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Environmental assessment of the food packaging waste management system in Spain: understanding the present to improve the future

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Keywords:.

ABSTRACT

One-way packaging can represent up to half the environmental impacts of the food value chain and thus optimising its management is essential. Collective selective waste collection managed by authorised organisations (Extended Producer Responsibility, EPR), with or without Deposit-Refund Systems (DRS) are alternatives implemented at European level to handle this problem. Since there is no single simple formula that can be applied to every waste management system, this case study is focused on the entire Spanish model of one-way food packaging waste management, from collection of each fraction in specific containers to final treatment, considering eight different materials. For the analysis, six different impact categories were considered: abiotic depletion potential, global warming potential, acidification potential, eutrophication potential, ozone depletion potential and photochemical ozone formation. Results reveal that the recycling stage is the main stage contributing to the environmental impacts, but the environmental savings related to the

recovery of materials in this stage compensates these loads and the system must be considered advantageous for the environment. By contrast, sorting plants present the lowest contributions and is the least significant stage. Significant environmental improvements (close to 10%) would be achieved by addressing the total bulk collection flow to mechanical-biological treatment and increasing the selective collection of light and glass packaging waste. This study can serve to identify common drivers that contribute significantly to the development of an integrated approach to waste packaging management and as baseline for comparison studies with alternative waste recovery technologies and systems.

1. Introduction

As a consequence of globalisation, an increasing distance between farmer and consumer is observed. Packaging systems are essential in this process to facilitate the protection, transport and storage of food products, while preserving food from being wasted (Wikström et al, 2018). Reducing food losses and waste (FLW) is a significant challenge to our society, which endangers long-term food security owing to the current pattern of resources exploitation by food systems. In fact, it is estimated that around one third of food production is lost or wasted along the supply chain (Gustavsson et al., 2011), which involves huge quantities of resources embedded in its production, as well as related environmental impacts. For example, food systems represent up to the 30% of final energy use, 70% of world's freshwater withdrawals and 20-30% of greenhouse gas emissions (Garnett, 2011, Vermeulen et al., 2012). Moreover, in a global context of increasing population, such figures are expected to increase by 50% by 2050 to satisfy population needs (Vora et al., 2017).

The behaviour of consumers plays a key role in FLW generation in industrialised countries, and so does the food packaging. For example, the Waste and Resources Action Program (WRAP) in UK showed that around 80% of FLW in households is thrown away in packs containing more than 25% of their contents, indicating that the packaging size is an important consideration for certain food products (Quested and Murphy, 2014). But other aspects of packaging contribute to FLW as well. Packaging failures can damage food during transportation and additional food degradation appears when inadequate packages do not provide proper protection against oxygen, moisture or microbes (Pauer et al., 2019). Moreover, the changing trend observed in the structure of households reveals that new consumption habits are being developed and, thus, it is key to ensure that the packaging sizes are able to meet such needs. In this scenario, facilitating handling functions such as unitization, apportionment, resealability and emptying must be considered crucial to minimize FLW (Heller et al., 2018). Therefore, packaging sizes should be optimised, so that food items are apportioned in suitable sizes to avoid overconsumption or wasted products, which usually have a greater environmental impact than the packaging itself (Wohner et al., 2019; Verghese et al., 2015; Silveniues et al., 2014).

Despite the fact that the implementation of one-way packaging has been easily accepted by consumers, it is worth to mention that it has reinforced the use and throw away culture. In fact, the amount of packaging waste is growing in European countries. For instance, the total domestic production of packaging waste in Spain has increased from 5.8 Mt in 1997 to 7.2 Mt in 2016, with similar trends in other UE countries (Eurostat, 2017). According to Simon et al. (2016), packaging can contribute up to 45% of the environmental impacts of the food value chain. Consequently, packaging waste is undoubtedly one of the challenges to put into practice the principles of the EU action plan for circular economy (EC, 2015a), since the employment of reusable packages and recovery of recycled materials can replace the production of single-use packages, which aims to decouple economic growth from resource consumption by maximising the use of waste (EMF, 2013).

The abandonment and losses in the environment of packaging waste have an impact on the biodiversity and involve significant socio-economic costs (EU, 2019). The millions of tonnes of plastic litter that are found in the oceans every year are one of the most visible and alarming signs, causing growing public concern and being object of numerous initiatives such as the European strategy for plastics in a circular economy (EC, 2018) and the EU Directive regarding the consumption of lightweight plastic carrier bags (EC, 2015b). To reduce such impacts on the environment, the Packaging and Packaging Waste Directive (EC, 1994) promotes prevention of the production of packaging waste and the development of reuse, recycling and recovery systems, before its final disposition. This directive enhances the selective collection of wastes, focusing the responsibility of its management on producers (Extended Producers Responsibility, EPR), and opening the possibility of promoting the use of “Deposit-Refund Systems” (DRS) in the EU member states, although under certain circumstances. On the one hand, while in Spain or France packaging producers have decided to apply for the EPR by means of the Green Dot System, covering all packaging materials and formats, in other countries such as Finland or Denmark, they have implemented DRS mainly for PET bottles, glass jars and cans. On the other hand, in Germany, Sweden or Norway there is a coexistence between EPR and DRS alternatives for certain types of packaging. In the light of the above, it can be said that there is no single simple formula that can be applied to every waste management system that will make that system optimal. The variation between geography, demographics, politics, legislation, management infrastructure and public opinion within countries makes a single solution impossible (McDougal et al., 2001). In this sense, this case study is focused on the current Spanish packaging waste management system, integrating all the possible waste treatment options included in the waste management hierarchy promoted by the EU (EC, 2008), after reducing and re-using strategies have been applied. The aim of this paper is identifying the advantages and disadvantages of the implemented scheme and contributing to the identification of common drivers to develop an integrated approach to waste management.

In recent decades, European policies foster life cycle thinking as a key tool in the field of waste management in order to facilitate the decision-making process towards sustainable development (UE, 2004; Vagt, 2007). The environmental impacts of food packaging have been widely studied in the literature using life cycle assessment (LCA). Some studies are focused just on the comparison of different packaging materials, like plastics, cardboard, glass, steel, aluminium or tin (Huang and Ma, 2004; Laso et al., 2017; Navarro et al., 2018; Gallego-Schmid et al. 2018), whereas others analyse particular applications of those materials. Examples of this are the analysis of different beverage application materials (Simon et al., 2016) or more specifically drinking water bottles, a case where the use of bioplastics such as polylactic acid (PLA) can offer advantages over the use of polyethylene terephthalate (PET) bottles in categories like global warming potential, but the use of pesticides during the production of raw materials can result in higher impact on human health (Papong et al., 2014, Gironi and Piemonte, 2011). Some other studies are addressed to specific materials such as plastics (Barlow and Morgan, 2013; Chilton et al., 2010; Toniolo et al., 2013), aluminium (Detzel and Mönckert, 2009; Niero and Olsen, 2016) or glass (Vellini and Savioli, 2009). Finally, others are limited to fixed system boundaries such as end of life stage (Foolmaun and Ramjeeawon, 2012). As far as the authors could acknowledge, there are no studies that consider all the life-cycle stages of the Spanish waste treatment model for food packaging. Therefore, this study aims to fill this gap, considering this work as baseline for further comparisons with alternative recovery technologies. The paper is structured as follows: Section 2 presents the life cycle assessment methodology including the goal of the study, the system description and the establishment of the system boundaries, the allocations rules, the life cycle inventory and the environmental impact categories used; Section 3 presents the results, including a sensibility analysis; and lastly, Section 4 provides the main conclusions of the study.

2. Methodology

LCA is applied according to the recommendations provided by the ISO 14040 and 14044 standards (ISO 2006a and 2006b). Environmental aspects are quantified all along the life cycle of a product or service and the associated potential impacts are evaluated (Navarro et al. 2018). This study aims to fill the gap through a consistent life cycle assessment framework in order to lay the groundwork for future comparisons with alternative recovery systems. Four phases are necessary for a complete study: (i) definition of the goal and scope; (ii) life cycle inventory – LCI; (iii) life cycle impact assessment – LCIA; and (iv) interpretation.

2.1. Goal and scope

The main objective of this study is to evaluate the environmental performance of the current system for managing packaging waste in Spain, which consists of the selective collection of the different types of one-way packaging. Therefore, the system function is the collection, management and recycling of one-way light and glass packaging.

2.1.1. Definition of the functional unit

To this end, it is necessary to define a functional unit (FU), to which inputs, outputs and environmental impacts will be referred (Laso et al. 2018). In this case, the FU is defined as the amount of one-way light and glass packaging collected, managed and recycled in Spain in 2014. The packaging included in the FU is composed of the following materials: steel, aluminium, polyethylene terephthalate (PET), high-density polyethylene (HDPE), film, mixed plastic, beverage carton and glass. Table 1 defines the reference flow and the amount collected in Spain in 2014 of the different types of packaging under study.

Table 1. Reference flow characterisation of the different types of packaging studied.

Type of packaging	Value (t)
Beverage carton	133,383
Steel	282,401
Aluminium	48,160
HDPE	116,208
PET	259,414
Film	139,825
Mixed plastic	154,045
Glass	1,367,285
TOTAL	**2,500,721**

2.1.2. System description

Within the system boundaries, the waste management process is divided into several stages, from the waste production at households to the last step of its treatment: (i) the disposal of each waste fraction into specific containers; (ii) collection and transportation; (iii) treatment in a transfer plant; (iv) recovery and transformation; (v) recycling process; and (vi) final treatment (see Figure 1).

A detailed description of the current Spanish system for the management of packaging waste is provided to perform the modelling. Within the set of waste management operations carried out under the framework of the system, all the activities included since the waste is generated in households until the last phase of its treatment is performed are considered: containerization, collection and transport, transfer, treatment and conditioning of waste, and recycling. Under the framework of the SCRAP, the collection of packaging collection and recovery routes that are directly managed or funded by Ecovidrio and Ecoembes have been considered: municipal selective collection through containers destined for such purpose, complementary selective collection of private scope made by authorized waste managers, and bulk collection of the residual fraction.

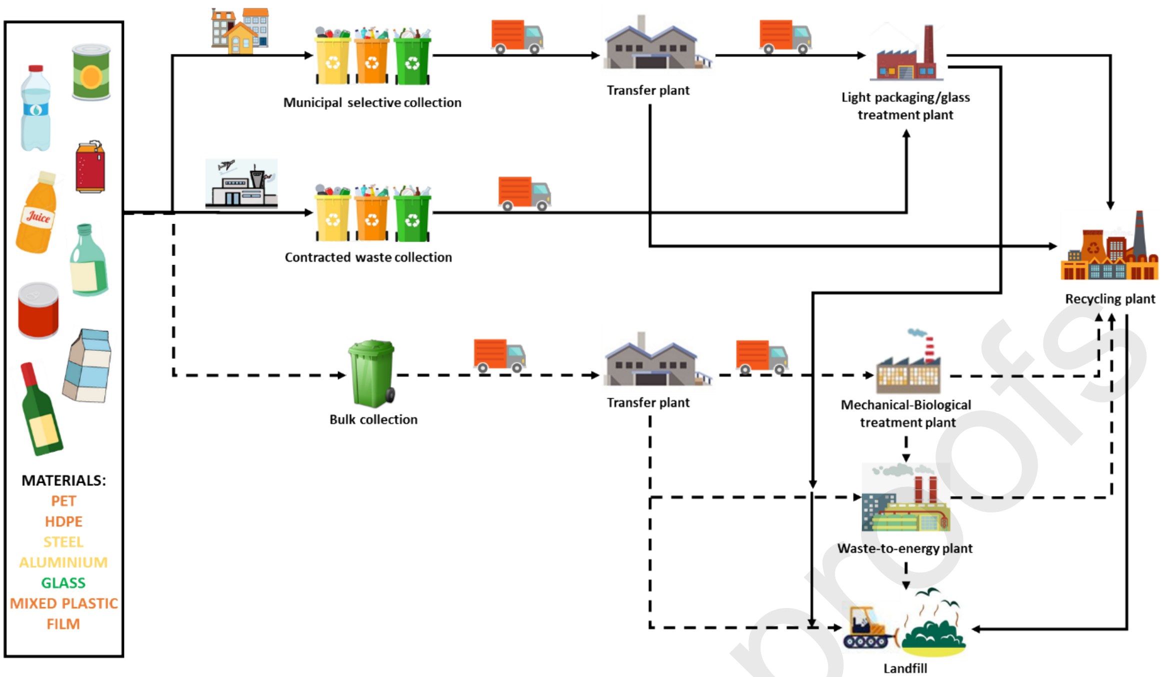

Figure 1. System boundaries overview.

Then, the different stages of the waste management process are explained in detail:

(i). Containerisation

This stage includes the manufacture of the containers (mainly HDPE, steel and fiberglass), cleaning and annual maintaining, as well as their transport to the final waste manager when their lifespan ends. This study only considered the multi-container system, but other minority collection systems are used in Spain, such as door-to-door (1.5%) and mobile pneumatic (0.7%), that are also used in Spain (Ecoembes, 2016; Iriarte et al., 2009).

(ii). Collection and transportation

There are three ways to collect and recover packaging: the municipal selective collection through the specific containers, the contracted waste collection through authorised waste managers and the bulk collection. For selective and bulk collection, this stage includes both the collection of waste and its transportation to the transfer plants; however, for contracted waste collection, waste is directly addressed to sorting plants. Three types of garbage truck were considered: (i) rear loaders, which have an opening at the rear with a lifting mechanism to automatically empty the container; (ii) side loaders, which are loaded from the side with the assistance of a joystick-controlled robotic arm used to automatically lift and tip wheeled containers into the truck´s hopper; and (iii) front loaders equipped with robotic arm with a claw to lift the container over the truck and empty the load into the truck´s hopper (Rosroca, 2017). All these garbage trucks have "pack-on-the-go-hydraulics" that lets the driver pack loads while driving, allowing faster route times.

(iii). Transfer treatment

A transfer station is an installation located close to the cities, where the garbage trucks relocate their load to higher-capacity vehicles. This action allows the optimisation of the waste transport to the treatment plants. Some transfer plants conduct a slight sorting with the aim to improve the quality of the waste sent to the treatment plant. However, this sorting has not been included in the study. Only transfer plants with compaction (to reduce waste volume) and without compaction have been considered.

(iv). Sorting, recovery and transformation treatments

The objective of this stage is the recovery of the recyclable materials, facilitating the use of the rejected materials as energy source or their disposal in a landfill.

- <u>Sorting plants for light packaging (SPLP).</u> These installations are specialised in the classification, either manually and/or mechanically, of the different types of light-weight packaging obtained from selective collection performed by the public and/or private sector. Firstly, garbage trucks are weighted, and the waste is unloaded for homogenisation. Materials likely to cause jams in the rest of the line (film sheets, cardboard, etc.) must be removed. Secondly, a trommel classifies waste by size by means, in order to separate light weight packaging and, after that, a ballistic separator separates the light-flat material (film and P/C) from the heavily rolling material (packages) based on density differences. Then, the different materials are sorted by means of different technologies. For instance, pneumatic suction is used to separate film material, magnetic separators to recover steel, optical separators to segregate PET, HDPE, beverage carton and mixed plastic materials and induction separators to remove aluminium. Finally, quality controls are performed before compaction. The rejected waste material is compacted or stored in containers for delivery to landfills or waste-to-energy (WtE) plants.

- Sorting plants for glass packaging (SPGP). In these installations, the cleaning, classification, either manually and/or mechanically, and crushing of the glass obtained from selective collection performed by public- and/or private-performed is conducted. Similarly to SPLP, garbage trucks are weighted and waste is unloaded for homogenisation. Then, glass waste is fed to magnetic separation to remove ferric materials. Manual separation collects the different types of glass (flint, green and amber) and then size classification is conducted. Waste material smaller than 10 mm is stored, while materials higher than 60 mm are crushed and sent to the beginning of the process. On the other hand, waste material between 10 mm and 60 mm are optically classified, distinguishing the "clean glass" from broken glass, ceramic, stones, porcelain and non-ferric material. The glass recovered is stored and the rejected materials are sent to the beginning of the process. Finally, quality controls are performed to obtain a recovered glass with the required quality to manufacture a new product.
- Mechanical-Biological treatment plants (MBTP). These automatized installations are responsible for the mechanical and biological treatment of the waste from the bulk collection. The former treatment includes the recovery of the recyclable materials (packaging, paper, carton and glass), while the latter is only applied to the organic fraction. Since the organic fraction is not included in the analysis, only packaging materials and the corresponding mechanical treatment was considered. Part of the packaging waste from bulk collection is directly addressed to final destination (i.e. landfill or incineration) without undergoing mechanical-biological treatment.

(v). Recycling process

The recovered materials in the SPLP, SPGP and MBTP are sent to the recycling plants to be transformed into new materials, closing the loop for packaging.

- Pre-treatment and recycling of steel. In Spain, there are three types of pre-treatment installations of steel: recovery (without fragmentation); recovery and fragmentation; and recovery and detinning. The recovered steel is sent to the steel casting where is mixed with different qualities by means of the electric arc technology.
- Pre-treatment and recycling of aluminium. Similarly to steel, aluminium requires a pre-treatment before melting and recovery. Aluminium melting can be conducted by two different procedures: melting and casting (45% of the total entering the recycling plant) or refining and casting (55% of the total entering the recycling plant).
- PET, HDPE, film and mixed plastic recycling. These materials are thermoplastics so they can be easily casted when heated. The mechanical recycling is composed by 5 processes: classification; crushing; cleaning; density separation and drying to obtain the recycling flakes. Regarding to HDPE and film recycling, after the drying step, they are extruded and transformed into pellets. On the other hand, the mixed plastic requires a first classification and separation step.
- Beverage carton recycling. These packaging are composed by three types of materials: paper (75%), polyethylene (20%) and aluminium (5%). In this study, it has been considered that only the paper fraction is recovered and the polyethylene and aluminium waste is sent to a landfill, which is the current situation in Spain.

(vi). Final treatment

The packaging waste which is not recovered in the sorting and recycling plants has two management options: waste-to-energy plants and/or landfilling.

2.1.3. Allocation

The waste management system described is a multi-functional process in which the recovery of energy and materials provides the system with additional functions. This situation was handled with system expansion by subtracting the function of the alternative system (material or energy production) from the system under study. Following the procedure recommended by ILCD (2011) when there is no certainty about the energy source that is being replaced, in our case electricity, the displaced electricity should be referred to the corresponding electricity mix. In this study, the electric power mix of Spain included in GaBi 2016 database, which can be considered representative for the 2012-2018 period, was used.

For materials recovery, the common practice assumes a 1:1 substitution ratio of recycled to virgin materials, which implicitly means that recycled materials replace the same amount of virgin materials with the same quality. This study used the alternative method proposed by Bala et al. (2015) based on two variables: the quantity of material effectively replaced-estimated through the average mix of virgin and recycled materials used in the market for the production of new goods- and the quality of the material obtained-estimated by means of a quality factor. More information is available in section S1 of the supplementary material (SM).

However, in processes such as sorting and recovery of waste materials or waste incineration, more materials than the assessed in this study are involved. In this case, it is also necessary to establish a method to allocate the energy and material consumption and the emissions and residues generated. A hierarchy to perform this impact allocation is defined by ISO 14044 (ISO, 2006a). Table 2 collects the different allocation methods applied. Some processes included in the packaging waste management cycle are simple, in the sense that all material and energy consumption, as well as process emissions are associated with the product (or flow) that enters or leaves that process. However, in processes such as the recovery of packaging in sorting plants or in energy recovery, systems can treat or select more types of goods or products (co-products) than those that interest for a particular study. This can be clearly exemplified in the case of incineration. To an incinerator can reach all waste streams collected in bulk containers. However, for this study, only the impact of the management of aluminium, steel, PET, HDPE, glass and briks was considered. Therefore, most of the allocations were based on the mass amounts, but causality relationships derived from chemical compositions or energy contents by the corresponding mass and energy balances were also considered.

Table 2. Allocation methods used in the different waste treatment stages.

STAGE	ACTION	ALLOCATION
Selective collection	Containers collection and transportation to the transfer of sorting plant	Mass, according to the average composition of the container
Sorting plant	Energy consumption	Mass
Incineration	Consumption of fuel and auxiliary materials	Mass
	Emissions of carbon and heavy metals compounds, sulphur, HCL, HF compounds and dioxins and furans	Causality, based on the carbon, chlorine, sulphur, fluorine, and heavy metal content of the corresponding elements
	Emissions of nitrogen compounds (NO_x, N_2O and NH_3) and particles	Mass
	Energy generation	Energy, based on the low heating value of each material
Landfill	Soil and diesel consumption and emissions linked to the disposal and compaction of the waste	Mass
	Water and energy consumption along the waste degradation	Mass
	Emissions to water and air, and inert along the waste degradation	Chemical composition of materials
	Emissions to air due to gas treatment and biogas burning	Waste composition
	Energy generation	Energy, based on the low heating value of each material
	Emissions to water and air, and sludge generation due to leachate treatment	Waste composition

2.2. Life Cycle Inventory

According to ISO 14044 (ISO, 2006b), the life cycle inventory (LCI) involves the compilation and quantification of inputs and outputs of the system under study throughout its life cycle. The LCI showed in Table 3 includes the material balance of the collection, treatment and recycling of packing wastes, including their effectiveness values, which definition is provided in Section S1 in the SM. All the data used to elaborate the LCI is based on the LCIs developed within the framework of LIFE+ FENIX project (2010-2013) and were updated with GaBi 2016 database within the ARIADNA project (2015-2017) (Fullana-i-Palmer et al, 2017). The life cycle inventories of the different subsystems were developed by different research groups of Spanish or Portuguese universities or technological institutes, as acknowledged hereafter.

Table 3. Summary of the material balance for the system under study (in tonnes).

	Beverage carton	**Metals**	**Plastics**	**Glass**
Input	**133,383**	**330,561**	**669,492**	**1,367,285**
Destination				
Selective collection	84,265	110,691	372,459	858,731
Bulk collection	48,730	219,329	294,447	504,413
Littering	388	541	2,586	4,142
SPLP				
Input	55,717	71,736	298,443	12,959
Output	44,832	65,325	236,904	3,499
Effectiveness	***0.80***	***0.91***	***0.79***	***0.27***
Rejection to incineration	1,742	1,026	9,846	1,514
Rejection to landfill	9,143	5,385	51,693	7,947
SPGP				
Input				880,679
Output				863,066
Effectiveness				***0.98***
Rejection to landfill				17,614
MBTP				
Input	37,229	167,567	224,957	385,371
Output	21,208	148,458	99,925	31,409

Effectiveness	***0.57***	***0.89***	***0.44***	***0.08***
Rejection to incineration	2,884	3,440	22,506	35,396
Rejection to landfill	13,137	15,670	102,526	318,566
CONTRACTED COLLECTION				
Input	28,548	38,955	74,016	151,446
Output	28,548	38,955	74,016	151,446
Effectiveness	***1.00***	***1.00***	***1.00***	***1.00***
Rejection	0	0	0	0
RECYCLING				
Input	94,589	276,459	410,845	953,100
Output	91,287	240,234	371,539	935,487
Rejection	3,302	36,224	39,307	17,614
INCINERATION				
Inputs	***9,255***	***25,302***	***60,324***	***84,829***
MSW to incineration	4,629	20,836	27,972	47,919
SPLP/SPGP Rejection	1,742	1,026	9,846	1,514
MBP Rejection	2,884	3,440	22,506	35,396
Outputs	***9,255***	***23,721***	***60,324***	***72,421***
Material recovery	0	23,721	0	72,421
Energetic valorisation	9,255	0	60,324	0
Effectiveness	***1.00***	***0.94***	***1.00***	***0.85***
Rejection (unrecovered)	0	1,581	0	12,408
LANDFILL				
Inputs	***29,152***	***53,561***	***195,736***	***410,043***
MSW to landfill	6,871	30,925	41,517	71,122
SPLP/SPGP Rejection	9,143	5,385	51,693	7,947
MBP Rejection	13,137	15,670	102,526	318,566
Incineration rejection	0	1,581	0	12,408

2.2.1. Containerisation

According to the *Fundación Centro Tecnológico Miranda de Ebro* inventory for bulk collection used as a baseline (CTME, 2012), there were no data available about the number of containers. These data were estimated according to the methodology showed in Section S2 in the SM. Table 4 shows the number of containers of each type. For light-weight packaging collection, 2,500 L capacity containers (in rural municipalities) and 2,400 L in urban municipalities made of fibreglass-reinforced polyester (6%), HDPE (64%), and steel (30%) were considered. For glass packaging, 2,700 L capacity containers made of fibreglass-reinforced polyester (100%) were assumed. A 7.5, 10, and 8 years lifetime was considered for the HDPE, steel and fibreglass containers, respectively.

Table 4 Number of containers of each type considered for the system under study.

	HDPE	Steel	Fibreglass
Selective collection	228,791	107,246	210,531
Bulk collection	17,495	8,747	17,495

2.2.2. *Collection and transport*

This model considers the type of truck used (rear, side or front), the hopper capacity, the number of containers, and the distance between the different route points (see Table 5). The model of a conventional truck for freight transportation from GaBi 2016 database and the European Reference Life Cycle Database (ELCD) was adapted to garbage truck characteristics.

Table 5. Characteristics of the transport of each type of waste: collected quantity, distance covered and average load.

	Quantity (tonnes)	Distance (km)	Load (%)
Selective collection			
Light packaging	606,956	123	10
Glass	708,495	137	26
Bulk collection	1,066,918	93	49
Contracted collection			
Light packaging	141,519	100	85
Glass	151,446	100	85

2.2.3. *Transfer plant*

There are no official data about the number of transfer plants in Spain; however, as reference, the *Laboratorio Nacional de Energía y Geología* states that in 2012 there were 236 transfer installations (LNEG, 2012). The percentage of packaging waste transported to transfer plants is displayed in Table 6, distinguishing between transfer installation with compaction system and without compaction. According to Ecoembes (Ecoembes, 2014) and Ecovidrio (Ecovidrio, 2014), the 21% of light-weight packaging recovered and the 53% of glass recovered is sent to transfer plants. Light packaging is subjected to a compaction process, whereas glass is stored without compaction. The transport from the transfer plant to the ultimate destination was modelled using a conventional truck for freight transportation from GaBi 2016 database. An 85% load was considered and 65 km and 157 km average distance were assumed for transport of light-and glass-packaging to the treatment plant, respectively. On the other hand, packaging from bulk collection is transported 50 km to the final destination, specifically: 9.5% to incineration, 14.1% to landfilling and 76.4% to MBTP.

Table 6. Distribution of packaging waste dealt with in transfer plants.

	Light-packaging from selective collection	**Glass from selective collection**	**Bulk collection**
Without transfer	79%	47%	
Transfer plant with compaction	21%		100%
Transfer plant without compaction		53%	

2.2.4. *Sorting plant for light-packaging (SPLP)*

The models developed by the *Instituto Andaluz de Tecnología* (IAT, 2012), which are based on data from all the Spanish sorting plants in 2010 and updated with GaBi 2016, were used to describe the packaging sorting plants. Water and energy consumption of the process were included in the inventory using survey data of a 25.5% sample of such plants.

According to Ecoembes, in 2014 in Spain there were 96 sorting plants, 54 automatized and 42 manual. Automatized plants addressed 77% of the total packaging waste recovered, while manual plants treated 23%. Table 6 displays the effectiveness recovery values (IAT, 2012; Ecoembes and Asplarsem, 2012). Additionally, a 0.27 average effectiveness of recovery was considered for glass packaging. Regarding the rejected fraction, 84% was assumed to be landfilled and 16% to be incinerated.

The transport of the rejected fractions to final destination was modelled using a conventional truck for freight transportation from GaBi 2016 database. A 50 km average distance and 85% load were considered. The same truck model was used for describing the transport of the recovered fractions to the pre-treatment, treatment and/or recycling plants.

2.2.5. *Sorting plant for glass packaging (SPGP)*

The environmental impacts of the glass treatment process were modelled using the inventories developed by *Universidad San Jorge* (USJ, 2012). The LCI includes the electricity consumption and the use of lubricating oils. The effectiveness of recovery was 0.98 (Ecovidrio, 2014) and the 100% of the rejected fraction was assumed to be landfilled. In this stage, 880,679 tonnes are estimated to enter the glass treatment plant per year, while 863,066 are recovered and 17,614 are rejected and landfilled.

The same truck model as in the previous stage was used to model the transport of the glass fractions to the landfill and to the recycling plant, considering 50 km and 56 km average distances, respectively, and 85% load.

2.2.6. *Mechanical-biological treatment plant (MBTP)*

The environmental impacts of the MBTP were modelled using the inventories developed by *Escola d'Enginyeria d'Igualada de la UPC* (EUETTI-UPC, 2012), which are based on data from Spanish MBTP associated to Ecoembes and survey data and were updated with GaBi 2016. The inventories are specific for each type of material and plant (manual or automatic). According to them, a 79% of packaging waste enters manual plants and the remaining is managed in automatic plants (EUETTI-UPC, 2012). The effectiveness values used are not empiric, but the theoretical ones that meet the material balance. According to MAGRAMA (2013), 82% of the rejected light-packaging fractions are sent to landfill and the remaining is incinerated, while a 90% of the rejected glass is sent to landfill and the remaining is incinerated.

The same truck model as in the previous stage was used to model the transport of the rejected fractions to final destination (landfill or recycling), considering 50 km average distance and 85% load. Regarding the transport of the recovered material to the pre-treatment, treatment and/or recycling plants, the distances assumed are displayed in Section S3 in the SM.

2.2.7. Treatment of packaging waste collected directly from high generators

To estimate the environmental impacts of the preparation of the packaging waste gathered directly from high generators (such as football stadiums or music festivals), the model of a manual packaging sorting plant was used (IAT, 2012). Effectiveness values equal to unity were considered for every material, assuming that losses are not generated.

The transport of the recovered materials to the pre-treatment, treatment and/or recycling plants was modelled using the same considerations and distances that those for the packaging sorting stage (see Section S3 in the SM).

2.2.8. Preparation for recycling and recycling processes

For plastics and metals, the inventories were built up using survey data from a 24% sample of the companies associated to Ecoembes and Sociedade Ponto Verde. The inventories include the consumption of energy, auxiliary materials and water; and the generation of rejected fractions and wastewater. In addition, the transport of the rejected fractions to the landfill was considered, assuming a 50 km average distance. The inventories of plastics recycling were developed by *Instituto Tecnológico del Plástico* (AIMPLAS, 2012) and are specific for PET, HDPE, film and plastic mix. On the other hand, the inventories for metals recycling were developed by *Instituto Tecnológico Metalmecánico* (AIMME, 2012) and describe the metal smelting by means of the technologies previously described until obtaining aluminium or steel ingots. The model considers that 30% of the metal goes through recovery installations, 30% is sent to recovery and fragmented installations and the rest goes to recovery and metal separation facilities (detinning in case of steel).

For glass, information provided by *Universidad San Jorge* (USJ, 2012) was used. It was built from data provided by a company with 3 plants in Spain and complemented with the glass treatment process of Ecoinvent (Hischier, 2007). The inputs of auxiliary materials were sourced from companies, while electricity and lubricating oils consumption were estimated from average data between the companies and Ecoinvent. The recycling process was modelled using data provided by 11 Spanish companies, considering the savings of water and energy (USJ, 2012).

Regarding beverage carton, a specific inventory developed by *Instituto Tecnológico del Mueble, Madera, Embalaje y Afines* (AIDIMA, 2012) was used. This inventory considered the recovery of 75% of the cardboard contained in this packaging.

The effectiveness values for the materials recovered through SPLP and MBTP were sourced from technical specifications by Ecoembes and Asplarsem (2012). For glass and for materials recovered through contracted waste collection, an effectiveness equal to unity was assumed (see Table 6).

2.2.9. Incineration

To account for the environmental impacts related to energy valorisation, the inventory developed by the group *SOSPROCAN of Universidad de Cantabria* (UC-SOSPROCAN, 2012) was used and updated with GaBi 2016. Data from the Spanish association of energetic valorisation (AEVERSU), characterization inputs provided by Ecoembes, and information available in Integrated Environmental Authorisations were used. The inventories were specific for each type of packaging material. In the case of plastics and beverage carton, the energy recovered in terms of electricity from the combustion of such materials was also considered, as explained in the allocation subsection. More information about this model can be found in Margallo et al. (2014).

2.2.10. Landfilling

The inventories developed by *Universidade de Santiago de Compostela* (USC, 2012) and updated with GaBi 2016 were used to estimate the environmental impacts of packaging landfilling. Inventories are based on experimental values and survey data of different landfill sites, complemented with bibliographic data. They are specific for the different packaging materials and adapted to the climatic and technical Spanish conditions. More information is available in Camba et al. (2014).

2.3. Limitation, assumptions and hypotheses of the study

This section details the main considerations that has been taken into account to describe the performance of the system and to model it.

The amount of packaging waste generated in 2014 was considered equivalent to the amount of packaging produced to cover the products launched on the market in 2014, facilitated by Ecoembes and Ecovidrio. The quantity of packs recovered through collection channels was estimated dividing the quantity recovered from packaging sorting plants by the effectiveness score for each material.

The percentage of containers of different materials (HDPE, steel and glass fibre) for municipal waste was similar to those for light waste packaging, for which data was available. The number of containers for municipal waste was assumed to be twice the one required for light waste packaging.

The distances covered in the transport of packaging waste through selective and bulk collections were considered the same for all types of materials. When no data were available, a 50 km average distance was assumed based on the average distances from the locations to the plants and landfill sites.

The electricity recovered through the incineration of containers was considered to displace the mix of electricity production in Spain. The material recovered through the recycling processes displaces the consumption mix of virgin / recycled material from the market. Regarding the percentages of displaced virgin material, a replacement quality factor of 1 was assumed for metals and glass, 0.91 for PET, 0.79 for HDPE, 0.59 for film and 0.48 for plastic mix.

2.4. Life Cycle Impact Assessment

The life cycle impact assessment (LCIA) phase was carried out using a mix of impact categories from different assessment methods following the recommendations provided by the Joint Research Centre of the European Commission (ILCD, 2011). Table 7 shows the environmental methods and impact categories employed in this study.

These categories were selected because they are based on the most recent scientific consensus. The characterization factors selected in this study are those recommended by the ILCD guide (2011), which are also being used by the pilot tests for the development of the environmental product footprints promoted by the European Commission. The only exception is the Photochemical Oxidant Formation category, since the characterization factors developed by the Impact 2002+ method (Jolliet et al., 2003) were preferred. This choice was made following technical reasons, as it can be considered a more suitable method for this study. This method is also included in the recommended methods of the ILCD and has enough scientific endorsement.

The extraction of the raw materials necessary to manufacture the packaging materials subject of this study contributes to the category of Abiotic Depletion Potential (fossils and minerals). Public concern for Climate Change has been growing in recent years in our society and this environmental impact is normally measured through the Global Warming Potential. The protection of the ecosystems and the quality of forests and soils is directly related to the exposure to substances such as sulphur and nitrogen oxides, which are directly related to the Acidification Potential and Eutrophication Potential. Another of the impact categories included refers to the Ozone Depletion Potential, considered within the Montreal Protocol about substances that deplete the ozone layer, which came into effect more than 30 years ago. The formation of tropospheric ozone threatens the protection of the environment, health and quality of life and it has been considered with the Photochemical Oxidant Formation impact category. Regarding the toxicity indicator (on water, marine or terrestrial environment), it has not been included in this analysis because the packaging considered must fulfil quality requirements that avoid the use of toxic substances. Water depletion as a resource has not been included in the analysis. This impact category depends on local conditions and adequate methodology to determine this impact is still under development and not agreed at international scientific level.

Table 7. Impact categories and environmental assessment methods considered.

Environmental indicator	Environmental method	Reference
Abiotic Depletion Potential (ADP)	CML 2002	Guinée et al. 2002
Global Warming Potential (GWP)	IPCC, 100-year time horizon	IPCC, 2013
Acidification Potential (AP) Eutrophication Potential (EP)	Accumulated Exceedande	Seppälä et al. 2006; Posch et al. 2008
Ozone Depletion Potential (ODP)	ReCiPe midpoint	Van Zelm et al. 2008
Photochemical Oxidant Formation (POF)	Impact 2002+	Jolliet et al. 2003

3. Results and discussion

Table 8 shows a summary of the overall impacts for the system under study. All the results are negative (good for the environment) due to the environmental savings (or credits) related to the recovery of energy and mass, which are higher than the environmental impacts associated to the collection and management of wastes. Therefore, results show that the current system of packaging waste management is beneficial for the environment. The environmental credits for energy and materials recovery are on average 2.7 times the environmental impacts of the system.

Table 8. Summary of the overall impacts for the system under study.

	Total
ADP (kg Sb eq.)	-19,886
GWP (t CO_2 eq.)	-678,376
AP (kmol H^+ eq.)	-2,754
EP (kmol N eq.)	-5,007
ODP (kg CFC^{-11} eq.)	-7.6
POF (kg C_2H_4 eq.)	-397,223

The results have been normalized using the normalization factors (NF) proposed for the Environmental Footprint and Life Cycle Assessment by the EU (Sala, et. Al, 2017), available for all the impact categories used except for POF. In particular, the relative NF per person have been used. The results presented in Figure 2 show, for instance, that the analyzed system avoids the extraction of raw materials (ADP) equivalent to the average ADP of 313,000 world's citizens, or to 80,000 citizens in the case of GWP.

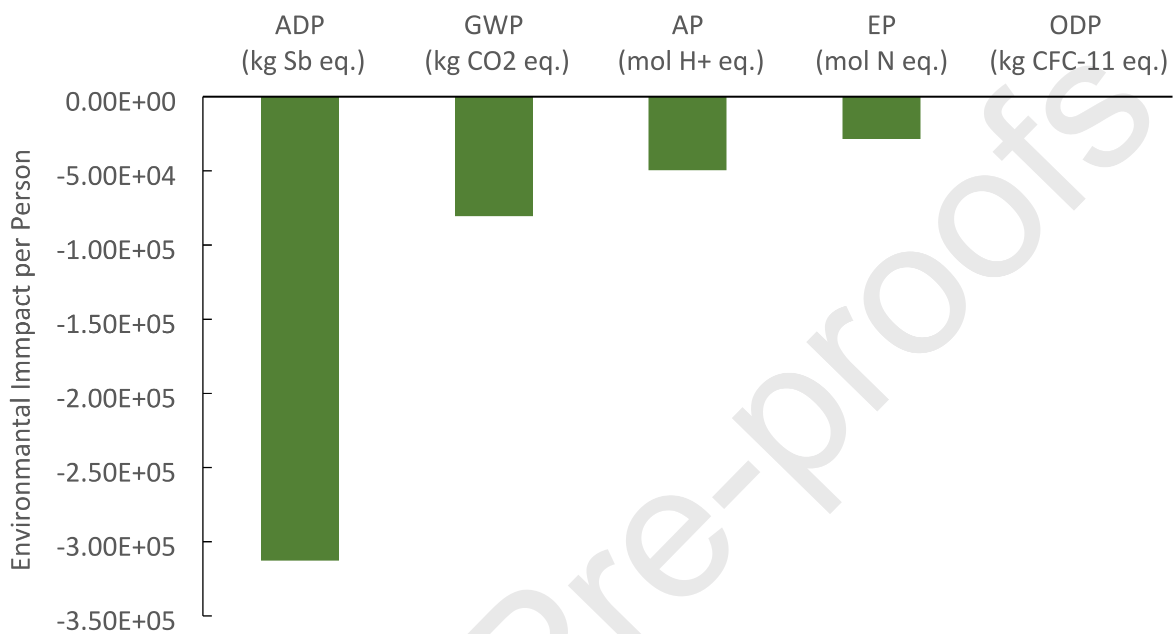

Figure 2. Normalized results of the environmental impacts.

3.1. Life cycle impacts

The contribution of the different stages to the different categories is shown in Figure 3. The recycling stage is the main contributor to all the environmental impacts, ranging from 51.49% for EP to 97.6% for ODP. Regarding the avoided environmental burdens, most of them are related to the recovery of materials, being the contribution of energy credits below 2% on average.

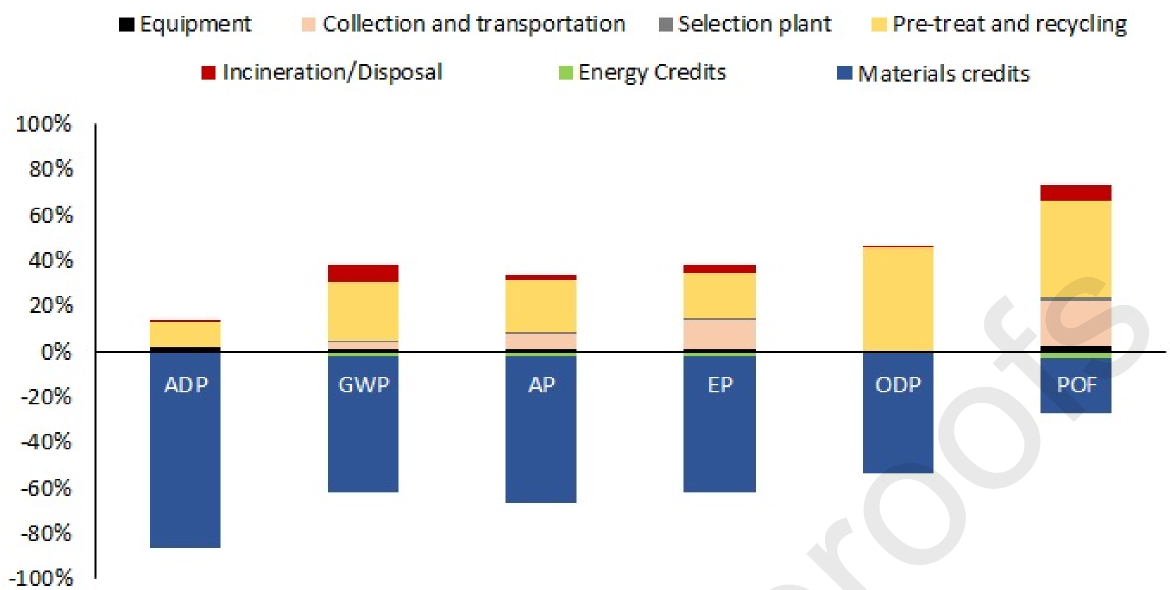

Figure 3. Contribution of the different stages to the different impact categories.

The collection and transport stage is the second main hotspot of the system for AP (20.1%), EP (34.3%) and POF (27.8%), only overcome by incineration and disposal for GWP (18.6%). Equipment and sorting stages represent the lowest contribution to the environmental impacts, ranging from 0.0 to 3.0% for all categories. An exception is observed for ADP, for which the equipment stage represents a 13.4%, due to the fiber glass containers used for glass selective collection.

3.2. Life cycle impacts per materials

Figure 4 displays the contribution of each packaging material under study to the different impact categories in terms of impact and credit, which highly depends on the quantity used of each material. It can be observed that glass is the packaging material whose management present the highest contribution for all impact categories, ranging from 44% for POF to 98% for ODP, except to ADP where metals are the greatest contributors (52%). On the other hand, briks and metals packages present the lowest contributions. Regarding brik packaging, its management contributes 6% to AP, 8% to EP and 5% to ADP, whereas metal packages produce the 14% of POF 10% of GWP and 0% of ODP.

As mentioned in section 2.1.3, the waste management system described is a multi-functional process in which the recovery of energy and materials provides the system with additional functions. Figure 4 also shows the contribution of each material to the credit for each impact category. In this case, glass and plastic are the packaging materials that present the highest credit due to their recycling. Glass credit ranges between 41% for AP to 83% for ODP, whereas the contribution of plastic is 78% for POF and 39% for GWP. Finally, metals reach the highest credit for ADP (84%).

Since credits can be obtained by means of energy and materials recovery, Figure 5 displays this difference in credit contribution for the different impact categories. As can be observed, for all impact categories and for all packaging materials, almost 100% of the credits are obtained by means of materials recovery. Glass and metals cannot be combusted and, for the rest, there are very few incineration facilities in Spain.

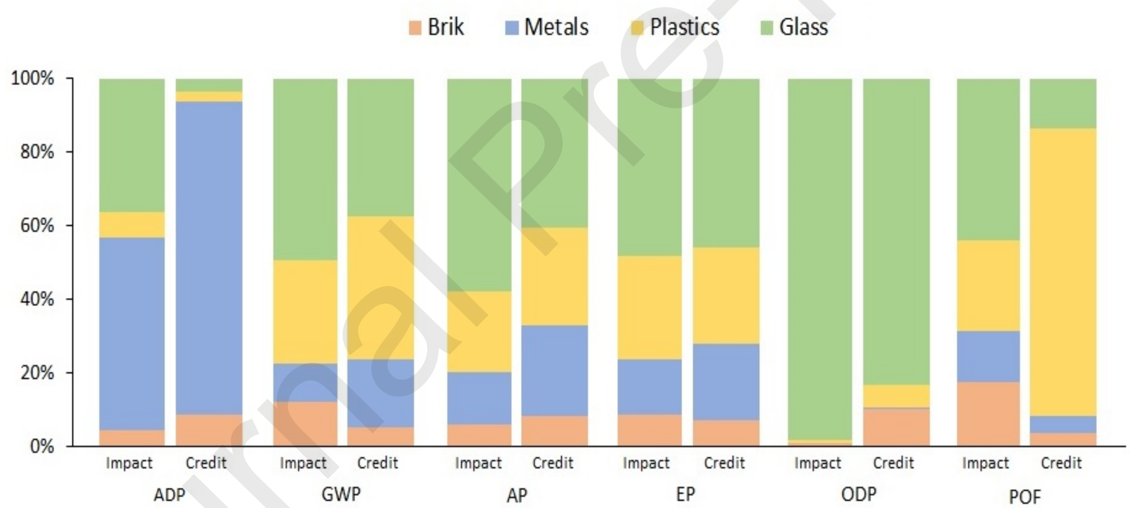

Figure 4. Contribution of each packaging material under study to the different impact categories in terms of impact and credit.

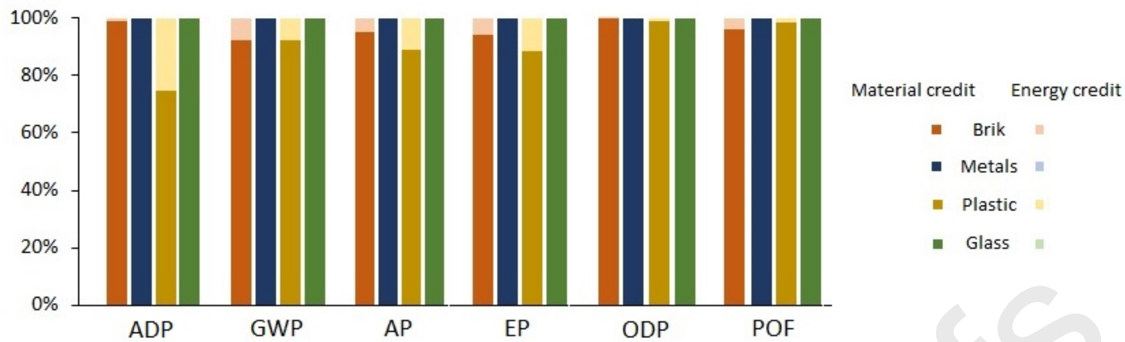

Figure 5. Credit distribution for the different impact categories.

3.3. Specific life cycle impacts per materials in each stage

Figure 6 displays the impact distribution for the different impact categories related to each of the packaging materials under study. As can be observed, pre-treatment and recycling stage is the most contributing in all impact categories for brik, metals and glass, whereas for plastics there is an exception for collection and transport stage, which is the main contributor to POF (44.4%) and EP (44.7%), and incineration, which is the main contributor to GWP (48.3%).

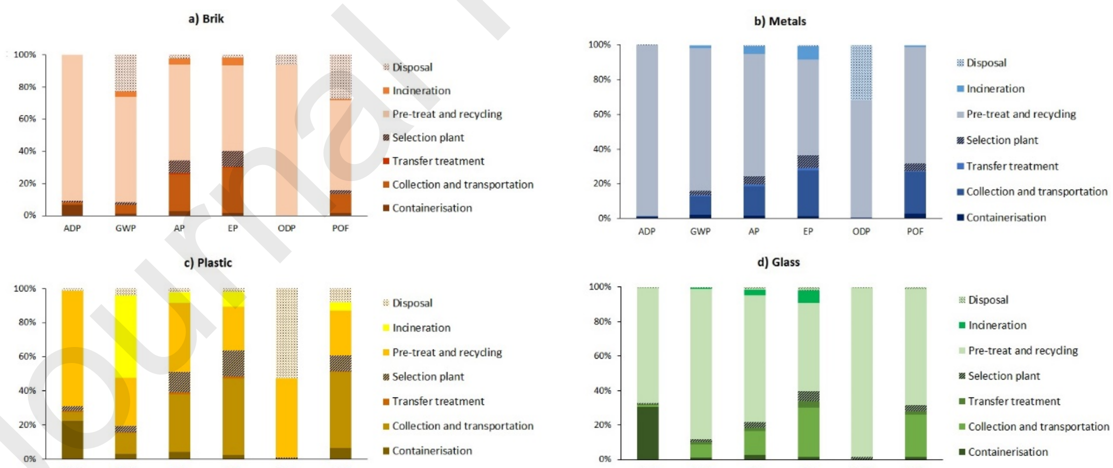

Figure 6. Contribution of the different stages of the life cycle to the different impact categories: a) brik packages, b) metal packages; c) plastic packages and d) glass packages.

3.4. Sensitivity analysis

The objective of this section is to determine the model robustness varying some variables that could affect to the trend of the results obtained (Guo and Murphy, 2012). The parameters varied are collected in Table 9 where the baseline and new scenarios shown.

Figure 7 shows the results of the sensitivity analysis compared to the baseline scenario. In the following sub-sections, each sensitivity analysis is assessed separately. Equation 1 has been applied to estimate the environmental impact variation, where EI_m represents the environmental impact with the modified parameter, while EI_b represents the environmental impact of the baseline scenario. Therefore, a value upper than zero implies that the analysed option is worst that the baseline scenario whereas a negative value seems to the modified option has lower environmental impact than the baseline scenario.

$$\Delta\, Environmental\ impact = \frac{(EI_m - EI_b)}{|EI_b|} \quad \text{Eq.1}$$

Table 9. Parameters and new scenarios assessed in the selectivity analysis.

	Parameter	Baseline scenario	New scenario
SA.1	Percentage of the selective collection of light packaging	37.6%	52.6%
SA.2	Percentage of the private selective collection of light packaging	12.5%	14.2%
SA.3	Percentage of automatic sorting plants	76.7%	100%
SA.4	Percentage of bulk waste sent to MBTP	76.4%	100%
SA.5	Percentage of glass selective collection	50.8%	70.0%
SA.6	Glass selection effectiveness in the MBTP	8.2%	15.0%
SA.7	All	-	-

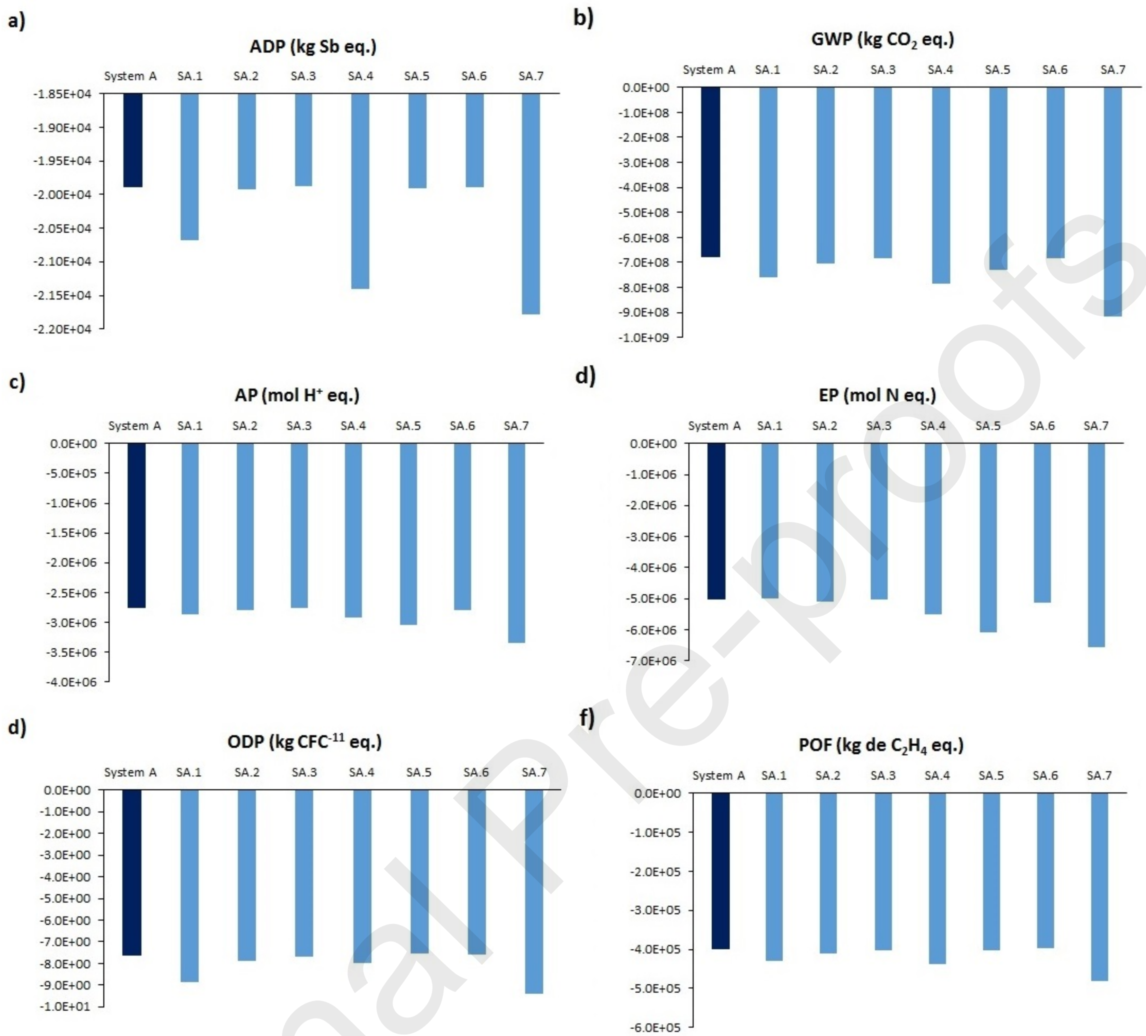

Figure 7. Sensitivity analysis results.

Table 10. Variation of the environmental impact of each sensitivity analysis to the different impact categories

	Unit	SA.1	SA.2	SA.3	SA.4	SA.5	SA.6	SA.7
ADP	kg Sb eq.	-4.0%	-0.2%	0.0%	-7.6%	0.1%	0.0%	-9.6%
GWP	kg CO_2 eq.	-11.9%	-3.8%	-0.9%	-15.5%	-7.3%	-0.9%	-34.9%
EP	mol N eq.	0.5%	-1.9%	-0.4%	-9.7%	-21.7%	-2.6%	-31.3%
AP	mol H^+ eq.	-4.0%	-1.6%	-0.3%	-6.0%	-10.7%	-1.2%	-21.3%
ODP	kg CFC^{-11} eq.	-16.7%	-3.8%	-0.9%	-4.6%	1.1%	0.1%	-23.5%
POF	kg C_2H_4 eq.	-7.9%	-3.1%	-1.1%	-10.5%	-1.2%	-0.2%	-20.9%

SA.1 Percentage of the selective collection of light packaging

This parameter has been modified based on data from the collection of the yellow container of the Autonomous Community of the Basque Country, which is the most effective community of Spain (52.6%). Therefore, the improvement percentage (15%) has been distributed between the different materials according to Table 11.

Table 11. Improvement percentage applied to each material from the modified collection of light packaging.

	Brik	Steel	Aluminium	HDPE	PET	Film	Mixed plastic
Improvement percentage	1.65%	4.62%	0.88%	1.64%	3.44%	0.80%	2.00%

Table 10 collects the relative results of the SA.1 respect to the baseline scenario. As can be observed, the increase of the selective collection percentage improves the environmental profile of the system under study for all impact categories, except to EP, between 4% (AP) and 16.7% (ODP).

SA.2 Percentage of the private selective collection of light packaging

The sensitivity analysis about this parameter has been based on the percentage variation of the private selective collection according to Ecoembes between 2014 and 2015, which increased from 12.5% to 14.2%. This increase has been distributed between the different materials as Table 12 shown.

Table 12. Improvement percentage applied to each material from the private selective collection of light packaging.

	Brik	**Steel**	**Aluminium**	**HDPE**	**PET**	**Film**	**Mixed plastic**
Improvement percentage	0.05%	0.50%	1.57%	1.70%	2.91%	2.80%	2.80%

As can be observed in Table 10 for SA.2, the increase of the private selective collection percentage did not represent a high saving with respect to the baseline scenario, ranging from 0.2% (ADP) to 3.8% (GWP and ODP).

SA.3 Percentage of automatic sorting plants

In this case, it has been considered that all sorting plants are automatic, applying their corresponding effectiveness. However, Table 10 shows that the increase of this parameter to 100% did not have a great influence in the results, between 0% (ADP) and 1.1% (POF). This is due to the relative low relevance of the sorting stage in the overall environmental impact, so the effectiveness of the automatic plants is not as important as other variables of the system.

SA.4 Percentage of bulk waste sent to MBTP

This sensitivity analysis considered that all waste recovered by means of bulk collection are sent to a MBTP, that is to say, there is no waste that is incinerated or landfilled directly as in the baseline scenario. Results collected in Table 10 show that there is an improvement between 4.6% (ODP) and 15.5% (GWP) of the different impact categories. An increase of the percentage of waste sent to MBTP implies the recovery of part of them by means of the packaging selection.

SA.5 Percentage of glass selective collection

This parameter has been modified based on data from the collection of the green container of the Autonomous Community of Navarra, which is the most effective community of Spain (70%). The increase of this parameter enhanced the environmental profile respect to the baseline scenario between 0.1% (ADP) and 21.7% (EP) (see Table 10).

SA.6 Glass selection effectiveness in the MBTP

The glass selection effectiveness in Spanish MBTP is very low (8.32%). In this sense, in the SA 6 it has been considered that 15% of total glass sent to these plants is recovered. According to Ecovidrio, this value is similar to those reached in MBTP from Cataluña. However, the increase of this parameter to 15% did not have a great influence in the results, between 0% (ADP) and 2.6% (EP) because the amount of glass recovered by means of bulk collection is low compared to selective collection (see Table 10).

SA.7 All

Finally, the SA.7 has joint all the considerations assessed in the previous analysis. With the application of these improvements, the environmental profile of the system enhanced from 9.6% (ADP) to 34.9% (GWP) (see Table 10).

4. Conclusions

The environmental impact of the packaging waste produced in Spain, and treated by means of the current and most representative waste management system operating in this country, results to be globally advantageous for the environment. This is mainly due to the fact that the zero burden criteria is applied (then, no environmental loads are charged to the waste produced) added to the environmental savings associated to the recovery of materials.

Results reveal that the recycling process itself is the main stage contributing to the negative environmental impacts of the system (although this is widely compensated by the environmental savings of the recovered materials that could not be recovered without this previous stage), followed by collection and transport stage. In contrast, equipment and sorting plants have the lowest contribution, with the exception of abiotic resource depletion, for which the manufacturing of containers becomes the second main hotspot. The outcomes of the sensitivity analysis suggest that significant environmental improvements would be achieved by simple changes in the system. For instance, if all the residual waste collected in bulk containers were sent to a mechanical-biological treatment plant, before being sent to final destination (landfill or incineration), and improvement of 15% in GWP regarding the current situation could be achieved. Likewise, if the average selective collection rates of light packaging were those of the Spanish region with the highest indexes (which is the Basque Country with a value of 52.6%) the improvement in GWP will be 12%.

This case study allows identifying common drivers that contribute significantly to the development of an integrated approach to waste packaging management at European level. As all the inputs of the model are parametrized, the same model could be adjusted and alternative scenarios analysed easily. For instance, the effects of a drastic decrease of the total amount of packaging waste produced or a significant increase in the amount of packaging waste from biological origin (bioplastics) could be studied.

In addition, this work must be considered as a valuable baseline for comparison studies with novel recovery systems, such as deposit-refund system (DRS), already implemented in other European countries and under discussion in Spain.

Abbreviation

AP	Acidification Potential
ADP	Abiotic Depletion Potential
ELCD	European Reference Life Cycle Database
EP	Eutrophication Potential
FU	Functional unit
FLW	Food loss and waste
FSC	Food supply chain
GWP	Global Warming Potential
HDPE	High-density polyethylene
IPCC	Intergovernmental Panel on Climate Change
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
MBTP	Mechanical-Biological treatment plants
ODP	Ozone Depletion Potential
PET	Polyethyleneterephtalate
POF	Photochemical Oxidant Formation
SDGs	Sustainability Development Goals
SPGP	Sorting plants for glass packaging

SPLP	Sorting plants for light packaging
WtE	Waste-to-energy
WRAP	Waste and Resources Action Program
WRD	Water resource depletion

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REFERENCES

AIDIMA, 2012. Instituto Tecnológico del Mueble, Madera, Embalaje y Afines. Inventario de Ciclo de Vida de la Clasificación y Reciclaje de Papel y Cartón. Documento confidencial del Proyecto LIFE+ FENIX, Giving Packaging a new Life! Mayo 2012.

AIMME, 2012. Life Cycle Inventory of Steel and Aluminium Recycling. Documento confidencial del Proyecto LIFE+ FENIX, Giving Packaging a new Life! Octubre 2012. Instituto Tecnológico Metalmecánico.

AIMPLAS, 2012. Instituto Tecnológico del Plástico. Life Cycle Inventory of Recycling of Household Plastic Packaging Waste. Documento confidencial del Proyecto LIFE+ FENIX, Giving Packaging a new Life! Abril 2012.

Bala, A., Raugei, M., Fullana-i-Palmer, P., 2015. Introducing a new method for calculating the environmental credits of end-of-life material recovery in attributional LCA. Int. J Life Cycle Assess. 20, 645-654.

Barlow, C. Y., Morgan, D.C., 2013. Polymer film packaging for food: an environmental assessment. Resour. Conserv. Recycl. 78, 74-80.

Camba, A., González-García, S., Bala, A., Fullana, P., Moreira, M.T., Feijoo, G., 2014. Modelling the leachate flow and aggregated emissions from municipal waste landfills under life cycle thinking in the Oceanic regions of the Iber. J. Clean. Prod. 67, 98-106.

Chilton, T., Burnley, S., Nesaratnam, S., 2010. A life cycle assessment of the closed-loop recycling and thermal recovery of post-consumer PET. Resour. Conserv. Recycl. 54(12), 1241-1249.

CTME, 2012. Centro Tecnológico de Miranda de Ebro. Life Cycle Inventory of Containers. Documento confidencial del Proyecto LIFE+ FENIX, Giving Packaging a new Life! Marzo 2012.

Detzel, A., Mönckert, J., 2009. Environmental evaluation of aluminium cans for beverages in the German context. Int. J. Life Cycl. Assess. 14(1), 70-79.

EC, 2018. Communication from the Commission: A European Strategy for Plastics in a Circular Economy – Strategy, COM/2018/028

EC, 2019. Good Environmental Status Online Report. https://ec.europa.eu/environment/marine/good-environmental-status/descriptor-10/index_en.htm (assessed 26 July 2019)

EC, 2015a. European Commission. Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the regions. Closing the loop - An EU action plan for the Circular Economy, COM/2015/0614 final.

EC, 2015b. Directive (EU) 2015/720 of the European Parliament and of the Council of 29 April 2015 amending Directive 94/62/EC as regards reducing the consumption of lightweight plastic carrier bags (Text with EEA relevance), OJ L 115

EC, 1994. European Parliament and Council Directive 94/62/EC of 20 December 1994 on packaging and packaging waste, OJ L 365

Ecoembes and Asplarsem, 2012. Diseño de la nueva de pago por selección de envases ligeros en plantas automáticas. Abril 2012. Available at: http://www.asplarsem.com/img/nfp_ecoembes_asplarsem.pdf.

Ecoembes, 2014. Modelos de Gestión de Residuos Urbanos en el marco de la Ley 22/2011. 6 de octubre de 2014. Available at: https://www.ecoembes.com/sites/default/files/archivos_estudios_idi/modelos_de_gestion_de_residuos_urbanos_en_el_marco_de_la_ley_22-2011.pdf

Ecoembes, 2016. Plantas de Selección de Envases Ligeros. Ed. Ecoembes. Madrid. Available at: https://www.ecoembes.com/es/sites/default/files/archivos_estudios_idi/plantas-de-33seleccion-de-envases-ligeros.pdf.

Ecovidrio, 2014. Informe de Sostenibilidad Ecovidrio 2014. Available at: https://www.ecovidrio.es/BlankSite/media/Docs/Memoria-2014.pdf?ext=.pdf.

EMF (Ellen MacArthur Foundation), 2013. Towards the circular economy. Opportunities for the consumer goods sector. https://www.ellenmacarthurfoundation.org/assets/downloads/publications/TCE_Report-2013.pdf

EUETTI-UPC, 2012. Escuela Técnica de Ingeniería. Universidad Politécnica de Cataluña. Life Cycle Inventory of Mechanical-Biological Treatment Plants for Municipal Solid Waste. Documento confidencial del Proyecto LIFE+ FENIX, Giving Packaging a new Life!

Eurostat, 2017. Generation of waste by waste category, hazardousness and NACE Rev. 2 activity. http://ec.europa.eu/eurostat/

Foolmaun, R. K., Ramjeeawon, T., 2012. Comparative life cycle assessment and life cycle costing of four disposal scenarios for used polyethylene terephthalate bottles in Mauritius. Environ. Tech. 33(17), 2007-2018.

Fullana-i-Palmer, P., Bala, A., Colomé, R., Ayuso, S., Díaz-Such, B., Muñoz, I., Retolaza, J.L., Ribas, J., Roca, M., Weidema, B. (2017). ARIADNA PROJECT. “Sustainability study on the introduction of a mandatory DRS for packaging in Spain: comparative environmental, social and financial analysis versus current situation” (in Spanish). Performed by ESCI-UPF, and funded by Ecoembes and Ecovidrio, 2015-2017.
Available at: https://www.esci.upf.edu/frontend/web/uploads/files/Estudio%20Ariadna%20para%20Espan%CC%83a_Final.pdf

Gallego-Schmid, A., Mendoza, J.M.F., Azapagic, A., 2018. Improving the environmental sustainability of reusable food containers in Europe. Sci. Total Environ. 628-629, 979-989.

Garnett, T., 2011. Where are the best opportunities for reducing greenhouse gas emissions in the food system (including the food chain)? Food Policy, 36 (SUPPL. 1), S23-S32.

Gironi, F., Piemonte, V., 2011. Life cycle assessment of polylactic acid and polyethylene terephthalate bottles for drinking water. Environ. Progress Sustain. Energy, 30(3), 459-468

Guinée, J.B., 2002. Handbook on Life Cycle Assessment – Operational Guide to the ISO Standards. Series: Eco-Efficiency in Industry and Science, Vol. 7. 2002, 708 p., Softcover. ISBN: 1-4020-0557-1.

Guo, M., Murphy, R.J., 2012. LCA data quality: sensitivity and uncertainty analysis. Sci. Total Environ. 435-436, 230-243.

Gustavsson, J., Cederberg, C., Sonesson, U., van Otterdijk, R., Meybeck, A., 2011. Global food losses and food waste: extent, causes and prevention. Rome: Food and Agriculture Organisation of the United Nations (FAO).

Heller, M.C., Selk, S.E.M., Keolian, G.A. (2018). Mapping the influence of food waste in food packaging environmental performance analysis. J. Ind. Ecol. 23, 480-495.

Hischier, R., 2007. Life Cycle Inventories of Packagings and Graphical Papers. Ecoinvent report Nº11, Swiss Center for Life Cycle Inventories, Dübendorf, 2007.

Huang, C., Ma, H., 2004. A multidimensional environmental evaluation of packaging materials. Sci. Total Environ. 324(1-3), 161-172.

IAT, 2012. Instituto Andaluz de Tecnología. Life Cycle Inventory of Lignt Containers Sorting Plants. Documento confidencial del Proyecto LIFE+ FENIX, Giving Packaging a new Life! Mayo 2012.

ILCD, 2011. Recommendations for life cycle assessment in the European context – based on existing environmental impact assessment models and factors. Joint Research Centre. ISBN: 978-92-79-17451-3.

Iriarte, A., Gabarrell, X., Rieradevall, J., 2009. LCA of selective waste collection systems in dense urban areas. Waste Manage. 29, 903-914.

ISO, 2006a. ISO 14044: Environmental management – Life cycle assessment- Requirements and guidelines. International Organization for Standardization, Geneva, Switzerland.

ISO, 2006b. ISO 14040: Environmental management – Life cycle assessment – principles and framework. International Organization for Standardization, Geneva, Switzerland.

Jolliet, O., Margni, M., Charles, R., Humbert, S., Payet, J., Rebitzer, G., Rosenbaum, R., 2003. IMPACT 2002+: a new life cycle impact assessment methodology. Int. J. Life Cycl. Assess. 8(6), 324-330.

Laso, J., Margallo, M., Fullana, P., Bala, A., Gazulla, C., Irabien, Á., Aldaco, R. (2017). When product diversification influences life cycle impact assessment: A case study of canned anchovy. Sci. Total Environ. 581-582, 629-639.

Laso, J., Margallo, M., Serrano, M., Vázquez-Rowe, I., Avadí, A., Fullana, P., Bala, A., Gazulla, C., Irabien, A., Aldaco, R. (2018). Introducing the Green Protein Footprint method as an understandable measure of the environmental cost of anchovy consumption. Sci. Total Environ. 621, 40-53.

LIFE+ FENIX - Giving Packaging a New Life. "Finding regional ENvironmental life cycle assessment Information on packaging waste management through fleXible sofwtare tools and databases". LIFE08 ENV/E/000135, Funded by the European Commission, 2010-2013.

Layman report: http://ec.europa.eu/environment/life/project/Projects/index.cfm?fuseaction=home.showFile&rep=file&fil=LIFE08_ENV_E_000135_LAYMAN.pdf

LNEG, 2012. Laboratório Nacional de Energia e Geologia. Life Cycle Inventory of Transfer Stations. Documento confidencial del Proyecto LIFE+ FENIX, Giving Packaging a new Life! Mayo 2012.

MAGRAMA, 2013. Memoria anual de generación y gestión de residuos. Residuos de competencia municipal. Available at: https://www.mapama.gob.es/es/calidad-y-evaluacion-ambiental/publicaciones/memoriadegeneracionygestiondecompetenciamunicipal2013_tcm30-185551.pdf.

Margallo, M., Aldaco, R., Irabien, A., Carrillo, V., Fischer, M., Bala, A., Fullana, P., 2014. Life cycle assessment modelling of waste-to-energy incineration in Spain and Portugal. Waste Manage. Res. 32(6), 492-499.

McDougall, F., White, P., Franke, M., Hindle, P., 2001. Integrated Solid Waste Management: a Life Cycle Inventory (2nd Edition). Blackwell Science (USA), ISBN 0-632-05889-7.

Navarro, A., Puig, R., Martí, E., Bala, A., Fullana, P., 2018. Tackling the Relevance of Packaging in Life Cycle Assessment of Virgin Olive Oil and the Environmental Consequences of Regulation. Environ. Manage. 62(2), 277-294.

Niero, M., Olsen, S. I., 2016. Circular economy: to be or not to be in a closed product loop? A Life Cycle Assessment of aluminium cans with inclusion of alloying elements. Resour. Conserv. Recycl. 114, 18-31.

Papong, S., Malakul, P., Trungkavashirakun, R., Wenunun, P., Chom-in, T., Nithitanakul, M., Sarobol, E., 2014. Comparative assessment of the environmental profile of PLA and PET drinking water bottles from a life cycle perspective. J. Clean. Prod. 65, 539-550.

Pauer, E., Wohner, B., Heinrich, V., Tacker, M. (2019) Assessing the environmental sustainability of food packaging: an extendedlife cycle assessment including packaging-related food losses and waste and circularity assessment. Sustainability, 11, 925.

Posch, M., Seppälä, J., Hettelingh, J.P., Johansson, M., Margni M., Jolliet, O. (2008). The role of atmospheric dispersion models and ecosystem sensitivity in the determination of characterisation factors for acidifying and eutrophying emissions in LCIA. Int. J. Life Cycle Assess., 13, 477–486.

Quested, T., Murphy, L., 2014. Household food and drink waste: A product focus. http://www.wrap.org.uk/content/household-food-drink-waste-%E2%80%93-product-focus

Rosroca, 2018. Available at: https://www.rosroca.es/es/.

Sala S., Crenna E., Secchi M., Pant, R. (2017). Global normalisation factors for the Environmental Footprint and Life Cycle Assessment, EUR (28984), Publications Office of the European Union, Luxembourg, ISBN 978-92-79-77213-9, doi:10.2760/88930, JRC109878.

Seppälä, J., Posch, M., Johansson, M., Hettelingh, J.P. (2006). Country-dependent Characterisation Factors for Acidification and Terrestrial Eutrophication Based on Accumulated Exceedance as an Impact Category Indicator. Int. J. Life Cycle Assess. 11(6), 403-416.

Silvenius, F., Grönman, K., Katajajuuri, J.-M., Soukka, R., Koivupuro, H.-K., Virtanen, Y. The role of household food waste in comparing environmental impacts of packaging alternatives. Packag. Technol. Sci. 27, 277-292.

Simon, B., Amor, M. B., Földényi, R., 2016. Life cycle impact assessment of beverage packaging systems: focus on the collection of post-consumer bottles. J. Clean. Prod. 112, 238-248.

Toniolo, S., Mazzi, A., Niero, M., Zuliani, F., Scipioni, A., 2013. Comparative LCA to evaluate how much Recycling is environmentally favorable for food packaging. Resour. Conserv. Recycl. 77, 61-68.

UC-SOSPROCAN, 2012. Unidad de Sostenibilidad de la Producción de Cantabria. Universidad de Cantabria. Life Cycle Inventory of Packaging Waste Incineration. Documento confidencial del Proyecto LIFE+ FENIX, Giving Packaging a new Life! Mayo 2012.

UE, 2004. Impact Assessment on the Thematic Strategy on the prevention and recycling of waste and the immediate implementing measures. Non-official document. DG Environment. Brussels.

USC, 2012. Universidad de Santiago de Compostela. Life Cycle Inventory of Landfilling. Documento confidencial del Proyecto LIFE+ FENIX, Giving Packaging a new Life! Diciembre 2012.

USJ, 2012. Universidad San Jorge. Life Cycle Inventory of Pretreatment and Glass Recycling. Documento confidencial del Proyecto LIFE+ FENIX, Giving Packaging a new Life! Diciembre 2012.

Vagt, H., 2007. D.3 Position paper. LCA for Sustainability Governance. Requirements. CALCAS (Co-ordination Action for innovation in Life Cycle Analysis for Sustainability) Project.

Van Zelm, R., Huijbregts, M.A.J., Den Hollander, H.A., Van Jaarsveld, H.A., Sauter, F.J., Struijs, J., Van Wijnen, H.J., Van de Meent, D., 2008. European characterisation factors for human health damage of PM10 and ozone in life cycle impact assessment. Atmosph. Environ. 42, 441-453.

Verghese, K., Lewis, H., Lockrey, S., Williams, H., 2015. Packaging's role in minimizing food loss and waste across the supply chain. Packag. Technol. Sci. 28, 603-620.

Vermeulen, S. J., Campbell, B. M., Ingram, J. S., 2012. Climate change and food systems. Annual Review Environ. Resour. 37, 195-222.

Vora, N., Shah, A., Bilec, M. M., Khanna, V., 2017. Food–Energy–Water Nexus: Quantifying Embodied Energy and GHG Emissions from Irrigation through Virtual Water Transfers in Food Trade, ACS Sustain. Chem. Eng. 5(3), 2119-2128.

Wikström, F., Verghese, K., Auras, R., Olsson, A., Williams, H., Wever, R., Grönman, K., Pettersen, M.K., Moller, H., Soukka, R., 2018. Packaging Strategies That Save Food: A Research Agenda for 2030. J. Ind. Ecol. DOI: 10.1111/jiec.12769.

Wohner, B., Pauer, E., Heinrich, V., Tacker, M., 2019. Packaging-related food losses and waste: An overview of drivers and issues. Sustainability 11, 264.

FIGURE CAPTIONS

Figure 1. System boundaries overview.

Figure 2. Normalized results of the environmental impact.

Figure 3. Contribution of the different stages to the different impact categories.

Figure 4. Contribution of each packaging material under study to the different impact categories in terms of impact and credit.

Figure 5. Credit distribution for the different impact categories.

Figure 6. Contribution of the different stages of the life cycle to the different impact categories: a) brik packages, b) metal packages; c) plastic packages and d) glass packages.

Figure 7. Sensitivity analysis results.

TABLE CAPTIONS

Table 1. Reference flow characterisation of the different types of packaging studied.

Table 2. Allocation methods used in the different waste treatment stages.

Table 3. Summary of the material balance for the system under study (in tonnes).

Table 4. Number of containers of each type considered for the system under study.

Table 5. Characteristics of the transport of each type of waste: collected quantity, distance covered and average load.

Table 6. Distribution of packaging waste dealt with in transfer plants.

Table 7. Impact categories and environmental assessment methods considered.

Table 8. Summary of the overall impacts for the system under study.

Table 9. Parameters and new scenarios assessed in the selectivity analysis.

Table 10. Variation of the environmental impact of each sensitivity analysis to the different impact categories.

Table 11. Improvement percentage applied to each material from the selective collection of light packaging.

Table 12. Improvement percentage applied to each material from the private selective collection of light packaging.

- A LCA of the current Spanish waste treatment model for food packaging was done.
- The analysis was done over the amount of packaging waste managed in Spain in 2014.
- The results can be used as a baseline to compare with alternative models.

- Relevant improvements can be achieved by sending all residual waste to MBTs.
- The main challenge is to increase the selective collection of light packaging.