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Environmental performance of plastic food packaging: Life cycle assessment extended with costs on marine ecosystem services

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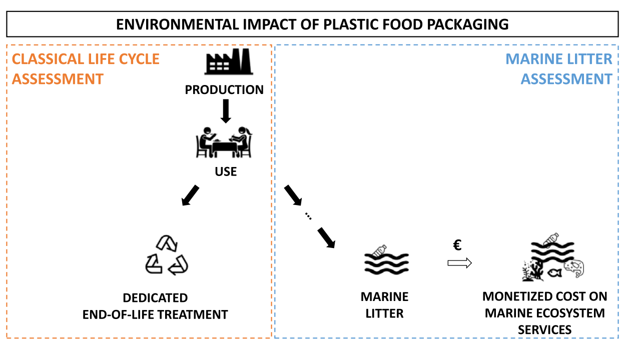
HIGHLIGHTS

- LCA does not enable evaluating all aspects of biobased biodegradable packaging.
- LCA is extended with assessing the impacts of marine litter by a new indicator.
- This indicator addresses the monetized costs on marine ecosystem services.
- The evaluation is performed for falafel using PHBV and one fossil-based packaging.
- The broad evaluation gives no clear preference for one packaging type.

GRAPHICAL ABSTRACT



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ABSTRACT

Packaging can play a substantial role in moving towards more sustainable food systems by affecting the amount of food loss and waste. However, the use of plastic packaging gives rise to environmental concerns, such as high energy and fossil resource use, and waste management issues such as marine litter. Alternative biobased biodegradable materials, such as poly(3-hydroxybutyrate-*co*-3-hydroxyvalerate) (PHBV) could address some of these issues. For a careful comparison in terms of environmental sustainability between fossil-based, non-biodegradable and alternative plastic food packaging, not only production but also food preservation and end-of-life (EoL) fate must be considered. Life cycle assessment (LCA) can be used to evaluate the environmental performance, but the environmental burden of plastics released into the natural environment is not yet embedded in classical LCA. Therefore, a new indicator is being developed that accounts for the effect of plastic litter on marine ecosystems, one of the main burdens of plastic's EoL fate: lifetime costs on marine ecosystem services. This indicator enables a quantitative assessment and thus addresses a major criticism of plastic packaging LCA. The comprehensive analysis is performed on the case of falafel packaged in PHBV and conventional polypropylene (PP) packaging. Considering the impact per kilogram of packaged falafel consumed, food ingredients make the largest contribution. The LCA results indicate a clear preference for the use of PP trays, both in terms of (1) impact of packaging production and dedicated EoL treatment and (2) packaging-related impacts. This is mainly due to the higher mass and volume of the alternative tray. Nevertheless, since PHBV has limited

Abbreviations: $AC_{ES\text{-}ML}$, Annual cost of marine ecosystem services loss due to marine litter; AV_{ES}, Annual value of marine ecosystem services; CEENE, Cumulative Exergy Extraction from the Natural Environment; ES, Ecosystem Services; EoL, End-Of-Life; EPT, Environmental persistence time of marine plastic litter; FLW, Food Loss and Waste; FU, Functional Unit; GLOPACK, Granting society with LOw environmental innovative PACKaging; INC, Incineration; LCA, Life Cycle Assessment; $LTC_{ES\text{-}ML}$, Lifetime costs on marine ecosystem services; MAP, Modified Atmospheric Packaging; OEFSR, Organisation Environmental Footprint Sector Rules; PHBV, Poly(3-Hydroxybutyrate-Co-3-Hydroxyvalerate); PP, Polypropylene; Pt, point; $RD_{ES\text{-}ML}$, Reduction in marine ecosystem services due to plastic litter; REC, Recycling; SI, Supplementary Information; SDR, Social Discount Rate; ST_{ML}, Current stock of marine litter.

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persistence in the environment compared to PP packaging, the lifetime costs for marine ES are about seven times lower, and this despite its higher mass. Although further refinements are needed, the additional indicator allows for a more balanced evaluation of plastic packaging.

1. Introduction

Reducing food loss and waste (FLW) is an important strategy in the quest for sustainable food supply chains (FAO, 2011; Kakadellis and Harris, 2020). It is known that packaging can play an active role in this. For instance, thanks to mechanical properties, packaging can prevent damage during transport or, thanks to its barrier properties, ensure the quality and shelf life of food products during storage (Lindh et al., 2016; Wikström et al., 2014). The latter, in turn, is related to the amount of FLW in the supply chain, as shown in previous research (Conte et al., 2015; Gutierrez et al., 2017; Lindh et al., 2016).

Notwithstanding the role of packaging in sustainable food consumption, there are several environmental concerns related to plastic packaging in particular (Pauer et al., 2019; Wohner et al., 2019). In the EU-28 (+Norway and Switzerland), around 40 % of plastics were used for packaging in 2019, with a large share intended for food applications (Plastics Europe, 2020). Although plastics are lightweight and characterised by high performance, their intensive use as food packaging and their very short lifespan lead to the generation of a huge amount of post-use plastics. Public awareness is emerging about the consumption of resources and energy associated with their production, waste management issues and the persistence of non-degradable plastic waste in the natural environment (Barboza et al., 2018; Wikström et al., 2014; Williams and Wikström, 2011).

It is estimated that global greenhouse gas emissions amounted to 1.7 Gt CO_2-eq in 2015 due to production, use and end-of-life (EoL) treatment of conventional plastic, and this is expected to increase to 6.5 Gt CO_2-eq in 2050 if there are no changes in strategies of plastic use compared to the situation in 2019 (Zheng and Suh, 2019). Regarding waste management, in 2018 in the EU-28 (+Norway and Switzerland), about 42 % of collected post-consumer plastic packaging waste was recycled, while almost 40 % was incinerated with energy recovery and the remainder was landfilled (Conversio Market and Strategy, 2020; Plastics Europe, 2020). Nevertheless, plastics recycling has clear environmental benefits compared to incineration with energy recovery (Civancik-Uslu et al., 2021). Though, a fraction of plastic waste is not properly collected and treated, but can be defined as mismanaged waste, being waste that is littered or not adequately disposed and leaks thus into the natural environment (Jambeck et al., 2015). Depending on precipitation, wind conditions and river density, this plastic is transferred from terrestrial to marine ecosystems where it accumulates as marine litter, together with litter directly disposed in marine ecosystems (Meijer et al., 2021; Lebreton et al., 2017; Jambeck et al., 2015). Given the large share of food packaging in total plastic use and its large application as single-use products, it is identified as a major contributor to marine litter (Andrady, 2015; Koutsodendris et al., 2008). Once in the marine environment, plastic waste is fragmented into smaller particles, but its degradation is limited, leading to accumulation in marine ecosystems (Andrady, 2011). There they can interact with organisms, reducing the intensity of marine ecosystem services (Beaumont et al., 2019). It is considered a major environmental threat due to its wide-ranging impacts, non-reversibility and global scale (Galloway et al., 2017; Villarrubia-Gómez et al., 2018).

These environmental concerns have led to the search for alternative food packaging materials where biodegradable biobased plastics seem promising for two reasons (Bishop et al., 2021). Firstly, it results in a reduced need for fossil fuels and may play a role in climate change mitigation (Guillard et al., 2018; Zheng and Suh, 2019). But the results are strongly affected by the type of EoL management, which depends on the region (e.g. waste collection) and advances in e.g. new recycling technologies (Pauer et al., 2019; Zheng and Suh, 2019). On top, many of the biobased plastics on the market use food resources, leading to pressure on land use (Guillard et al., 2018; Zheng and Suh, 2019). Secondly, their biodegradable nature is crucial in combating plastic litter, although many factors, such as material properties or physico-chemical conditions, affect the biodegradability in the natural environment (Kliem et al., 2020). This use of biobased biodegradable plastic packaging is also in line with the goals set by EU policies and legislation. For instance, plastic marine litter should be reduced by 50 % by 2030, as stated in the Zero Pollution Action Plan (European Commission, 2022). Next, as part of the Green Deal, the European Commission adopted the new Circular Economy Action Plan, in which circularity of plastics is one of the targets to reduce pressure on natural resources. This will promote reuse, recycling and other forms of recovering of plastic waste to keep materials in the economy for as long as possible (European Commission, 2023). A communication on a policy framework for biobased, biodegradable and compostable plastics is published to, among other things, improve understanding of the materials and achieve a common understanding in the EU. However, this is not legally binding. Although biomass used as feedstock for bioplastic production must meet the requirements of the EU sustainability criteria for bioenergy, there is currently no comprehensive law applying to biobased, biodegradable and compostable plastic (European Commission, 2022, 2023).

To compare the environmental sustainability of these biobased biodegradable alternatives and fossil-based non-biodegradable plastic packaging over their entire lifespan (i.e. from raw material extraction to EoL fate), life cycle assessment (LCA) can be used. But although the internationally standardized methodology has proven to be a useful tool, LCA still faces challenges to comprehensively analyse the environmental performance of packaged food products.

Firstly, often in LCA studies of packaged food, the direct impacts caused by the production processes and the EoL treatment of the packaging are considered, while indirect impacts such as packaging-related FLW are often neglected (Molina-Besch et al., 2019). The choice of functional unit (FU) (i.e. reference of comparison) is essential to ensure that the full food supply chain and according FLW are included, e.g. “food eaten” (Notarnicola et al., 2017; Wikström et al., 2014). Secondly, the long-term fate and associated environmental burden of mismanaged plastic waste ending up in the natural environment (soil, freshwater but also seas and oceans) has not yet been embedded in classical LCA (Woods et al., 2016). However, research is ongoing. For instance, Civancik-Uslu et al. (2019) introduced a littering potential indicator that can be used for comparative assessments of plastic bags. In Saling et al. (2020) and Woods et al. (2019), effect modelling (e.g. eco-toxic effects on organisms) is included but the high requirements in terms of data needs (amount and details) limit the implementation in standard food packaging LCA studies. Next, the international working group MARILCA (MARine Impacts in LCA), supported by the Life Cycle Initiative hosted by UN Environment, proposed in Woods et al. (2021) a framework to include marine litter impacts in LCA. Although the framework aims to advance clear quantification by indicators, no characterisation factors are proposed yet. Finally, Maga et al. (2022) presented a method to quantify plastic emissions in LCA. They focus on the fate of plastic materials in the environment, based on their residence time. However, the method does not yet allow quantifying the probability of plastic exposure and the impact of plastic on species, i.e. exposure and effect factors are still missing.

In this study, an approach to comprehensively analyse and compare the environmental sustainability of packaged food using different plastic packaging (e.g. biobased, fossil-based and/or biodegradable) is introduced. Life cycle assessment is used to evaluate the impacts of packaging production and dedicated EoL treatment, as well as the packaging-related impacts on packaged food (e.g. food loss and waste). Additionally, the likelihood and impact of the plastic's release into the natural environment must be covered as well. Therefore, a new indicator is proposed to address the impact of plastic litter on the marine ecosystem, i.e. lifetime costs on marine ecosystem services. By accounting for the biodegradability, the environmental persistence of plastics is taken into account.

This evaluation approach is applied to a case study in which organic falafel is packaged in a fossil-based non-biodegradable tray on the one hand and a biobased biodegradable plastic tray on the other. The alternative packaging consists of Poly(3-hydroxybutyrate-*co*-3-hydroxyvalerate) (PHBV) material. According to Rivera-Briso and Serrano-Aroca (2018), this material is a potential candidate to substitute fossil-based polymers, although much research is still ongoing to improve its properties. Furthermore, PHBV-based pellets are promising in terms of environmental performance at both pilot and industrial scale (Boone et al., 2021; Nhu et al., 2021). This results in the question of whether the same is true for PHBV-based packaging. This will be investigated using the following steps. First, the results of the packaged falafel will be discussed. Second, the focus will be on the impact of the packaging, considering its production, functionality, and waste or improper management at EoL, which is associated with marine litter impacts.

2. Materials and methods

2.1. Comprehensive comparison of the environmental sustainability of plastic packaged food

In order to compare the environmental sustainability of a particular food product packaged in different packaging options, the interrelationship between food and packaging is of utmost importance. In other words, all life cycle stages must be taken into account and the evaluation must enable to cover the impact of the packaging on the amount of FLW (Fig. 1). Therefore, three main phases are distinguished, i.e. production, use and end-of-life phase. The latter includes both the dedicated EoL treatment and EoL fate (i.e. leakage into the environment). The packaging production and specific EoL treatment are clear direct consequences of the chosen packaging itself. At the use phase, packaging-related impacts are noticed. They imply the impacts of packaging along the supply chain as well as the impacts of packaging on food. First, at the use stage, food is processed and packaged. Utility consumption (i.e. related to the packaging line) as well as losses of packaging and food, might be different for different packaging materials. Then the packaged food is distributed, sold at supermarkets, brought home and prepared by consumers. While the mass of the packaging affects the transport costs, the volume of the packaging might affect utility consumption at distribution, retail and storage at households, since its allocation is time- and volume-based according to the Organisation Environmental Footprint Sector Rules (OEFSR): Retail (Quantis, 2018). Finally, the packaging might affect the amount of FLW by e.g. losses during transport or by affecting shelf-life and the corresponding amount of FLW. Hence, the food lost due to imperfect packaging must also be produced, processed, transported, prepared and treated at EoL. They are again presented by the main phases: food production, use and EoL phase of FLW. Consequently, to allow for a careful comparison between packaging materials, the full life cycle of packaged food should be considered.

However, while the focus is often on the proper treatment of waste, one should consider all pathways. According to the terminology of Conversio Market and Strategy (2020), the different options of dedicated EoL treatment and EoL fate are recycling, energy recovery, landfilling, improper disposal and leakages. Waste that is dumped in unsanitary and unauthorized

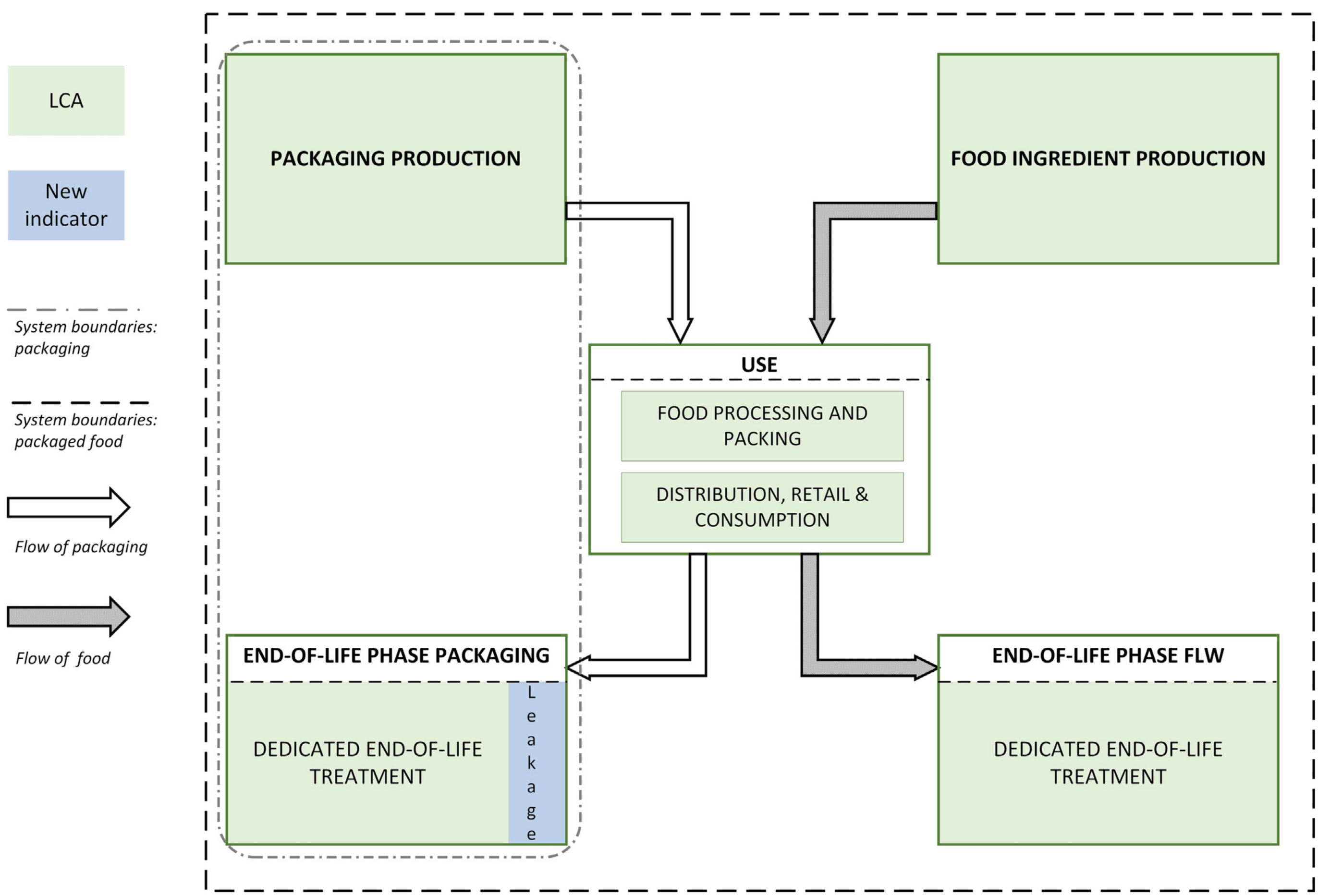

Fig. 1. Simplified process scheme of the life cycle of packaged food, with an indication of potential evaluation by classical LCA and the newly developed indicator. This indicator only focuses on the marine litter impact. The system boundaries for the evaluation of (1) the packaged food product and (2) only the packaging (i.e. primary packaging) are indicated. FLW = food loss and waste; LCA = life cycle assessment.

dumpsites but stays concentrated in one area is considered under the category "improper disposal". The category "leakages" accounts for the fraction of plastic waste that is spread into natural environments, resulting in the environmental concern of marine litter. While LCA is a well-developed methodology for evaluating the environmental performance of a product, the environmental burden of plastics released into the natural environment is, however, not yet embedded in classical LCA. Therefore, an indicator to account for the impact of plastic leakage is needed, implying the probability to become plastic litter and the effect of plastic on the marine ecosystem while considering the persistence of the plastic. All different steps are schematically presented in Fig. 1 with an indication of what can be evaluated by LCA and what is measured by the newly developed indicator, which is presented in Section 2.4.

2.2. Case study description

In this study, the environmental impact of packaged falafel produced and sold at supermarkets in Belgium is considered. The primary packaging consists of a tray and a film and one pack contains 150 g of food. Two options of primary packaging are compared: (1) a traditional fossil-based non-biodegradable packaging made of pure polypropylene (PP) (tray) and PP multilayer (film) and (2) a biobased biodegradable packaging (tray and film) as an alternative. The latter was developed in the context of the H2020 project GLOPACK (Granting society with LOw environmental innovative PACKaging, grant 773375), in which PHBV packaging materials are produced from fruit pulp waste. The PHBV tray has another geometry due to limitations in available moulds. It could be assumed that trays with the same depth as the PP trays could be produced in the future, although this was not done in the context of this project (Table 1).

The food production and packaging take place in La Vie Est Belle, a Belgian food company supplying vegetarian food products. Based on the first experimental tests (including sensory quality, microbiological and chemical assessment) carried out on falafel in conventional PP and alternative PHBV-based packaging, no reduction in shelf life is reported (Vermeulen, 2021). Consequently, the shelf life and corresponding amount of FLW are assumed the same for the two packaging options.

The considered EoL treatment of the PHBV packaging is incineration with energy recovery, because this packaging material is currently not allowed in the Belgian plastic collection system towards recycling nor in the residual waste fraction towards composting (OVAM, personal communication; VLACO, 2023). With regard to the conventional packaging, the packaging is collected towards recycling, but while the tray is considered to be recycled, the film is ending up in a residue stream in the recycling scheme towards incineration, based on Civancik-Uslu et al. (2021). The composition, mass, dimensions and considered EoL treatment of the primary packaging materials are summarized in Table 1.

2.3. Life cycle assessment

LCA, according to the ISO standards ISO 14040/14044 (ISO, 2006a, 2006b), is used to evaluate (1) the impact of packaged falafel, (2) the impacts of packaging by production and dedicated EoL treatment and (3) the packaging-related impacts of packaged food by production and consumption.

Table 1
Composition, mass, dimensions and dedicated end-of-life (EoL) treatment of conventional and alternative primary packaging. INC, incineration; REC, recycling; PHBV, Poly(3-hydroxybutyrate-*co*-3-hydroxyvalerate); PP, polypropylene.

	Packaging component	Composition	Mass (g/kg falafel)	Dimensions tray (mm × mm × mm)	EoL scenario*
Conventional	Tray	PP	61.3	187 × 137 × 25	REC
	Film	multilayer PP	10.0		INC
Alternative	Tray	PHBV	120.5	187 × 137 × 37	INC
	Film	PHBV	13.3		INC

* includes collection and sorting (in case of recycling scenario) as well.

2.3.1. Goal and scope

The goal of the LCA study is to analyse the environmental performance of falafel packaged in trays containing 150 g of food. Two primary packaging options are compared: a conventional PP-based tray and film and a PHBV-based alternative.

The FU corresponds to the household consumption of 1 kg packaged falafel. In this way, the impact of FLW is taken into account in the analysis. When analysing the packaged food, all life cycle stages, going from the packaging production to the end-of-life treatment of food loss waste and packaging waste, are included (Fig. 1).

2.3.2. Data inventory

Data for the foreground system are mainly collected in collaboration with the manufacturers involved in the GLOPACK project, completed with data from scientific literature, reports or life cycle inventory databases. Data for the background processes are retrieved from ecoinvent v3.8, using the software SimaPro v9.4. An overview of the data collection is given per life cycle stages. Quantitative data and assumptions made throughout the different stages are listed in Supplementary Information (SI), Part A.

2.3.2.1. Packaging production. The production of packaging materials consists of two main steps: pellet production and conversion of pellets into packaging. The pellet production for the alternative packaging implies PHBV production and compounding. To obtain the packaging materials, the PHBV pellets are pre-dried and then converted into films by extrusion. To obtain trays, extruded sheets are thermoformed. The production process is schematically presented in SI, Fig. A1. The same production steps are valid for the conventional packaging: pellet production is followed by extrusion (film) or extrusion and thermoforming (tray).

Mass and energy data on PHBV pellet production are retrieved from Nhu et al. (2021). The main input is agro-waste, therefore considered zero-burden. This is used as feedstock to produce biobased biodegradable PHBV relying on a three-stage mixed microbial culture process (Nhu et al., 2021; Silva et al., 2022). In their studies, data were collected at a pilot level for several scenarios. Upscaling factors of inputs and outputs to an industrial scale were provided based on literature and expert knowledge, among others, and then validated via mass flow analysis (Nhu et al., 2021).

The data related to conversion of pellets into packaging are collected in collaboration with Coopbox Group s.p.a., an Italian company that produces packs and packaging for food products. PHBV tray production is not yet operational on an industrial scale. To compile the data inventory (i.e. material and energy consumption and emissions), the production of polystyrene trays are used as a proxy, being the best alternative according to experts within Coopbox Group s.p.a. It is assumed that also PHBV films are produced, although they were not produced within the GLOPACK project.

With respect to the conventional packaging, data are retrieved from ecoinvent v3.8. Secondary and tertiary packaging are assumed equal for both cases. Information on production and mass per FU are summarized in SI, Tables A1–A3.

2.3.2.2. Food ingredient production. Regarding the agricultural production of the main ingredients, data on the composition of falafel, the origin and transport of each food ingredient are supplied by the Belgian food company La Vie Est Belle. Data on the production of the food ingredients for falafel are retrieved from the ecoinvent v3.8, except sunflower oil, for which data from Schmidt (2015) is used (SI, Table A4).

2.3.2.3. Use

2.3.2.3.1. Food processing and packing. The production of falafel takes place in the food company La Vie Est Belle which provided information on all inputs and outputs related to the manufacturing and packing of falafel (e.g. electricity, water, frying oil, etc.) (SI, Fig. A2, Table A5). As this happens on one production line, utility consumption could not be defined separately for food processing and packing. The packaging of the food ingredients (which are often delivered in bulk) is excluded from the analysis.

After falafel processing, the food is packaged into a tray and film (primary packaging), and a cardboard sleeve around it. Then, they are put in plastic boxes on pallets and wrapped with foil. The amount of FLW at La Vie Est Belle is 1.7 %, while it is assumed that the losses of packaging during packing are negligible (La Vie Est Belle, personal communication). Based on expert opinions, utility consumption is assumed equal for both packaging types (La Vie Est Belle, personal communication).

2.3.2.3.2. Distribution, retail, use. Data on the consumption of energy and materials and the emitted waste flows at distribution, retail and consumer level, are retrieved from the OEFSR: Retail (Quantis, 2018). Data on transport to the distribution are supplied by the food company, while transport by consumers is modelled according to the OEFSR. In analogy with the study of Gruber et al. (2016), the proxy that one passenger car transports 10 kg of retail goods is used as the allocation key. Data for preparing the falafel at home are based on the instructions on the label and the OEFSR (Quantis, 2018). The amount of FLW at the distribution and retail level is retrieved from the retail sector in Belgium for a similar product and corresponds to 2 % (Boone et al., 2021), while 5 % is lost at the consumer level as indicated by the OEFSR: Retail for processed meals (Quantis, 2018). Considering the same shelf life for both packaging materials (Section 2.2) and no difference in e.g. possibility to empty the packaging, it is assumed that the amount of FLW is equal for both packaging materials.

2.3.2.4. Dedicated end-of-life treatment of packaging. The amount of losses of packaging materials was mentioned in Section 2.3.2.3 and summarized in SI, Table A7. The results from Civancik-Uslu et al. (2021) and Huysveld et al. (2022) are used to model the collection, sorting and recycling of fossil-based plastics in Belgium, however, results are updated to ecoinvent v3.8 to be consistent with the other calculations. Data for the incineration of the PHBV packaging and EoL treatment of secondary and tertiary packaging are retrieved from ecoinvent v3.8.

2.3.2.5. Dedicated end-of-life treatment of food loss and waste. The following EoL scenarios for FLW are used: incineration with energy recovery (50 %), composting (25 %) and anaerobic digestion (25 %), as proposed by Quantis (2018). By-products of incineration (i.e. electricity, heat and ashes) are assumed to substitute for electricity, heat from natural gas on the European market, and gravel, respectively, while the substitution factors are retrieved from ecoinvent v3.8. With respect to the substitution of compost, the recommendations of Tonini et al. (2020) are used. Biogas produced by anaerobic digestion is assumed to substitute for biogas available on the European market. The amount of FLW per stage was mentioned in Section 2.3.2.3 and summarized in SI, Table A7.

2.3.3. Impact assessment

The environmental impact is assessed using two different methods. To quantify the resource-related impact, the Cumulative Exergy Extraction from the Natural Environment (CEENE) is used (Alvarenga et al., 2013; Dewulf et al., 2007), which is the recommended method by UNEP-SETAC (Berger et al., 2020) and Liao et al. (2012) to account for and characterize resource use. Exergy is the maximum amount of useful work that can be obtained from a resource when it is brought into equilibrium with the environment. It is expressed in one unit: joules of exergy and measures both quantity and quality of natural resources (Dewulf et al., 2007). This way, CEENE accounts for the cumulative amount of exergy contained in the resources deprived of nature to produce the final product.

The Environmental Footprint (EF) 3.0, launched by the European Commission, is used to calculate the midpoint impact category climate change and the single score, which is an aggregated score at endpoint. Climate change (unit kg CO_2-equivalent) is the most widely evaluated impact category in environmental sustainability analyses of the food supply chain (Vidergar et al., 2021). Moreover, the impact on climate change was evaluated in all 44 LCA studies on bio- and fossil-based plastic packaging considered in Bishop et al. (2021). The single score, expressed in point (Pt) is easy to interpret due to the aggregation of sixteen midpoint impact categories into one comprehensive overview.

The selection of these three indicators allows for an inclusive assessment of packaging. First, the indicators address the environmental impact from both the emission and resource perspectives. Second, both midpoint and endpoint assessment is performed.

2.3.4. Sensitivity analysis

An evaluation of the influence of variations of input parameters on the outcome is accomplished through a sensitivity analysis. This analysis enables to identify the extent to which the included parameters affect the impact results. Therefore, Oracle's Crystal Ball software, which relies on a Monte Carlo analysis, is applied and all input parameters are varied (10,000 iterations) within a triangular distribution of -10 % and $+10$ % of the original value, which allows the identification of the parameters that influence the results the most (Thomassen et al., 2019).

2.4. Measuring the impact of leakage

The impact of plastic food packaging leakage on the marine environment is measured by the lifetime costs on marine ecosystem services (ES), considering three components: (1) the potential contribution of packaging to marine litter formation, (2) the annual costs of loss of marine ES due to the existing stock of marine litter and, (3) the environmental persistence of the plastic food packaging, which varies according to the plastic material. The potential contribution of packaging to marine litter formation is measured by multiplying its mass by its probability of reaching the marine environment, taking into account the treatment options for collected plastic packaging waste. Together with the environmental persistence, this is a key parameter for calculating the lifetime costs on marine ES due to the release of plastic into the natural environment. The stepwise procedure used to come to the final evaluation is explained in the next sections and is schematically presented in Fig. 2.

2.4.1. Potential contribution to marine litter

The potential contribution of a particular food packaging to marine litter is obtained by multiplying its mass by the probability of collected plastic food packaging waste becoming marine litter. Both the geographic area where the food packaging waste is generated (Meijer et al., 2021) and the polymer type that underpins its treatment influence this probability (Conversio Market and Strategy, 2020; Erni-Cassola et al., 2019). The probability is obtained by dividing the amount of marine litter from food packaging by the total amount of food packaging waste generated in the particular region. The amount of marine litter is defined based on Meijer et al. (2021) who provide country-specific probabilities that mismanaged plastic waste reaches the marine environment. The relative importance of each treatment for plastic food packaging waste is determined for the region under study but also for other regions importing plastic food packaging waste. A detailed explanation of the followed methodology and data sources to define the probability are presented in SI, part B. Importantly, the starting point for the evaluation is collected waste, as for the European case, reliable data on the fraction of uncollected waste discarded by consumers could not be found.

2.4.2. Lifetime costs on marine ecosystem services

A new indicator is proposed to evaluate the impact of food packaging on marine ES. ES are the benefits people derive from the functioning of ecosystems, which can be expressed in monetary terms at their most endpoint level (Millennium Ecosystem Assessment, 2005). The indicator relies on the annual costs of marine ES loss, integrated over the duration of the packaging's persistence in marine conditions and on its potential contribution to marine litter.

The methodology from Beaumont et al. (2019) is adopted to estimate the annual cost of marine ES loss for a given mass of plastic waste entering the marine environment (i.e. obtained by multiplying the mass of the functional unit by the probability of plastic waste entering the marine

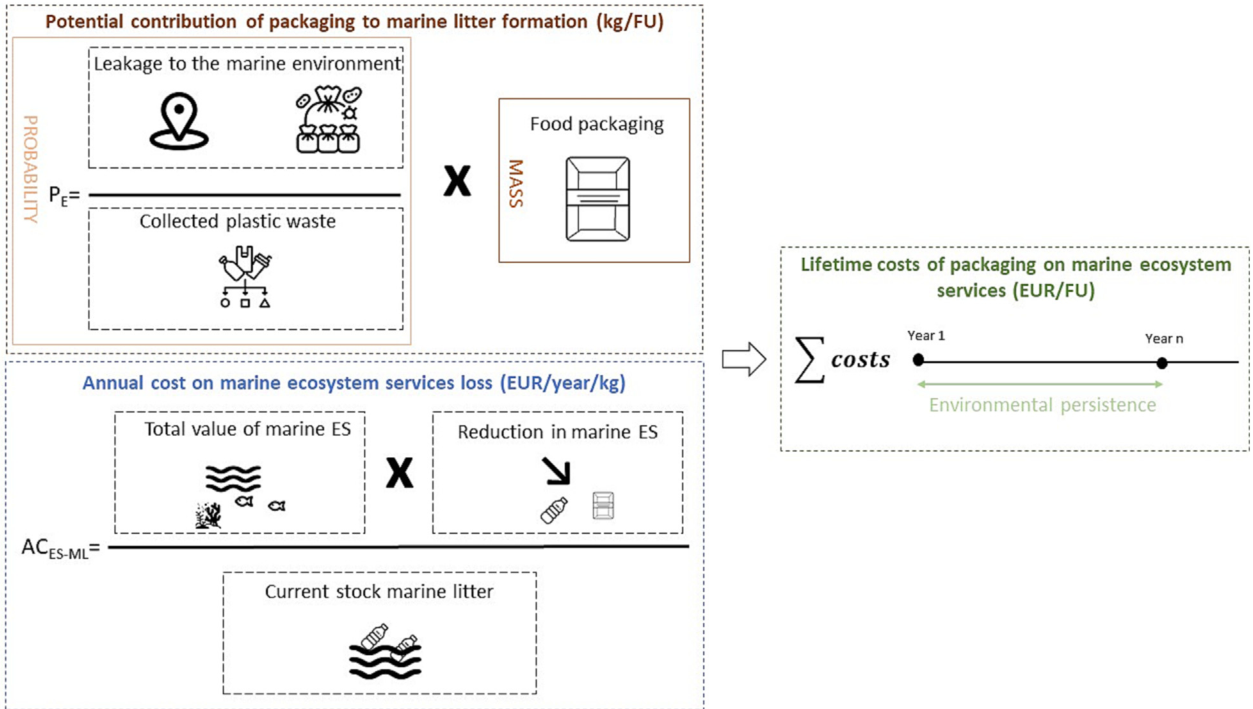

Fig. 2. Schematic representation of the steps to assess the monetary impact of plastic food packaging on marine ecosystem services (ES). P_E is the probability of food packaging to become marine litter; $AC_{ES\text{-}ML}$ are the annual costs of ecosystem services due to marine litter (figures used from freeicon.io).

environment). The total value of marine ES has been estimated at 49.7 trillion USD_{2007} per year (Costanza et al., 2014), equivalent to 69.2 trillion EUR_{2007} per year, assuming the average exchange rate EUR/USD in 2007 (Eurostat, 2021a). This value is revalued to 84.9 trillion EUR_{2020} per year based on the 2020-price level, relying on the annual European harmonized index of consumer prices (Eurostat, 2021b). It is estimated that the 2011-stocks of marine litter reduced ES supply by 1 to 5 % that year (Beaumont et al., 2019). The annual cost of marine ES loss is reported per kilogram of plastic waste entering the marine environment and is computed according to Eq. (1) (Beaumont et al., 2019).

$$AC_{ES-ML} = \frac{AV_{ES} \times RD_{ES-ML}}{ST_{ML}} \tag{1}$$

AC_{ES-ML} is the annual cost of marine ES loss due to marine litter (EUR_{2020}/yr x kg plastic litter), AV_{ES} is the annual value of marine ES (EUR_{2020}/year), RD_{ES-ML} is the reduction in marine ES due to plastic litter (%) and ST_{ML} is the current stock of marine litter (kg).

In view of the uncertainty related to the proportion of marine ES loss due to littering (1–5 % in 2011) (Beaumont et al., 2019) and the marine litter stocks (75–150 million tonnes in 2011) (McKinsey & Company and Ocean Conservancy, 2015; Jang et al., 2015), two contrasted scenarios are considered. The first scenario (i.e. best case) assumes a 1 % reduction in marine ES as a result of 150 million tonnes of marine plastic litter in 2011. In contrast, the second scenario (i.e. worst case) assumes 5 % of marine ES reduction due to 75 million tonnes of marine plastic litter in 2011.

However, AC_{ES-ML} does not vary with biodegradability. Therefore, the annual costs are discounted over the environmental persistence period of marine plastic litter to determine the lifetime costs following the methodology proposed by DeWit et al. (2021) (Eq. (2)).

$$LTC_{ES-ML} = \frac{AC_{ES-ML}}{(1 + SDR)^{EPT}} \tag{2}$$

$LTC_{ES\text{-}ML}$ represents the lifetime costs on marine ES per kilogram of plastic litter entering the marine environment (EUR_{2020}/kg marine plastic litter), SDR is the social discount rate and EPT is the environmental persistence time of marine plastic litter.

A social discount rate of 4 % per annum is selected according to EU recommendations (European Commission, 2017). The environmental persistence time depends on the food packaging under study. In the case of non-biodegradable packaging, a perpetual annuity formula is used to assess the lifetime costs. This implies that 99.7 % of the lifetime costs are incurred during the first 150 years.

This is valid for the case of the non-biodegradable plastic PP, as degradation is negligible compared to PHBV from a human time perspective (UNEP, 2015). The literature on the degradation of PHBV in marine conditions reports lifetimes of less than one to more than two years to reach complete degradation (Meereboer et al., 2021; Kliem et al., 2020; Deroiné et al., 2015). The variation in the degradation time could be explained by the lack of standards for the experimental design. Considering the range of degradation times reported so far, this study assumes that PHBV packaging is fully degraded in the marine environment within two years.

3. Results and discussion

First, the environmental performance of packaged falafel is discussed, considering the several life cycle stages. Second, emphasis is placed on the comparison of the two primary packaging materials. A distinction is made between the impact due to (1) packaging production and dedicated EoL treatment and (2) packaging-related impacts of packaged food. Finally, the new indicator is used to define the impact of leaked packaging on the marine ecosystem. As indicated in Section 2.3.3, the resource footprint, carbon footprint and single score are calculated. The results of the latter are presented here, while the two others are given in SI, part C.

3.1. The impact of packaged falafel

The total environmental impact of the household consumption of 1 kg packaged falafel (FU) is equal to 480 μPt when using the current PP packaging and 530 μPt for the alternative PHBV-based packaging. The relative

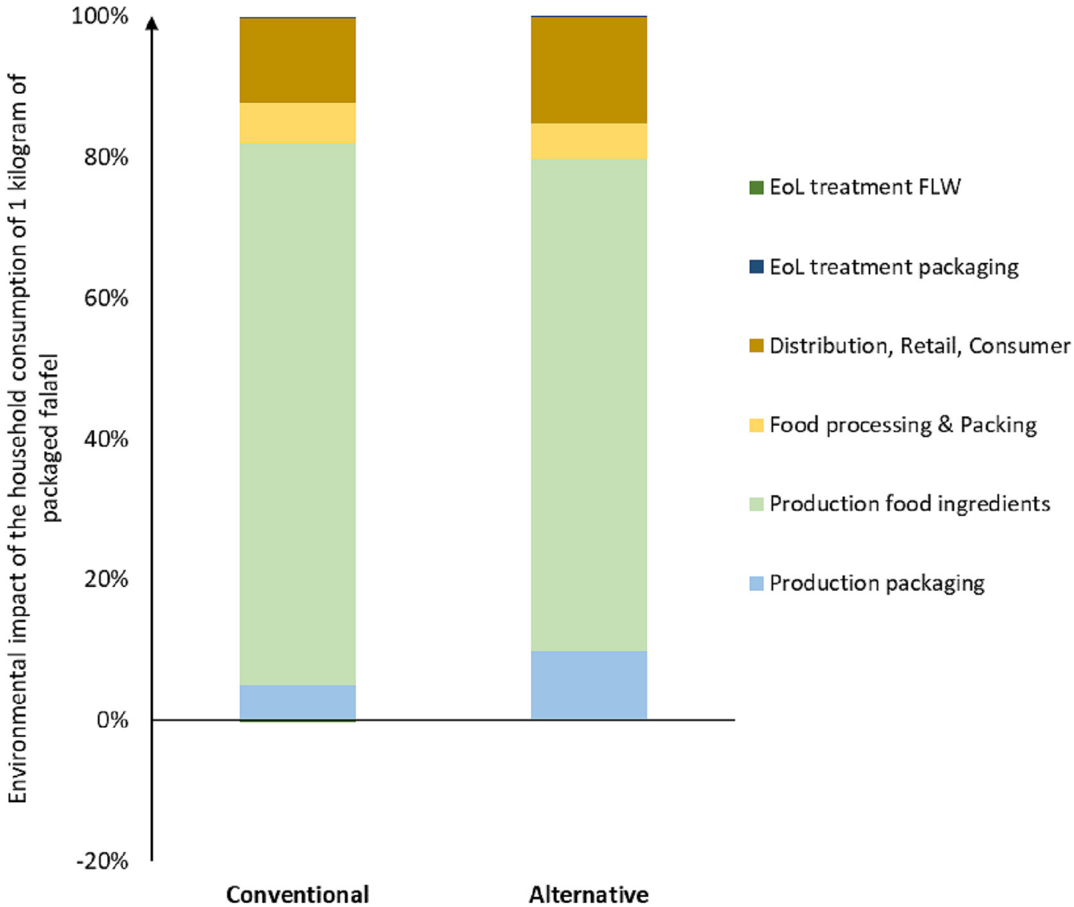

Fig. 3. Contribution of the different life cycle stages to the environmental impact (EF single score) considering the conventional and alternative packaging. The functional unit (FU) corresponds to the household consumption of 1 kg packaged falafel. Total impact is equal to 480 (conventional) and 530 (alternative) μPt per FU.

contribution of the different life cycle stages to the total impact is presented in Fig. 3. The largest contribution is made by the food ingredient production and transport (371 μPt). The other life cycle stages with a significant contribution to the total impact of the conventional case are, in descending order, distribution, retail and consumer (12 %), food processing and packaging (6 %), and packaging production (5 %). In the case of the alternative packaging, the main contributing stages in descending order of share are distribution, retail and consumer (15 %), packaging production (10 %) and food processing and packing (5 %). The discrepancy between both is mainly due to the primary packaging production and disposal, which accounts for more than 50 % of the difference in impact.

First, the main contributing categories are studied. In the next section, the differences between the two cases are thoroughly discussed, being subdivided into the impact of packaging production and dedicated EoL treatment on the one hand, and the packaging-related impacts, on the other hand.

Fig. 4 zooms in on the main contributing categories (e.g. packaging, utilities such as water and energy and auxiliaries (e.g. chemicals)) by life cycle stage. The results are presented only for the conventional case, as the differences between the two cases are discussed in Section 3.2. Furthermore, the impact results for both the conventional and alternative case are given in SI, Table C1.

The production of primary packaging (i.e. tray and film) has the largest share (76 %) in the total impact of the packaging production. Secondary and tertiary packaging together have a contribution of 23 % in which the share accountable to the cardboard sleeve is highest, followed by the production of wooden pallets (which are reused with a limited lifespan) and wrapping foil. The impact of the top foils on the plastic crates is negligible as well as the impact of transport of the material to the food company, as is also indicated by Molina-Besch et al. (2019). The absolute impact of food ingredient production (stage 2) corresponds to 365 μPt/FU. Chickpeas contribute the most, which also corresponds to the largest share by mass (56 %). In contrast, garden peas, onion and parsley count together for one third of the mass, while the share in impact for those three together is only 7 % (SI, Fig. C3, Table A4).

Utilities and auxiliaries are responsible for the environmental impact at the food company. Mainly sunflower oil used to fry falafel is causing the impact for this step (i.e. a share of 72 %). Next, cleaning and disinfecting media accounts for 14 % of the environmental burden, in ascending order due to the use of cleaning chemicals, the consumption of tap water and softening with salt, and finally the treatment of wastewater. The use of energy is responsible for 13 % of the impact, while the contributions of lubricating oil and modified atmosphere packaging gases (MAP) are minimal (SI, Fig. C3). This is consistent with the study by Mouron et al. (2016) in which frying oil is identified as hotspot during fried food processing.

Energy consumption is the main contributor (77 %) to the next stage (distribution, retail & consumption). Transport accounts for 14 %, most of which can be attributed to retailer-to-consumer transport. The distribution among different means of transport is based on the OEFSR Retail (Quantis, 2018), while in accordance with the study by Gruber et al. (2016), it is assumed that one passenger car transports 10 kg of retail goods. However, more research would be needed to get a better view of transport from retail to consumer. The impact of auxiliaries (6 %) is mainly defined by the consumption of soap and oil for preparing falafel. Water consumption and treatment, used for cleaning and in the dishwasher, accounts for 3 % of the burden. The effect of refrigerants' use (distribution centre) is minor.

Regarding EoL treatments, the net impact is presented, which corresponds to the burdens of the treatment and the avoided burden. With respect to the EoL of FLW, the impact is very small, resulting from the total FLW along the food supply chain which is equal to 94 g per kg of packaged

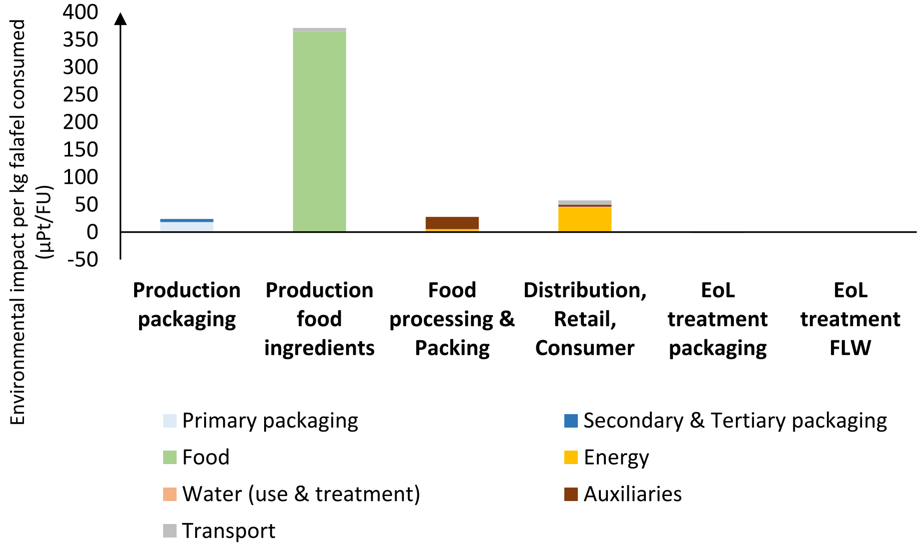

Fig. 4. Environmental performance of packaged falafel per kg consumed at home (functional unit, FU), using the conventional PP packaging. Impacts of contributing categories are expressed per life cycle stage.

falafel (Section 2.3.2). Primary, secondary and tertiary packaging have a share in the avoided burden of 70, 14 and 16 %, respectively.

3.2. Comparing the environmental performance of two packaging materials

3.2.1. Impacts related to primary packaging production and dedicated end-of-life treatment

Fig. 5 shows the environmental impact per household consumption of 1 kg packaged falafel for both conventional (i.e. PP) and alternative PHBV-based primary packaging production and dedicated EoL treatment. Packaging production consists of two main phases: pellet production and conversion of these pellets into the desired packaging, i.e. trays and films (SI, Fig. A1). Regarding the EoL treatment, both the burdens related to the treatment and the avoided burdens are presented.

The largest net impact is for the alternative packaging, corresponding to 41.8 (tray) and 2.8 µPt (film) per FU, compared to 14.8 and 2.5 µPt per FU for the conventional tray and film, respectively. The environmental impact is mainly determined by the packaging production, to which the pellet production contributes the most. For the alternative packaging, this share is 78 to 88 % (tray and film, respectively), while for the conventional tray and film, the proportion amounts to 80 and 83 %, respectively. The PP pellets are clearly more environmentally friendly than the PHBV pellets, namely 181 µPt for 1 kg of PP versus 238 µPt for 1 kg of PHBV pellets. Regarding the latter, PHBV powder production has the highest share in the

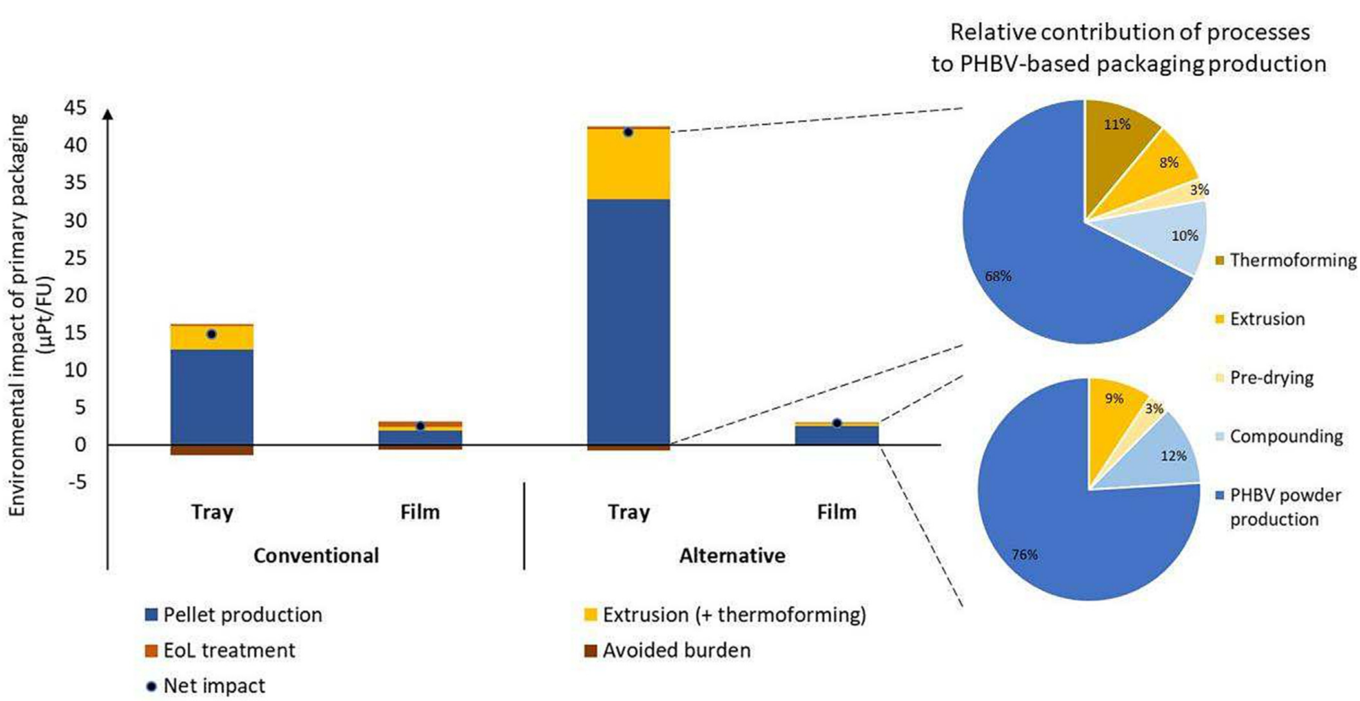

Fig. 5. Environmental impact of the production and End-of-Life (EoL) of the primary packaging (e.g. tray and film) per household consumption of 1 kg packaged falafel (functional unit, FU), considering the conventional PP-based and the alternative PHBV-based packaging.

environmental impact, due to nutrient and electricity consumption, among others. The addition of milled fibre in the pellets could lower the environmental impact, but this could lead to technical constraints in the conversion of pellets into packaging materials (Boone et al., 2021). It is important to note that because only small production volumes are required, the feedstock (fruit waste) can be used directly, without high transport costs or specific storage conditions. This motivates the assumption that the feedstock is burden-free. However, when large production volumes would be needed, this should be further investigated and could negatively influence the environmental sustainability performance of the alternative packaging.

Also, processing of the pellets into trays and films by pre-drying, extrusion and thermoforming (in the case of trays) is much less environmentally beneficial in the case of the alternative packaging. Firstly, pre-drying is required, whereas it is not in the case of PP, but the impact of this process is very small. Secondly, tray production is still a new process and improvements in extrusion and thermoforming can be obtained over time. Thirdly, primary data are used in this case, while secondary data are retrieved from ecoinvent v3.8 for the PP trays.

The high difference in impact between the production of the two cases is reinforced by the higher mass of packaging in the alternative case (Table 1). To ensure the same environmental impact for the alternative tray as for the conventional one (with different geometry), the mass of the former would have to be reduced by 62 %. However, this sharp reduction in thickness and related mass could lead to changes in barrier properties (Boone et al., 2021), potentially affecting the shelf life and hence the amount of FLW. If it is assumed that PHBV trays with the same geometry can be produced in the future, the difference in impact between the two types of trays would become smaller. However, the difference in density (i.e. ~0.90 and 1.24 g cm^{-3} for PP and PHBV, respectively (David et al., 2020)) and the challenge of producing PHBV-based trays, as this is still a new process, would still remain. Therefore, the conclusion of the largest environmental impact for the production of the alternative trays would still be valid.

The considered EoL treatment is incineration for the alternative packaging and recycling for the conventional tray, while the PP film is assumed to be incinerated. The avoided burden implies energy and gravel for incineration and regranulates for recycling. To account for the reduction in the effective quality of the end product and potential fields of application - which is closely related to the potential market size - correction factors (market substitution and technical feasibility) are applied (Huysveld et al., 2022). The disposal impact (EoL treatment and avoided burden) is lower (greater negative) for the conventional, i.e. −1.1 μPt per FU than for the alternative tray (−0.4 μPt per FU), although less material is treated. However, the value of avoided products from recycling is higher than that from incineration (Civancik-Uslu et al., 2021). The impact of the EoL treatment of the two types of film is more in line with each other, but is very small compared to the total impact of the primary packaging.

Thanks to recycling of conventional primary packaging instead of incinerating, the impact of the stage "end-of-life of packaging" (i.e. EoL treatment and avoided burden) is reduced by 88 %. However, important to keep in mind is that this end-of-life stage only accounts for 0.3 % of the total impact. But this means that a significant reduction in EoL environmental impacts would probably be obtained for the alternative packaging as well. Further research on the recyclability of PHBV-based packaging is therefore needed (OVAM, personal communication).

3.2.2. Packaging-related impacts of packaged food

Packaging-related impacts can occur at all considered life cycle stages. However, because secondary and tertiary packaging, packaging losses and food losses along the life cycle are assumed equal for both cases, there is no difference in impact for production and EoL treatment of packaging and food. Indeed, as no difference in primary packaging losses is assumed either, there is only a difference in direct impacts of production and EoL treatment (Section 3.2.1). Furthermore, also food processing is not affected by changing the packaging. The discrepancy in impact (Table 2) between both is mainly due to the different dimensions which affect the utility consumption (i.e. energy use (cooled storage) and water (cleaning)) at distribution, retail and consumer. Namely, a volume-time allocation approach is used to calculate the consumption, based on the recommendations of the OEFSR Retail (Quantis, 2018). This is also the case for the refrigerant consumption. Next to refrigerants, MAP gases are classified into the category auxiliaries. Also here, the volume is decisive for the amount used per pack of falafel. Finally, the higher mass for the PHBV packaging results in a higher transport cost.

Table 2
Difference in packaging-related impacts between the conventional and alternative case.

	Δ impact alternative and conventional (μPt/FU)		
	Production packaging	Food processing & packing	Distribution, retail, consumer
Energy	0.0	0.0	21.0
Water (use & treatment)	0.0	0.0	0.1
Auxiliaries	0.0	0.1	0.0
Transport	1.4	0.0	0.4

3.2.3. Packaging impact due to leakage

As a first step, annual costs based on Eq. (1) are evaluated for two scenarios: 1) a best case that assumes a 1 % reduction in marine ES due to a large stock of marine plastic litter (150 million tonnes) and 2) a 5 % reduction due to a half as large marine litter stock (worst case). For the best case scenario, a loss of 5.7 EUR per year and per kg plastic litter on marine ES is obtained, while this value is ten times bigger (56.7 EUR per year per kg plastic litter) for the worst case scenario. These annual costs are discounted in Eq. (2) to assess the lifetime costs of marine plastic litter, in this way taking into account the environmental persistence of the plastic. Considering that PHBV litter fully degrades after two years, the lifetime costs per kilogram of PHBV litter vary from 10.7 EUR_{2020} (best case scenario) to 106.8 EUR_{2020} (worst case scenario). On the other hand, the lifetime costs for 1 kg of PP (non-biodegradable) litter varies from 141.5 EUR_{2020} for the best case scenario to 1415.3 EUR_{2020} for the worst case scenario.

In line with the LCA study, the lifetime costs on marine ES of the packaging are assessed per household consumption of 1 kg packaged falafel (FU), in this way also including the functionality of the packaging materials. The focus is put on the tray, being the main component of the primary packaging. The lifetime costs are obtained by multiplying the mass of the tray used per 1 kg of falafel consumed by the probability to become marine litter (0.02 %, SI, part B) and, by the lifetime costs per kilogram of marine plastic litter. In the best case scenario, the potential costs per FU are estimated to be 1.9×10^{-3} and 2.8×10^{-4} EUR_{2020} for PP and PHBV packaging, respectively; while for the worst case scenario, these costs reach 1.9×10^{-2} and 2.8×10^{-3} EUR_{2020} for PP and PHBV packaging, respectively (Table 3). In light of the potential marine plastic litter formation, PHBV packaging performs about two times worse than PP packaging

Table 3
Potential marine litter formation (g) and lifetime costs on marine ecosystem services (ES) (EUR_{2020}) per household consumption of 1 kg packaged falafel (functional unit, FU) in conventional (PP) and alternative (PHBV) packaging, assuming waste collection in EU-28. P_E, the probability for plastic waste collected in EU-28 to become marine litter.

	Conventional packaging	Alternative packaging
Mass of the tray per FU (g)	65.9	129.5
P_E (%)	2.0E−02	2.0E−02
Potential marine litter formation (g/FU)	1.3E−02	2.6E−02
Lifetime costs on marine ES, best case scenario (EUR_{2020}/FU)	1.9E−03	2.8E−04
Lifetime costs on marine ES, worst case scenario (EUR_{2020}/FU)	1.9E−02	2.8E−03

because of its higher mass required to pack the same amount of food. However, since the environmental persistence of PHBV packaging is limited compared to that of PP packaging, the lifetime costs on marine ES are about seven times lower than for PP packaging and this despite its higher mass.

The results highlight the importance of proper management of plastic waste to prevent leakages and hence, marine plastic litter formation. Furthermore, lightweight and biodegradable materials are preferred for their lower marine litter formation potential and environmental persistence, limiting their impacts on marine ES. On the other hand, it is important to keep in mind the limitations of this study while interpreting the results. Although this is still a simplified assessment, it opens discussion and addresses an important criticism of LCA for plastics.

3.3. Critical look at the evaluation of the environmental impact of plastic packaging

3.3.1. Sensitivity analysis of the performed life cycle analysis

With respect to the parameters that most influence the environmental impact per FU (SI, Fig. D1), the chickpea content predominates, and this for both packaging options. This corresponds to the fact that the total environmental impact is mainly determined by the production of food ingredients, in which chickpeas account for more than 78 % of the impact.

To a much smaller extent, the environmental impact of 1 kg consumed packaged falafel in a conventional tray is influenced by the storage volume of packaged falafel and the sunflower oil content. In the case of the alternative tray, the variables that contribute most to the variation in total impact, after chickpea content, are the mass of the tray and the storage volume of packaged falafel using alternative packaging. Again, the share of the latter two in the total variance is small compared to the chickpea content. The volume of the packaging is used to calculate utility consumption at distribution, retail and consumer. Minimizing the volume would therefore reduce the impact. Currently, the volume of the packaging is different for the two cases due technical constraints during the experimental set-up, but it can be assumed that the same volume can be obtained for the alternative tray. The allocation key is based on the OEFSR guidelines (Quantis, 2018). If data were available, other allocation methods could be applied, such as the amount of stock in the distribution centre or supermarket, and compared to this result. With respect to the mass of the alternative tray, difficulties to reduce the impact were discussed in Section 3.2.1.

To understand the influence of other key parameters affecting the variability of alternative tray production, a sensitivity analysis of the alternative trays per kg material is performed (SI, Fig. D2). From this, it can be concluded that the two parameters use of NH_4Cl and use of $NaHCO_3$ during the PHBV powder production have the largest influence on the overall impact, each having a share of about 38 %. These parameters are followed by electricity consumption during several sub-processes, e.g. PHBV production, compounding and thermoforming (SI, Fig. A1).

Finally, the results of the sensitivity analyses for the main contributing stages are presented in SI, section D (Figs. D3–D6).

3.3.2. Evaluation of the impact of plastic food packaging on marine ecosystems

The introduced indicator allows a first insight into the added value of biodegradable packaging by including the effect of leakage into the natural environment. Although still a simplified assessment, it brings quantification, opens up discussion and addresses a major criticism of plastic packaging LCA. However, future research could focus on refinements as suggested in the following sections.

3.3.2.1. Fate of plastic waste in Europe. Evaluating the probability of plastic waste collected in the EU becoming marine litter is key to determine the lifetime costs on marine ES of the packaging under study. However, this relies on multiple assumptions due to the difficulty of gathering reliable data on specific European plastic food packaging waste. The fraction of plastic waste discarded by consumers (i.e. not collected) is not included (SI, Fig. B1). Though, accounting for direct leakage to the environment due to bad consumer behaviour will slightly change the results, even though the contribution to the total leakage in developed countries is limited (Kedzierski et al., 2020). Furthermore, the potential formation of marine litter is affected by the polymer type because of the following assumptions. The fate (i.e. in terms of export and treatment options) of plastic food packaging waste is assumed to be similar to the average fate of plastic waste due to limitations related to recent estimates of European food packaging waste exports. Moreover, the fate of PHBV and PP packaging material is considered to be equal in this study. Because experiments indicate that PHBV is biodegradable in soil, freshwater or sand (Deroiné et al., 2015; Meereboer et al., 2021), some of the PHBV packaging found in terrestrial ecosystems may be degraded before reaching the marine environment. In further research, it might be interesting to explore the possibility of integrating the fate factors of Maga et al. (2022) to refine this methodology. Finally, the likelihood that mismanaged plastic waste ends up in the oceans is also relying on the polymer type, which determines its density and thus its dispersion potential (via wind, precipitation and rivers). Since PHBV has a higher density than PP, the mobility of empty PHBV packaging will be lower than that of the equivalent PP packaging. Further refinement is needed to include the dispersion potential of the polymer type, as introduced by Civancik-Uslu et al. (2019) for relative assessments of plastic bags.

3.3.2.2. Polymer-specific impacts. In terms of lifetime costs on marine ES, persistence in the marine environment is the only input parameter to distinguish between different food packaging types. In other words, the impact on the marine ecosystem is assumed to be comparable among polymer types (i.e. annual costs) and is integrated over varying degradation times in marine conditions to assess the polymer-specific lifetime costs. The annual costs of marine plastic litter considered in this study rely on the proportion of marine ES loss (1–5 %) for the current level of marine plastic litter. This share stems from a global evidence study that considers marine litter as a whole (Beaumont et al., 2019). However, the hazards of marine plastic debris to ecosystem processes vary with polymer-specific aspects such as density (influencing the sedimentation rate of plastic particles and thus the interaction time with marine organisms), incorporation of toxic chemicals, capacity to adsorb persistent organic pollutants and fragmentation rate into micro- and nanoplastics (Andrady, 2015; Ballerini et al., 2018; Galloway et al., 2017). The inclusion of polymer-specific impacts on marine organisms and hence, on marine ES, represents a potential improvement of this indicator.

3.3.2.3. Regional perspective in assessing ecosystem services value. The evaluation of the lifetime costs of marine plastic litter relies on the methodology introduced by DeWit et al. (2021). However, their results do not coincide with those obtained in this study for two reasons, highlighting the importance of considering the region-specific perspective when interpreting the results of this new indicator.

The first reason has to do with determining the present value of global marine ES. The authors report the lifetime costs of one tonne of non-biodegradable marine plastic litter in USD 2019 based on the USD 2007 estimate of global marine ES from Costanza et al. (2014), which was converted to USD 2019 according to the US consumer prices index (US Bureau of Labor Statistics, 2022). Similarly, the current study relies on the global marine ES value (USD 2007) (Costanza et al., 2014), which, however, is first converted to EUR 2007 according to the EUR/USD average exchange rate in 2007 prior to its actualization to EUR 2020 using the European harmonized index of consumer prices (Eurostat, 2021a, 2021b). Hence, global marine ES are valued at 84.9 trillion EUR 2020 per year. In contrast, when the global value of marine ES considered in DeWit et al. (2021) (in USD 2019), is converted to USD 2020 considering the US consumer prices index and further converted to EUR 2020 considering the annual average exchange rate EUR/USD, a value of only 54.3 trillion EUR 2020 per year is obtained. The 36 % lower value is due to diverging consumer prices indices in the US and the EU (i.e. inflation), which are not fully offset by the exchange rates between the two currencies; the latter being influenced by additional macroeconomic descriptors. This demonstrates the importance of the country/region-specific perspective (i.e. price level) in the estimation of ES value and raises concerns about the use of a single and standard currency to avoid bias in the comparison across studies. Furthermore, the relevance of the consumer price indices for adjusting ES value from 2007 to 2020 (or 2019 in DeWit et al., 2021) is debatable: it would be more appropriate to consider the consumer price index of each country where a specific ES is consumed to determine its current value. This approach would estimate an (annual) average consumer price index based on the consumer price indices of each country, weighted by the country-specific contribution to global ES demand.

The second reason for the deviation with Dewit et al. (2021) arises from the social discount rate considered for discounting the annual cost of marine plastic litter (Eq. (2)). While a rate of 2 % is used in DeWit et al. (2021), a 4 % rate is considered in the current study, following a European perspective (European Commission, 2017). This higher social discount rate gives less weight to the annual cost of marine plastic litter: the lifetime costs of marine litter when using a 4 % social discount rate is half that when using a 2 % discount rate (ceteris paribus).

3.3.2.4. Discounting impacts on ecosystem services. Discounting future loss of ES gives less weight to costs supported by future generations, which raises intergenerational equity issues because of the time preference aspect included in the discount rate (Koetse et al., 2018; Padilla, 2002). The use of declining discount rates has been studied to better account for future environmental impacts. Besides intergenerational equity aspects, this reasoning assumes that environmental impacts will be valued more in the future due to increasing environmental concerns, higher spending on environmental restoration, higher income levels and scarcer natural environments (Koetse et al., 2018; Azar and Sterner, 1996; Weitzman, 1994). Furthermore, varying discount rates can be used depending on the specific ES under consideration. In this approach, future costs for ES with limited substitutability and high demand are discounted at a lower discount rate than others due to their relative importance (Zhu et al., 2019). In the current study, relying on a global estimate for ES, a constant and unique discount rate was applied. To compare plastic food packaging with impacts on different ES over varying lifetimes, a set of ES-specific and varying discount rates could be applied.

3.3.3. Towards a comprehensive analysis of the environmental sustainability of food packaging

In this study, a broad analysis of food packaging is offered by a well-considered choice of FU to pay attention to FLW and by including all life cycle stages of the packaging. In this sense, both direct and indirect impacts are included. This is in line with the review of Pauer et al. (2019). They introduced a methodological framework to compare the sustainability of food packaging focusing on direct environmental impacts due to packaging production, use and disposal, indirect impacts and circularity. In their study, indirect impacts refer to the amount of FLW at the consumer level due to, for example, difficulties in emptying the packaging. The evaluation in this study goes even beyond the suggestions of Pauer et al. (2019) and classical LCA by covering also the likelihood and impact of plastic's release into the natural environment. On the other hand, circularity is not considered in this study because the focus is put on the environmental impacts of packaging and not on the contribution to a circular economy.

In this study, there is no difference assumed in FLW due to the fact that the shelf life was the same in the performed experiments. However, although approaches to correlate shelf-life and FLW exist (e.g. Conte et al., 2015), it might be challenging to gather all required data for a classical LCA study. Next, more research is needed into all aspects of packaging influencing the amount of FLW. For instance, Cooreman-Algoed et al. (2022) show that the impact of consumer behaviour on the amount of FLW and related environmental burden is substantial. Therefore, data on the influence of the type of food packaging on the amount of FLW by the consumer would be of added value.

Finally, the results of the classical LCA and the evaluation of the lifetime costs on marine ecosystems are offered separately. To come to an integrated assessment, a clear cause-effect chain should be elaborated, as the first steps are achieved in Woods et al. (2021). However, also the option of a new midpoint impact category ‘marine ecosystem services’ could be investigated. Only when the same level of elaboration of the cause-effect chain would be achieved, the results could be integrated. One way could be to monetarize all impacts, as proposed in the study of Taelman et al. (2023).

4. Conclusion

Packaging can play a primary role towards more sustainable food supply systems, but there are several environmental concerns associated with food packaging, in particular plastic packaging, such as the use of fossil fuels and improper management that contribute to the release of plastic into the natural environment. This has promoted research into alternative materials, such as biobased and/or biodegradable plastic food packaging, e.g. PHBV packaging. To answer the question of whether these innovative packaging materials are more environmentally sustainable, there is a need for a comprehensive assessment. Therefore, this study introduces an evaluation in which LCA is extended with marine litter assessment.

When considering the full life cycle of packaged falafel, it is clear that the contribution of food production to the total impact is predominant. It accounts for more than 70 % in both cases considered. Also, utility consumption at distribution, retail and consumer plays an important role. When using the conventional packaging, packaging production accounts for 5 % of the impact, while for the alternative case, the contribution amounts to 10 %. The environmental impact of production and dedicated EoL treatment of packaging, calculated by LCA, is expressed per kilogram of falafel consumed at home. Here, the conventional packaging is preferred above the alternative, however, improvement in the new production technology of PHBV pellets might reduce the impact. The high mass of the PHBV tray, required to enable thermoforming and to acquire the same functionality, strengthen the difference in the result. With respect to the end of life, the results are rather conservative as research on the recyclability of PHBV packaging is ongoing, which might affect the results. However, one should keep in mind that the data for the production of alternative trays are based on proxies, as the production of PHBV and the extrusion and thermoforming of this material are not yet implemented at industrial scale. Further improvements over time are therefore possible.

Also for the packaging-related impacts, the balance is in favour of the conventional packaging due to the mass and volume difference affecting transport and utility consumption at distribution, retail and consumer. To emphasize the importance of including the function of packaging to preserve food, the functional unit ‘food consumed’ is considered. In the case of falafel, no changes in shelf life are reported. Other impacts of packaging type on FLW (e.g. protection during transport) are not taken into account in this study but could be interesting for further research.

Finally, additional information on the environmental performance of plastic food packaging is provided by evaluating the impact of plastic leakage due to improper management on marine ecosystems. In other words, the indicator “lifetime costs on marine ecosystem services” enables to monetize the benefits associated with biodegradable plastic food packaging. This results in the following conclusion: the higher weight of PHBV packaging, leading to higher potential marine litter formation, is compensated by its lower environmental persistence, leading to a lower impact on marine ecosystem services.

However, up to now, the introduced indicator ‘lifetime costs on marine ecosystem services’ is only applied to two materials: polypropylene and PHBV. To enlarge the applicability, the costs should be computed for a range of plastics, including both fossil-based, non-biodegradable and (biobased) biodegradable plastics. Furthermore, it would be relevant to not only include the collected waste to calculate the contribution to marine litter, but also uncollected waste discarded by consumers. Finally, other improvements could be made in future with respect to e.g. the fate of plastics in the natural environment, impact on organism, varying discount rates over time. Therefore the proposed indicator is only a first step to bring insight into the impact of plastic use on the marine environment. So although it is only a first step to providing insight into the impact of plastic use and associated marine litter, the proposed indicator nevertheless brings a broader perspective to the environmental assessment.

The contrasting results for the environmental impact evaluated by LCA and marine littering highlight the importance of considering additional aspects to LCA impact categories when benchmarking the environmental performance of plastic food packaging. This enhances the relevance of the broad evaluation to assess plastic food packaging in the function of a major environmental threat (i.e. plastics accumulation in natural environments) besides additional benefits such as limiting burdens shifting among (non-)LCA impact categories.

CRediT authorship contribution statement

Lieselot Boone: Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Project administration, Software, Validation, Visualization, Writing – original draft, Writing – review & editing. **Nils Préat:** Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Validation, Visualization, Writing – original draft, Writing – review & editing. **Trang T. Nhu:** Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Software, Validation, Writing – review & editing. **Fabio Fiordelisi:** Investigation, Validation, Writing – review & editing. **Valérie Guillard:** Methodology, Validation, Writing – review & editing. **Matthias Blanckaert:** Investigation, Validation, Writing – review & editing. **Jo Dewulf:** Conceptualization, Methodology, Supervision, Validation, Visualization, Writing – review & editing.

Data availability

The data used for the calculations are in aggregated form in the Supplementary information.

Declaration of competing interest

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

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2023.164781.

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