented as "blackbox" data, which do not provide sufficient transparency subject to high-quality interpretation for all 16 impact categories [18]. The available transparency and data allowed us to compare only five impact categories, namely, climate change, particulate matter, photochemical ozone formation, terrestrial eutrophication, and abiotic depletion (fossil fuels), as shown in Figure 13.7.

From Figure 13.7, it can be seen that for all the seven cases, from cradle to grave, most of the bio-based products offer environmental benefits in two impact categories: *climate change* and *resource depletion fossil fuels*. All the seven bio-based products have higher impacts on *particulate matters* compared to the petrochemical counterparts. The higher impact of the particulate matter of bio-based products differs substantially from case to case. Bio-based PET has over six times higher impacts on particulate matter than the petrochemical PET because of the combustion of sugarcane trash in the biomass cultivation and harvesting phase. It should be noted that the burning cane trash will be soon phased out in Brazil. In the case of PLA food packaging films and cups, the particulate matter impacts are two times as much as the impact of their PP counterparts. PLA has a slightly higher energy requirement in the manufacturing phase and has much less environmental credits from the EoL phase compared to PP. For the remaining cases, the particulate matter is only marginally higher (3–8%) for the bio-based products than for their petrochemical counterparts.

For the remaining two impact categories, namely, photochemical *ozone formation* and *terrestrial eutrophication*, some bio-based products do not offer impact reductions, whereas for some other cases, substantial impact reductions are observed.

Table 13.5 Comparing cradle-to-grave environmental impacts of the seven bio-based products with their petrochemical counterparts and results indicated as environmental impact median savings of seven bio-based products (negative values stand for savings).

	Environmental impact categories	Cradle-to-grave environmental *savings* of bio-based products compared to petrochemical products, median values for seven cases	
		With EoL EU mix	With intended EoL
Lower	Climate change	−11% (−81% to 38%)	−61% (−87% to −14%)
	Resources depletion, fossil fuels	−45% (−72% to −7%)	−33% (−72% to −3%)
Higher	Particulate matter	94% (2% to 603%)	110% (6% to 597%)
Case specific	Photochemical ozone formation	23% (±79%)	19% (±79%)
	Terrestrial eutrophication	70% (−77% to 70%)	72% (−77% to 146%)

The comparisons between the bio-based product systems with *intended EoL* are listed in the second column. Savings are without taking into account the effect of iLUC.

Reasons are case-specific and no generic causes can be identified. It is advised to look into these impacts case by case for specific policy implications.

Table 13.5 shows that the choice of EoL has a strong influence on the overall impact savings of the bio-based products. The influence is especially significant for climate change impact. Treated with intended EoL, the bio-based products on average could offer more than 60% of the GHG emission savings compared to their petrochemical counterparts. This indicates a great potential of low carbon bio-based products if the EoL waste management is implemented appropriately.

13.5 Lessons Learned from the Case Studies and Looking Forward to a Circular Bio-Based Economy

From the seven case studies, we conclude that the bio-based products offer important environmental benefits to combat climate change and to reduce the limited fossil fuel resources, despite different assumptions about EoL waste management and alternative approaches of inventory modeling (e.g. to include the effect of iLUCs). This conclusion is in line with the findings from many previous LCAs [30–34].

However, like many LCA studies, the presented research struggled with limitation on data availability and LCA methodological issues. The uncertainties caused by these two issues should be taken into account when the results are interpreted and policy advices are given.

For the bio-based products, one of the key uncertainties is caused by unavailable data in the EoL waste management phase because there is little information about the new materials in the EoL waste streams. In the BIO-SRI project, this was dealt with by comparing average mix and an idealistic scenario (i.e. intended EoL). Even though many data were still absent, for example, assumptions needed to be made to anticipate the real compositions of different plastic items including unexpected additives and unknown food residues. The extremely heterogeneous waste collection, labeling, and management systems across different EU member states also made it challenging to come up with a representative average for a European level policy advice.

For the comparison with the reference cases based on petrochemical plastics, *data transparency* appeared to be the most crucial issue. The Eco-profiles published by PlasticsEurope are probably the most representative LCI data to assess the impacts of average EU polymer productions. However, the data were packed in a form of "black box" LCI because the information from real production facilities is often treated as commercial secrets that are carefully protected by the sector. The "packed" LCI data provided limited insights, for example, into allocation choices and data uncertainty interpretation. If the petrochemical polymers were to be used as the benchmarks for future policies of bio-based plastics, this benchmark needs to be more transparent to offer a prudent interpretation.

Next to the data availability and transparency issues, several *methodological issues* need to be tackled linked to the LCA of plastics. A few of those are highlighted below.

- The impact of land use changes remains a research hot spot in the field of LCA. In the case studies, attempt was made to assess the impacts from iLUC using a deterministic model. Well acknowledged data from the Intergovernmental Panel on Climate Change (IPCC) and the FAO were used as inputs. However, the method still relies on choices (e.g. share of intensification and expansion; type of intensification) that can greatly affect the results. The current LCA guidelines and norms do have the consensus on how iLUC should be modeled for bio-based production.
- *Littering* is nowadays one of the key environmental concerns both in the public and on the EU policy level, yet the exact amount (and relative share) of plastics that are littered is very uncertain and will largely differ among different countries, user behaviors, and end use applications. Also, the impacts of littering and microplastics are currently not properly captured in the toxicity-related impact categories (eco-toxicity and human toxicity, see Chapter 3) or impact categories related to eco-system services. There are methodological hurdles in impact assessment models. The fate, exposure, and effect models for macro- and microplastics (including additives) are still in the initial stage. Physical impacts (e.g. entanglement, ingestion of larger plastic particles, inhalation of micro- and nano-sized particles, and the effects), chemical impacts (e.g. because of microplastic formation, microplastics as carriers of other chemicals, and unknown impact of additives), and biological impacts (e.g. microplastics as carriers of germs/alien species) are largely unknown and are still not included in LCA. Therefore, for the time being, this is going to remain a major uncertainty in LCA of plastics. It thus will also remain a difficult trade-off to compare, e.g. the impacts of the feedstock production of biodegradable plastics (such as the bio-based starch film and clips) to the EoL impacts of their fossil counterparts. Having said all these, *bio-based and non-biodegradable plastics* will cause similar problems as their fossil counterparts in terms of the risk and the impact of littering.

- One general issue that is not taken into account in current LCA methodology is the *timing of emissions*, i.e. whether a GHG gas is emitted during or very soon after the use phase of whether this occurs only a long time after the production and use of the product. In the seven case studies investigated, average lifetimes are all very short (typically less than one year), but especially for more long-lived biomaterials and plastics (e.g. used in construction or automobile), this could lead to significant additional benefits of bio-based products. The issue of timing of emissions has been extensively discussed in the literature for bioenergy-related applications (especially, woody biomass for electricity and heat production, see e.g. the "carbon debt" described by Lamers and Junginger [35]) and should also be investigated in more detail for bio-based plastics.

Finally and, possibly, most importantly, the LCA analyses are typically bound to the present stage of technology and social/political choices (EoL). They, thereby, fail to show the potential of future technologies over the mature ones. The bio-based plastic industry is still in its initial stage. Most commercialized products are still "young" compared to the petrochemical counterparts. There are many ongoing efforts to optimize the manufacturing process of bio-based production. The efforts would likely lead to a lower environmental impact for the future bio-based products than today. The potential (decrease in) environmental impacts of bio-based plastics due to technological progress and the exogenously changing energy system should be assessed in the future. Ultimately, the assessment studies conducted today provide a starting point for the policy debate. Looking forward, the bio-based plastics will play an important role in reducing the environmental impacts, particularly in the battle of combating climate change. It is still not a silver bullet. Adopting bio-based plastics on a large scale in the future requires to understand the potential environmental trade-offs. LCA, which is still not perfect, does provide a good insight in the potential trade-offs. Toward a sustainable bio-based economy, both benefits and trade-offs of the innovative bio-based products need to be taken seriously in R&D decisions, individual consumer decisions, and ultimately policy decisions.

13.A General Structure of Classification and Characterization in LCIA, using the example of 16 Impact Categories Recommended by the EC EF (Environmental Footprint) Impact Assessment Methods

Name of mid-point impact category	Unit	Examples of inventory substances classified to this category	LCIA characterization models recommended by the EF method[a)]
Climate change	kg CO_2 equiv	CO_2, CH_4, N_2O, and F-gases, in total > 200 substances identified	IPCC fifth assessment report [36]
Ozone depletion	kg chlorofluorocarbon (CFC) – 11 equiv	CFCs, hydrofluorocarbons (HFCs), etc., in total 23 substances	World Metrological Organization (WMO) model [37]

Name of mid-point impact category	Unit	Examples of inventory substances classified to this category	LCIA characterization models recommended by the EF method[a)]
Human toxicity, non-cancer effects	CTUh	In total >15 k substances	USETox model
Human toxicity, cancer effects	CTUh	In total >9 k substances	USETox model
Particulate matter	kg PM2.5 equiv	Ammonia, SO_2, NO_2, NO_x, PM2.5, PM10, and regional specific impacts	RiskPoll model [38]
Ionizing radiation human health	kBq U235 equiv	Uranium 235, uranium 238	[39]
Photochemical ozone formation	kg non-methane volatile organic compounds (NMVOC) equiv	SO_2 (regional specific impacts), volatile organic compounds	CML model [40]
Acidification	Molc H^+ equiv	Ammonia, SO_2, NO_x, etc., in total 194 substances	CML model [41]
Terrestrial eutrophication	Molc N equiv	Ammonia, NO_x, regional specific impacts	CML model [41]
Freshwater eutrophication	kg P equiv	Phosphate, phosphoric acid, and phosphorous	ReciPe model [42]
Marine eutrophication	kg N equiv	Ammonia, NO_x, nitrates, nitrites, and regional specific impacts	ReCiPe model [42]
Freshwater eco-toxicity	CTUe	Organics, inorganics, and metals	USETox model
Land transformation	kg C deficit	Agriculture, forest, grass land occupations, and transformation in total 160 items	Soil organic matter model [43]
Water use	m^3 equiv	Regional specific impacts for surface water and groundwater	AWARE model [44]
Resource depletion, mineral	kg Sb equiv	Minerals and metals	CML model [45]
Resource depletion, fossil fuels	MJ equiv	Coal, crude oil, natural gas, and uranium	CML model [45]

a) This is an ongoing research area. The impact models are regularly updated (with the same impact categories). Always check the latest version of the models when performing an LCA.

13.B Normalization and Weighting Factors Recommended by the EF (Environmental Footprint) Method [12, 19, 46], Latest Update: May 2020

Impact categories	Unit for normalization factors	Normalization factors	Weighting factors (%)
Climate change	kg CO_2 equiv/person	8.10E+03	21.06
Particulate matter	Disease incidences/person	5.95E−04	8.96
Water use	m^3 water equiv of deprived water/person	1.15E+04	8.51
Resource use, fossils	MJ/person	6.50E+04	8.32
Land use	pt/person	8.19E+05	7.94
Resource use, minerals, and metals	kg Sb equiv/person	6.36E−02	7.55
Ozone depletion	kg CFC-11 equiv/person	5.36E−02	6.31
Acidification	mol H+ equiv/person	5.56E+01	6.20
Ionizing radiation, human health	kBq U-235 equiv/person	4.22E+03	5.01
Photochemical ozone formation – human health	kg NMVOC equiv/person	4.06E+01	4.78
Eutrophication, terrestrial	mol N eq./person	1.77E+02	3.71
Eutrophication, marine	kg N equiv/person	1.95E+01	2.96
Eutrophication, freshwater	kg P equiv/person	1.61E+00	2.80
Human toxicity, cancer	CTUh/person	1.69E−05	2.13
Eco-toxicity, freshwater	CTUe/person	4.27E+04	1.92
Human toxicity, non-cancer	CTUh/person	2.30E−04	1.84

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