Pre-print of: Sevigné-Itoiz, et al. “Contribution of plastic waste recovery to greenhouse gas (GHG) savings in Spain” in Waste Management, vol. 46 (Desember 201), p. 557-567. Sent to Elsevier. The final version is available at DOI 10.1016/j.wasman.2015.08.007

The contribution of plastic waste recovery to greenhouse gases (GHG) savings in Spain

Eva Sevigné-Itoiz [1,2], Carles .M. Gasol[1,3] , Joan Rieradevall[1,2], Xavier Gabarrell[1,2]*

[1]Sostenipra, Department of Chemical Engineering, Universitat Autònoma de Barcelona (UAB) Building Q UAB - 08193 Cerdanyola del Vallès (Barcelona), Spain.

[2]Institute of Environmental Science and Technology (ICTA), Universitat Autònoma de Barcelona, (UAB) Building C Campus UAB - 08193 Cerdanyola del Vallés (Barcelona), Spain

[3]Inèdit Innovació s.l., Research Park of the UAB, Carretera de Cabrils, km. 2, IRTA, 08348 Cabrils, Spain

*Correspondence to: eva.sevigne@uab.cat/esevigne@gmail.com

Abstract: This paper concentrates on the quantification of greenhouse gases (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 more feasible plastic waste recovery option to decrease the GHG emissions. We integrate the methodologies of material flow analysis (MFA) for a time period of thirteen years and consequential life cycle assessment (CLCA). The study focused on Spain as a representative country for Europe. Our results show that in order to improve resource efficiency and avoid more GHG emissions, the option for plastic waste management are with or against the waste hierarchy depending on the quality of the recovered plastic. Result also show that there is an increasing trend of plastic waste export to recycle, mainly to China, what reduce the GHG benefits from recycling and suggest that a new focus should be introduced to take into account the split between local recycling and exports.

KEYWORDS: plastic waste recycling, material flow analysis, life cycle assessment, plastic waste quality, greenhouse gases (GHG), Spain

1. INTRODUCTION

Given the versatile properties of plastics, such as it being lightweight, durable and strong, the world production and usage of plastics has increased sharply (Hong, 2012), from 1.5 million tonnes (Mt) in 1950 to 288 Mt in 2012, and it has estimated that global plastic production could triple by 2050 (European Commission, 2013; Plastic Europe, 2013). However, plastics, as materials, are generating environmental problems along its whole life cycle. Firstly, in order to produce plastic products greenhouse gas (GHG) emissions are produced. Secondly, the characteristics that make plastic so useful also makes waste management problematic (European Commission, 2013) and only a small fraction of plastic waste is at present recycled due to contamination and technical limitations (Briassoulis et al., 2013; Hong, 2012). Thirdly, 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, about 80% of which is plastic (European Commission, 2011a; European Commission, 2013). Once in the environment, particularly in the marine environment, plastic waste can persist for hundreds of years (Kaps, 2008).

Hence, considerable concern has been focused on plastic waste management. At European level, since 1994 several objectives for plastic waste recycling and recovery have been set (European Commission, 1994; European Commission, 2000; European Commission, 2004) and last Waste Framework Directive (European Commission, 2008) has established a 22.5% target for packaging plastic waste which must be reached by all EU Member States by 2020 (Plastics Recyclers Europe, 2012). One important aspect to consider, however, is that the 22.5% target is based on the amount of packaging plastic waste collected rather than the final packaging plastic waste recycled (BIO Intelligence

Service, 2013). Thus, although one EU member reaches the target, it does not imply same amount of plastic waste recycled within the country. In fact, in recent years, plastic waste exports have increased dramatically, both within the EU and even more so to third countries due to 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 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; European Commission, 2012).

Furthermore, the last Waste Framework Directive (European Commission, 2008) also establishes the waste hierarchy of prevention, preparing for reuse, recycling, other recovery and disposal; but 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 to fulfill with the waste hierarchy is that options for 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 down-cycling of plastics for cheaper and less demanding applications (JRC, 2012b). In this case, the energy recovery has been presented as a better environmental option although it is against the waste hierarchy (Astrup, 2009; Eriksson and Finnveden, 2009). So quality, application and market challenges appear to limit plastic waste recycling, what suggest that it is not clear which are the best options to improve plastic waste management while reducing the GHG emissions. For instance, if the efforts should be done to increase reuse, the energy recovery, to improve the technology of sorting and recycling or to promote the use of cheaper and less demanding applications of recycled plastics.

The aim of this study is evaluate different options in order to identify the limitation and opportunities of plastic waste recovery with the objective of decrease the GHG emissions. Spain was selected as a case study to evaluate different options because it is the fifth European plastic consumer (Plastic Europe, 2013), but considering its plastic waste management, Spain is in the middle range of European averages (ANARPLA, 2013; Plastic Europe, 2013). Thus, Spain could be representative of the current context but also it could be representative of the consequences of following the trends and strategies of other European countries in relation to the increasing or decreasing the GHG emissions of plastic waste management. countries.

Addressing these questions can be conducted with dynamic material flow analysis (MFA), which evaluate the flows and stocks of materials within a system defined in space and time (Brunner and Rechberger 2004), and allow observing variability over time and determining possible changes in waste management trends. Therefore, dynamic MFA could be the base to project future waste management trends in Spain if management remains the same. However, if waste policies are promoted to achieve European targets, different waste management trends should be followed. Thus, in order to calculate the increase or decrease of GHG emissions several scenarios could be defined. GHG emissions could be calculated with consequential life cycle assessment (CLCA) which quantify and describe how the environmental impacts will change in response to possible decisions (i.e., increase collection or energy recovery) (Reinhard and Zah, 2009). Similar studies were already conducted to evaluate the GHG emissions derived from recycling of paper-cardboard waste and aluminum old scrap in Spain (Sevigné-Itoiz et al., 2014a; Sevigné-Itoiz et al., 2014b).

2. MATERIALS AND METHODS

In the following sections, the methodologies used for the MFA (2.1) and for the CLCA (2.2) are explained. More detail is available in Appendix A and Appendix B.

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 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 of disposal in landfill, 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 takes place (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. Along its life cycle, we considered the following life cycle stages (LCs): first raw materials are extracted and transformed into virgin plastics, then plastics products from virgin plastics and recycled plastics are manufactured, then the products are used, and finally, they become wastes that have to be managed. Figure 1 presents the system boundaries of the Spanish plastic cycle and shows every flow and stock that has to be determined, including importations and exportations and losses into the environment. In Spain, packaging plastic waste is collected selectively through containers in the street applying a Green Dot System (GDS) as well as through others selective ways (i.e., agriculture plastic waste) but there is also an important fraction that is collected within the refuse fraction, latter namely plastic waste refuse fraction. The plastic waste selective collected is sent to sorting plants where it is classified as recovered plastic for recycling or incineration with energy recovery. Nevertheless, it should be noticed 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 (i.e., 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). In this study we evaluated both options of substitutions with virgin plastics and wood.

LCs represented in Figure 1 as rectangles with solid lines can be disaggregated in several types which are represented below as rectangles with discontinuous lines. For example, virgin plastics production were classified as high density polyethylene (HDPE), low density polyethylene (LDPE), polypropylene (PP), polystyrene (PS), expanded polystyrene (EPS), polyvinyl chloride (PVC), polyethylene terephthalate (PET) and ethyl vinyl acetate (EVA). Remaining virgin plastics (i.e., engineering, polyurethanes, etc) were categorized as others. Capital letters in brackets refer to LCs (i.e., virgin plastic production [B]) while lower case letters refer to material flows (i.e., recycled plastics (f)).

2.1.2. Flows and stocks estimations

The system under study concerns 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, with flows detailed in Figure 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 plastic products consumption 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, calculated by combining statistics with coefficients and deduced using the mass balance. Details on data collection, sources and explanations of assumptions are given in Appendix A.

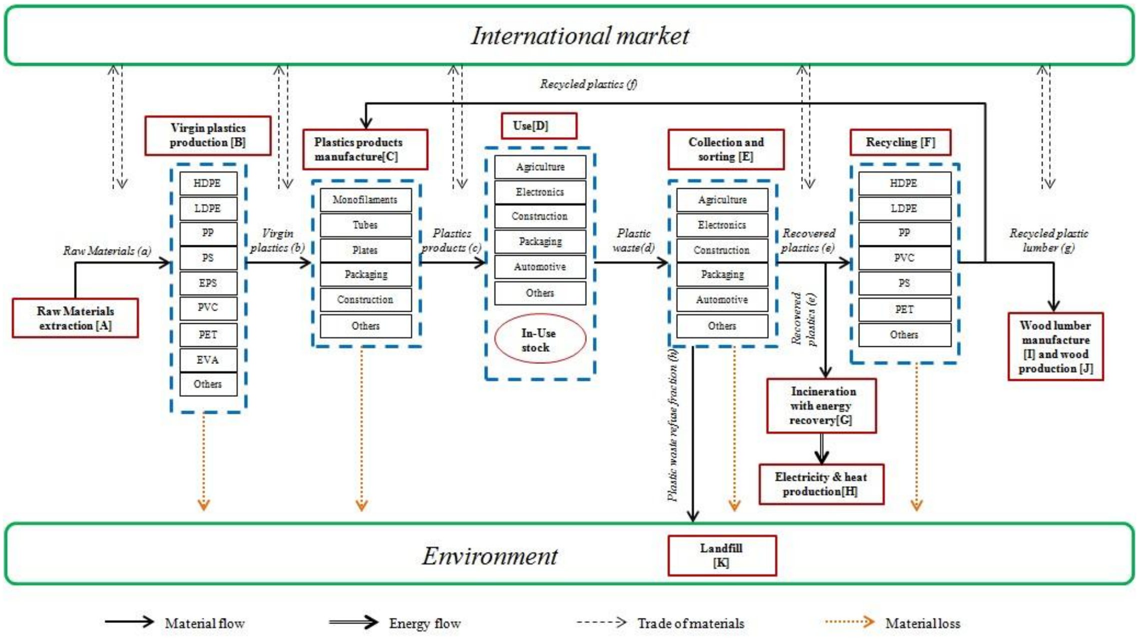

Figure 1: Spanish plastic cycle system boundaries

2.2. Consequential Life Cycle Assessment (CLCA)

2.2.1. Scope and system boundaries

Even though the MFA takes into account the whole life cycle of plastics as shown in Figure 1, for the GHG quantifications we focused only into the GHG emissions from the plastic waste selective collected for recycling and energy recovery. Both recoveries (i.e., materials and energy) are a case of multifunctional product, and in CLCA the allocation problem derived is avoided by system expansion (Weidema et al., 2009). In this paper the recycling activities were modeled based on the guidelines and assumptions of the study of Schmidt (Schmidt, 2012) which conclude that the market for recycled plastics waste and virgin plastics are not considered as two different markets. Hence, the marginal effect of plastic waste collection for recycling will be that virgin plastics are affected (Schmidt, 2012). Nevertheless, as explain 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 also for the GHG quantifications. We considered that recovered plastic fraction that is sent the incineration with energy recovery will avoid the marginal electricity and heat production.

Therefore, for the GHG quantifications we considered the LCs of collection and sorting [E], recycling [F], incineration with energy recovery [G], virgin plastics production [B], wood lumber manufacture [I], raw materials extraction [A], wood [J] and electricity and heat production [H]. Although the GHG emissions of landfill [K] were outside of the scope of this paper, we took into account the processed plastic waste from sorting and recycling facilities which undergo to landfill (indicated in Figure 1 as loss flow).

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

The FU has been defined as the increase of 1 ton of plastic waste selective collected in Spain for recycling and energy recovery. Inventory data and assumptions are presented in Tables B1-B3 in Appendix B where it is explained the assumptions of source of data. 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

Eight scenarios were evaluated and discussed in relation to the GHG emissions. These scenarios were defined based on the MFA results and literature review, so more detailed information is found in section 3.2.

3. RESULTS

The following sections present the results for the dynamic MFA (sections 3.1.1 to 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 results for the GHG quantifications for the Baseline scenario and the alternative scenarios (sections 3.3.1 and 3.3.2).

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

3.1.1. From virgin plastics to plastics products

Figures 2 and 3 show the results of consumption of virgin plastic, recycled plastic and applications by sector from 1999 to 2011. Virgin consumption grew steadily from 1999 to 2007 exceeding 6,000,0000 tonnes and then decreased substantially until 2009 probably due to the economic crisis which have affected all industries and sectors, especially the construction sector. However, recycled plastic consumption has increased year by year from a percentage contribution of 5% in 1999 to 12% in 2011.

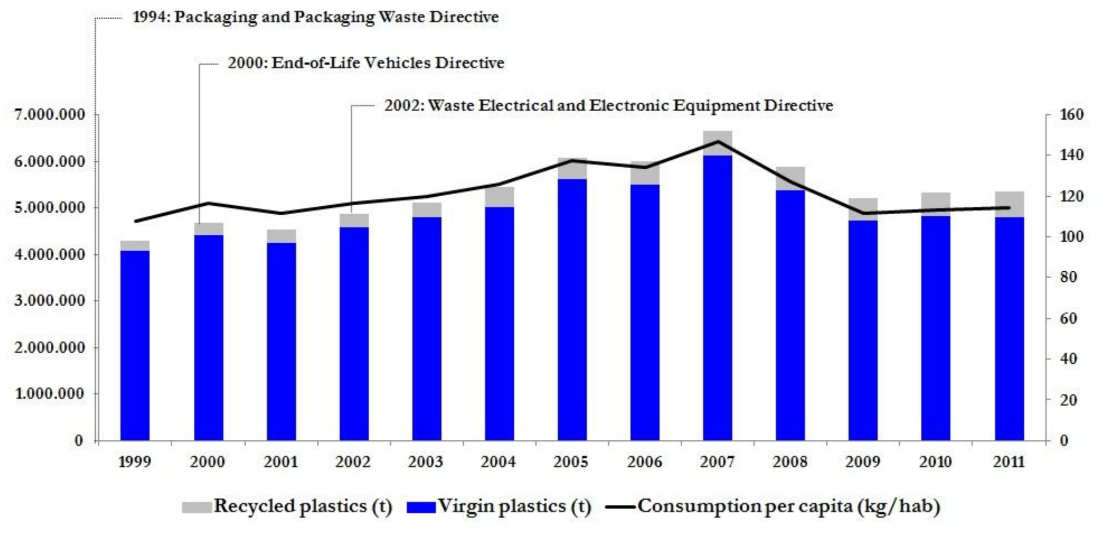

Figure 2: Consumption of virgin plastic and recycled plastics in tonnes from 1999 to 2011 in Spain in the left axis and overall consumption per capita in the right axis from 1999 to 2011 in Spain

Table 1 summarizes production, consumption, main application and recycling by type of polymer for 2011. Main application of PE, PP and PET is packaging (Plastic Europe, 2011) and as it is shown in Figure 3, it is clear that in packaging is the largest single sector for plastics what justifies the higher share of PE, PP and PET consumption (46% in 2011). In fact, packaging consumption followed trends of virgin plastic consumption. PVC, PS and EPS are mainly used in the construction sector (Plastic Europe, 2011) and its consumption follows the trend for this sector. Production and consumption data from 1999 to 2010 are available in Tablas A1 and A2 in Appendix A.

Table 1: Production, consumption, main application and recycled in percentage per type of plastic for year 2011

	Production (%)	Consumption (%)	Main application by sector (virgin plastic)*	Recycled (%)
HDPE	8	16	Packaging	24
LDPE	13	16	Packaging	29
PP	18	16	Packaging	4
PVC	11	8	Construction, others	5
PS	3	3	Construction	2
EPS	1	1	Construction	na
PET	14	13	Packaging	22
EVA	2	0	na	na
Others	30	28	na	14

**based on data from Plastic Europe, 2013*

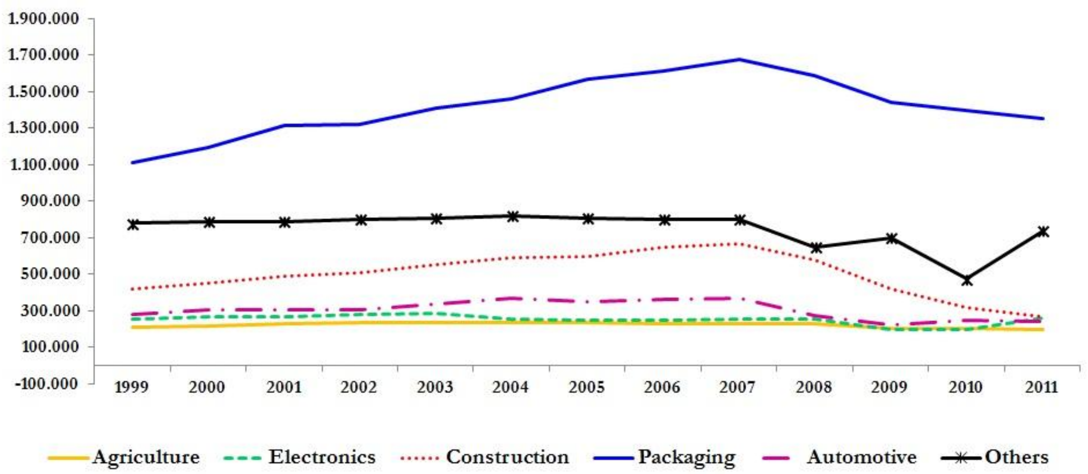

Figure 3: Consumption of plastics products in tonnes from 1999 to 2011 in Spain by main application sector

3.1.2. Plastic waste management

Table 2 shows 1) total plastic waste generated and its treatments, 2) packaging plastic waste generated and its treatment and 3) the rest of plastic waste generated, such as agriculture plastic waste, and its treatment from 1999 to 2011. For generation and collection data, official statistics have been used whenever these have been available from Spanish or European authorities and waste management companies. Where required some assumptions were used to complete the picture (see Appendix A for more detail). Total plastic waste generated includes the plastic waste selective collected though containers in the street and in others ways of selective collection as well as the plastic waste collected within the refuse fraction.

Plastic waste generation, recycled, incinerated and landfilled increased up to 2007 and then decreases slowly following the observed trend for plastic consumption. However, while in 1999 12% of plastic waste was sent to recycled, 7% was sent to incineration and 81% was landfilled; in 2011, recycling increased up to 38%, incineration up to 15%

and landfilled decreased up to 47%. We considered that total packaging products consumed were generated as a waste within the same year; thus, we can observe that if packaging is the main application for plastics, it is also the largest plastics waste stream arising (around 64 %). However, if in 1999 more than 67% of packaging plastic waste selective collected was sent to landfill, in 2011 decreased up to 49%. Complementary, recycling and energy recovery of packaging plastic waste have increased their presence reaching 26% and 25% of end-of life treatments in 2011, respectively. Data of plastic waste collection and plastic waste recycling from 1999 to 2011 for the others sectors considered in this study (agriculture, electronics, construction, automotive and others) are presented in table A3 and table A4 in Appendix A.

As summarizes in Table 1, main polymer recycled was LDPE (29%) followed by HDPE (24%) and PET (22%) due to the recycling of packaging plastic waste which is the main plastic product consumed and main plastic waste collected and recycled. Although PP consumption accounts for 15%-18% of total consumption, PP has 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 it normally is very contaminated with other materials (JRC, 2012b). Their contributions in the recycling fraction are 4% and 5%, respectively. Around 14% of plastic recycled was classified as others which are a mix of polymers. In this study we consider that this mix of polymers is used to produce the RPL in substitution with wood.

Table 2: Total plastic waste generated and its treatment, packaging plastic waste and rest of plastic waste generated and its treatment in tonnes from 1999 to 2011 in Spain

	Plastic waste (tonnes)				Packaging plastic waste (tonnes)				Rest of plastic waste (tonnes)		
	Generated	**Recycled**	**Incinerated**	**Landfilled**	**Generated**	**Recycled**	**Incinerated**	**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

(*) There is only data for packaging plastic waste incinerated with energy recovery. We assumed same data for plastic waste incinerated with energy recovery.

3.1.3. International trade

Figure 4 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 higher 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 is also a lack of plastics products, which were imported mainly from Germany and France (datacomex, 2013), but from 2008 there was an important decrease of the commercial balance indicating that there was less demand of plastic virgin plastic and plastic products, which were exported. In the case of plastic waste, from 2002 there has been an excess of supply which has been exported, and this export of plastic waste has increased over the years. In this regard, is important to highlight that the main destination of the plastic waste has changed, and if in 1999 main destination was intra Europe (mainly Portugal and France), in 2011 around 90% of the plastic waste was sent to Asia (China and Hong Kong) (datacomex, 2013) positioning Spain as the fifth European exporter (Anarpla, 2013).

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

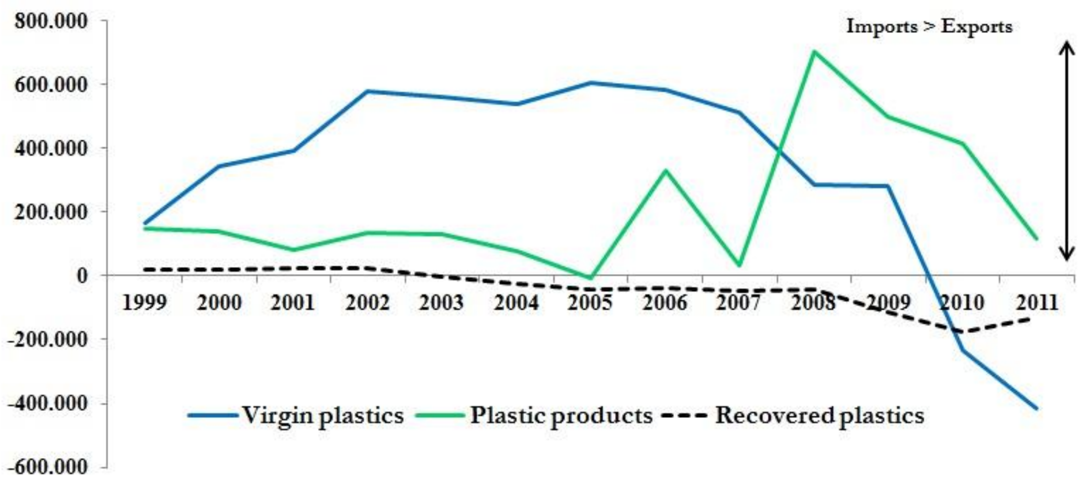

Figure 4: Commercial balances (imports minus exports) for virgin plastic, plastic products and recovered plastics (packaging plastic waste) in Spain from 1999 to 2011 in tones

3.1.4. In-use stocks

Because the lifetime of many plastics products can be between less than one year (packaging), and more than 50 years (construction), there has been an accumulation of plastics products in use (Kaps, 2008). There are no data available for Spain before 1999; therefore, our stock calculations are underestimated. However, in 2011, we calculated an in-use stock since 1999 of 27,034,084 tonnes of plastics products, which represents approximately 8 years of supply of plastic at 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 have been an accumulation of plastics products in landfill due to the plastic waste within the refuse fraction as well as the processed plastic waste from sorting and recycling processes. We have considered that all these losses would end up in landfills but probably a fraction would end up in the marine environmental. Our estimation for 2011 is 30,542,493 tonnes of plastics accumulated the environment between 1999 and 2011.

3.1.5. Spanish plastic life cycle in 2011

Figure 5 presents the Spanish plastic life cycle in 2011, as it is the most representative year of the current situation.

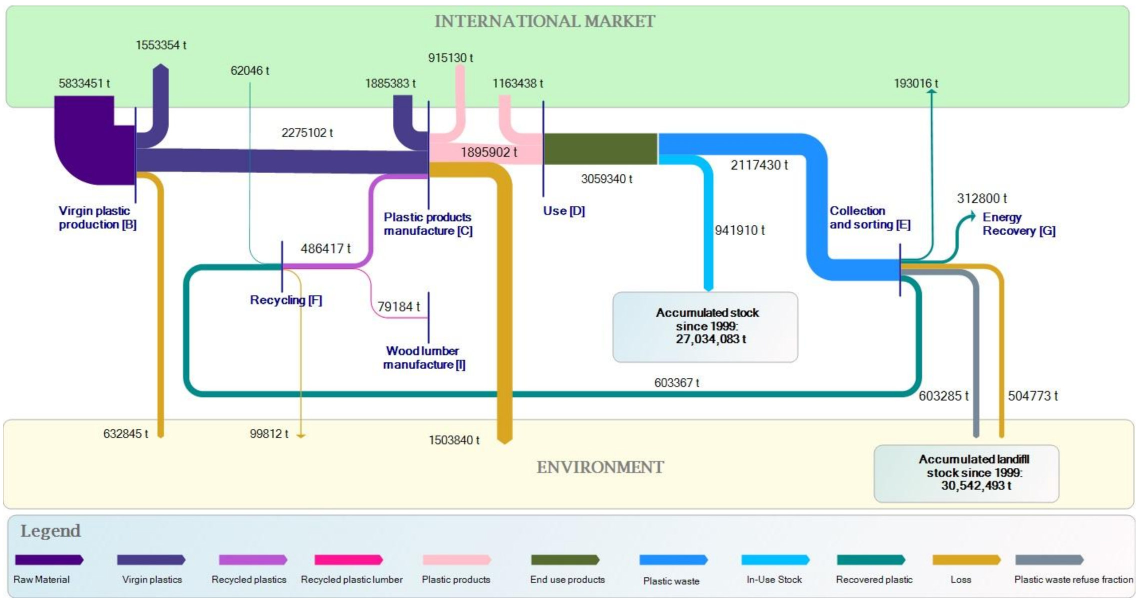

Figure 5: Spanish plastic life cycle for 2011

3.2. Scenarios of analysis: trends in plastic waste management

If we take into account the waste management performance between 1999 and 2011, despite a 22% growth for plastic waste generation, the quantity going to landfill has declined by about 23%. The plastic waste management has shown an incredible improvement mainly derived from the packaging and packaging waste law (Law 11/1997, 1997) which has contributed to an increase in the collection of 66% since 1999. The remaining post-consumer plastic waste have also experienced significant improvements as a result of the implementation of specific legislation for EOL vehicles and electronic products but their contribution is still low (Royal Decree 1383/2000; 2000; Royal Decree 208/2005; 2005). However, though these improvements, there is a huge potential for higher collection rates in Spain and several scenarios were projected to evaluate future plastic waste management possibilities and their GHG emissions derived.

Table 3 summarizes data for these scenarios. We took into account the quantity of recovered plastic waste sent to recycling (referred as management); the ratio substitution between recycled plastics and virgin plastics/wood (referred as quality); the percentage of recycled plastic use for plastic purposes (referred as application); and the percentage of recovered plastic waste exported (referred as market). Baseline scenario evaluated the situation in Spain in 2011 and it was the reference scenario. Values highlighted in grey showed the modifications in relation to this Baseline scenario.

Table 3: Data and assumptions for the Baseline scenario and the alternative scenarios used for the GHG emissions evaluation

	Management	**Quality**	**Application**	**Market**
	Recycling (%)*	Ratio substitution	Virgin plastic (%)	Global (%)
Baseline scenario	71	1:1	86	29
Alternative scenarios				
Scenario Man1	90	1:1	86	29
Scenario Man2	50	1:1	86	29
Scenario Qua1	71	1:0.5	86	29
Scenario App1	71	1:1	100	29
Scenario App2	71	1:1	50	29
Scenario Mar1	71	1:1	86	0
Scenario Mar2	71	1:1	86	100

***** Percentage over recovered plastic after sorting plants

If trends remain the same, in following years we can expect an increase of plastic waste collection due to better plastic waste management. Thus, **Scenario Man1** evaluates an increase of the recovered plastic sent to recycling up to 90%. In addition, plastic consumption in last decade has entailed an accumulation of plastic products and through the MFA we calculated that there is an in-use stock of around 27 million tonnes. So, we can also project an increase of plastic waste collection due to the in-use stock achieving its end-of-life, mainly from construction and electronic sectors. Considering that the construction and electronic sectors have higher presence of PP and PVC more difficult to recycle (JRC, 2012b; BIO Intelligence Service, 2013), we can also expect an increase of the energy recovery treatment and RPL production. **Scenario Man2** evaluates an increase of energy recovery up to 50% similar to Northern European countries and **Scenario App1** evaluates an increase of the application of recycled plastic to produce RPL up to 50%, what means that 50% of the recovered plastic is used to substitute virgin polymers and 50% to substitute other products (i.e., wood lumber). In addition, most recycling processes involve loss of quality which may lead to a need for extra secondary plastic in the final products to obtain a quality identical with products of virgin plastic (Astrup et al., 2009). Although we assumed that the substitution ratio is 1:1, it is commonly less than 1. The literature review ranges the ratio substitution from a 0.5 to 1 (Lazarevic et al., 2010; OECD, 2010; Hong R, 2012). This loss of quality was evaluated in **scenario Qua1**.

Another consequence of an increase of plastic waste collection could be an increase of plastic waste exported, which in fact has already projected for Europe (European Commission, 2011a; Shonfield, 2008; WRAP, 2011; JRC, 2012b, BIO Intelligence Service, 2013). Thus, **scenario Mar2** evaluated that all recovered plastic is recycled globally with 10% sent to Europe and 90% to China. In order to evaluate the consequences of a circular economy of plastic waste, **Scenario Mar1** evaluated that all recovered plastic is recycling in Spain. We have used for both scenarios same classification of plastic waste by type of polymer than in 2011 (see section 3.1.2 and 3.1.3).

Finally, in order to observe the influence of the marginal electricity mix in the GHG results, we have conducted a sensitivity assessment considering the use of average electricity mix produced from the mix of power sources in 2011. Detail information of the marginal electricity mixes and average electricity mixes are explained in Appendix B.

3.3. GHG quantifications of recycling

3.3.1. Baseline scenario

Figure 6 presents the material flows for the Baseline scenario per the functional unit taken into account the plastic waste recycling by type of polymer for the Spanish recycling and in the case of the recovered plastic export, we took into account data presented in section 3.1.3 by type of polymer.

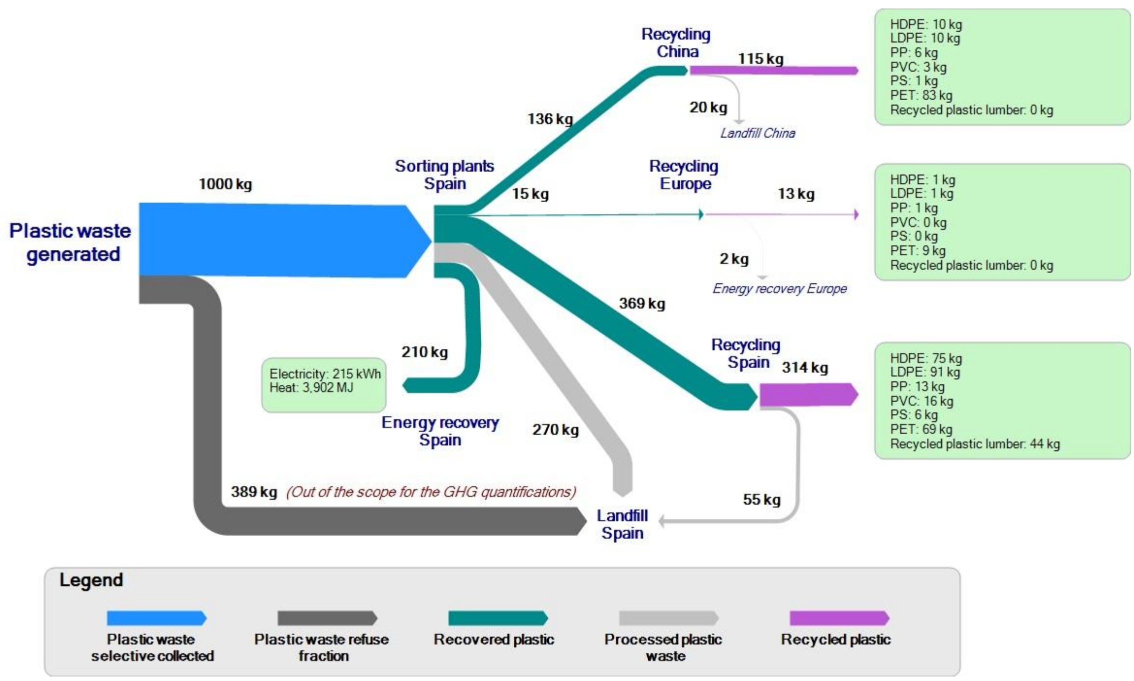

Figure 6: Material flows for the Baseline scenario

Table 4 present the GHG quantifications for the Baseline scenario taking into account the considerations and assumptions presented in Table 2 and the LCI data presented in Appendix B. The GHG results show that 81% of the emissions due to collection, sorting, recycling and incineration took place in Spain, while the rest 19% were emitted abroad from which 18% correspond to the international transport of plastic waste export, 2% correspond to the recycling process in Europe and 80% to recycling process in China. Although 71% of recovered plastic is sent to recycling, the GHG emissions are considerable lower (167 kg CO_2 eq. t^{-1}) than the GHG emissions from the incineration process (497 kg CO_2 eq. t^{-1}, without the energy recovery) even with the GHG emissions derived for the international recycling.

Table 4: GHG emissions generated per tonne of plastic waste collected for the Baseline scenario

Baseline scenario kg CO_2 eq. t^{-1} plastic waste selective collected	
Collection and sorting [E]	
Collection & sorting	101
Recycling [F]	
International transport	27
Recycling in Spain	21
Recycling in Europe	3
Recycling in Asia	117
Incineration [G]	
Incineration	497
Total collection, sorting, recycling & incineration= [E]+[F]+[G]	**766**
Raw materials [A][J] and production [B][I] avoided	
HDPE (20%)	-143
LDPE (23%)	-183
PP (4%)	-32
PVC (4%)	-37
PS (2%)	-22
PET (37%)	-371
Wood lumber (10%)	-3
Electricity and heat avoided [H]	
	-315
Total raw material and production avoided= [A]+[J]+[B]+[I]+[H]	**-1,108**
GHG quantification for plastic waste recycling= [E]+[F]+[G]-([A]+[J]+[B]+[I]+[H])	**-342**

Besides, the GHG comparison of recycling between the three countries show that although 71% of plastic waste is recycled in Spain, 2.9% is recycled in Europe and the remaining 26.1% is recycled in China, the GHG emissions in China are higher than those of Spain, which is explain for a higher the electricity consumption for recycling in China but also because of the marginal electricity mixes considered for the recycling process. Asia has more contributions from coal primary energy what increases the GHG emissions showing that highest GHG emissions due to recycling.

Regarding the primary production avoided, the recycling of PET implies the highest GHG savings while the production of recycled plastic lumber leads to an increase of the GHG emissions. These results correspond to the percentage of recycled plastics which were indicated in sections 3.1.2. However, in order to assess which type of plastic implies the highest GHG quantifications, we evaluate the GHG emissions when it is considered that 1 tonne of each plastic waste is collected, sorted and recycling with same assumptions as in Baseline scenario (i.e., 29% of export rate and 1:1 substitution). The production of PS and PET has the highest CO_2 eq. emissions; thus, the GHG quantifications for their recycling have the highest GHG benefits (-2,166 kg CO_2 t^{-1} and -1,617 kg CO_2 t^{-1}, respectively). It is followed by PVC, LDPE, PP and HDPE (-1,281 kg CO_2 t^{-1}, -1,187 kg CO_2 t^{-1}, -1,109 kg CO_2 t^{-1} and-1,081 kg CO_2 t^{-1}, respectively). In the case of the GHG quantifications due to the production of the recycled plastic lumber, the results are positive (222 kg CO_2 t^{-1}), what indicates that the GHG emissions due to collection, sorting and recycling are higher than those saved from the wood lumber production. In the case of the incineration process with energy recovery, we have also obtained that although electricity and heat is recovered in substitution with the marginal production, the GHG emissions from the process are positive (181 kg CO_2 eq. t^{-1}), due to the low efficiencies for electricity and heat production.

3.3.2. Alternative scenarios and sensitivity assessment

The assessment of the alternative scenarios leads to the GHG results presented in Figure 7 in kg of CO_2 eq. per tonne of plastic waste collected, and the variation in brackets in respect the Baseline scenario (-342 kg CO_2 eq. t^{-1}).

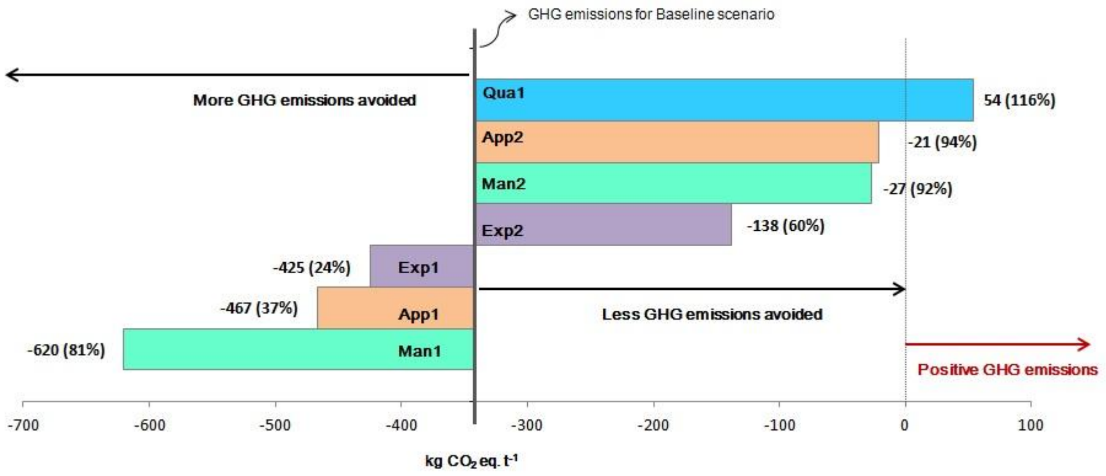

Figure 7: GHG emissions for the alternative scenarios (kg CO_2 eq. t^{-1})

Scenario Qua1 leaded to the worst GHG results. In fact, the whole process would generate 54 kg of CO_2 eq. because the GHG emissions from collection, sorting and recycling remain the same but the GHG benefits from the avoided primary productions are reduced to 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 is 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 purpose or energy recovery is not clear in the sense of GHG emissions. On the other hand, **Scenario Mar2** showed that this option avoids more GHG emissions than the **scenario Man2** because the GHG emissions due to the international transport and the international recycling are lower than those from the incineration process. Contrary, if all recovered plastic is recycled in Spain (and probably within Europe) as evaluated in **scenario Mar1**, higher GHG benefits are obtained because the GHG emissions from the international stages are avoided. With no doubt, the best results are obtained for the **scenario Man1** which evaluates an increase of the recovered plastic sent to recycling.

These results indicate that the mechanical recycling for virgin plastics substitution is better 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, same trends were obtained for all scenarios but the GHG emissions are lower for all the scenarios, except for **scenario Man2** and **scenario Qua1**. This means that less 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%), when 50% of 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. Results are presented in Figure 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 comparing to feedstock recycling, energy recovery or landfill (Ross and Evans, 2003; Perugini et al., 2005; Dodbiba and Fujita., 2004; Foster, 2008; Shoenfield, 2008; Astrup et al., 2009; Lazarevic et al., 2010; Erikson and Finnveden, 2009; Hong R, 2012). Mechanical recycling shows a clear advantage compared with other options because the GHG emission saving is largely derived from the avoided products which are accounted as virgin plastics with the substitution based on 1:1. 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 and the best results were obtained for

scenario Man1 (-620 kg CO_2 eq. t^{-1}). However, GHG results were also highly dependent on 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}). Actually, the evaluation of the GHG results per tonne of type of recovered plastic waste, pointed out that the recycling of plastic waste for 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., aluminium or steel).

In literature it has been suggested 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 other study (Erikson and Finnveden, 2009) which concluded that the GHG emissions avoided in energy recovery are highly dependent on the electricity and heat production efficiencies, so for higher ratios, higher GHG benefits should be obtained (Erikson and Finnveden, 2009). Nevertheless, other aspects could influence the decision between recycling and energy recovery. For example in Denmark with an energy recovery around 60% (Plastic Europe, 2013), the environmental benefit of incineration is debated because the recovered energy is likely to substitute fossil fuels-based energy and thus contribute to increase the renewable energy sources in the mix, but also because plastic waste is traded on a world market what could increase the environmental burden of the plastic waste recycling (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 (-269 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 materials 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 higher GHG burden which means that GHG savings from recycling are higher (Friedrich and Trois, 2013). Other studies also pointed out the difficulties in modelling 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 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 in order to save more GHG emissions, the best plastic waste management is mechanical recycling (-620 kg CO_2 eq. t^{-1}). However, contamination was pointed out as one of the most important parameters limiting plastic waste recycling what suggest that collection and sorting are very important processes. Therefore, to promote an increase of the recycling rate seems not realistic if other strategies and actions are not promoted in parallel. In addition, export of plastic waste for recycling presented more GHG emissions reductions than energy recovery; however, there are uncertainties in modelling recycling of plastic waste in a global market. So if the possibility to improve the plastic waste quality is not feasible, efforts should be put on improve the electricity efficiencies of energy recovery plants.

REFERENCES

Asociación Nacional de Recicladores de plástico (ANARPLA), Accessed December 2013. Boletines mensuales 2011-2012-2013; Madrid, http://www.anarpla.com/

Astrup T, Fruergaard T, Christensen TH. Recycling of plastic: accounting of greenhouse gases and global warming contributions. Waste Manage Res 2009; 27(8): 763-72

BIO Intelligence Service, 2013. Study on an increased mechanical recycling target for plastics. Final report prepared for Plastics Recyclers Europe, http://www.plasticsrecyclers.eu/sites/default/files/Study%20on%20an%20increased%20mechanical%20recycling%20target%20for%20plastics_BIOIS.pdf

Briassoulis D, Hiskakis M, Babor E. Technical specifications for mechanical recycling of agricultural plastic waste. Waste Manage 2013; 33(6): 1516-30

Brunner PH, Rechberger H, 2004. Practical Handbook of Material Flow Analysis. Lewis Publisheres, Boca Raton, London, New York and Washington DC

Christensen TH, Bhander G, Lindvall H, Larsen AW, Fruergaard T, Damgaard A, et al. Experience with the use of LCA-modelling (EASEWASTE) in waste management. Waste Manage Res 2007; 25(3): 257-62

Dodbiba G, Fujita T. Progress in separating plastic materials for recycling. Phy Sep Sci Engin 2004; 13(3-4): 165-82

Eriksson O, Finnveden G. Plastic waste as a fuel: CO2 neutral or not?. Energy Environ Sci 2009; 2(9): 907-14

Estadísticas del comercio exterior español (DataComex). Ministerio de Economía y Competitividad (http://datacomex.comercio.es Accessed October 2013)

European Commission, 1994. Directive 94/62/EC. On packaging and packaging waste. Official Journal of the European Union, 1994. L 365: 10-23

European Commission, 2000. Directive 2000/53/EC. On end-of-life vehicles. Official Journal of the European Union. 2000, L 269: 34-43

European Commission, 2004. Directive 2004/12/EC. On packaging and packaging waste. Official Journal of the European Union 2004. L 047: 26-32

European Commission, 2008. Directive 2008/98/EC. On waste and repealing certain Directives. Official Journal of the European Union 2008. L 312: 3-30

European Commission, 2011a. Plastic waste in the environment, Bio-inteligence Service and AEA Technology; Paris, http://ec.europa.eu/environment/waste/studies/pdf/plastics.pdf

European Commission, 2011b. Roadmap to a Resource Efficient Europe, Communication from the commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions; Brussels http://ec.europa.eu/environment/resource_efficiency/about/roadmap/index_en.htm

European Commission, 2012. Manifesto for a resource-efficient Europe, European Commission (EC); Brussels http://europa.eu/rapid/press-release_MEMO-12-989_en.htm

European Commission, 2013. Green paper on a European strategy on plastic waste in the Environment, European Commission; Brussels, http://ec.europa.eu/environment/waste/pdf/green_paper/green_paper_en.pdf

Foster S, 2008. Domestic Mixed Plastics Packaging Waste Management Options, WRAP; London,

http://www.wrap.org.uk/content/domestic-mixed-plastics-waste-management-options

Friedrich R, Trois C. GHG emission factors developed for the recycling and composting of municipal waste in South African municipalities. Waste Manage 2013; 33(11): 2520-31

Hischier R, 2007. Life cycle inventories of packaging and graphical papers. Ecoinvent report Nº 11, Swiss Centre for Life Cycle Inventories, Dübendorf, Switzerland

Hong R, 2012. Plastic Waste Recycling and Greenhouse Gas Reduction Taking. Copenhagen as an example from life cycle assessment perspective. Master Thesis. Department of Development and Planning. Aalborg University, Denmark

Joint Research Centre (JRC), 2012a. Supporting Environmentally Sound Decisions for Waste Management. A technical guide to Life Cycle Thinking (LCT) and Life Cycle Assessment (LCA) for waste experts and LCA practitioners, JRC; Luxembourg,: http://publications.jrc.ec.europa.eu/repository/bitstream/111111111/22582/1/reqno_jrc65850_lb-na-24916-en-n%20_pdf_.pdf

Joint Research Centre (JRC), 2012b. End-of-waste criteria for waste plastic for conversion. Technical proposal. Second working document, JRC; Seville (Spain), http://susproc.jrc.ec.europa.eu/activities/waste/documents/Plastics2ndworkingdoc23may2012.pdf

Kaps R, 2008. Comparative environmental evaluation of plastic waste management at national level on example of Polish and Austrian systems. Thesis for the degree of Doctor of Philosophy, Poznan University of Economics, Faculty of Commodity Science, Poznan, 2008

Laurent A, Bakas I, Clavreul J, Bernstad A, Niero M, Gentil E, Hauschild MZ, Christensen TH. Review of LCA studies of solid waste management systems – Part I: Lessons learned and perspectives. Waste Manage 2014; 34(3): 573-88

Lazarevic D, Aoustin E, Buclet N, Brandt N. Plastic waste management in the context of a European recycling society: Comparing results and uncertainties in a life cycle perspective. Resour Conserv Recy 2010; 55(2): 246-59

Law 11/1997. On packaging and packaging waste. Official Journal of Spain 2001. Nº 99, 13270-13277

Merrild H, Larsen AW, Christense TH. Assessing recycling versus incineration of key materials in municipal waste: The importance of efficient energy recovery and transport distances. Waste Management, 2012, 32 (5), 1009-18

Organization for Economic Co-operation and Development (OECD), 2010. OECD Global Forum on Enviromental. Focusing on sustainable materials management. Materials case study 4: Plastics; Belgium,
http://www.oecd.org/env/waste/46156805.pdf

Perugini F, Mastellone ML, Arena U. A Life Cycle Assessment of Mechanical and Feedstock Recycling Options for Management of Plastic Packaging Wastes. Environ Prog 2005; 24(2): 137-54

Plastic Europe, 2011. Plásticos-Situación en 2011. Análisis de la producción, la demanda y la recuperación de plásticos en Europa en 2010. Plastic Europe, Brussels. Available from: http://www.plasticseurope.es/industria-del-plastico/datos-de-mercado.aspx

Plastic Europe, 2013. Plastics - the Facts 2012. An analysis of European latest plastics production, demand and waste data, Plastic Europe; Brussels, http://www.plasticseurope.org/documents/document/20131018104201-plastics_the_facts_2013.pdf

Plastics Recyclers Europe, 2012. How to boost plastic recycling and increase resource efficiency? Strategy paper of plastics recyclers Europe, Plastic Recyclers Europe; Brussels, http://www.plasticsrecyclers.eu/sites/default/files/EuPR%20Strategy%20Paper%202012_0.pdf

Reinhard J, Zah R. Global environmental consequences of increased biodiesel consumption in Switzerland: consequential life cycle assessment. J Clean Prod 2009, 17, S46-S56

Ross S, Evans D. The environmental effect of reusing and recycling a plastic-based packaging system. J Clean Prod 2003; 11(5): 561-71

Royal Decree 1383/2002. On end-of-life vehicles. Official Journal of Spain 2003. Nº 3, 185-191

Royal Decree 208/2005. On waste electrical and electronic equipment (WEEE). Official Journal of Spain 2005. Nº 49, 7112-7121

Schmidt JH, 2012. Plastberegner.dk-LCA tool for plastics converters in Denmark. Documentation of the tool and database. 2.0 LCA consultants, Aalborg, Denmark

Sevigné-Itoiz E, Gasol CM, Rieradevall J, Gabarrell X. Environmental consequences of recycling aluminum old scrap in a global market. Res Conserv Recy, 2014a, 89; 94-103

Sevigné-Itoiz E, Gasol CM, Rieradevall J, Gabarrell X. Methodology of supporting decision-making of waste management with material flow analysis (MFA) and consequential life cycle assessment (CLCA): case study of waste paper recycling. J Clean Prod 2014b; In press.

Shonfield P, 2008. LCA of Management Options for Mixed Waste Plastics, WRAP; London, http://www.wrap.org.uk/content/domestic-mixed-plastics-waste-management-options

Waste & Resource Action Programme (WRAP), 2011. Market Situation Report-Spring 2011. The Chinese markets for recovered paper and plastics-an update, WRAP; London, http://www.wrap.org.uk/sites/files/wrap/China%20-%202011%20update.pdf

Weidema BP, Ekvall T, Heijungs R, 2009. Guidelines for applications of deepened and broadened LCA. Deliverable D18 of work package 5 of the CALCAS project