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Contribution of plastic waste recovery to greenhouse gas (GHG) savings in Spain

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ABSTRACT

This paper concentrates on the quantification of greenhouse gas (GHG) emissions of post-consumer plastic waste recovery (material or energy) by considering the influence of the plastic waste quality (high or low), the recycled plastic applications (virgin plastic substitution or non-plastic substitution) and the markets of recovered plastic (regional or global). The aim is to quantify the environmental consequences of different alternatives in order to evaluate opportunities and limitations to select the best and most feasible plastic waste recovery option to decrease the GHG emissions. The methodologies of material flow analysis (MFA) for a time period of thirteen years and consequential life cycle assessment (CLCA) have been integrated. The study focuses on Spain as a representative country for Europe. The results show that to improve resource efficiency and avoid more GHG emissions, the options for plastic waste management are dependent on the quality of the recovered plastic. The results also show that there is an increasing trend of exporting plastic waste for recycling, mainly to China, that reduces the GHG benefits from recycling, suggesting that a new focus should be introduced to take into account the split between local recycling and exporting.

1. Introduction

Given the versatile properties of plastics, such as being lightweight, durable and strong, the world production and usage of plastics has increased sharply (Hong, 2012) from 1.5 million tons (Mt) in 1950 to 299 Mt in 2013. It has been estimated that global plastic production could triple by 2050 (European Commission, 2013; Plastic Europe, 2015). However, plastics, as materials, are generating environmental problems along their entire life cycles. First, to produce plastic products, greenhouse gas (GHG) emissions are generated. Second, the characteristics that deem plastic so useful materials also make waste management problematic (European Commission, 2013), and presently, only a small fraction of plastic waste is recycled due to contamination and technical limitations (Briassoulis et al., 2013; Hong, 2012). Third, there is a considerable accumulation of plastic wastes in the environment. For example, waste patches in the Atlantic and the Pacific oceans are estimated to be in the order of 100 Mt, approximately 80% of which is plastic (European Commission, 2011a, 2013). Once in the environment, particularly in the marine ecosystem, plastic waste can persist for hundreds of years (Kaps, 2008).

Hence, considerable concerns have been focused on plastic waste management. At the European level, several objectives for plastic waste recycling and recovery have been set since 1994 (European Commission, 1994, 2000, 2004). The last Waste Framework Directive (European Commission, 2008) established a 22.5% target for packaging plastic waste recycling that must be reached by all EU Member States by 2020 (Plastics Recyclers Europe, 2012). However, in recent years, plastic waste exports have increased dramatically, both within the European Union (EU) and even more so to third countries due to a demand from fast-growing Asian economies. For example, in 2012, between 32% and 55% of plastic waste collected for recycling in the EU was exported (2.0–3.5 Mt), mostly for recycling in China (BIO

Abbreviations: CE, circular economy; EoL, end-of-life; EU, European Union; EPS, expanded polystyrene; FU, functional unit; GHG, greenhouse gas; HDPE, high density polyethylene; LC, life cycle; LCA, life cycle assessment; LDPE, low density polyethylene; MFA, material flow analysis; MT, million tons; PP, polypropylene; PS, polystyrene; PVC, polyvinyl chloride; PET, polyethylene terephthalate; RPL, recycled plastic lumber.

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Intelligence service, 2013). This situation poses several challenges to the recent EU proposals of resource efficiency and of a circular economy (CE) (European Commission, 2012), where waste is regarded as a valuable resource within Europe (European Commission, 2011b, 2012).

Furthermore, the last Waste Framework Directive (European Commission, 2008) also established the waste hierarchy of prevention in preparing for reuse, recycling, other recovery and disposal; however, it also allows specific waste streams to depart from the waste hierarchy when justified by life cycle thinking and life cycle assessments (LCA) (JRC, 2012a; Lazarevic et al., 2010; Laurent et al., 2014). In this regard, one important limitation with the waste hierarchy is that options for the use of recycled plastic depend on the quality and polymer homogeneity of the material (JRC, 2012b). If the polymer is clean and contaminant-free, it can be used to substitute virgin plastic; however, if the polymer is mixed with other polymers, the options for marketing materials often involve downcycling of plastics for less expensive and less demanding applications (JRC, 2012b). In this case, energy recovery has been presented as a better environmental option.

Therefore, the quality, application and market challenges appear to limit plastic waste recycling, which suggests that the best options to improve plastic waste management while reducing the GHG emissions are not clear. The aim of this study is evaluate different options in order to identify the limitations and opportunities of plastic waste recovery with the objective of decreasing GHG emissions. Addressing these questions can be conducted with dynamic material flow analysis (MFA), which evaluates the flows and stocks of materials within a system defined in space and time (Brunner and Rechberger, 2004) and allows determining changes in waste management trends.

Spain was selected as the case study because it is the fifth highest European plastic consumer (Plastic Europe, 2015), but considering its plastic waste management, Spain is in the middle range of European averages (ANARPLA, 2013; Plastic Europe, 2015). Therefore, the current situation and determine the best strategy to follow can be evaluated. For instance, if efforts should be made to reach the European targets (60% for recovery and 22.5% for recycling), to increase energy recovery, to improve the technology of sorting and recycling or to promote the use of less expensive and less demanding applications of recycled plastics. Thus, several scenarios were defined based on the MFA results and projections. GHG emissions can be calculated with consequential life cycle assessment (CLCA), which quantifies and describes how the environmental impacts will change in response to possible decisions (i.e., increased collection or energy recovery) (Reinhard and Zah, 2009). Similar studies have already been conducted to evaluate the GHG emissions derived from the recycling of paper-cardboard waste and aluminum old scrap in Spain (Sevigné-Itoiz et al., 2014, 2015).

2. Materials and methods

In the following sections, the methodologies used for the MFA (Section 2.1) and for the CLCA (Section 2.2) are explained.

2.1. Dynamic material flow analysis (MFA)

2.1.1. Scope and system boundaries

Plastic waste can be classified as pre-consumer waste (also known as post-industrial waste or industrial scrap), which refers to waste generated during converting or manufacturing processes, or as post-consumer waste, which is produced by material consumers after its use. The MFA is focused on post-consumer plastic waste and on the current end-of-life (EOL) options in Spain, including disposal in landfills, incineration with energy recovery and recycling. Plastic recycling may follow two routes: mechanical recycling, where the plastic waste is converted to new plastic products, and chemical recycling (also called feedstock recycling), where a certain degree of polymeric breakdown occurs (JRC, 2012b). However, recycling plastic as chemical feedstock in industrial processes is negligible in Spain and is not discussed in this paper.

The temporal and spatial boundaries of the MFA were defined as years 1999–2011 and Spain, respectively. Fig. 1 presents the system boundaries of the Spanish plastic cycle. The following life cycle stages (LCs) were considered (from A to K regarding Fig. 1): first, raw materials are extracted and transformed into virgin plastics; after which the plastic products from virgin plastics and recycled plastics are manufactured; next, the products are used, and finally, they become wastes that have to be managed.

In Spain, plastic waste is collected selectively, but an important fraction is collected within the residual waste (referred in this paper as residual plastic waste). The selectively collected plastic waste is sent to sorting plants to eliminate the impurities but also to separate the plastic waste itself into the different plastic polymer categories and/or colors (JRC, 2012). After sorting, recovered plastic can be sent to recycling or energy recovery, while the losses are sent to a landfill. Nevertheless, it should be noted that if recovered plastics are clean and consist of only one plastic type, the recycled plastic substitutes for virgin plastic, but if the plastic wastes are contaminated and/or are a mix of different plastic types, the recycled plastic is used for products that often could be made of other materials (e.g., garden furniture). This recycled mix is known as recycled plastic lumber (RPL). In such cases, the substituted material is not virgin plastic but may be wood for the production of wood lumber (Astrup et al., 2009). As a substitute for treated wood, RPL products offer the advantages of being low maintenance, do not require painting or staining, and are impervious to rot and wood-eating insects (EPIC, 2003).

2.1.2. Flows and stocks estimations

The system under study concerned only material flows and the calculation of both stocks and flows, which is then based only on the principle of mass conservation. For each LC, the total flows entering the LC should equal the total flows leaving it. The flows are detailed in Fig. 1. All these flows were then classified into five groups: (1) trade flows; (2) loss flows; (3) transformation flows that transform raw materials to virgin plastics, from virgin plastics to plastic products, and from consumed plastic products to recovered plastic after its use; (4) recycling flows of plastic waste and (5) energy flows. Each flow was calculated in three ways depending on the data availability; it was calculated directly based on statistics, by combining statistics with coefficients and deduced using the mass balance. The capital letters in the brackets refer to LCs (i.e., virgin plastic production [B]), while the lowercase letters refer to material flows (i.e., recycled plastics (f)). Data collection, sources and explanations of assumptions are summarized in Table 1. In addition, more details are provided in Appendix A.

2.2. Consequential life cycle assessment (CLCA)

2.2.1. Scope and system boundaries

Even though the MFA takes into account the entire life cycle of plastics, as shown in Fig. 1, for the GHG quantifications the focus was only on the GHG emissions from the selectively collected plastic waste for recycling and energy recovery. The following LCs were considered: collection and sorting [E], recycling [F], incineration with energy recovery [G], virgin plastic production [B], wood lumber manufacture [I], raw material extraction [A], wood [J] and electricity and heat production [H].

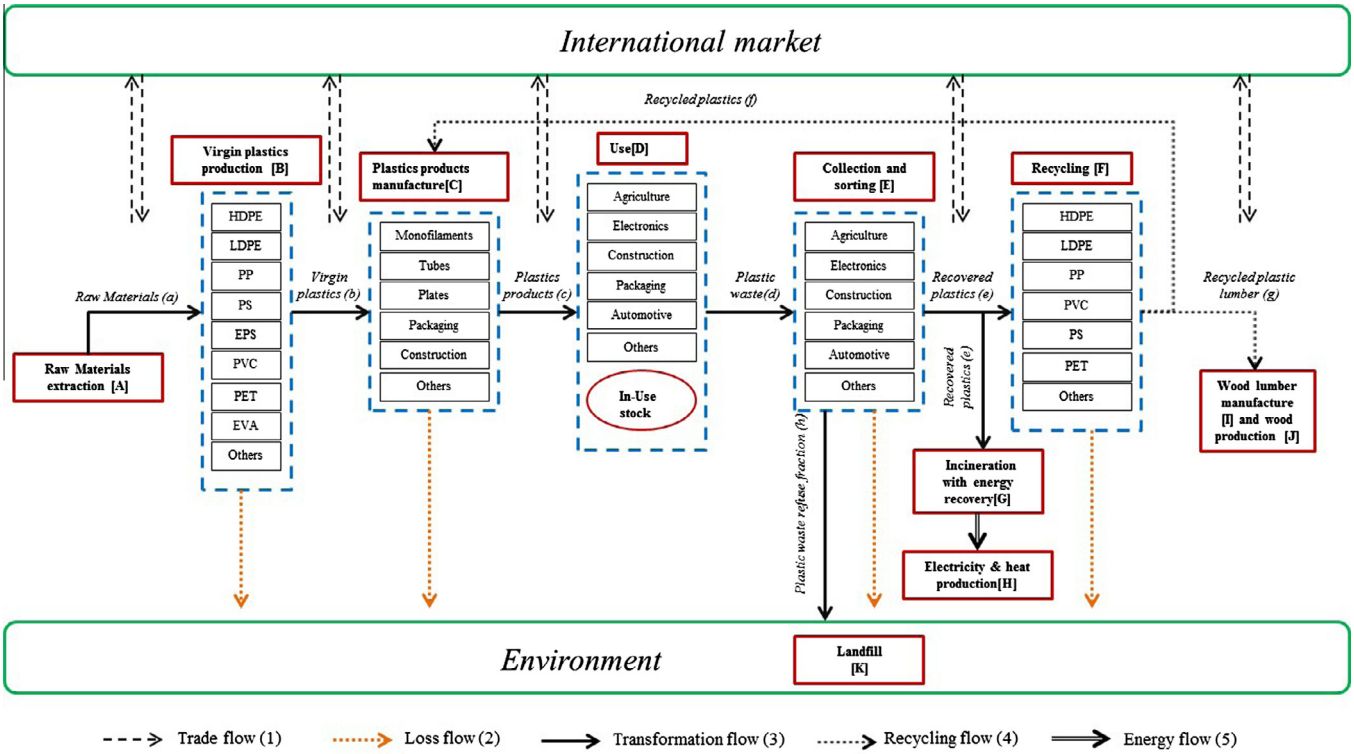

Fig. 1. Spanish plastics system: Boundaries, flows and stocks.

Table 1
Source of data and main assumptions for MFA.

LCs	Source data of production	Source data of trade	Assumptions
Raw materials	–	DataComex (2013)	There is no oil in Spain and all plastic is produced from imported oil. We assumed that 4% of the imported oil goes to make plastics
Virgin plastic production	INE (2013)	DataComex (2013)	Average content of additives for mass balance: 20%
Plastic products	INE (2013)	DataComex (2013)	
Use	Cicloplast (2009)	–	Most of the final products may serve in the use stage for a long time and will not be consumed, so, an in-use stock of plastics will gradually form and enlarge in a defined geographical area such as a city or a country. Thus, the in-use stock in year j can be calculated as the difference between total plastic products consumed (production + import – export) in year j, plus the in-use stock of previous year minus plastic waste generated in year j
Collection and sorting	ANARPLA (2013) and Eurostat (2013)	DataComex (2013)	Residual plastic waste was calculated based on statistics of municipal solid waste (MSW). According to ANARPLA (2013), 64% of packaging plastic waste is collected selectively through containers in the street, thus, the remaining 36% is thrown away within the residual waste Sorting material efficiency: 27% (based on mass balance) (2011)
Recycling	CEP (2012) and ANARPLA (2013)	DataComex (2013)	Material efficiency of recycling plants: 85% (JRC, 2012)
Wood lumber production and wood manufacture	Cicloplast (2009) and Sathre (2007)	DataComex (2013)	
Energy recovery	Eurostat (2013)	–	Energy recovery efficiency: 71% electricity and 14% heat (Schmidt, 2012)
Landfill	Mass balance	–	Total plastic waste sent to landfill is the sum of plastic waste losses from sorting and recycling and residual plastic waste

Both recoveries (i.e., materials and energy) are the case of a multifunctional product, and in CLCA, the derived allocation problem is avoided by system expansion (Weidema et al., 2009). As the use of waste plastics will not affect the amount of plastics collected for recycling, the more recycled plastics are used, the less plastic will be available for other actors in the market. Hence, the marginal effect of collected plastic waste for recycling will be that virgin plastics are affected (Schmidt, 2012), so recycled plastics could substitute virgin plastics. Nevertheless, as explained in Section 2.1.1, depending on the quality of the recovered plastic, it can be used as a substitute of virgin polymers or wood lumber, and both types of substitution were considered in the GHG quantifications. It was considered that the recovered plastic fraction sent to incineration with energy recovery will also avoid the

marginal electricity and heat production (see Section 2.2.2 for more details). Although the GHG emissions of a landfill [K] were outside of the scope of this paper, the plastic waste loss from sorting and recycling facilities which is sent to landfills was taken into account (indicated in Fig. 1 as plastic waste loss flow).

2.2.2. Functional unit, life cycle inventory and life cycle impact assessment

The functional unit (FU) was defined as the increase of 1 ton of selective plastic waste collected in Spain for recycling and energy recovery. The inventory data and assumptions are presented in Table 2, and more detailed explanations are provided in Appendix B.

The marginal electricity production was modeled following previous recommendations by Schmidt et al. (2011) and Schmidt and Thrane (2009). According to MFA results, a fraction of the plastic waste is sent to recycling in Europe and China. Thus, the marginal electricity was calculated differentially by country (or region) for all of the processes involved in this study. For Spain, data and projections from the Ministry of Industry, Energy and Tourism (Minetur, 2011) was used. These projections established that the structure of electricity generation in Spain will continue to evolve over the forecast period in the same way it has in recent years, with a reduction in the amount of oil and coal in the generation mix, a slight increase in natural gas usage and greater growth of renewable energy and hydroelectric generation (Minetur, 2011). For Europe and China, data and projections from Schmidt and Thrane (2009) was used. Table 3 lists the electricity mixes for each country.

SimaPro 7.3.3 software was used for the environmental evaluation, together with the "IPCC 2007 GWP 100a" method, which only considers the impact category of GWP expressed in CO_2 eq. units.

2.2.3. Scenarios and sensitivity assessment

Six scenarios were evaluated and discussed in relation to the GHG emissions. These scenarios were defined based on the MFA results and literature review on the quantity of recovered plastic waste sent to recycling (referred to as management), the ratio substitution between recycled plastics and virgin plastics/wood (referred to as quality), the percentage of recycled plastic use for plastic purposes (referred to as application), and the percentage of exported recovered plastic waste (referred to as market). Table 3 summarizes a list of scenarios and more detailed information is provided in Section 3.2.

Table 3
List of scenarios for the GHG emissions calculations.

Name of scenario	Description
Scenario Man1	Evaluates an increase in the quantity of recovered plastic sent to recycling (Management 1)
Scenario Man2	Evaluates an increase in the quantity of recovered plastic sent to energy recovery (Management 2)
Scenario Qua1	Evaluates a decrease in the quality of recycled plastic to substitute virgin plastics (Quality 1)
Scenario App1	Evaluates an increase in the application of recycled plastic to produce RPL (Application 1)
Scenario Mar1	Evaluates the recycling of all recovered plastic waste in Spain (Market 1)
Scenario Mar2	Evaluates the recycling of all recovered plastic waste outside of Spain (Market 2)

3. Results

The following sections present the results for the dynamic MFA (Sections 3.1.1–3.1.5), the Baseline scenario and the alternative scenarios, which were defined from the MFA results, the trends projected for Europe and literature review (Section 3.2), and the GHG quantification results (Section 3.3).

3.1. Dynamic MFA of plastics from 1999 to 2011 in Spain

3.1.1. From virgin plastics to plastics products

Figs. 2 and 3 show the results of the consumption of virgin plastic, recycled plastic and applications by sector from 1999 to 2011. Virgin consumption grew steadily from 1999 to 2007, exceeding 6,000,000 tons, and then decreased substantially until 2009, likely due to the economic crisis that affected all industries and sectors, especially the construction sector. However, recycled plastic consumption increased year by year from a percentage contribution

Table 2
Inventory data and main assumptions for CLCA.

LCs	Inventory data source	Assumptions
Raw materials and virgin plastics production	Hischier (2007)	–
Collection, sorting and international transport	Rives et al. (2010), Retorna (2011), Itene (2008) and Spielmann et al. (2007)	All distances were calculated using the EcoTransIT tool (EcoTransIT, 2012) Data regarding the electricity and diesel consumption in sorting plants were provided by waste managers in Spain
Recycling	Shonfield (2008), Astrup et al. (2009), Chen et al. (2011) and Schmidt (2012)	Electricity consumption for Spain and Europe: 229 kW h t^{-1} Electricity consumption for China: 575 kW h t^{-1} Material recycling efficiency: 85% Plastic waste losses management: landfill in Spain and China, incineration with energy recovery in Europe
Wood lumber production and wood production	Sathre (2007) Astrup et al. (2009)	–
Energy recovery	Doka (2003a)	Energy recovery efficiency: 71% electricity and 14% heat (Schmidt, 2012) *Marginal electricity substitution:* Spain: Natural Gas (94%), hydropower (6%) Europe: Hard coal (28%), natural gas (46%), hydropower (29%) China: Hard coal (82%), natural gas (2%), nuclear (5%9, hydropower (11%) *Marginal heat substitution:* Spain: Natural gas Europe: Natural gas
Landfill	Doka (2003b)	–

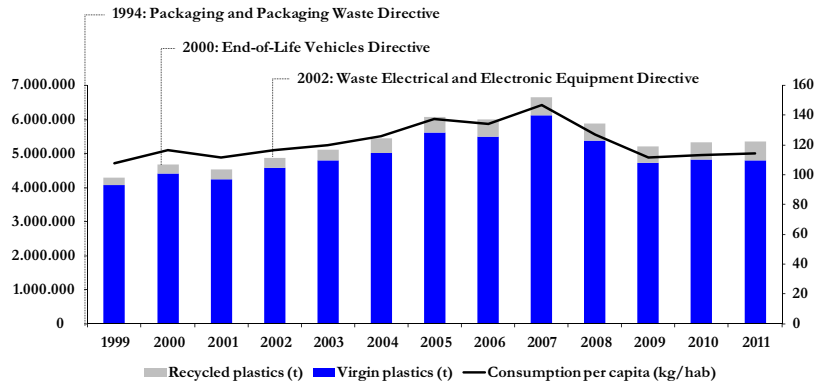

Fig. 2. Consumption of virgin plastic and recycled plastics in ton from 1999 to 2011 in Spain on the left axis and overall consumption per capita from 1999 to 2011 in Spain on the right axis.

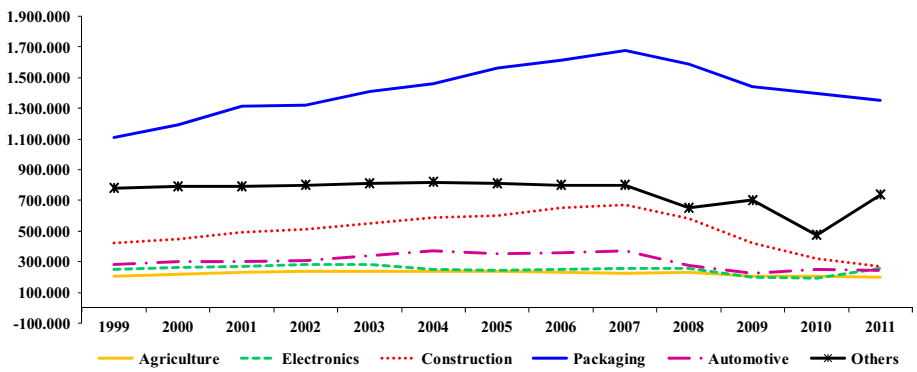

Fig. 3. Consumption of plastic products in ton from 1999 to 2011 in Spain by main application sector.

of 5% in 1999 to 12% in 2011. Production and consumption data disaggregated by type of virgin plastic from 1999 to 2010 are available in Tables A1 and A2 in Appendix A.

3.1.2. Plastic waste management

Table 4 shows (1) the total generated plastic waste and its treatments, (2) the generated packaging plastic waste and its treatments and (3) the remaining generated plastic waste, such as agriculture plastic waste, and its treatments from 1999 to 2011. The total generated plastic waste includes the selectively collected plastic waste as well as the residual plastic waste.

Generated, recycled, incinerated and landfilled plastic waste increased up to 2007 and then decreased slowly following the observed trend for plastic consumption. However, in 1999, 12% of plastic waste was sent to be recycled, while 7% was sent to incineration and 81% was landfilled; in 2011, recycling increased up to 38%, incineration up to 15% and landfilled decreased to 47%. As assumption, the total consumed packaging products were generated as a waste within the same year; thus, if packaging was the main application for plastics, it was also the largest plastic waste stream (approximately 64%). However, if in 1999 more than 67% of the selectively collected packaging plastic waste was sent to landfills, that amount decreased to 49% in 2011. Similarly, recycling and energy recovery of packaging plastic waste increased their presence, reaching 26% and 25% of EOL treatments in 2011, respectively.

Fig. 4 presents the recycling by polymer type in 2011. The main recycled polymer was low density polyethylene (LDPE) (29%), followed by high density polyethylene (HDPE) (24%) and polyethylene terephthalate (PET) (22%), due to the recycling of packaging

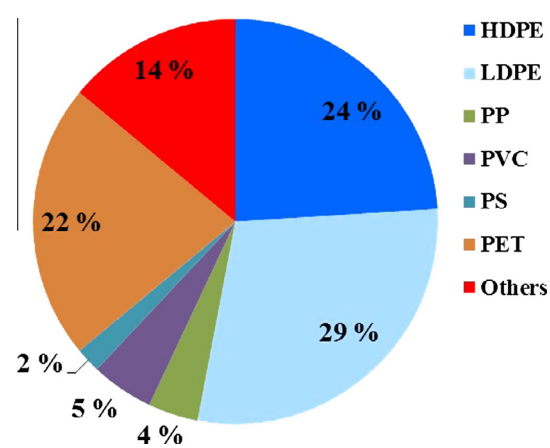

Fig. 4. Plastic waste recycling by type of polymer in 2011.

plastic waste, which is the main consumed plastic product and main collected and recycled plastic waste. Polypropylene (PP), polyvinyl chloride (PVC) and polystyrene (PS) are mainly used in the construction sector (Plastic Europe, 2011). However, PP had a small contribution in the waste streams as it is difficult to quickly identify and separate from other polymers, hampering its effective recovery as a separate stream (JRC, 2012b). In the case of PVC, its small contribution is due to the low collection rates and low efficiency of recycling, as PVC is normally contaminated with other materials (JRC, 2012b). The contributions of PP and PVC in the recycling fraction were 4% and 5%, respectively. Approximately 14% of recycled plastic was classified as others, which represents a mix of

Table 4
Total plastic waste generated and its treatment, packaging plastic waste and rest of plastic waste generated and its treatment in ton from 1999 to 2011 in Spain.

	Plastic waste (ton)				Packaging plastic waste (ton)				Rest of plastic waste (ton)[a]		
	Generated	Recycled	Incinerated[a]	Landfilled	Generated	Recycled	Incinerated[a]	Landfilled	Generated	Recycled	Landfilled
1999	1,735,938	200,200	125,310	1,431,726	1,111,000	159,984	125,310	825,706	624,938	39,772	606,020
2000	1,864,531	268,900	130,900	1,484,970	1,193,300	205,248	130,900	857,152	671,231	64,368	627,818
2001	2,057,813	279,000	182,000	1,620,015	1,317,000	234,426	182,000	900,574	740,813	28,375	719,441
2002	2,060,938	303,700	183,700	1,599,881	1,319,000	258,524	183,700	876,776	741,938	45,704	723,105
2003	2,198,906	329,000	204,216	1,664,568	1,407,300	280,053	204,216	923,031	791,606	61,332	741,537
2004	2,286,211	420,810	219,858	1,621,748	1,463,175	294,098	219,858	949,219	823,036	127,004	672,529
2005	2,445,781	463,311	209,700	1,731,609	1,565,300	324,017	209,700	1,031,793	880,481	138,668	700,026
2006	2,523,438	497,409	279,400	1,710,485	1,615,000	361,760	279,400	973,840	908,438	136,457	736,645
2007	2,623,438	525,931	248,000	1,804,425	1,679,000	391,207	248,000	1,039,793	944,438	134,388	764,452
2008	2,476,563	500,483	247,000	1,865,073	1,585,000	386,740	247,000	951,260	891,563	113,268	733,813
2009	2,245,556	482,893	248,000	1,411,018	1,442,916	383,816	248,000	811,100	811,640	99,799	599,917
2010	2,183,889	515,674	306,000	1,186,864	1,397,689	408,125	306,000	683,564	786,200	108,108	503,281
2011	2,117,430	565,601	312,800	1,108,058	1,355,155	439,070	312,800	603,285	762,275	126,660	504,773

[a] Statistics for incineration are only available for packaging plastic waste, so we assumed same quantity of packaging plastic waste sent to incineration and plastic waste sent to incineration. Therefore, there is no data available of rest of plastic waste sent to incineration and no data was included on the table.

polymers. In this study, it is considered that this mix of polymers is used to produce the RPL in substitution with wood. Data on the plastic waste collection and the plastic waste recycling from 1999 to 2011 for the other sectors considered in this study (agriculture, electronics, construction, automotive and others) are presented in Tables A3 and A4 in Appendix A.

3.1.3. International trade

Fig. 5 represents the commercial balance, defined as the difference between imports and exports of plastic products; thus, lines above the horizontal axis indicate that there were more imports than exports. Between 1999 and 2009, Spain experienced a lack of virgin plastic, which was imported mainly from Germany and France (DataComex, 2013). During the same period, there was also a lack of plastic products, which were imported mainly from Germany and France (DataComex, 2013), but from 2008, there was an important decrease in the commercial balance, indicating that there was less demand for plastic virgin plastic and plastic products, which were exported. Since 2002, also plastic waste started to be exported, and the amount of export of plastic waste increased over the years. In this regard, it is important to highlight that the main destination of the plastic waste changed: in 1999, the main destination was intra Europe (mainly Portugal and France), whereas in 2011, approximately 90% of the plastic waste was sent to Asia (China and Hong Kong) (DataComex, 2013), positioning Spain as the fifth largest European exporter of plastic waste (Anarpla, 2013).

Table A5 in Appendix A presents the export trade by type of polymer (PE, PS, PVC, PP and others). Until 2010, the highest plastic waste imported and exported was PE, followed by PVC, PS and PP; however, in 2010 and 2011, the highest plastic waste traded was classified as others. In this regard, there is no more detailed disaggregation on this classification; because PET is one of the main plastics consumed in the packaging sector, which is also the main rising plastic waste stream, the typology "others" was assumed to correspond to PET wastes.

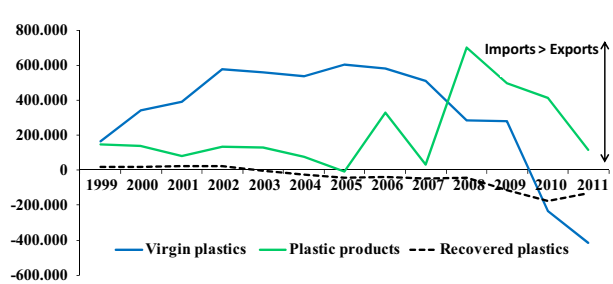

Fig. 5. Commercial balances (imports minus exports) for virgin plastic, plastic products and recovered plastics (packaging plastic waste) in Spain from 1999 to 2011 in tonnes.

3.1.4. In-use stocks

Because the lifetime of many plastic products can be between less than one year (packaging) and more than 50 years (construction), there has been an accumulation of plastic products in use (Kaps, 2008). There are no data available for Spain before 1999; therefore, the stock calculations are underestimated. However, in 2011, an in-use stock since 1999 of 27,034,084 tons of plastic products was calculated, which represents approximately 8 years of supply of plastic at the current consumption rates. Therefore, in subsequent years, this in-use stock will be an important source of plastic waste. In addition, during the same period (1999–2011), there was an accumulation of plastic products in landfills due to the plastic waste within the residual waste, as well as the plastic waste loss from sorting and recycling processes. All these losses would end up in landfills, but a fraction would likely end up in the marine ecosystem. The total plastic waste accumulated in landfills between 1999 and 2011 was 30,542,493 ton, calculated as the sum of the plastic waste losses from sorting and recycling and the residual plastic waste contained in the residual waste.

3.1.5. Spanish plastic life cycle in 2011

Fig. 6 presents the Spanish plastic life cycle in 2011, as it is the most representative year of the current situation. Approximately 50% of produced virgin plastics were exported, while 30% were imported. More than 30% of consumed plastic products were stocked, and approximately 50% of generated plastic waste was collected within the residual waste and sent to landfills. Approximately 12% of selectively collected plastic waste was exported, 21% was sent to energy recovery, and 40% was sent to recycling, from which 86% was used as substitutes of recycled polymers. There is no information regarding further treatments for the material losses from virgin plastic production and plastic products manufacture, but as assumption, these losses were sent to landfills. However, these amounts were not taken into account for the plastic waste accumulated in landfills presented in Section 3.1.4 (30,542,493 ton).

3.2. Scenarios of analysis: trends in plastic waste management

The waste management performance between 1999 and 2011 showed that despite a 22% growth for plastic waste generation, the quantity going to landfills declined by approximately 23%.

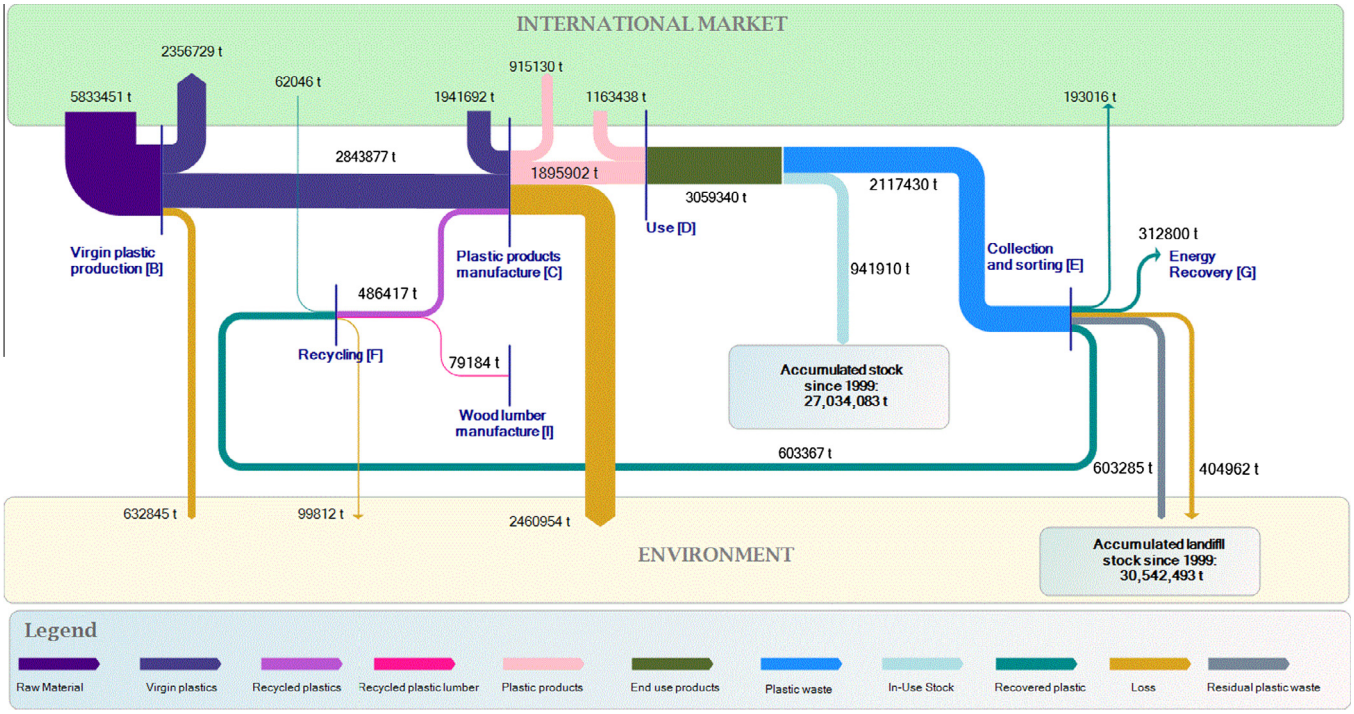

Fig. 6. Spanish plastic life cycle for 2011.

Table 5
Data and assumptions for the Baseline scenario and the alternative scenarios used for the GHG emissions evaluation.

	Percentage of plastic waste collected sent to recycling in Spain (%)	Ratio substitution	Percentage of recycled plastic to substitute virgin plastic (%)	Percentage of plastic waste collected sent to recycling internationally (%)
Baseline scenario	37 (21% to energy recovery)	1:1	86 (14% wood lumber substitution)	15
Alternative scenarios				
Scenario Man1	56 (2% to energy recovery)	1:1	86 (14% wood lumber substitution)	15
Scenario Man2	8 (50% to energy recovery)	1:1	86 (14% wood lumber substitution)	15
Scenario Qua1	37 (21% to energy recovery)	1:0.5	86 (14% wood lumber substitution)	15
Scenario App1	37 (21% to energy recovery)	1:1	50 (50% wood lumber substitution)	15
Scenario Mar1	52 (21% to energy recovery)	1:1	86 (14% wood lumber substitution)	0
Scenario Mar2	0 (21% to energy recovery)	1:1	86 (14% wood lumber substitution)	52

In each scenario, 27% of plastic waste collected is sent to landfill as plastic waste loss.

The plastic waste management showed an improvement mainly derived from the packaging and packaging waste law (Law 11/1997, 1997), which contributed to a collection increase of 66% since 1999. The remaining post-consumer plastic wastes also experienced significant improvements as a result of the implementation of specific legislation for EOL vehicles and electronic products, but their contribution was still low (Royal Decree 1383/2002, 2003; Royal Decree 208/2005, 2005). However, though these improvements, there is a large potential for higher collection rates in Spain, and several scenarios were projected to evaluate future plastic waste management possibilities and their GHG emissions were derived. The Baseline scenario evaluated the situation in Spain in 2011, and it served as the reference scenario. In this year, for every ton of plastic waste collected; 21% was sent to energy recovery, 37% was sent to recycling in Spain, 15% was sent to recycling internationally and 27% was sent to landfill as plastic waste loss. Values highlighted in gray showed the modifications in relation to this Baseline scenario. Table 5 summarizes the data for these scenarios.

In the following years, an increase in plastic waste collection due to better plastic waste management can be expected. Thus, **Scenario Man1** evaluates an increase (up to 56%) of the recovered plastic sent to recycling in Spain. In addition, plastic consumption in the last decade entailed an accumulation of plastic products, and through the MFA, the calculation showed that there is an in-use stock of approximately 27 million tons. Thereby, an increase in plastic waste collection due to the in-use stock achieving its EOL can also be projected, mainly from construction and electronic sectors. Considering that the construction and electronic sectors have a higher presence of PP and PVC, which are more difficult to recycle (JRC, 2012b; BIO Intelligence Service, 2013), an increase in plastic waste sent the energy recovery treatment and/or RPL production can also be projected. **Scenario Man2** evaluates an increase (up to 50%) of energy recovery, similar to Northern European countries. **Scenario App1** evaluates an increase (up to 50%) in the application of recycled plastic to produce RPL. This scenario indicates that 50% of the recycled plastic could substitute for virgin polymers and 50% could substitute for different products (i.e., wood). As assumption, the recycled plastics substitute virgin plastics (or wood) without any loss of quality, so the substitution ratio is 1:1 (1 kg of recycled plastic replaces 1 kg of virgin plastic). However, most recycling processes involve a loss of quality due to organic contamination

but also due, for example, to the macromolecular degradation or the presence of minute quantities of incompatible polymers. This requires the addition of virgin polymers to recover, fully or partially, the original properties. In this case, the substitution ratio is less than 1, because 1 kg of recycled plastic is not equivalent to 1 kg of virgin plastic. A literature review revealed that this substitution ratio ranges from a 0.5 to 1 (Lazarevic et al., 2010; OECD, 2010; Hong, 2012). The influence of the substitution ratio was evaluated in **scenario Qua1**.

Another consequence of an increase in plastic waste collection could be an increase in plastic waste exported, which in fact has already been projected for Europe (European Commission, 2011a; Shonfield, 2008; WRAP, 2011; JRC, 2012b; BIO Intelligence Service, 2013). Thus, **scenario Mar2** evaluated the situation where all recovered plastic is recycled globally, with 10% sent to Europe and 90% to China. To evaluate the consequences of a circular economy of plastic waste, **Scenario Mar1** assumed that all recovered plastic is recycled in Spain.

Finally, to observe the influence of the marginal electricity mix in the GHG results, a sensitivity assessment was conducted considering the use of an average electricity mix produced from the mix of power sources in 2011. Detailed information on the marginal electricity mixes and average electricity mixes are provided in Appendix B.

3.3. GHG quantifications of recycling

3.3.1. Baseline scenario

Fig. 7 presents the material flows for the Baseline scenario per functional unit.

Table 6 presents the GHG quantifications for the Baseline scenario based on data in Fig. 6 and Table 2. Although 52% of plastic waste collected was sent to recycling and only 21% to incineration, the GHG emissions for recycling were considerably lower (141 kg CO_2 eq. t^{-1}) than the GHG emitted due to the incineration process (497 kg CO_2 eq. t^{-1}). This is explained by a higher energy consumption of the incineration process. In fact, the GHG balance of incineration is positive (181 kg CO_2 eq. t^{-1}), although the electricity and heat avoided was considered. In this case, the positive balance was due to the low efficiencies for electricity and heat production.

Table 6
GHG emissions generated per tonne of plastic waste collected for the Baseline scenario (kg CO_2 eq. t^{-1}).

	GHG Emitted	GHG Avoided
Collection and sorting		
Collection and sorting	101	–
International transport	27	–
Recycling		
Recycling in Spain	21	−517
Recycling in Europe	3	−27
Recycling in Asia	117	−247
Incineration		
Incineration	497	−315
Total	766	−1108
GHG quantification for plastic waste recovery	−342	

The GHG comparison of recycling among the three regions showed that although 37% of plastic waste collected was recycled in Spain, 1.5% was recycled in Europe and the remaining 13.6% was recycled in China, the GHG emissions emitted in China are higher than those in Spain. This can be explained by the electricity consumption for recycling in China (see Table 2) and by the marginal electricity mixes considered for the recycling process. China has more primary energy contributions from coal, which has the highest GHG emissions for recycling.

To assess which type of plastic implies the highest GHG quantifications, the GHG emissions for 1 ton of each plastic waste were taken into account. The production of virgin PS and PET has the highest CO_2 eq. emissions (3453 kg CO_2 t^{-1} and 2677 kg CO_2 t^{-1}, respectively), followed by PVC, LDPE, PP and HDPE (2223 kg CO_2 t^{-1}, 2070 kg CO_2 t^{-1}, 1959 kg CO_2 t^{-1} and 1920 kg CO_2 t^{-1}, respectively). In the case of wood lumber, the production of one ton emitted 77 kg CO_2 eq. Therefore, the GHG balance or the RPL is positive. If the recycling of 1 ton of RPL would emit 141 kg of CO_2 eq., the GHG emissions due to collection, sorting and recycling would be higher than those saved from wood lumber production.

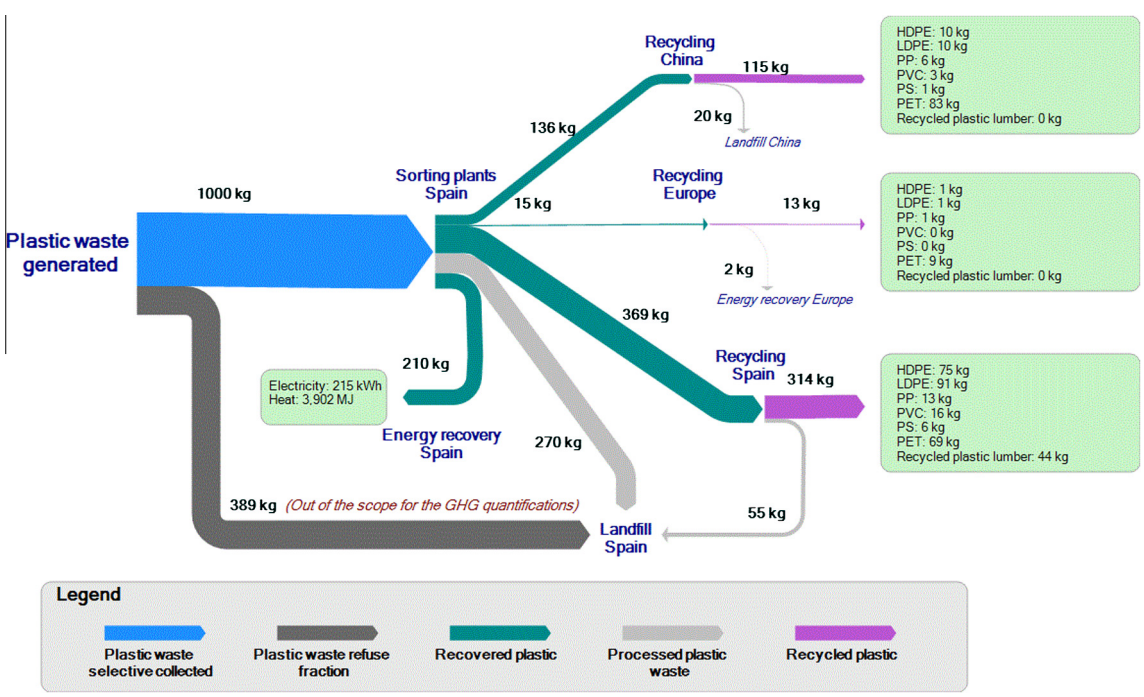

Fig. 7. Material flows for the Baseline scenario for year 2011.

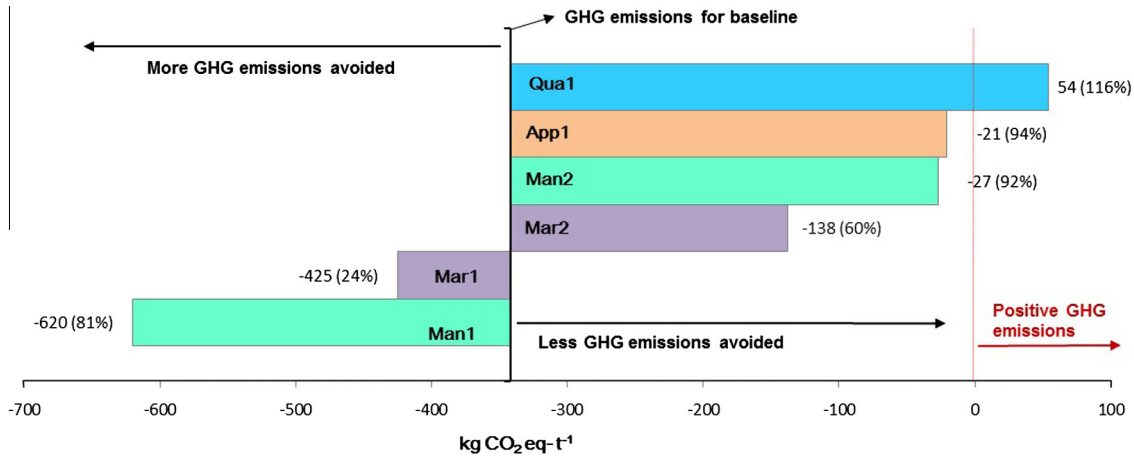

Fig. 8. GHG emissions for the alternative scenarios (kg CO_2 eq. t^{-1}).

3.3.2. Alternative scenarios and sensitivity assessment

The assessment of alternative scenarios leads to the GHG results presented in Fig. 8 in kg of CO_2 eq. per ton of plastic waste collected, and the variation in brackets refer to the Baseline scenario (−342 kg CO_2 eq. t^{-1}).

Scenario Qua1 had the worst GHG results. In fact, the entire process would generate 54 kg of CO_2 eq. because the GHG emissions from collecting, sorting and recycling remain the same, but the GHG benefits from the avoided primary productions were reduced by half. **Scenario App1** and **scenario Man2** also showed an important reduction of the GHG benefits. In both cases, the balance between the GHG emitted and the GHG avoided was nearly zero, −21 kg of CO_2 eq. t^{-1} and −27 kg of CO_2 eq. t^{-1}, respectively. These results suggest that the preference between recycling for non-plastic purposes or energy recovery is not clear in the sense of GHG emissions. On the other hand, **Scenario Mar2** showed that this option would avoid more GHG emissions than **scenario Man2**, because the GHG emissions due to the international transport and the international recycling were lower than those from the incineration process. In contrast, if all recovered plastic is recycled in Spain (or likely within Europe) as evaluated in **scenario Mar1**, higher GHG benefits would be obtained, because the GHG emissions from the international stages would be avoided. With no doubt, the best results were obtained for the **scenario Man1**, which evaluates an increase in the recovered plastic being sent to recycling. These results indicate that the mechanical recycling for virgin plastic substitution is the best environmental option.

Finally, when the marginal electricity mix was substituted for the average mix in 2011 for the countries (or regions) involved in the study, the same trends were obtained for all scenarios, but the GHG emissions were lower for all scenarios, except for **scenario Man2** and **scenario Qua1**. This finding indicates that fewer GHG emissions were avoided if the average mix was considered. The GHG quantification for the Baseline scenario was −337 kg CO_2 eq. t^{-1}, a difference of less than 2%. The highest difference was obtained for the **scenario Man2** (over 40%), where 50% of the recovered plastic was sent to energy recovery, while the lowest was obtained for the **scenario Mar1** (<1%), when all recovered plastic is recycled in Spain. The results are presented in Fig. B1 in Appendix B.

4. Discussion

Generally, in LCA studies, mechanical recycling of plastic waste is offered as the best option for end-of-life treatments compared to feedstock recycling, energy recovery or landfills (Ross and Evans, 2003; Perugini et al., 2005; Dodbiba and Fujita, 2004; Foster, 2008; Shonfield, 2008; Astrup et al., 2009; Lazarevic et al., 2010; Eriksson and Finnveden, 2009; Hong, 2012; Rigamonti et al., 2014). Mechanical recycling shows a clear advantage compared with other options, because the GHG emission savings are largely derived from the avoided products that are accounted as virgin plastics with a 1:1 substitution. The results of this study were in accordance with those presented in the literature, and the GHG quantifications showed that mechanical recycling was the best environmental option from a GHG perspective, with the best results obtained for **scenario Man1** (−620 kg CO_2 eq. t^{-1}). However, the GHG results were also highly dependent on the ratio of substitution with virgin plastic. In fact, quality considerations have the highest influence on the GHG benefits of recovery, which were evaluated in **scenario Qua1** (54 kg CO_2 eq. t^{-1}) and **scenario App1** (−21 kg CO_2 eq. t^{-1}). In fact, the evaluation of the GHG results per ton of type of recovered plastic waste indicated that the recycling of plastic waste for the substitution of other materials, such as wood, provided no GHG savings at all. It should be realized that materials other than wood may also be substituted (Astrup et al., 2009), so the GHG savings could be different depending on the substituted product (i.e., aluminum or steel). Nevertheless, in most cases, recycled plastic, such as HDPE or PP, are not currently replacing the same virgin plastic, and as concluded by Rigamonti et al. (2009), the assumptions made about which materials are replaced by the recycled materials are very important, and more research should be conducted. Likely, **scenario App1** may represent the current situation for Spain.

In this regard, it has been suggested in the literature that for a mixture of different plastic types, plastic waste should be used for energy utilization (Astrup et al., 2009; OECD, 2010). However, the results for **scenario Man2** (−27 kg CO_2 eq. t^{-1}) suggested that the related GHG improvements are not clear. This limitation was also highlighted in another study (Eriksson and Finnveden, 2009) that concluded that the GHG emissions avoided in energy recovery are highly dependent on the electricity and heat production efficiencies; thereby, for higher ratios, higher GHG benefits should be obtained (Eriksson and Finnveden, 2009). Nevertheless, other aspects could influence the decision between recycling and energy recovery. For example, in Denmark, where the energy recovery is approximately 70% (Plastic Europe, 2015), the environmental benefit of incineration is promoted, because the recovered energy is likely to increase the renewable energy sources in the mix and plastic waste is traded on a world market that could increase the environmental burden of the plastic waste recycling and thus decrease the GHG benefits (Merrild et al., 2012).

In this regard, the evaluation of **scenario Mar1** and **scenario Mar2** confirmed this approach, and higher GHG emissions were

avoided for regional recycling (−425 kg CO_2 eq. t^{-1}) than for global recycling (−138 kg CO_2 eq. t^{-1}). However, the GHG emissions from this option should be evaluated in more detail, because the LCI data for the virgin plastic production was based on ecoinvent data from eco-profiles for the European plastic industry (Hischier, 2007). Data for energy and material consumptions are aggregated, so it was not possible to evaluate and adapt the inventory profiles depending on the country to take into account the production efficiencies, technologies, distances of transport, marginal technologies, etc. This issue was also highlighted by Friedrich and Trois (2013) in a recent study of GHG emissions from waste recycling in South Africa. They concluded that the use of European data on plastic production underestimated their GHG results because the South African electricity mix has a higher GHG burden, which means that the GHG savings from recycling should be higher (Friedrich and Trois, 2013). Other studies also highlighted the difficulties in modeling material recycling in a global market due to the lack of data for recycling facilities, which is predominantly sourced from developed countries and might not be representative of the facilities in developing economies (Christensen et al., 2007; Lazarevic et al., 2010).

5. Conclusions

In the coming years, plastic waste collection in Spain will increase due to better plastic waste management and the in-use stock at its EOL. This study demonstrates that for Spain to save more GHG emissions, the best plastic waste management is mechanical recycling (−620 kg CO_2 eq. t^{-1}). However, the results are highly dependent on the replacement of recycled plastic, and more research is needed to represent the complex reality of plastic recycling. In addition, contamination was identified as one of the most important parameters limiting plastic waste recycling, which suggests that collection and sorting are very important processes. Therefore, promoting an increase in the recycling rate is not realistic if other strategies and actions are not promoted in parallel. In this regard, there is currently an open debate between supporters of the current management system for packaging, which is handled through a Green Dot System (GDS), and supporters of a change towards a Container Deposit Scheme (CDS). The latter argue that a transition to a CDS system could be key to increasing collection, quality and recycling rates similar to other European countries (i.e., Germany) (Retorna, 2011). Finally, the exportation of plastic waste for recycling presented more GHG emission reductions than energy recovery; however, there are uncertainties in modeling the recycling of plastic waste in a global market. Therefore, if the possibility of improving the plastic waste quality is not feasible, efforts should be focused on improving the electricity efficiencies of energy recovery plants. This study was limited to the estimation of GHG emissions because of their current high priority in EU policies; however, other impact categories should be assessed to have more information for waste management decisions and to determine if conclusions behind this manuscript would maintain.

Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.wasman.2015.08.007.

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