Potential savings of resources and greenhouse gas emissions from waste management: a case study of Spain in a global economy

By

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They asked him,
"hey, where's this bus going?"
and he said, "well, I'm really not sure."
"well then, how will you know where to get off?"
and he said, "the place with the most allure."

(I love the unknown, Clem Snide)

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Abbreviations

ACP	aluminium-containing product
Al_2O_3	alumina
BEKP	bleached *Eucalyptus* kraft pulp
$B_{production}$	biogas production
C	carbon
CDS	container deposit scheme
CE	circular economy
CED	cumulative energy demand
C_f	characterization factor
$C_{fbiogas}$	emission factor of biogas combustion in the landfill
$C_{fcombustion}$	emission factor of diesel consumption
$C_{fprod\text{-}d}$	emission factor of diesel production
$C_{fprod\text{-}e}$	emission factor of electricity production
CF	fraction of carbon in dry matter
CH_4	methane
CLCA	consequential life cycle assessment
CO_2	carbon dioxide
CP	compost production
Dc	diesel consumption at plant
dm	dry matter
DMC	direct material consumption
E_c	Electricity consumption
eq	equivalent
EP	energy production
EAA	European Aluminium Association
EC	European Commission
EEA	European Environmental Agency
ELV	end-of-life vehicle
EOL	end-of-life
EPS	expanded polystyrene
EU	European Union
EF	emission factor of treatment plant
EVA	ethyl vinyl acetate
FCF	content of fossil carbon in total carbon
FU	functional unit
g	gram
GDP	gross domestic product
GDS	green dot system
GHG	greenhouse gases
GWP	global warming potential
h	hour
HDPE	high density polyethylene
IE	industrial ecology

IAI	International Aluminium Institute
IPCC	Intergovernmental Panel on Climate Change
Kt	kilo tonnes
kWh	kilowatt hour
LC	life cycle stage
LCA	life cycle assessment
LCI	life cycle inventory
LCIA	life cycle impact assessment
LDPE	low density polyethylene
LHV	low heating value
MBT	mechanical and biological treatment
MFA	material flow analysis
MIT	Massachusetts Institute of Technology
MJ	mega Joule
MR	materials recovered
Mt	million tonnes
MSW	municipal solid waste
N_2O	nitrous oxides
OF	oxidation factor
PB	paper and board
PET	polyethylene terephthalate
PP	polypropylene
PS	polystyrene
PVC	polyvinyl chloride
RR	recovery rate
RER	recovery export rate
RIR	recovery import rate
RUR	recovery utilization rate
SSOF	source-separated organic fraction
SW	solid waste
t	tonne (1000 kg)
TE	total exports
TI	total imports
tkm	tonne kilometer

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Finally, to my unconditional: Fernando, parents, sisters, brothers in law and especially to my nephew Alvaro, although you stay a few years until you can read I want you to know, that you are the most beautiful thing in the world.

Summary

Our societies need solutions to reduce resource consumption as well as greenhouse gas (GHG) emissions. Identifying waste as a valuable resource can help reduce resource consumption and consequently GHG emissions. Therefore, focus has originally been placed on recovery and recycling. However, waste managers and researchers have recently highlighted the importance of wastes traded in global markets. This global perspective could affect the savings of GHG emissions attributed to recycling. Thus, the goal of this thesis is firstly to calculate and evaluate the GHG emissions from municipal solid waste (MSW) management in Spain. Secondly a special focus is put on the GHG emissions of recycling considering the market and the international trade, specifically for waste paper, aluminium old scrap and plastic waste. A new tool called CO2ZW®, consequential life cycle assessment (CLCA) and material flow analysis (MFA) were applied for the GHG and resource assessments.

The application of the CO2ZW® to evaluate the MSW management confirmed that there is a high potential for climate change mitigation in Spain through the increase of material recovery along with reducing disposal to landfills. The application of MFA showed that there is a considerable accumulation of paper, aluminium and plastics products which in coming years will become waste but the increase in waste supply will probably be exported. Moreover, under the assumption that recycling avoids raw material production, it was also determined that recycling avoids GHG emissions. However, waste can be recycled in Spain or abroad, and recycling can substituted global or Spanish raw material production. The GHG emissions varied in each case. The most competitive global productions of virgin pulp, primary aluminium and virgin plastic were identified as the base scenarios under CLCA. The Spanish national productions were assessed as alternative scenarios. Results showed that the most competitive processes generate more GHG emissions as they are more inefficient and they are often located in countries with high hard coal content in their electricity mixes. Therefore, if these processes are avoided by recycling, more GHG emissions are mitigated than if the Spanish processes are avoided. In addition, increasing the export of waste decreases the GHG benefits for all scenarios evaluated except for the aluminium old scrap export.

The results not only help researchers to evaluate the GHG emissions from waste management but also can be used by producers, waste managers and waste politicians to evaluate and propose the best strategy to reduce the resource consumption and the GHG emissions.

Resumen en castellano

Nuestras sociedades necesitan soluciones para reducir el consumo de recursos, así como de gases de efecto invernadero (GHG en inglés). La identificación de los residuos como un recurso valioso puede ayudar a reducir el consumo de recursos y en consecuencia las emisiones de GHG. Por lo tanto, el enfoque ha sido originalmente puesto en la recuperación y el reciclaje. Sin embargo, los gestores de residuos y los investigadores han destacado la reciente importancia de los residuos comercializados en mercados globales. Esta perspectiva global podría afectar los ahorros de emisiones de GHG atribuidas al reciclaje. Por lo tanto, el objetivo de esta tesis es en primer lugar calcular y evaluar las emisiones de GHG de la gestión en España de los residuos sólidos municipales (MSW, en inglés). En segundo lugar un enfoque especial se pone en las emisiones de GHG del reciclaje teniendo en cuenta el mercado y el comercio internacional, especialmente para los residuos de papel, aluminio y plástico. Una nueva herramienta llamada CO2ZW ®, el análisis de ciclo de vida consecuencial (CLCA en inglés) y el análisis del flujo de materiales (MFA en inglés) se aplicaron para las evaluaciones de GHG y de recursos.

La aplicación de la CO2ZW ® para evaluar la gestión de los MSW confirmó que existe un alto potencial para la mitigación del cambio climático en España a través del aumento de la recuperación material junto con la reducción de la eliminación de los vertederos. La aplicación del MFA demostró que existe una considerable acumulación de productos de papel, aluminio y plástico que en los próximos años se convertirá en residuos pero el incremento de la oferta de residuo probablemente será exportado. Por otra parte, bajo el supuesto de que el reciclaje evita la producción de materia prima, se determinó también que el reciclaje evita emisiones de GHG. Sin embargo, los residuos pueden ser reciclados en España o en el extranjero, y el reciclaje puede sustituir la producción mundial o la española. Las emisiones de GHG varían en cada caso. Las producciones mundiales más competitivas de pulpa virgen, aluminio primario y plástico virgen fueron identificadas como los escenarios base bajo el CLCA. Las producciones nacionales españoles fueron evaluados como escenarios alternativos. Los resultados mostraron que los procesos más competitivos generan más emisiones de GHG, ya que son más ineficientes y a menudo se encuentran en países con alto contenido de carbón en su mix eléctrico Por lo tanto, si estos procesos se evitan mediante el reciclaje, más emisiones de GHG son mitigados que si se evitan los procesos españoles. Además, el aumento de la exportación de residuos disminuye los beneficios de GHG en todos los escenarios evaluados con excepción de la exportación de chatarra de aluminio.

Los resultados no sólo pueden ayudar a los investigadores para evaluar las emisiones de GHG de la gestión de residuos, pero también pueden ser utilizados por los productores, gestores de residuos y los políticos de residuos para evaluar y proponer la mejor estrategia para reducir el consumo de recursos junto con las emisiones de GHG.

Resum en català

Les nostres societats necessiten solucions per reduir el consum de recursos, així com els gasos d'efecte hivernacle (GHG, en anglès). La identificació dels residus com un recurs valuós pot ajudar a reduir el consum de recursos i en conseqüència les emissions de GHG. Per tant, l'enfocament ha estat originalment col·locat en la recuperació i el reciclatge. No obstant això, els gestors de residus i els investigadors han posat en relleu la recent importància dels residus comercialitzats als mercats globals. Aquesta perspectiva global podria afectar els estalvis d'emissions de GHG atribuïdes al reciclatge. Per tant, l'objectiu d'aquesta tesi és en primer lloc calcular i avaluar les emissions de GHG de la gestió a Espanya dels residus sòlids municipals (MSW, en anglès). En segon lloc un enfocament especial es posa en les emissions de GHG del reciclatge tenint en compte el mercat i el comerç internacional, especialment per als residus de paper, alumini i plàstic. Una nova eina anomenada CO2ZW ®, l'anàlisi de cicle de vida consecuencial (CLCA, en anglès) i l'anàlisi del flux de materials (MFA, en anglès) es van aplicar per a les avaluacions de GHG i de recursos.

L'aplicació de la CO2ZW ® per avaluar la gestió dels MSW va confirmar que existeix un alt potencial per a la mitigació del canvi climàtic a Espanya a través de l'augment de la recuperació material juntament amb la reducció de l'eliminació dels abocadors. L'aplicació del MFA va demostrat que existeix una considerable acumulació de productes de paper, alumini i plàstic que en els propers anys es convertirà en residus però l'augment de l'oferta de residus probablement serà exportat. D'altra banda, sota el supòsit que el reciclatge evita la producció de matèria primera, es va determinar també que el reciclatge evita emissions de GHG. No obstant això, els residus poden ser reciclats a Espanya o a l'estranger, i el reciclatge pot substituir la producció mundial o l'espanyola. Les emissions de GHG varien en cada cas. Les produccions mundials més competitives de polpa verge, alumini primari i plàstic verge van ser identificades com els escenaris basi sota el CLCA. Les produccions nacionals espanyols van ser avaluats com a escenaris alternatius. Els resultats van mostrar que els processos més competitius generen més emissions de GHG, ja que són més ineficients i sovint es troben en països amb alt contingut de carbó en el seu mix elèctric Per tant, si aquests processos s'eviten mitjançant el reciclatge, més emissions de GHG són mitigats que si s'eviten els processos espanyols. A més, l'augment de l'exportació de residus disminueix els beneficis de GHG en tots els escenaris avaluats amb excepció de l'exportació de ferralla d'alumini.

Els resultats no només poden ajudar als investigadors per avaluar les emissions de GHG de la gestió de residus, però també poden ser utilitzats pels productors, gestors de residus i els

polítics de residus per avaluar i proposar la millor estratègia per reduir el consum de recursos juntament amb les emissions de GHG

Preface

This thesis has been developed within the research group Sostenipra (Sustainability and Environmental Prevention) as part of the "Environmental Science and Technology" PhD program of the Institute of Environmental Science and Technology (ICTA) of the Universitat Autònoma de Barcelona (UAB) from October 2011 to May 2014.

The research analyses the GHG emissions of the waste management in Spain with special focus on the GHG quantifications of recycling. The aim is to evaluate opportunities for waste management to contribute to climate change mitigation and resource savings by assessing quality, technological and market constraints. The research attempts to highlight the need for quality tools and proper GHG quantifications of recycling by considering the current market trends traditionally excluded from LCA studies but necessary for decision support processes for waste management and resource saving.

The dissertation is divided into four sections and nine chapters, with the main contents summarized below. For clarity, the structure of the thesis is outlined in Figure A which can be used during the reading of the manuscript as a *thesis map*.

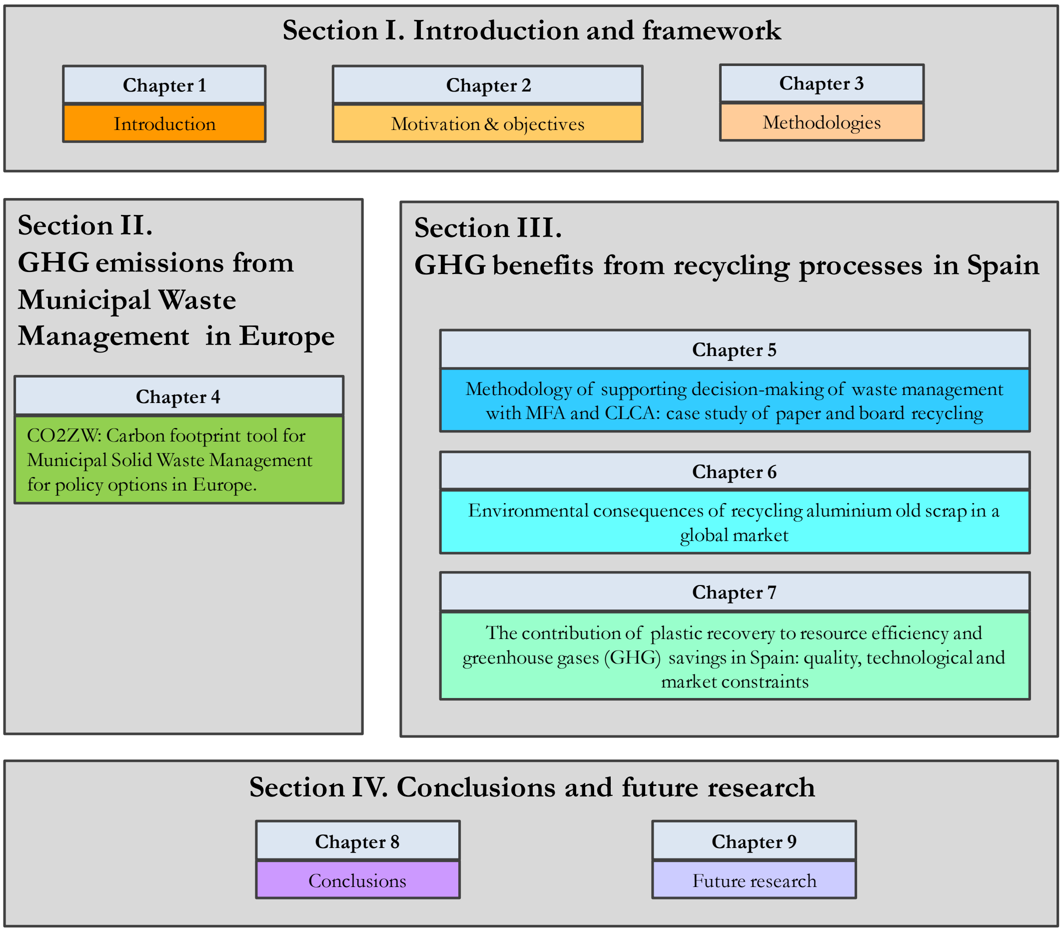

Figure A: Flow chart of the thesis structure

Section I. Introduction and framework

Section I is composed of three chapters. **Chapter 1** presents the problems of traditional linear systems of production and introduces the concepts of Industrial Ecology (IE) and Circular Economy (CE) in order to achieve more circular systems in which waste is a valuable resource. In addition, it introduces the topic of waste management and GHG emissions and highlights the current uncertainties and challenges in the GHG quantifications of recycling. Finally, Spain is presented as a representative case study of European trends. **Chapter 2** presents the motivation of this thesis and the objectives are listed. **Chapter 3** gives an overview of the methodologies used and the study systems considered throughout the research: intergovernmental panel on climate change (IPCC) for direct GHG emissions, MFA and CLCA.

Section II. GHG emissions from Municipal Waste Management in Europe

Section II is composed of **Chapter 4** [*CO2ZW®: Carbon footprint tool for municipal solid waste management for policy options in Europe: inventory of Mediterranean countries*]. This chapter presents, describes, compares and evaluates a new tool called CO2ZW® developed by Sostenipra and able to calculate GHG emissions from MSW in Europe. The CO2ZW® includes default data for Italy, Slovenia, Greece, Spain and Europe. The comparison with other tools shows that the CO2ZW®includes all key life cycle stages of collection & transport, sorting facilities, biological treatments, incineration treatments and landfill deposition. In addition, it incorporates and allows the modification of the principal parameters discussed in literature affecting the calculation of the GHG emissions; such as the differentiation of biogenic and fossil carbon (C) content or the C sequestration in landfill. Besides, the CO2ZW® tool includes the two approaches more used and proven in the international scientific framework of GHG calculations from landfills, the IPCC methodology (which calculates the present and past emissions for a given year) and the LCA approach (which calculate the future emissions derived from the current waste disposal). However, through the evaluation of the CO2ZW®, the need to further investigate GHG quantifications of recycling processes was detected. The results showed that these parameters are the most sensitive to the calculations and also because there was no previous quantification for Spain. Therefore, the following chapters delve into the quantification of GHG emissions from recycling. This chapter includes an addendum as there is a new version of CO2ZW® tool including default data from Catalonia and a new section to calculate the GHG emissions from waste management in industrial areas.

Section III. GHG quantifications from recycling processes in Spain

Section III consists of three chapters and is focus on GHG quantifications of recycling of paper and board waste, aluminium old scrap and plastics waste from all waste streams (i.e., construction). **Chapter 5** [*Methodology of supporting decision-making of waste management with MFA and CLCA: case study of paper and board recycling*] presents a methodological framework to consider the market effects on the quantification of GHG emissions of recycling by integrating the methodologies of MFA and CLCA. We applied the methodological framework to the paper and cardboard recycling system in Spain. The results showed that there is an increasing tendency to import virgin pulp from other countries and to export collected paper, mainly to China. While increasing the export of recovered paper, the GHG benefits of recycling are lost. **Chapter 6** [*Environmental consequences of recycling aluminium old scrap in a global market*] evaluates with the same methodological framework the GHG quantifications of old scrap aluminium recycling. For this material, more GHG emissions are avoided while increasing the export of old scrap and the same tendency to import raw material and export waste is observed. **Chapter 7** [*The contribution of plastic recovery to resource efficiency and greenhouse gases (GHG) savings in Spain: quality, technological and market constraints*] uses the integration of MFA and CLCA, however, data for the calculations are aggregated so the study is more focused on evaluating the current market, technological, quality and management limitations of plastic waste recycling and the GHG balance. The results suggest that at current recycling rates, the quality of plastic waste is more important for GHG benefits than the quantity collected. The options for plastic waste management are with or against the waste hierarchy depending on the quality of the recovered plastic. In order to save more GHG emissions, the best options from high to low savings are mechanical recycling for quality recovered plastic, export of recovered plastics, energy recovery and mechanical recycling for low quality recovered plastic.

Section IV. Conclusions and future research

Section IV concludes this dissertation and it consists of two chapters. **Chapter 8** provides a general discussion of most important results and gives general conclusions from the research presented. General discussion is organized into three sections: those related to waste and climate changes; those related to the CO2ZW® tool; and those related to the GHG quantifications of recycling. Finally, **Chapter 9** presents some recommendations for further work, to continue evaluating opportunities of waste as resource and climate change mitigation.

Note: Chapters 4 to 7 each present an article that is either published or accepted or under review in a peer-reviewed indexed scientific journal. Therefore these chapters follow the format of an article and include: abstract, introduction, methodology, results, discussion and conclusion. All chapters include the co-authors of the manuscripts and in the case of the chapters based on submitted or soon to be submitted manuscripts, the co-authors reflect those participated at the time of submission of the thesis.

Dissemination

The thesis is mainly based on the following published and accepted papers:

- Sevigné Itoiz E, Gasol CM, Farreny R, Rieradevall J, Gabarrell X. CO2ZW®: Carbon footprint tool for municipal solid waste management for policy options in Europe. Inventory of Mediterranean countries. Energy Policy 2013, 56, 623-32
- Sevigné Itoiz E, Gasol CM, Rieradevall J, Gabarrell X. Environmental consequences of recycling aluminium old scrap in a global market. Accepted to appear in Resources, Conservation & Recycling (13 May 2014)

The following manuscripts that have been submitted and are under review are also part of this thesis:

- Sevigné Itoiz E, Gasol CM, Rieradevall J, Gabarrell X. Methodology of supporting decision-making of waste management with MFA and CLCA: case study of paper and board recycling.
- Sevigné Itoiz E, Gasol CM, Rieradevall J, Gabarrell X. The contribution of plastic recovery to resource efficiency and greenhouse gases (GHG) savings.

Moreover, the following oral communications and posters presented to congresses and conferences also form part of this doctoral work:

- Sevigné Itoiz E, Oliver-Solà J, Gasol CM, Mitjans V, Rieradevall J, Gabarrell X. Comparative LCA of container Deposit Scheme and Green Dot System for PET bottles, cans and beverage carton waste in Spain. 2011. Poster. Life Cycle Management International Conference (LCM). Berlin (Germany)
- Sevigné Itoiz E, Gabarrell X, Farreny R, Gasol CM, Rieradevall J, Villalba G, Font X, Starr K, Quirós R, Rovira MR, Artola A, Sánchez T, Suarez ME, Blanquez P. Potencial de reducció d'emissions de GEI a través de la gestió de residus i aigua residuals. 2012. Poster. Reunió de membres del Grup d'Experts sobre el Canvi Climàtic a Catalunya (GECCC). Monestir de les Avellanes (Catalunya)
- Sevigné Itoiz E, Gasol CM, Farreny R, Rieradevall J, Gabarrell X. Greenhouse gases generated in the waste management of Barcelona. The CO2ZW® tool applied to cities. 2013. Oral communication. Symposium on ecoinnovation in the Sudoe region. Toulouse (France)
- Gabarrell X, Sevigné Itoiz E, Gasol CM, Rieradevall J. Material and environmental metabolism of paper and board recycling in Spain: international trade and credits of

CO_2. 2013. Oral communication. 7th International Society of Industrial Ecology Biennal Conference (ISIE). Ulsan (Korea)

- Sevigné Itoiz E, Gasol CM, Rieradevall J, Gabarrell X. GWP benefits from paper and board recycling with consequential LCA: Spanish case study considering the interaction with global markets. 2013. Oral communication. 6th International Conference on Life Cycle Management (LCM). Gothenburg (Sweden)
- Quirós R, Sevigné Itoiz E, Gasol CM, Farreny R, Rieradevall J, Gabarrell X. Quantification and validation of GHG emissions from Municipal Waste Management with CO2ZW® ® tool. 2013. Oral communication. International Solid Waste Association World Congress (ISWA). Vienna (Austria)
- Gabarrell X, Sevigné Itoiz E, Gasol CM, Rieradevall J. Créditos de CO_2 de la gestión de residuos en España y su interacción con los mercados globales: papel y aluminio. 2013. Oral communication. III Simpósio sobre Resíduos Sólidos. Sao Carlos (Brazil)
- Gabarrell X, Sevigné Itoiz E, Gasol CM, Rieradevall J. The role of dynamic perspectives to model future scenarios for attributional and consequential life cycle assessments. 2014. Oral communication. 24th Annual meeting SETAC Europe. Basel (Switzerland)
- Gabarrell X, Sevigné Itoiz E, Gasol CM, Rieradevall J. New approach to track the international trade of material flows within the GHG quantification of recycling. Oral communication. 11th International Society for Industrial Ecology (ISIE) Socio-Economic Metabolism section conference and the 4th ISIE Asia-Pacific conference Melbourne (Australia)

Furthermore, during the pre-doctoral master's degree, research focusing on environmental fields other than the topic of this thesis was also carried out. This parallel research was published in peer-reviewed journals:

- Sevigné Itoiz E, Gasol CM, Brun F, Rovira L, Pagés JM, Camps F, Rieradevall J, Gabarrell X. Water and energy consumption of *Populus spp.* bioenergy systems: A case study in Southern Europe. Renewable and Sustainable Energy Reviews 2011 15(2): 113-40
- Sevigné Itoiz E, Fuentes-Grünewald C, Gasol CM, Garcés E, Alacid E, Rossi S, Rieradevall J. Energy balance and environmental impact analysis of marine microalgal biomass production for biodiesel generation in a photobioreactor pilot plant. Biomass & Bioenergy 2012; 39: 324-35

- Gasol CM, Salvia J, Serra J, Anton A, Sevigné Itoiz E, Rieradevall J, Gabarrell X. A Life Cycle Assessment of biodiesel production from winter rape grown in Southern Europe. Biomass & Bioenergy 2012; 40: 71-81
- Farreny R, Oliver-Solà J, Escuder-Bonilla S, Roca-Martí M, Sevigné E, Gabarrell X, Rieradevall J. The metabolism of cultural services. Energy and water flows in museums. Energy and Buildings 2012; 47: 98-106

Finally, between the completion of the master's and the beginning of this thesis, other research focusing on environmental fields other than the topic of this thesis was also carried out during November 2010 and August 2011 at the Institute of Agriculture and Food Research and Technology (IRTA) of the regional Government of Catalonia, which is also part of research group Sostenipra. This parallel research was published in peer-reviewed journals:

- Sevigné Itoiz E, Fantke P, Juraske R, Kounina A, Antón Vallejo A. Deposition and residues of azoxystrobin and imidacloprid on greenhouse lettuce with implications for human consumption. Chemosphere 2012; 89(9): 1034-41
- Fantke P, Wieland P, Juraske R, Shaddick G, Sevigné Itoiz E, Friedrich R, Jolliet O. Parameterization Models for Pesticide Exposure via Crop Consumption. Environmental Science & Technology 2012; 46(23): 12864-72

SECTION I

Introduction and framework

Chapter I -

Introduction

1- Introduction

Firstly, **Chapter 1** presents an overview of the problems related with the unsustainable way of production and consumption which have relied on an increasing consumption of resources and waste generation which ends up in landfill. Secondly, it is explained how waste can be regarded as a valuable resource to recirculate to the economy based on the principles and tools of IE and CE. Thirdly, it is explained that waste management generates GHG emissions but GHG emissions can be avoided through recovery and recycling. Thus, the recirculation of waste is also seen as a way of decreasing GHG emissions. Fourthly, a number of tools and methodologies for waste GHG quantifications are currently available but there are some uncertainties and current challenges to take into account. Finally, Spain is presented as a case of study.

This chapter is structured as follows:

1.1 The problems of linear systems of production and consumption

1.2 Looking for solutions through resource efficiency and circular systems

1.3 The contribution of waste management to climate change

1.4 Tools for GHG quantifications from waste management

1.5 Spain as a case study

1.1. The problem of linear systems of production and consumption

Our life is based on natural resources in the form of materials, water and energy, as well as the land available to us on Earth. Without the constant use of natural resources, neither our economy nor our society could function. Nature provides humans with all resources necessary for life: energy for heat, electricity and mobility; wood for furniture and paper products; cotton for clothing; construction materials for infrastructures; food and pure water for a healthy diet. Natural resources have always been the material basis of societies and their economic systems. However, in human history, the annual per capita level of resource consumption changed dramatically from around 1 tonne per year in hunter-gatherer societies to 15-30 tonnes in modern industrialized nations (Kernegger and Giljum, 2009).

The 20th century was a time of remarkable progress for human civilization and driven by scientific and technological advances, we became more efficient in our resource consumption. However, thanks to population growth at global scale we are now consuming more resources than ever. Current growth trends suggest that this consumption could increase to 140 billion tonnes by 2050 (UNEP, 2011; Green Alliance, 2011). The Earth has only finite resources, and the consequences of these production and consumption patterns had profound material and environmental impacts such as over-exploitation, scarcity of resources, climate change, pollution, land-use change and loss of biodiversity which rose toward to top of the list of major international concerns (UNEP, 2011). In addition, during the 20th century, our global system of production and consumption has become predominantly linear based on extraction, production, consumption and disposal of waste (see Figure 1.1). Therefore, the other side of the story has been an increasing waste generation ending up in landfills. At present, annual total solid waste generation worldwide is approximately 17 billion tonnes and it is expected to reach 27 billion by 2050. About 1.3 billion tonnes are currently MSW generated by world cities, which are anticipated to generate up to 2.2 billion tonnes by 2025 (Laurent et al., 2014).

Figure 1.1: Linear system of production based on extraction, production, consumption and disposal

Furthermore, the resource challenge of the next 20 years will be quite different from any we have seen in the past. The new middle class of consumers is expected to grow (especially in China and India), the cost of new supply is expected to increase (i.e., the average real cost per oil well which has doubled over the past decade) or the resources prices is expected to be higher than at any point over the past century (Zils, 2013). It is clear that there is a lot of work to do and several challenges are besetting the 21st century. This situation urges the transition to another way of production and consumption with less resource consumption and less waste generation. How we achieve sustainable global patterns of resource use will be a major economic and environmental challenge of the 21st century (Green Alliance, 2011).

1.2. Looking for solutions through resource efficient and circular systems

As a main driver for these problems, natural resource use currently features prominently on the environmental policy agenda both in Europe and in other world regions (Giljum, 2008). Leaders increasingly understood that making progress towards a more sustainable economy requires an absolute reduction in resource use at a global level (UNEP, 2011). Resource efficiency (*defined as the systematic reduction in the quantity of resource employed to produce goods and services in the economy*) and decoupling (*defined as the use of less resource per unit of economic output and reducing the environmental impact of any resources that are used*) have been proposed as key determinants of economic success and human to well-being in the 21st century (Aldersgate Group, 2012).

While it is important to reduce the flow of new materials into the production process and make production processes more efficient, we also need to reduce the material loss and waste generated throughout production and consumption (EEA, 2014a). Consequently, the recirculation of waste through reuse, recovery or recycling has also been a key issue. Extracting fewer materials and using existing resources would help avert some of the impacts created along the chain (EEA, 2014a)

Traditionally waste has been considered as an inconvenient and unwanted by-product of society and industry, and although several attempts were conducted during the 20th century to change this vision, mostly remained marginal (Erkman, 2002). However, in 1989 Robert Frosch and Nicholas Gallopoulos wrote an article (Frosch and Gallopoulos, 1989) which essentially constituted the birth of the field of IE. In this article they compared the industrial approach to the use of materials and energy to that of nature (Harper and Graedel, 2004). The main idea was to compare industrial ecosystems with natural ones as a solution to move to more cyclic use of resources and materials, marking a shift away from thinking about waste as an unwanted burden to seeing it as a valued resource (see Figure 1.2).

Figure 1.2: Partial closed system of production based on recycling at the end of life

Since the publication of the article, the scientific field of IE has grown quickly and it was officially recognized at the National Academy of Engineering in 1992. In 1997, eight years after the Frosch and Gallopoulos paper, the first issue of the Journal of IE was published (owned by Yale University and published by The Massachusetts Institute of Technology (MIT) Press). The start of this journal can be seen as an official recognition by the academic community of the new field of IE (Erkman, 2002). In 2001, the International Society for IE was announced and biannual conferences have been held since then. In June 2013, the new constitution of the Swiss Canton of Geneva went into force, including an article (161) in which the State shall respect the principles of IE (Cst-GE, 2014).

More recently, the CE concept is gaining strength. The main idea is that *"greater circularity in the economy has the potential to mitigate the impacts of primary extraction, processing and production as well as disposal since waste would become a valuable input to another process, and products could be repaired, reused or upgraded instead of thrown away (Chatman House, 2012)"*. The CE integrates the IE principles from a broader perspective to support resource optimization but also to move away from an accumulation society (ESA, 2013).

Both IE and CE consider waste as a valuable resource that might be reintroduced into the economy. In addition, waste prevention and improvements in product design have also been proposed as main contributors to CE (BIO Intelligence Service, 2011). The best way to reduce the environmental impacts of waste is to prevent it in the first place. Besides, redesigning products and production processes could help minimize waste and turn the unused portion into a resource (EEA, 2014a). In an ideal world, almost everything would get re-used, recycled or recovered to produce other outputs. Figure 1.3 represents the main idea that lead to CE.

Figure 1.3: Circular system of production based on the concept of CE over the whole chain of production and consumption

Policy makers and other stakeholders are starting to appreciate the scale of the opportunities available by switching to a CE (ESA, 2013). China's leadership, inspired by Japanese and German Recycling Economy Laws, has formed a CE initiative that has major strategic importance worldwide and recently has adopted a new law for the CE promotion (NPSCS, 2008). Various EU strategies and legislation, such as Europe 2020, the Flagship initiative for a Resource-Efficient Europe, the Manifesto for a Resource Efficient Europe, the Waste Framework Directive or the 7th Environment Action Programme, are already in place and try to instil sustainability in key economic activities in a long-term transition perspective (EEA, 2014a). It is clearly stated that in a world with growing pressures on resources and the environment, the EU has no choice but to go for the transition to a resource-efficient and ultimately regenerative CE (European Commission, 2012b).

Catalonia, in the South of Europe, has been one of the first European areas which have included the principle of contribution to the CE in the new plan of waste management for 2013-2020. It is a way to recognize the need to use resources as efficiently as possible to reduce all type of waste, and change patterns of production, consumption and waste management in order to contribute to fully competitive economies and sustainable future (ARC, 2013).

1.3. The contribution of waste management to climate change

Every time a rotten lettuce is thrown in the bin, a broken toy discarded, or industrial scrap carted away, resources are being used up. This all contributes to the environmental pressures on our planet and once a product is thrown away and becomes waste, a whole new set of impacts are involved in treating it (European Commission, 2005). Thus, improved waste management is an essential element in efforts to make Europe more resource efficient (EEA, 2013b). Waste is defined as "*any substance or object which the holder discards or intends or is required to discard (European Commission, 2008)*". Therefore, waste is generated among different sources: in households, at commercial activities, at industries, at agriculture sites, in construction and mining and from the generation of energy (EEA, 2014b).

MSW is defined as *"waste collected by or on behalf of municipal authorities, or directly by the private sector (business or private non-profit institutions) not on behalf of municipalities. The bulk of the waste stream originates from households, though similar wastes from sources such as commerce, offices, public institutions and selected municipal services are also included. It also includes bulky waste but excludes waste from municipal sewage networks and municipal construction and demolition waste (Eurostat, 2014a)".* Special focus has been put into MSW as in Europe constitutes only around 10-14 % of total waste generated, but because of its complex character and its distribution among many waste generators, environmentally sound management of this waste is complicated (EEA, 2014c). Therefore, strict regulations to reduce MSW generation and improve MSW management were implemented in recent decades.

The past 30 years have been characterized by a change in complexity and scope of European waste management policies through a series of environmental action plans and a framework of legislation (European Commission, 2010). However, the amount of waste continued to increase and the nature of waste itself changed, partly due to the use of hi-tech products. As a result, in the 2000s waste management problems have increasingly been perceived as complex and several Directives have been adopted focused on waste treatment and on waste streams policies (i.e., European Commission, 1994; European Commission, 2000a; European Commission, 2000b). Besides, the 2005 Thematic Strategy on Waste Prevention and Recycling resulted in the revision of the Waste Framework Directive. The revision brings a modernized approach to waste management, marking a shift away from thinking about waste as an unwanted burden to seeing it as a valued resource in accordance with the IE and CE principles. The Directive introduced a new 50 % recycling target for MSW and also introduces a five-step waste hierarchy where prevention is the first option, followed by reuse, recycling and other forms of recovery, with disposal such as landfill as the last resort (European Commission, 2008). In addition, life cycle thinking (LCT) which seeks to identify the environmental improvement opportunities at all stages across the life cycle from raw materials to landfill (JRC, 2011) can be used to complement the waste hierarchy in order to make sure that the best overall environmental option is identified (JRC, 2011).

The benefits of shifting MSW management up the waste hierarchy are not limited to more efficient resource use and a reduced waste burden on the natural environment. Better waste management also offers a way to cut GHG emissions (EEA, 2013b), thus, in parallel, MSW management policies have been closely related to climate policies (Bogner et al. 2007). MSW management generates GHG emissions, also referred as direct GHG emissions. The most significant of which are the methane (CH_4) gas produced in landfill which is mostly released during the breakdown of organic matter. Collection and transport of waste generate GHG emissions due to the use of fuel and from the infrastructure. Biological treatments including composting and anaerobic digestion generate carbon dioxide (CO_2), CH_4 and nitrous oxide (N_2O). Relevant gases emitted during incineration include CO_2, CH_4 and N_2O but normally, emissions of CO_2 from waste incineration are more significant than CH_4 and N_2O emissions. In addition, waste management activities have upstream activities which are needed for running waste management operations (i.e., fuel or ancillary materials) and downstream activities due to recovered materials and energy from these operations which can be supplied back to the economic cycle offsetting primary resources (Gentil, 2011). As a result, all the GHG emissions generated over the operating activities, upstream activities (referred as indirect emissions) and downstream activities (referred as avoided emissions) may be taken into account. Figure 1.4 represents graphically the different types of emissions from waste management.

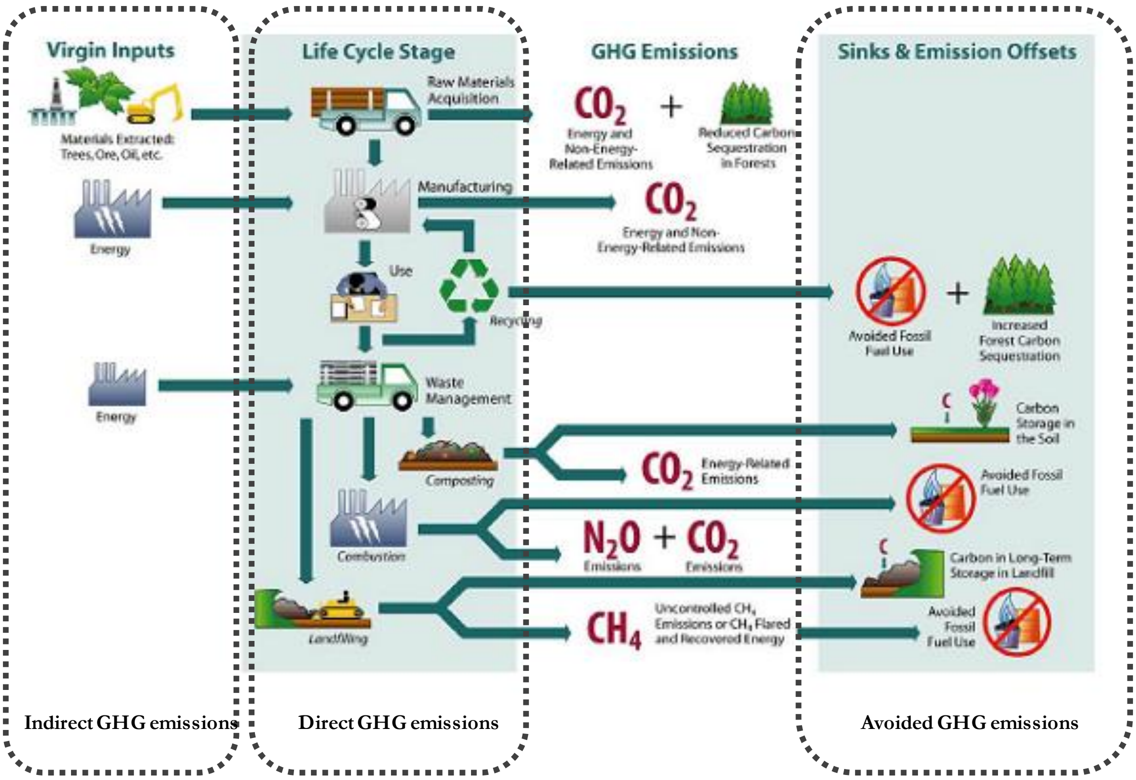

Figure 1.4: Types of GHG emissions from MSW management from LCA perspective

Source: Adapted from US EPA, 2006

The relationship between waste management and GHG emissions has been enhanced based on the idea that treatment and disposal of waste produce significant amounts of direct and indirect GHG emissions (IPCC, 2006) but proper waste management can avoid GHG emissions due to controlled composting of organic waste or by waste recycling through the conservation of raw materials and fossil fuels (Bogner et al., 2007). This is important since the environmental consequences of waste management often depend more on the impacts on surrounding systems than on the emissions from the waste management system itself (Ekvall, 2000). For example, at European level, it is anticipated that the net emissions of GHG from MSW management are reduced equivalent to about 55 million tonnes (Mt) of CO_2 in the late 80s to the equivalent of 10 Mt of CO_2 yearly maximum volume 2020 (see Figure 1.5).

It is expected that the amount of MSW sent to management facilities continue to increase which will increase the direct GHG emissions due to its treatment. However, as recycling and incineration rates will also increase, the avoided GHG emissions will increase which will decrease the net GHG emissions. Recycling would contribute to 75% of total GHG emission reduction and incineration at nearly 25% in 2020 (EEA, 2008).

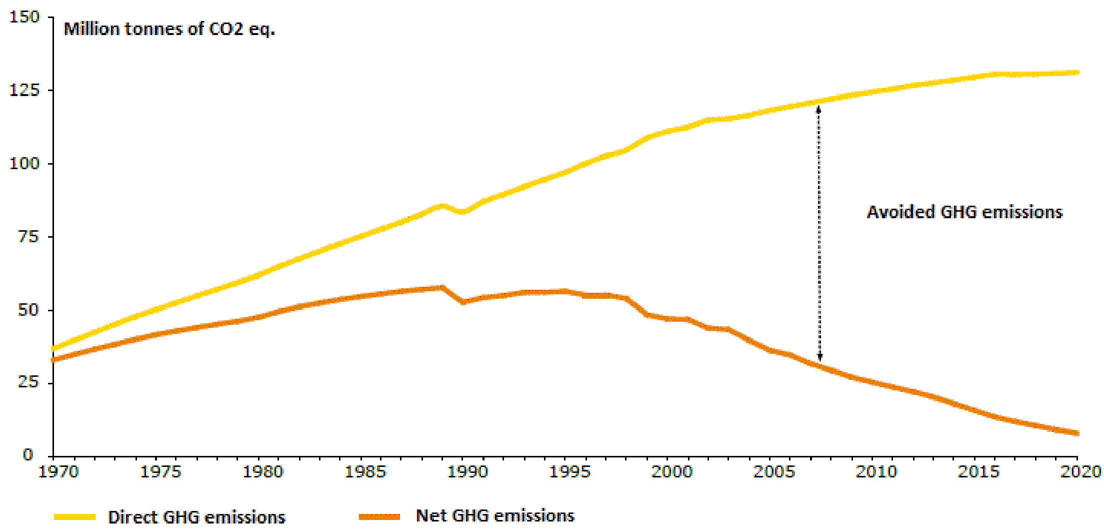

Figure 1.5: European direct GHG emissions and avoided GHG from MSW management
Source: EEA, 2008

1.4. Tools for GHG quantification from waste management

The accounting, reporting and modeling of GHG emissions started to be implemented on a global scale since the inception of the Kyoto Protocol (1997). Different protocols have been developed since then such as the IPCC guideline methodology for waste management activities (Bogner et al., 2007; IPCC, 2006) and jointly with the Life cycle assessment (LCA) are currently the principal GHG quantification methods. The IPCC protocol has been widely used to report the national GHG emissions from waste management (IPCC, 2006), however, with this protocol it only is possible to calculate the direct GHG emissions from disposal, biological treatment and incineration (IPCC, 2006). It is limited to the direct GHG emissions because, historically, the waste management sector was broadly constituted of disposal (open dumping, landfilling) and mass burn incineration, without energy recovery.

Also, because upstream and downstream activities are accounted for in other sectors (i.e., energy) and therefore including them would lead to double counting (Gentil, 2011). This constitutes a limitation in comparison with the LCA methodology which allows calculating the indirect GHG emissions and the avoided GHG emissions. Thus, this broad system perspective has made the LCA a powerful tool for the environmental comparison of different options for waste management. Because of this, quantifying the GHG emissions from MSW has been crucial and LCA has gained in acceptance as a tool for MSW management planning and policy-making. The applications of LCA studies and tools have been promoted to help in the decision-support processes (Smith et al., 2001; JRC, 2011). Both methodologies are used in this thesis so more detailed descriptions are presented in **Chapters 4, 5, 6, and 7**.

1.4.1.Uncertainties and limitations behind the GHG quantifications

Two important aspects regarding the GHG quantifications have been widely discussed with often conflicting interpretation; the Global Warming Potential (GWP) values and the carbon cycle. Under the Kyoto protocol it was decided to use the values of GWP for converting the various GHG emissions into comparable CO_2 equivalents. The GWP integrates the radiative force (how much heat is trapped by a GHG in the atmosphere) of a substance over a chosen time horizon and relative to that of CO_2 whose GWP was standardized to 1. These values are very dependent on metric type and time horizon (Myhre et al., 2013) because a gas which is quickly removed from the atmosphere may initially have a large effect but for longer time periods as it has been removed becomes less important, thus, GWPs were calculated over a specific time interval, commonly 20, 100 or 500 years (Myhre et al., 2013).

In addition, these values have been updated over the years due to new estimates of lifetimes, impulse response functions and radiative efficiencies. First values appeared in the IPCC Second Assessment Report from 1995 (IPCC, 1995) and for instance the GWP for CH_4 was 21 for a time horizon of 100 years. Since then, three more assessment reports have been published with the aim of assessing scientific, technical and socio-economic information concerning climate change, its potential effects, and options for adaptation and mitigation; the IPCC Third Assessment Report (IPCC, 2001), the IPCC Fourth Assessment Report (IPCC, 2007) and in May of 2014 the IPCC Fifth Assessment Report (IPCC, 2014a) has been presented. The GWP values have been updated and in the last assessment report, for instance, the GWP for CH_4 value is 34 (Myhre et al., 2013). The GHG accountings are dependent on the GWP values.

Furthermore, for biodegradable materials (i.e., paper) the carbon will have been absorbed from the atmosphere by photosynthesis during plant growth. If this carbon is released again as CO_2 during the treatment process then the carbon re-enters the natural carbon cycle. These emissions are reported as biogenic CO_2 (Smith et al., 2001) and the IPCC methodology ignore the contribution of biogenic CO_2. However, it is argued that the plant growth does not occur evenly over years and seasons. Therefore it could be several years before a flux of biogenic CO_2 emitted instantaneously from a process (i.e. combustion of biogenic carbon) is re-captured through plant growth. In addition, the atmosphere does not differentiate between a molecule of biogenic CO_2 and a molecule of fossil derived CO_2 and the key theme is climate change and how to mitigate it, not differentiation of carbon sources (UNEP, 2011). That's why some models do not quantify these biogenic CO_2 emissions, some quantify these biogenic CO_2 emissions but do not consider the emission to contribute to GWP, and some quantify these biogenic CO_2 emissions and consider that emissions contribute to GWP (Christensen et al., 2009).

On the other hand, in some organic materials, particularly plastics, the carbon originates from fossil carbon reserves laid down many millions of years ago. Combustion of fossil fuels releases the stored carbon into the atmosphere as fossil-derived CO_2 and these emissions are reported as fossil CO_2 and have the usual GWP of one. However, for almost all treatment options, not all of the carbon released from organic materials during the treatment process is returned to the atmosphere; if the carbon is sequestered then it could be argued that a sink for carbon has been created (Smith et al., 2001) (i.e., landfill of plastic waste). This raises the issue of how this carbon should be accounted for, when comparing the treatment options in terms of climate change. There is on-going debate as to whether this type of carbon sink will be included under the Kyoto Protocol. At present, the topic of carbon storage in soils is being considered for inclusion but the issue of landfills as a carbon sink has not been raised (Smith et al., 2001; Myhre et al., 2013).

Recently, it is gaining relevance the discussion around the use and application of the biostabilized (also known as grey compost) produced at mechanical and biological treatment plants (MBT). Currently, the grey compost produced is not suitable for agriculture application and in most cases ends up in landfills. Therefore, new applications for its use are under research. It is also under discussion if this grey compost production should be accounted or not within the overall recycling rates.

1.4.2. Challenges of GHG quantification of recycling in a global economy

In the framework of LCA, how to account for the environmental impacts of recycling have extensively discussed and discussion have traditionally relied on the methodological problems of multifunctionality as the product to be recycled has two functions (co-function): firstly the function(s) the product is primarily made for and secondly the function of providing secondary resources for use in subsequent life cycles/systems (ILCD, 2011). This characteristic leads to problems of how to allocate the environmental impacts between the co-functions and many approaches have been suggested. In fact, recycling allocation can also be examined within the larger discussion of attributional life cycle assessment (ALCA) or CLCA. ALCA, which can be thought of as LCA in its traditional form, describes how the primary production, the waste pretreatment, recycling steps and waste landfilling have to be shared between the first and second life cycle. CLCA expands the system of study by including the possible direct and indirect effects what means to evaluate the consequences of recycling; for instance if the recycled material can replace virgin material or recycled material, if it can replace different types of material or no material at all (Ekvall and Weidema, 2004). Both methodologies are presented in more detail in **Chapter 3**.

In addition, nowadays there is an increased demand of resources from developing countries what combined with greater activity and speculation in commodities markets. This has resulted in high and volatile prices for many resources, which has also resulted recently in an increase of trade across the world. As an example, Figure 1.6 (Zils, 2013) illustrates these issues along the whole chain for a typical product in England but which can represent a common situation in Europe.

The product is used in England while the raw materials and manufactured products are extracted and imported from developing countries. After consumption, the valuable components are commonly recycled back outside of the country and these decisions are related to economic criteria rather than environmental criteria. Where waste moves across borders it can enable access to recycling or disposal options that are unavailable or more costly in the source country, meaning lower financial costs for waste management. Equally, trade can increase the opportunities to use waste as a valuable input to production, avoiding the need to draw on virgin resources and thereby enhancing the resource-efficiency of the economy as a whole (EEA, 2012). However, market effects have been traditionally excluded in LCA (James, 2012) but all these movements might involve GHG emissions not only due to transportation but also due to the consequences produced in the countries of import or export of raw materials and/or waste, what could reduce (or increase) the GHG benefits of recycling. In addition, to reduce the dependency of imports is one of the challenges of CE.

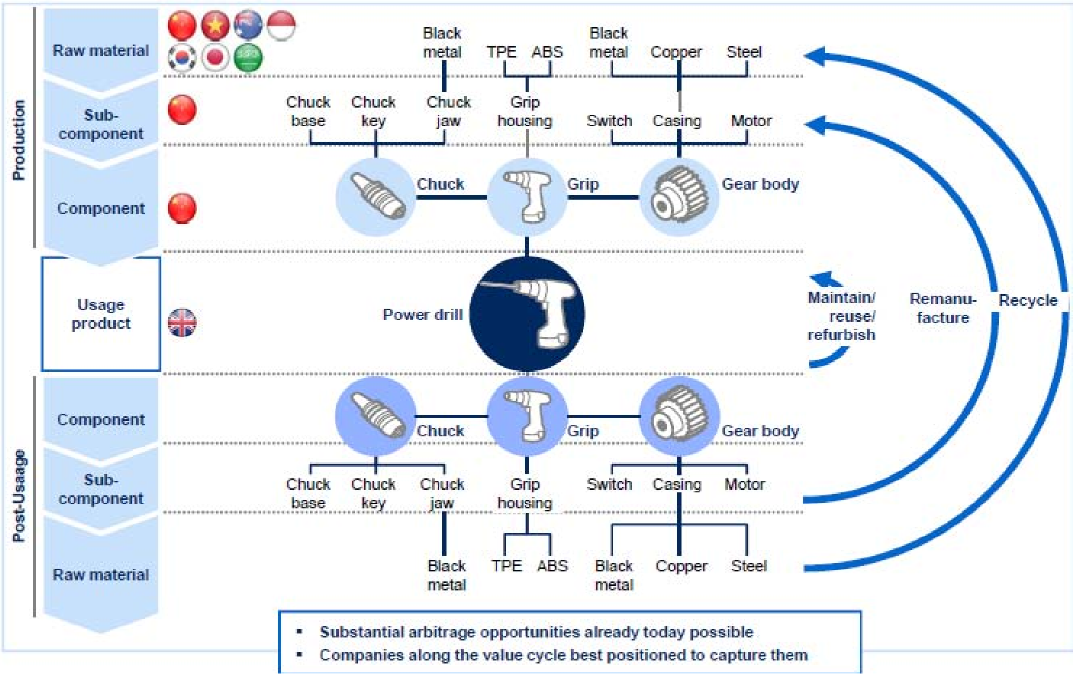

Figure 1.6: Complex and multi-layered bill of materials reflect geographical dispersion of today's manufacturing

Source: Zils, 2013

1.5.Spain as a case study

Spain is located on the Iberian Peninsula in southwestern Europe and since 1986 it is a member state of the European Union. Its mainland is bordered to the south and east by the Mediterranean Sea; to the north and north east by France, Andorra, and the Bay of Biscay; and to the west and northwest by Portugal and the Atlantic Ocean. In 2012 Spain's mixed capitalist economy was the fifth largest in the European Union (EU) with its per capita income slightly above the EU average (Euro Challenge, 2012). However, the growth of the industry sector has been slow due to the financial crisis. In fact, until 2008 it had been regarded as one of the most dynamic within the EU, attracting significant amounts of foreign investment, but the industry and construction sectors have been affected with decline in production and consumption (Euro Challenge, 2012). In 2012, Spain's economic structure was principally dominated by its service sector, which accounted for about 66.5% of its GDP. After the service sector, Spain's industry sectors had been the second largest contributor to the economy for about 15.6% followed by the construction sector for about 8.4%. Since Spain is a region of limited natural resources, hence with a weak primary economic sector which has a contribution to the gross domestic product (GDP) for about 2.5% (INE, 2013).

1.5.1. Resource consumption in Spain

Tracking the resource efficiency or economies is one way of understanding whether we are progressing towards sustainable development. An indicator often used for resource efficiency is the total amount of materials directly used by an economy (measured as domestic material consumption (DMC)) in relation to economic activity (measured as GDP). It provides an indication to whether decoupling between the use of natural resources and economic growth is taking place (BIO Intelligence Service, 2011; Sendra, 2008). Figure 1.7 presents the Spanish DMC between 2000 and 2011. However, one important aspect previously highlighted is that whilst industrialized countries have become more resource efficient in production terms, decreasing domestic raw material extraction have been compensated by resource imports from other world regions (see Figure 1.6). Industrialized countries are thus increasingly becoming dependent on imports of natural resources (Giljum, 2008; Kernegger and Giljum, 2009). Therefore, the indicators of total imports (TI) and total exports (TE) should be observed together with the DMC indicator to evaluate the resource consumption in Spain (see Figure 1.7Figure 1.8). DMC has increased up to 2007 and since this year has decreased in 44% with lower values than in 2000. DMC per citizen was over the European average but from 2007 started to decreased very quickly and in two years was under the European average and continued to decrease.

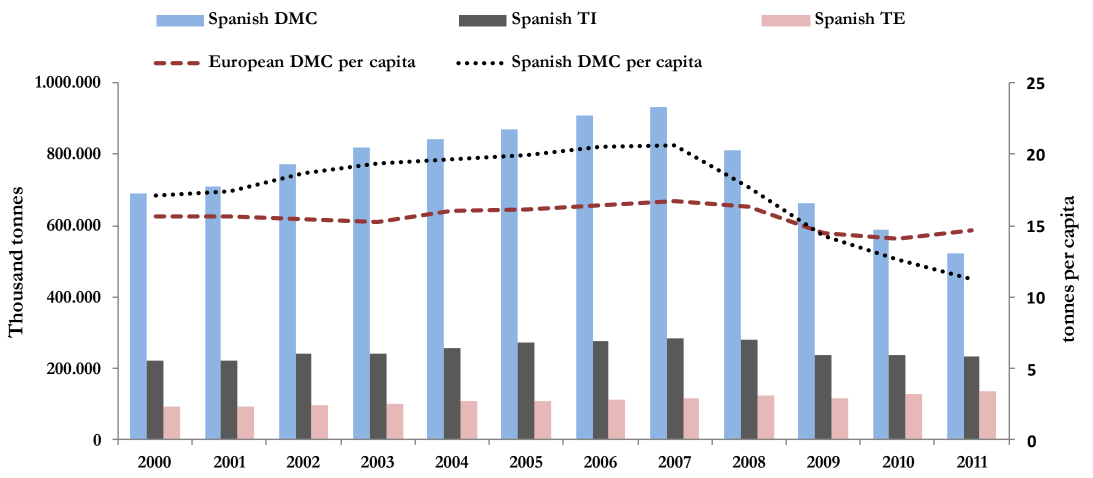

Figure 1.7: Domestic material consumption (DMC), total imports (TI) and total exports (TE) in thousand tonnes in Spain from 2000 to 2011in the right axis; and European and Spanish DMC per capita in the left axis

1.5.1.1. Resource trade in Spain

Both TI and TE remain more or less constant over the period, which suggest that we are consuming more imports than ever. In fact, trade plays an important role in the Spanish economy and in 2012 accounted for around 33% of the GDP (OECD, 2013). Besides, in recent years and as consequence of the crisis and the increase demand of resources from developing countries, there has been a change in the destination countries for exports from EU countries to non EU countries, and if in 2000 28.2% of exports were to non EU countries, in 2012 this percentage has increased up to 34% (Agencia Tributaria, 2012; Pajares, 2003).

1.5.2. MSW generation and MSW management in Spain

Regarding the MSW generation, the Spanish waste legislation framework has been developed in parallel to the European legislation since the first Spanish Waste Law in 1985 which forced municipalities to approach the problem of waste and to take measures for protecting the environment (EEA, 2013a). Several waste streams legislation have been implemented (Law 11/1997, 1997; Royal Decree 1383/2000; 2000; Royal Decree 208/2005; 2005) also with MSW plans for the periods 2000-2006 and 2008-2015 towards the reduction of waste in landfills and the maximizing of recovery and recycling to find a more circular system of materials. In July 2011 the new law (Law 22/2011, 2011) on waste and contaminated soils came into force, transposing the Waste Framework Directive (European Commission, 2008) into Spanish legislation and adopting all related targets and objectives within the waste hierarchy (EEA, 2013a).

Despite this wide waste legislation, however, the MSW generation increased from 1995 to 2007 and then started to decrease, probably to the crisis (see Figure 1.8). The Spanish generation per capita had been over the European average so in 1995, every European citizen generated 474 kg of MSW on average while 510 kg in Spain. This amount was increased at a maximum of 514 kg and 646 per person in 2003 in Europe and Spain, respectively, and decreased up to 492 kg per European citizen and 464 kg per Spanish citizen in 2012.

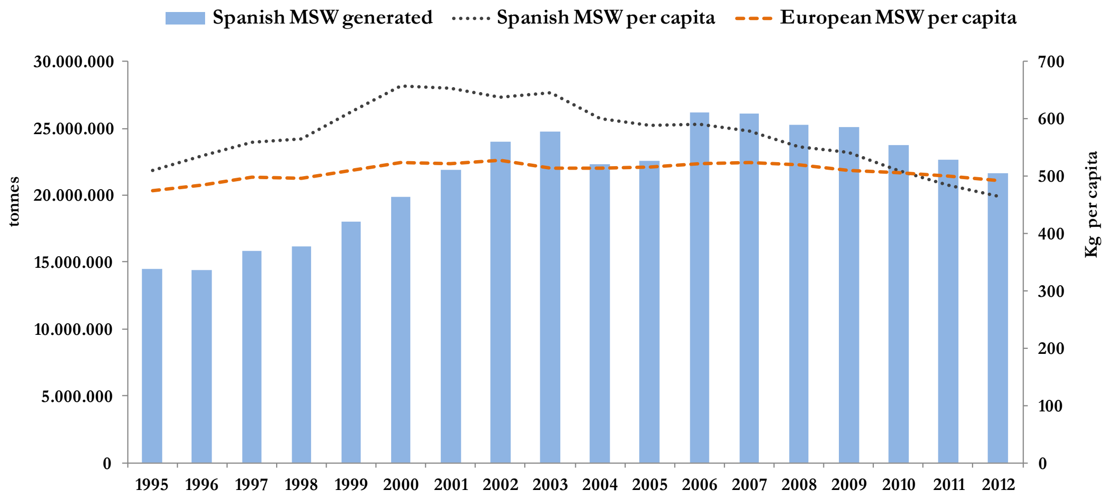

Figure 1.8: Spanish MSW generation per capita and European MSW generation per capita in kg per capita from 1995 to 2012 in right axis, and Spanish MSW generation in tonnes in left axis

In addition, if the MSW treatments are regarded (see Figure 1.9), in 2012 Spain had worst MSW management than the European average with 63% of MSW ended up in landfills, 17%

recycled, 10% incinerated and 10% sent to composting or anaerobic digestion (Eurostat, 2014b).

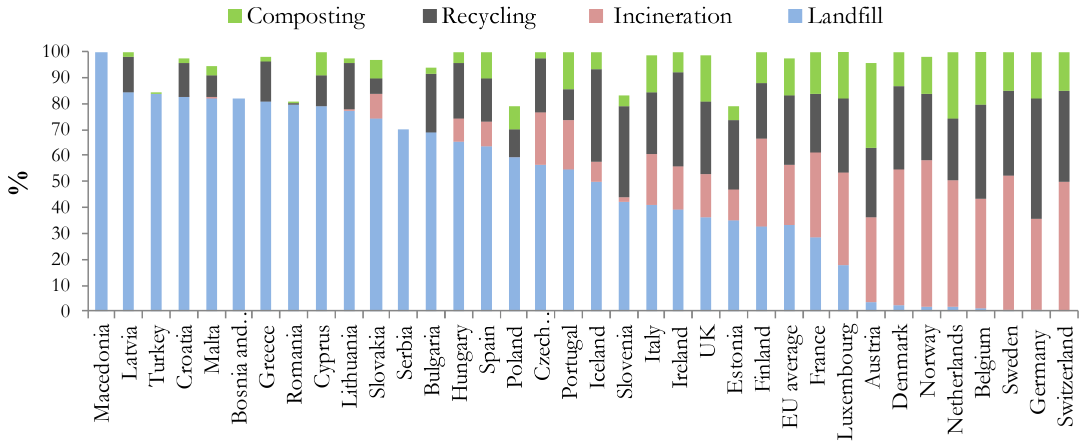

Figure 1.9: MSW waste treatments in 2012 in percentages (%) for EU countries (EU+27)

It can also be observed that there are important differences between the EU countries (Eurostat, 2014b): twelve countries have deposited in landfill more than 70% of its MSW; seven countries are between 70% and 50% (including Spain); seven countries are between 50% and 25%; and only nine countries have deposited in landfill less than 25% up to 0% for Switzerland.

1.5.2.1. Waste trade in Spain

Ever more waste is crossing EU borders, moving between Member States and to and from non-EU countries. Indeed, the growth in cross-border waste trade during recent years has been remarkable. Exports of waste iron and steel, and copper, aluminium and nickel from Member States doubled between 1999 and 2011, while waste precious metal exports increased by a factor of three and waste plastics by a factor of five, specially to Asia (EEA, 2012). Statistics of MSW trade in Spain are not aggregated and information is available by component of MSW such as plastic waste or glass waste but different reports have suggested that Spain has followed similar European trends of waste trade with more export of waste, especially to non EU countries (ANARPLA, 2013; EAA, 2012b; IVEX, 2010).

Chapter 2-

Motivation and objectives of this dissertation

2- Motivation and objectives of this thesis

This chapter presents the motivation that led the development of this thesis and presents the main goal and the specific objectives that may arise.

2.1.Motivation of this thesis

The waste management sector is expected **to reduce resource consumption jointly with the reduction of GHG emissions through the recirculation of waste**. Thus, accurate assessments are required in order to identify which are **the best strategies and opportunities** regarding waste quality, technology and waste market trends. **Spain has been chosen as a case study because it is an industrialized country and it could be representative for Europe**. Its evolution in recent years in terms of resource consumption, waste generation and waste management as well as trade of resources and waste followed the European trends with values over the European average. Moreover, the crisis has profoundly affected all sectors of industry, and recently opportunities from the waste management sector to create green jobs and contribute to the development of a CE have been pointed out (ARC, 2013).

The environmental impacts of MSW management have previously been addressed and several works were developed in Spain with different tools, objectives and perspectives which have relied in numerous scientific papers and reports. From a LCA perspective, studies have been focused on the evaluation of one life cycle stage (LC) (Ciroth et al., 2002a; Ciroth et al., 2002b; Iriarte et al., 2009; Colon et al., 2010; Martínez-Blanco et al., 2010; Rives et al., 2010; Inédit, 2011; Colon et al., 2012; Starr et al., 2012), on the comparison of waste management options or technologies (Güereca et al., 2006; Güereca et al., 2007; Cadena et al., 2009), or on assess waste management systems by including all LCs in the evaluation (Muñoz et al., 2004; Bovea and Powell, 2006; Bovea et al., 2010). Other works were focused on the evaluation of ecodesign of packaging products (González-García et al., 2011; Rieradevall et al., 2005).

Moreover, in Catalonia, a recent work was conducted to evaluate the GHG emissions from waste (Vicent and Gabarrell, 2005). In this study, the necessity of reduce waste in landfills and the opportunities to tackle climate change through proper waste management were pointed out. Recently, more focus have been started to put on this relationship between waste and climate change and only a few papers and reports have been published (Gabarrell et al., 2010; Rubio-Romero et al., 2009; FFA, 2012; Villalba et al., 2012; Starr et al., 2014). The possibility of quantifying GHG emissions as an important element in understanding the problem of waste and GHG emissions is highlighted (Gabarrell et al., 2010; FFA, 2012).

Despite these works, a tool that calculates the GHG emissions of the MSW management is necessary as a first step in the understanding of the problem and taking into account different MSW management options and treatments. Moreover, accurate tools and GHG accounting can help in the decision's process. In this regard, several models based on the LCA methodology are already available but most of them have been developed in North of Europe or North America using local data (Gentil et al,. 2010; Den Boer et al., 2007; Eriksson and Bisaillon, 2011; Tunesi, 2011). The GHG quantifications depend on its context or local specificities what prevents a consistent generalization of GHG results (Laurent et al., 2014). Therefore, with the aim of supporting Spanish waste management policies and identify the potentialities of resource savings and GHG savings, **it is necessary to concern a methodological framework and to develop a tool that includes local inventories to assess waste management systems and strategies in Spain**.

Although LCA is recognized as a valuable method for assessing the avoided impacts of recycling systems, there is still debate over methodologies. Moreover, **regarding the current situation of international trade and global markets, the CLCA approach by taking into account the market effects seems more appropriate to evaluate the GHG impacts of recycling**. Nevertheless, the complexity of the current situation also requires tracking and making visible material flows of raw materials, products and waste so that the international trade or the direct and indirect effects can be more readily identified. This is the main purpose of MFA. So, the integration of MFA with CLCA could be a good methodological framework to consider and to evaluate the whole picture.

In this regard, MFA studies to tackle material flows to and from Spain were evaluated in previous studies (Sendra, 2008; Hoque, 2012) and some were focused on waste flows but at regional level (Fragkou, 2009; Font et al., 2012). In addition, there are a few studies addressing the recovery of organic matter fraction in MSW to produce compost in substitution to fertilizers (Martínez-Blanco et al., 2009; Martínez-Blanco et al., 2013; Quirós et al., 2013; Martínez-Blanco et al., 2014), but there is no study which evaluates the GHG quantification of recycling processes **by CLCA approaches nor by the integration of MFA with CLCA.** These studies are necessary to have accurate GHG quantifications to decision support processes, waste management strategies and future scenarios in the current context.

2.2.Objectives of this thesis

The goal of this thesis is firstly to **calculate and evaluate the GHG emissions from municipal solid waste (MSW) management in Spain**. **Secondly a special focus is put on the GHG emissions of recycling considering the market, the quality, the technology and international trade.** The purpose is to evaluate the potentiality of the waste sector to contribute to climate change mitigation and resource savings.

In order to achieve this main aim, several goals are outlined:

1. To present a new tool named CO2ZW® to quantify the GHG emissions from local, regional or national MSW management from an IPCC and LCA approach by including direct, indirect and avoided GHG emissions (**Chapter 4**)
2. To determine the methodological advantages and disadvantages for GHG quantifications of the CO2ZW® in relation to other available tools (**Chapter 4**)
3. To determine what are the most sensitive parameters for the GHG emissions regarding waste data and composition, waste collection, waste treatment and waste recycling (**Chapter 4**)
4. To evaluate with the CO2ZW® tool the MSW management system in Spain and evaluate alternative scenarios to reduce the GHG emissions (**Chapter 4**)
5. To provide a methodology framework for including the market effects on the GHG quantifications of recycling (**Chapter 5**)
6. To evaluate the usefulness of this methodology framework to consider the GHG emissions derived of the international trade (**Chapter 5, 6 and 7**)
7. To verify the applicability of the developed methodology framework by applying to paper and cardboard as representative of a resource coming from renewable sources (**Chapter 5**)
8. To verify the applicability of the developed methodology framework by applying to aluminium old scrap as representative of a resource coming from ore sources (**Chapter 6**)
9. To verify the applicability of the developed methodology framework by applying to plastic waste as representative of a resource coming from fossil sources (**Chapter 7**)
10. To quantify the in-use stock aluminium products and plastic products accumulated in Spain over the years studied (**Chapter 6 and 7**)

11. To calculate future waste generation due to the in-use stock achieving its EOL (**Chapter 6 and 7**)
12. To determine whether the methodological framework is useful to define future waste management and resource consumption strategies (**Chapter 5, 6 and 7**)

Chapter 3-

Methodologies

3- Methodologies

This section presents the main methodological aspects that have been involved in the development of the thesis. First of all, main considerations of the methodology from IPCC to calculate the direct GHG emissions from waste management are explained. Secondly, main aspects and types of MFA methodology are described. Thirdly, main phases regarding LCA are presented and as this thesis is focus in CLCA, definitions and differences between ALCA and CLCA are also explained. Figure 3.1 explains graphically which methodologies are used in each Chapter.

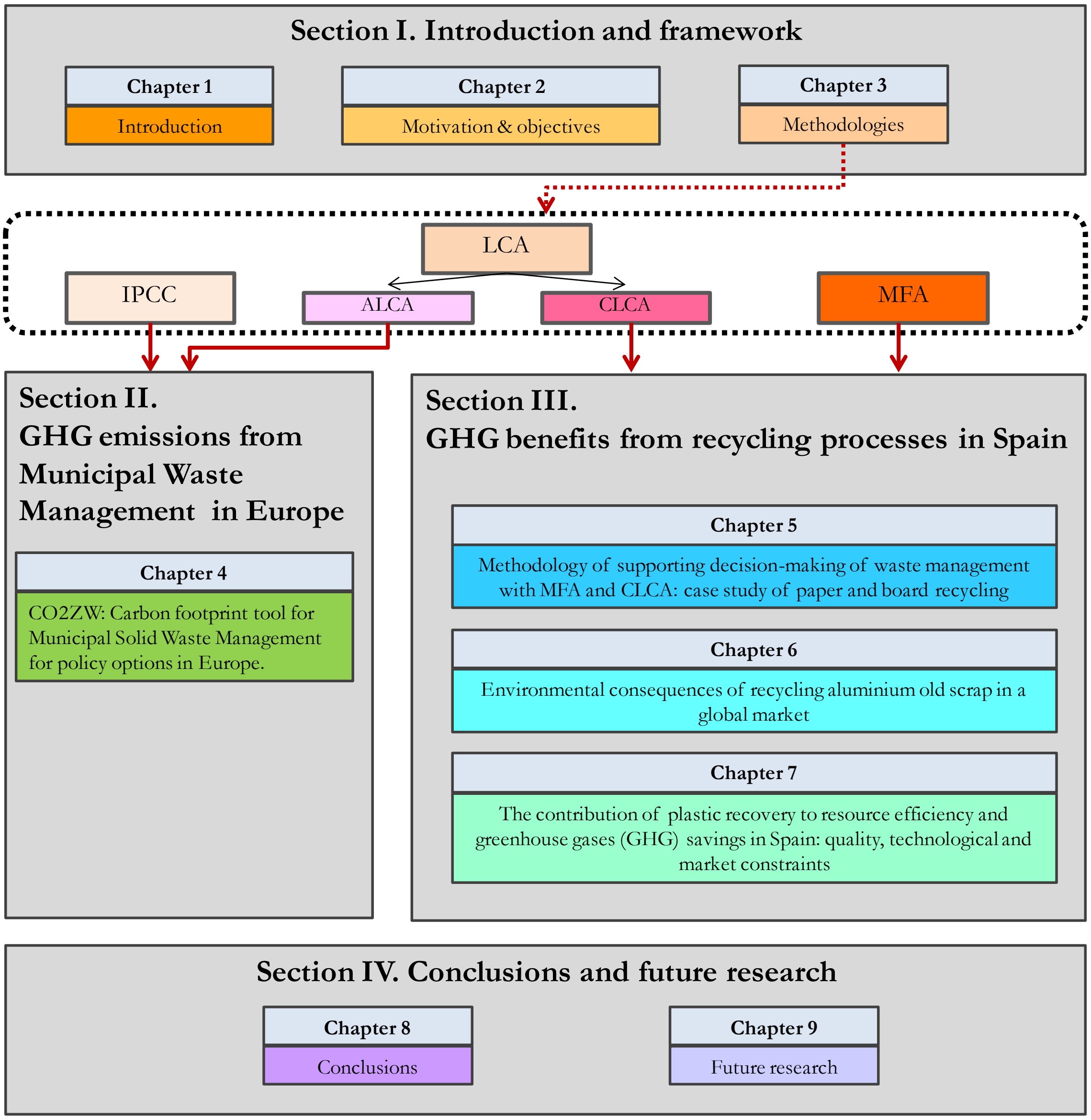

Figure 3.1: Methodologies used in this thesis

3.1.Direct GHG emissions: IPCC and LCA

According to IPCC, GHG are defined as "*gases that absorb radiation at specific wavelengths within the spectrum of radiation (infrared radiation) emitted by the Earth's surface and by clouds. Gases in turn emit infrared radiation from a level where the temperature is colder than the surface. The net effect is a local trapping of part of the absorbed energy and a tendency to warm the planetary surface. Water vapour (H_2O), carbon dioxide (CO_2), nitrous oxide (N_2O), methane (CH_4) and ozone (O_3) are the primary greenhouse gases in the Earth's atmosphere (IPCC, 2014b)".*

Direct GHG emissions are produced in the LCs of collection, transport, sorting, recycling, incineration, biological treatments and landfill (see Figure 1.4). In this thesis direct GHG emissions were calculated through combined application of the IPCC methodology and/or the LCA methodology. In all cases GWP values from the 4th assessment report (IPCC, 2007) were used as the 5th assessment was recently published (May 2014). In addition, the IPCC methodology considers that the CO_2 emissions from biomass sources including the CO_2 in landfill gas, the CO_2 from composting, and CO_2 from incineration of waste biomass are not taken into account in the GHG inventories as these are covered by changes in biomass stocks in the land-use, land-use change and forestry sectors (Bogner et al., 2007). In order to be consistent, the CO_2 emissions from biogenic sources in waste management were not taken into account in this thesis.

In **Chapter 4** more detail information of the methodology followed and adapted from the IPCC protocol and LCA methodology can be found. The indirect GHG emissions and avoided GHG emission are calculated through the methodology of LCA which is explained in following section 3.3.

3.2. Material accounting: MFA

MFA is based on accounts in physical units (usually in terms of tonnes) quantifying the inputs and outputs of those processes and can be defined as *"a systematic assessment of the flows and stocks of materials within a system defined in space and time (Brunner and Rechberger 2004)"*. The subjects of the accounting are chemically defined substances (for example, carbon or carbon dioxide) on the one hand and natural or technical compounds or 'bulk' material (for example, coal or wood) on the other hand (Brunner and Rechberger 2004).

It uses the principle of mass balancing to analyze the relationships between material flows (including energy), human activities (including economic and trade developments) and environmental changes. In general terms, MFA studies comprise the following three-step procedure: (a) definition of the system, (b) quantification of the overview of stocks and flows, and (c) interpretation of the results. All three steps involve a variety of choices and specifications, each of which depends on the specific goal of the study to be conducted (Van der Voet, 2002).

The type of MFA best suited for any particular case depends on the issues of concern and the questions being addressed. According to (Bringezu and Moriguchi, 2002), one can distinguish two broad groups comprising three types of analysis each. The focus can be on the flow of a particular item such as a chemical substance, material or product within certain firms, sectors or regions (Type I); or the focus can also have a larger scope such as a firms, industrial sector or geographic region associated with substances, materials or products (Type II) (Bringezu and Moriguchi, 2002). MFA can be applied to a wide range of economic, administrative or natural entities, studying the flows of materials within the global economy or the economy of a region or country, within an economic activity within a city, river basin or ecosystem, a firm or a plant (OECD, 2008).

The framework of the MFA and its potential to evaluate the effects and opportunities of different waste management policies and resource use policies can be observed in following Figure 3.2.

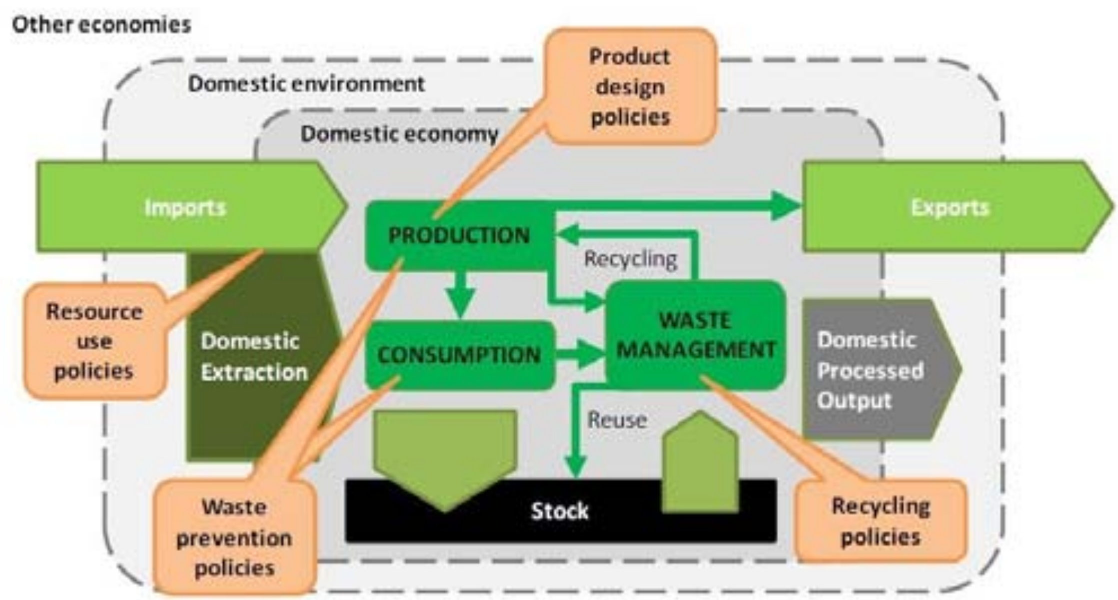

Figure 3.2: An overview of where different resource efficiency and waste management policy areas contribute to materials flows

Source: BIO Intelligence Service, 2011

MFA have been applied in **Chapters 5, 6 and 7**. More detail information of the methodology followed, source data, assumption and explanations can be found in the mentioned chapters and in the Supplementary Information related to the chapters.

3.3. Environmental accounting: LCA

3.3.1. General overview of LCA

The LCA is one of the new methodological tools used to assess the sustainability of products, processes and services. The Society for Environmental Toxicology and Chemistry (SETAC) defined LCA as "*an objective process to evaluate the environmental burdens associated with a product, process, or activity by identifying energy and materials used and wastes released to the environment, and to evaluate and implement opportunities to affect environmental improvements. The assessment includes the entire life cycle of the product, process or activity, encompassing extracting and processing raw materials; manufacturing, transportation and distribution; use, re-use, maintenance; recycling, and final disposal (SETAC, 1993)*".

LCA is a tool to assess the environmental impacts and resources used throughout a product's life cycle, i.e., from raw material acquisition (cradle), via production and use phases, to waste management (grave) (ISO, 2006). There are four phases in an LCA study which are summarizes in Figure 3.3.

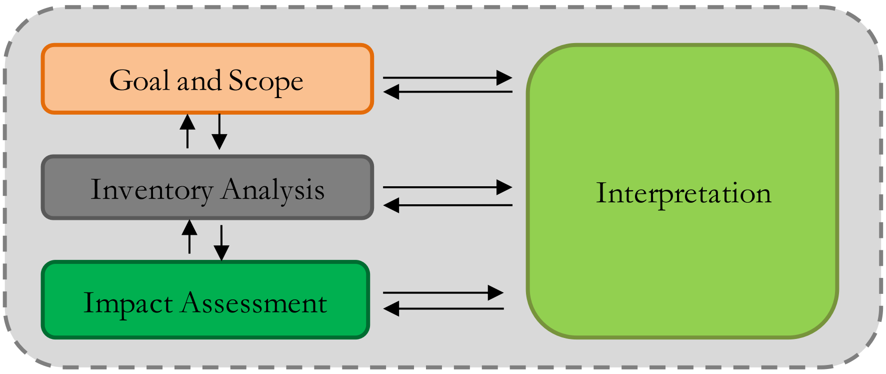

Figure 3.3: The phases of an LCA according to ISO 14040 (2006)

First, the goal and scope includes the reasons for carrying out the study, the intended application, and the intended audience (ISO, 2006). It is also the place where the system boundaries of the study are described and the functional unit (FU) is defined (i.e., one tonne of MSW collected for its treatment). Second, the life cycle inventory (LCI) is an inventory of input/output data with regard to the system being studied. It involves collection of the data necessary to meet the goals of the defined study (ISO, 2006). The result from the LCI is a compilation of the inputs (resources) and the outputs (emissions) from the product over its life-cycle in relation to the FU (Finnveden et al., 2009). Third, the life cycle impact assessment (LCIA) is expected to evaluate the potential environmental impacts transforming hundreds of inventory inputs and outputs into a few impact categories, thus attempting to understand these impacts. It is aimed at understanding and evaluating the magnitude and significance of the potential environmental impacts of the studied system (ISO, 2006). Finally, the fourth phase is the interpretation, in which the results from the previous phases are evaluated in relation to the goal and scope in order to reach conclusions and recommendations (ISO, 2006).

The impact categories considered in this dissertation are Global Warming Potential (GWP) and Cumulative Energy Demand (CED). The description of these impact categories are shown in Table 3.1.

Table 3.1: Environmental impact categories description

Impact category	Description
GWP Global warming potential	Climate change is related to emissions of GHG into air. The characterization model as developed by the IPCC. Factors are expressed as for time horizon of 100 years. Unit: kg CO_2 eq.
CED Cumulative energy demand	It aims to investigate the energy use throughout the life cycle of a good or a service. This includes the direct as well as the indirect uses. The characterization factors were given for the energy resources divided in: non renewable, fossil and nuclear, renewable, biomass, wind, solar, geothermal and water. Unit: MJ

3.3.2. Approaches in LCA

In the core of LCA studies, two main approaches are distinguished: the attributional and the consequential approach. ALCA is defined as *"system modelling approach in which inputs and outputs are attributed to the FU of a product system by linking and/or partitioning the unit processes of the system according to a normative rule (Sonnemann and Vigon, 2011)"* while CLCA is defined as *"system modelling approach in which activities in a product system are linked so that activities are included in the product system to the extent that they are expected to change as a consequence of a change in demand for the FU (Sonnemann and Vigon, 2011)".*

CLCA describes how environmental impacts will change in response to possible decisions (Ekvall and Weidema, 2004). The different focuses of ALCA and CLCA are reflected in several methodological choices (Tillman, 2000) which are summarizes in Table 3.2. One of the most important is the choice between average and marginal data used in the LCI. Marginal data means identify which technology or process is affected due to a change in demand. For example, several technologies contribute to generate the electricity mix of a country such as coal, nuclear or wind, but not all technologies are equally able to contribute to a change in demand (i.e., an increase in electricity demand) since there are some constraints that can limit the capacity of response. For example, natural gas technology would easily adapt to a higher electricity demand while nuclear technology would not, thus in this case, natural gas should be a marginal technology. As opposite, average data means that all technology would be equally affected by a change in demand, and would contribute proportionally to a change in demand. ALCA uses average data and CLCA uses marginal data.

The other one choice is related to the allocation problem. In ALCA, the co-production is treated by applying allocation factors what means divided the LCI between the different co-products, while in CLCA the co-production is treated by expanded the system under study to include additional life cycles and products affected by a change of physical flows in the respective life cycle and the consequences are evaluated.

The **ALCA** approach is used in **Chapter 4** because this approach is more useful for identifying opportunities for reducing emissions within the life cycle or supply chain, through improvements in processing efficiency or new technologies and for quantifying actual emissions from the consumption of goods and services.

However, a **CLCA** approach is basically concerned with identifying the cause and effect relationship between possible decisions and their environmental impacts (Mathiesen et al., 2009). So, in this thesis, we have followed this perspective as it is more appropriate to assess the consequences of recycling in the sense that recycling implies that the waste is recirculate to the industrial process and thus, other process are displaced (i.e., raw material production). In addition, it is more suitable to evaluate the consequences and the effects in other countries and economies. **CLCA** have been applied in **Chapters 5, 6 and 7**. More detail information of the methodology followed, source data, assumption and explanations can be found in the mentioned chapters and in the Appendix related to the chapters.

3.3.3. SimaPro

The software program SimaPro 7.3.3 was used as the LCA modelling and analysis tool. SimaPro is a well-known, internationally accepted and validated tool and since its emergence in 1990, it has been used in a large number of LCA studies by consultants, research institutes, and universities. In **Chapter 5**, the SimaPro tool is applied in order to estimate the GWP and CED of recycling waste paper. In **Chapters 6 and 7**, the SimaPro tool is applied to estimate the GWP of aluminium old scrap recycling and waste plastic recycling.

Table 3.2: Methodological differences between ALCA and CLCA *(based on Brander et al., 2008)*

	ALCA	CLCA
System boundary	The processes and material flows directly used in the production, consumption and disposal of the product.	All processes and material flows which are directly or indirectly affected by a marginal change in the output of a product (e.g. through market effects, substitution, use of constrained resources etc).
LCI	Average data	Marginal data
Allocation	Allocation between the system that generates the waste and to the one that uses the secondary good.	System expansion to reflect the consequences of the recycling.
Market effects	Not consider the market effects of the production and consumption of the product	Consider the market effects of the production and consumption of the product.
Uncertainty	Low uncertainty because the relationships between inputs and outputs are generally stochiometric	Medium/high uncertain because it relies on models that seek to represent complex socio-economic systems that include feedback loops and random elements

SECTION II

GHG emissions from Municipal Solid Waste

Chapter 4-

CO2ZM: Carbon footprint tool for municipal solid waste management for policy options in Europe: inventory of Mediterranean countries

4- CO2ZW®: Carbon Footprint Tool for Municipal Solid Waste Management for Policy Options in Europe. Inventory of Mediterranean Countries

based on the following paper: Eva Sevigné Itoiz, Carles M. Gasol, Ramón Farreny, Joan Rieradevall and Xavier Gabarrell (2013). CO2ZW®: Carbon footprint tool for municipal solid waste management for policy options in Europe: inventory of Mediterranean countries. Energy Policy, 56, 623-632

The CO2ZW® tool is available from www.sostenipra.ecotech.cat after previous registration.

Abstract

In the frame of the European project titled "Zero Waste" (1G-MED08-533), a tool has been developed called CO2ZW® for estimating the GHG emissions for the management of MSW at the municipal, regional or national levels with small amounts of input data. The objective of this paper is to demonstrate that the CO2ZW® tool allows us to inventory and monitor GHG emissions from MSW following the IPCC guidelines for national inventories and the principles of LCA. The CO2ZW® tool includes the key stages and parameters for calculating GHG emissions and includes several advantages regarding the implementation of the default values of the Mediterranean European countries, an improvement in accessibility (online free access) and two approaches for calculating GHG emissions from landfills. The results of this paper show that for countries with medium and high rates of deposition, implementation of the European policies limiting waste in landfills can contribute to mitigate climate change in a remarkable way. With the CO2ZW® tool, it is possible to evaluate waste management choices depending on waste management infrastructures and waste policies, along with the quantification of GHG emissions from MSW management, which is essential to guide waste policy options and climate change solutions.

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Reference link

SECTION III

GHG benefits from recycling processes in Spain

Chapter 5-

Methodology of supporting decision-making of waste management with MFA and CLCA: case study of waste paper recycling

5- Methodology of supporting decision-making of waste management with MFA and CLCA: case study of waste paper recycling

based on a manuscript by: Eva Sevigné Itoiz, Carles M. Gasol, Joan Rieradevall and Xavier Gabarrell.

Abstract

LCA studies on waste management have been promoted as they can help policy makers choose the best environmental options. However, as this study reflects, the increasing globalization of raw materials and waste makes the optimization of waste management strategies and policies quite challenging. Therefore, new approaches are needed in order to identify the consequences of markets on the current waste management systems. This paper concentrates on market effects on the quantification of GHG emissions of recycling processes. The aim is to generate a comprehensive assessment of GHG emissions as a consequence of increasing the amount of material collected for recycling. Consequential life cycle assessment (CLCA) is an effective methodological framework for addressing GHG quantifications within market, but to properly perform and assess all of the market links between raw materials and waste, it is necessary to determine the cause-effect chains made up of physical flows. Thus, we propose integrating the methodologies of MFA (MFA) and CLCA. We applied these methodologies to the paper and cardboard recycling system in Spain. The GHG results varied between -36 kg CO_2 eq. and -317 kg CO_2 eq. per ton of waste paper collected, depending on the quantity of waste paper exported and the source of marginal pulp considered. The cumulative energy demand (CED) was also calculated as complementary indicator. Similar trends as for GHG emissions were obtained. The future GHG quantifications should be based on the flows described by MFA analysis and should be quantified using CLCA because methodologies that accurately account for GHG are necessary for decision-making.

5.1. Introduction

Waste recycling is thought to offer some of the most significant GHG emissions savings in waste management practices (Friedrich and Trois, 2011). Thus, the recycling objectives set by waste policies (i.e., Directive 2004/12/EC (European Commission, 2004)) are based on the notion that increase collection rates will increase recycling rates and thereby increase the GHG savings. However, the rationale for and understanding of the consequences of these measures are often incomplete, and attempts to promote recycling levels beyond the market-clearing level must address the consequences and the importance of market behavior (Blomberg and Söderholm, 2009). Similarly, recycling may be largely driven by government objectives to divert waste from landfills. Therefore, the market for recycled material is only partially connected to market drivers because some market participants are legally obligated to participate and are not motivated by profit maximization (Angus et al., 2012). This creates interesting dynamics, and even in recessions, the supply of recycled materials will continue to flow into the market despite a lack of demand (Angus et al., 2012). Therefore, the supply and demand of waste can be in disequilibrium, and in some cases, the export of waste can serve as a significant adjustment mechanism for such imbalances (Stromberg, 2004).

For example, the Chinese demand for paper products has grown by approximately 10% per year since 1995 (FAO, 2012), accounting for more than half of the worldwide increase in demand (WRAP, 2007). Conversely, in recent years, in Europe and North America, the production and consumption of paper products have decreased, while waste paper collected has increased (FAO, 2012). Therefore, on one hand China is highly dependent on the importation of fibers to produce sufficient pulp for its paper production (NEP, 2009) and on the other hand, there has been an excess of supply of waste paper in Europe and North America which has resulted in large flows to Asia (NEP, 2009). This may cause a possible imbalance not only economic but also environmental (OECD, 2000, Rodrigue et al., 2001, ITENE, 2008) and the consequences should be considered. Moreover, due to globalization, the local consumption of goods and resources in European countries depends increasingly on countries outside Europe and the local use of resources in the European countries is stabilizing through increased resource use in other parts of the world (Reinhard and Zah, 2009). In the case of virgin pulp, imports to Europe have increased from 33% in 1995 up to 40% in 2011 while the European production has stagnated (FAO), what suggest that most competitive virgin pulp is displacing higher-cost fibers in the market, such as the European fibers (Hawkins Wright, 2011). Thus, it is essential to consider how waste and raw material fits into a bigger economic picture (Gadner, 2013), one that suggests that market mechanisms should be included in the GHG quantifications of recycling and considered when making waste policies.

LCA studies have been promoted to provide an informed and science-based support to a more environmentally sustainable decision-making in waste management (JRC, 2011) and several LCA studies have been published on MSW management systems (Björklund and Finnveden, 2005; Cleary, 2009; Rives et al., 2010; Lazarevic et al., 2010; Wang et al., 2012). Many of them focus exclusively on the internal flows of a production system, without considering the effects that the system and its final flows may have on other related economic systems. This perspective, known as the attributional LCA approach (ALCA), has been predominant in life cycle thinking, but this perspective does not account for the consequences that increased waste collections or increased virgin pulp imports may generate on GHG emissions. In this regard, a more recent approach, named CLCA, includes additional life cycles and products affected by a change of physical flows in the respective life cycle (Reinhard and Zah, 2009). The consequential approach seeks an environmental assessment that takes the evaluation a step further, in order to analyze how physical flows and, therefore, environmental burdens, may vary in response to changes with market implications in a specific life cycle beyond the foreground system (Vazquez-Rowe et al., 2014). In this regard, CLCA is a more effective methodology for address the GHG quantifications because it provides a modeling approach that seeks to describe the consequences of decisions (i.e., to increase waste paper collection) when processes are linked via market mechanisms (Weidema et al., 2009) and allows the limits of the system to be expanded beyond national boundaries.

However due to globalization, the relations between the production, use and waste are each time more complex and geographically dispersed. Thus, in order to properly perform and assess all the market links between raw material, product and waste, it is essential in first place, to establish the cause-effect chains made up of physical flows (Sandén and Karlström, 2007). In this sense, MFA has demonstrated its potential to evaluate the interaction between material flows, economy and the environment. Besides, with MFA dynamic perspectives, it is also possible to observe variability over time and determine possible changes in trends in raw materials and waste markets (Brunner and Rechberger, 2004; Moriguchi, 2009; Mathieux and Brissaud, 2013). Therefore, this paper proposes to integrate the methodologies of MFA and CLCA for assessing market effects on the GHG quantifications of recycling activities with the aim of helping to make better decisions. On one hand, this integration would help assessing the amount of the waste generated and consumed in one country, the trade along the whole production chain or the origin and destination of the products traded. On the other hand, the integration would also help evaluating the consequences in previous years, such as if the increase of waste collection implied in an increase of recycling or if an increase of consumption relied in an increase of production or whether these changes have consequences outside of studied area. With all this information it is possible to project more realistic scenarios to assess the future consequences and quantify the GHG emissions derived. In this study, we have applied the integration of both methodologies to evaluate the Spanish paper and cardboard recycling system, and we evaluate the increase of waste paper collection in Spain.

5.2.Methodology

The methodology proposed in this study consists of two steps: conducting a dynamic MFA to monitor trends and changes in the dynamics of raw materials, products and waste, and integrating MFA results in consequential LCI modeling to quantify the GHG of recycling. In the following sections, the methodologies followed for the quantification of flows and stocks (5.2.1) and the GHG savings (5.2.2) are explained.

5.2.1.MFA

5.2.1.1. Scope and system boundaries

In this study, MFA was used in its simplest form, and only the material mass flows were studied. The temporal and spatial boundaries were the years 2006 to 2011 and Spain, respectively. The life cycle of paper and board (PB) is composed of eight principal LCs: wood crops [A], wood chip production [B], virgin pulp production [C], PB manufacturing [D], PB product production [E], use [F], waste management (collection and sorting) [G], and recycling [H]. Every LC produces products that are classified as follows: wood [a], wood chips [b], virgin pulp [c], PB papers [d], PB products [e], waste paper [f], refuse waste paper [g], recovered paper [h], and recovered fiber [i]. Additionally, we considered the auxiliary materials (a. materials) [j] necessary for PB paper production. Figure 5.1 presents the system boundaries of the Spanish PB life cycle and shows every flow that has to be determined, including importations and exportations and losses. Some of the products were classified into several subtypes which are represented in Figure 5.1 as rectangles with discontinuous lines. For example, virgin pulp production [c] is classified as chemical pulp, mechanical pulp and semi-chemical pulp. Capital letters in brackets refer to LCs (i.e., virgin pulp production [C]) while lower case letters refer to material flows (i.e., virgin pulp (c)). In addition, in the Appendix A, Table A.1 summarizes the classification and indices used in this study.

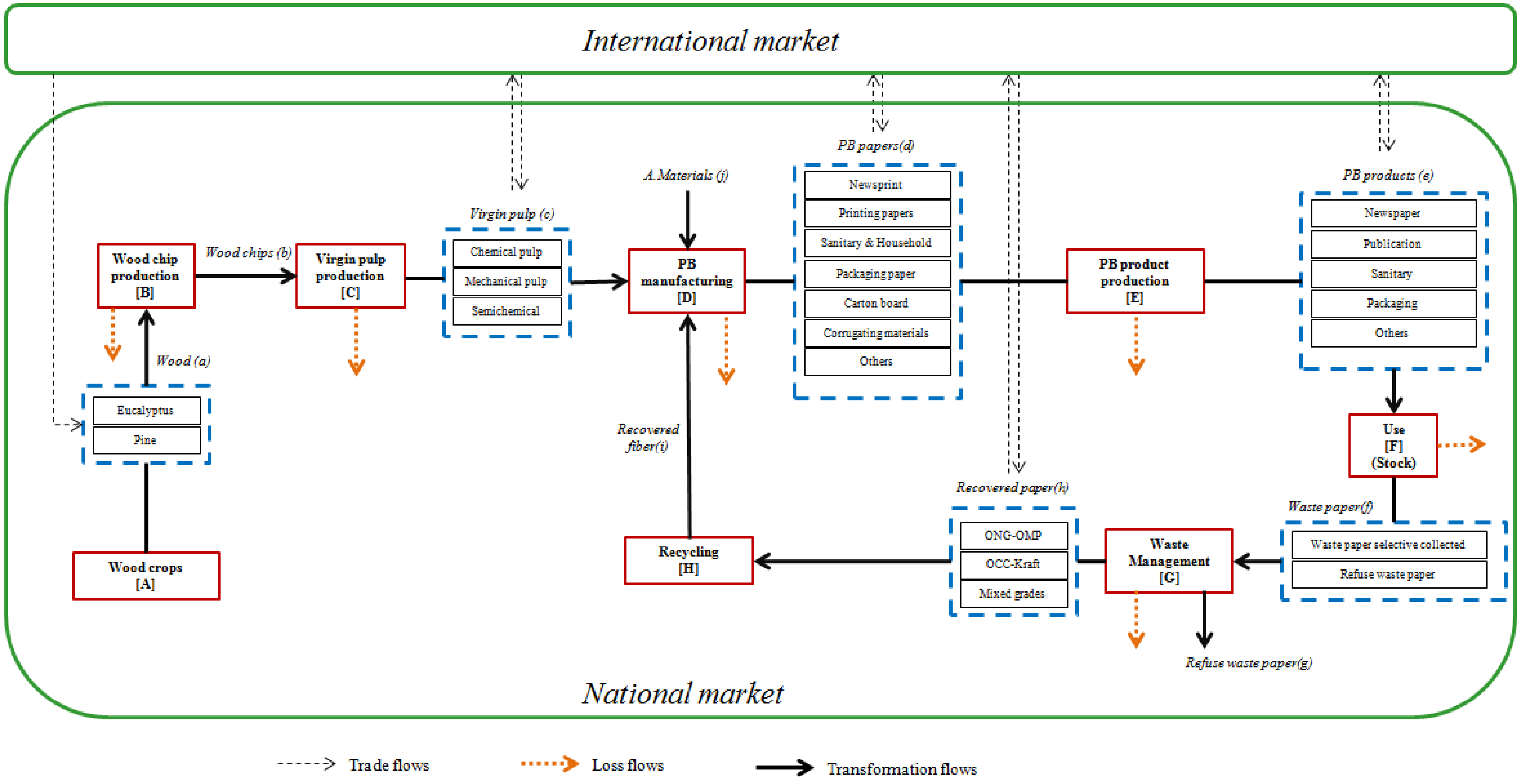

Figure 5.1: Spanish PB life cycle system boundaries

5.2.1.2. Accounting methods of flows and stocks

The system under study concerns only material flows and their calculation are based 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 inFigure 5.1. The total input of each process should be equal to production, stock and loss of each process. Production of each process plus imports minus exports should be equal to consumption, and should be equal to the input of the next process. Figure 5.2 summarizes the mass balance, where LC=LC; i=indicator for LC; j=indicator for the studied years; $INPUT_{i,j}$=product demanded by LC i in year j; $production_{i,j}$=product produced in LC i in year j; $stock_{i,j}$= product stocked in LC i in year j; $Loss_{i,j}$=product discarded from LC i in year j; $Import_{i,j}$=product imported and generated from LC i in year j; $Export_{i,j}$=product exported and generated from LC i in year j; and $consumption_{i,j}$=product consumed from LC i in year j.

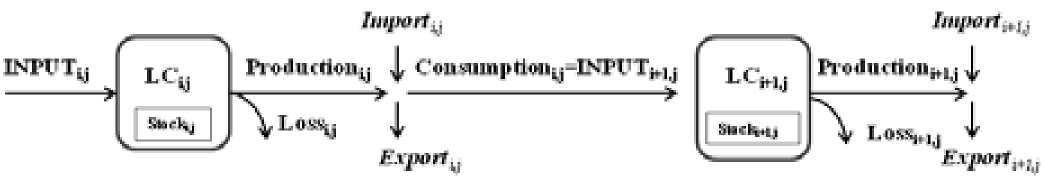

Figure 5.2: Schematic diagram of mass balance for each LC

Each flow is calculated in three different ways depending on the data availability: directly based on statistics, combining statistics with coefficients, and deducing by mass balance. Details on the data collection, sources and assumption explanations are provided in the Appendix A.

5.2.1.3. Performance indicators

In Spain waste paper is divided into groups that are either selectively collected waste paper – here referred to as waste paper selective collected – or refuse waste paper, which is waste paper that is contained within the refuse fraction (see Figure 5.1). Through sorting plants, waste paper selective collected is classified as recovered paper. To evaluate the performance of waste paper collection, sorting and recycling, the following indicators were defined: recovered rate (RR), recovered utilization rate (RUR), recovered import rate (RIR) and recovered export rate (RER). The indicators are calculated using the following equations (Eqs. 5.1-5.4).

$$\text{Recovery Rate (RR)} = \frac{\text{waste paper selective collected}}{\text{PB products consumed}} \cdot 100 \qquad \text{[Eq. 5.1]}$$

$$\text{Recovered Utilization Rate (RUR)} = \frac{\text{recovered paper consumed}}{\text{PB product production}} \cdot 100 \qquad \text{[Eq. 5.2]}$$

$$\text{Recovered Import Rate(RIR)} = \frac{\text{recovered paper imported}}{\text{recovered paper produced}} \cdot 100 \qquad \text{[Eq. 5.3]}$$

$$\text{Recovered Export Rate(RER)} = \frac{\text{recovered paper exported}}{\text{recovered paper produced}} \cdot 100 \qquad \text{[Eq. 5.4]}$$

5.2.2. CLCA

5.2.2.1. Scope and system boundaries

In consequential modeling, recycling is considered a treatment activity in which the material for treatment (i.e., recovered paper) must be treated before it is converted into a by-product (i.e., recovered fiber) that can displace another product or process through system expansion (Weidema et al., 2009; Schmidt and Dalgaard, 2012). In this study we considered that recycling recovered paper avoids virgin pulp production. Because recovered fibers are of lower quality than virgin fibers, we argue that the fibers would be replaced with short virgin fibers from hardwood species. Once the process avoided by recycling is identified, the second key issue in consequential LCI modeling is the identification of the affected technology, also referred as marginal (Weidema et al., 2009). This marginal technology appears to be the most sensitive to changes in market demand (i.e., the type of virgin pulp production that is affected by recycling). The geographical location of this technology does not have to be in the same country as the studied system (here, Spain) (Schmidt and Dalgaard, 2012). Reinhard et al. identified the marginal supply of wood pulp by assessing the scale and time horizon, the limits of the market, the trends in the volume of the market and the competitiveness of different suppliers following the guidelines of Weidema et al. (Weidema et al., 2009; Reinhard et al., 2010). The assessment concludes that the virgin wood pulp market is global and increasing so the marginal supplier is to be identified on the global market. Concerning the hardwood pulp, Eucalyptus plantation forests in Brazil or Indonesia represent the marginal supply of bleached hardwood kraft pulp (BHKP) (Reinhard et al., 2010) (also referred to as bleached eucalyptus kraft pulp (BEKP)). This type of pulp is the most sensitive to supply and demand dynamics for virgin pulp in the global market. Therefore, in this study, the BEKP production in Brazil was chosen as the marginal virgin pulp production. For more details on this identification, see (Reinhard et al., 2010).

5.2.2.2. FU

The results of the MFA have allowed observing the supply and demand of recovered paper to and from Spain. We have observed that, in recent years, the increase in Spanish waste paper collection belied an increase in the export of recovered paper because the internal demand for recovered paper in Spain decreased, so the FU has been defined as an increase of 1 ton of waste paper collected in Spain for recycling in Spain and internationally. Furthermore, 1 ton of recovered fiber is not equivalent to 1 ton of virgin pulp and we assumed that the equivalence ratio is 0.8:1 (virgin pulp: recovered fiber) (Ekvall, 2000).

5.2.2.3. Scenarios

Taken into account the marginal identification of Reinhard et al. (Reinhard et al., 2010) and the assumption that recycling will avoid virgin pulp, we have considered that an additional ton of waste paper collected in Spain for recycling (nationally and internationally) will avoid 0.80 ton of marginal virgin pulp production (BEKP in Brazil). Besides, as stated above, system boundaries in CLCA are not limited to the evaluated production system as in ALCA studies (Vazquez-Rowe et al., 2014). The MFA has revealed that the main export destinations of recovered paper are China and the Netherlands. Consequently, the system limits were expanded to include these flows and their consequences. We assumed that international recycling will also avoid the marginal BEKP production because the recovered paper is traded internationally. We have defined this scenario as Baseline scenario.

Spain is a traditional BEKP producer (ASPAPEL, 2012), and it would be logical to suggest that collecting more waste paper in Spain for recycling would avoid Spanish BEKP production. Therefore, we performed an alternative analysis based on this suggestion referred as Spanish scenario. We also considered that the recovered paper that is exported will avoid also the marginal BEKP production from Brazil. As a summary, in the Baseline scenario, the increase of waste paper collected for recycling avoids BEKP production in Brazil when recycling occurs in Spain and internationally. In the Spanish scenario, the increase in waste paper collected for recycling avoids BEKP production in Spain when recycling occurs in Spain and avoids BEKP production in Brazil when recycling occurs in China and the Netherlands. For each scenario, the GHG quantifications are calculated as the sum of the positive (emitted) or negative (avoided) GHG emissions associated with each stage. The following LCs were included within the boundaries of Spain: waste management [G]; recycling [H]; virgin pulp production [C], wood chip production [B]; wood cultivation [A]; and transport to all facilities. The stages in China and the Netherlands included recycling and transport. The stages in Brazil included wood cultivation and wood chip and virgin pulp production with the corresponding transport. The data inventory sources and assumption explanations are given in the Appendix A, and Figure 5.3 schematically summarizes both scenarios.

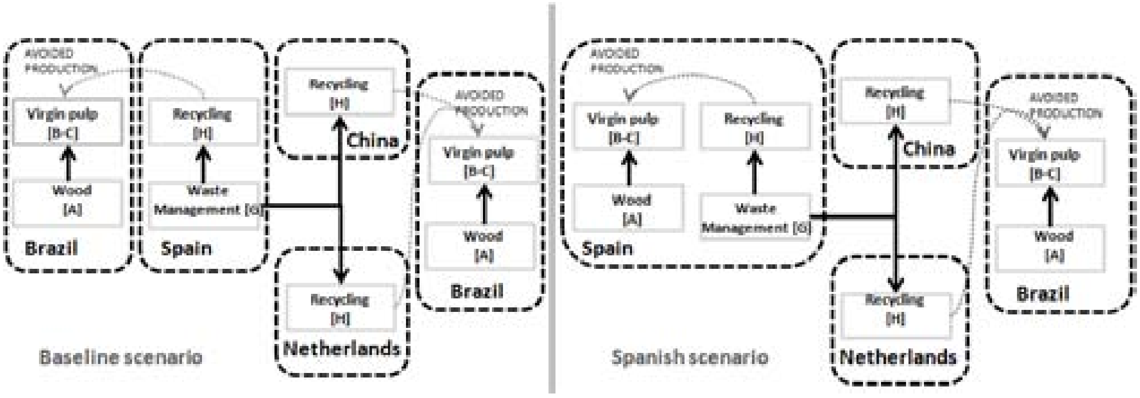

Figure 5.3: **A schematic representation of the baseline scenario and the Spanish scenario where the black arrows represent the product flows and their transport (i.e., wood, virgin pulp or recovered paper); the grey arrows represent the avoided production, which is the virgin pulp production from Eucalyptus wood (i.e., BEKP) and the transport avoided; and the dotted lines represent the country in which each stage has occurred.**

5.2.2.4. LCIA

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

5.3.Results

5.3.1.Spanish paper MFA from 2006 to 2011

The following sections present the results for the MFA (section 5.3.1.1 to 5.3.1.3). Firstly, the Spanish PB life cycle in 2006 is presented. Afterwards, trend for trade flows, transformation flows and loss flows for wood cultivation, virgin pulp production, PB manufacturing, PB product production and use LCs are evaluated from 2006 to 2011. Finally, the waste paper flows and recovered paper flows and recycling are presented and also evaluated between 2006 and 2011.

5.3.1.1. Spanish PB life cycle in 2006

Figure 5.4 presents the flows, stocks and losses included in each life cycle phase of paper in 2006. The LC in grey boxes [A to H] and product flows [a to j] as arrows within the national borders of Spain are included within the inner circle, whereas the arrows to and from the outer circle represent movements to and from the international markets. First it can be observed is that 29% of wood consumed was imported as well as 47 % of virgin pulp while at the same time 48% was exported. In addition, 67% of total PB papers were produced from recovered fiber recycled in Spain but more than 47% of PB papers consumed were imported. Around 24% of PB products consumed in 2006 were also imported and more than 90% were generated as waste paper on the same year while less than 5% was stocked. Around 50% of waste paper was classified as recovered paper, 4% was exported, 19% of recovered paper consumed in Spain was imported and 41% was sent directly to landfill. The Figure 5.4 was drawn using the e-Sankey ® software.

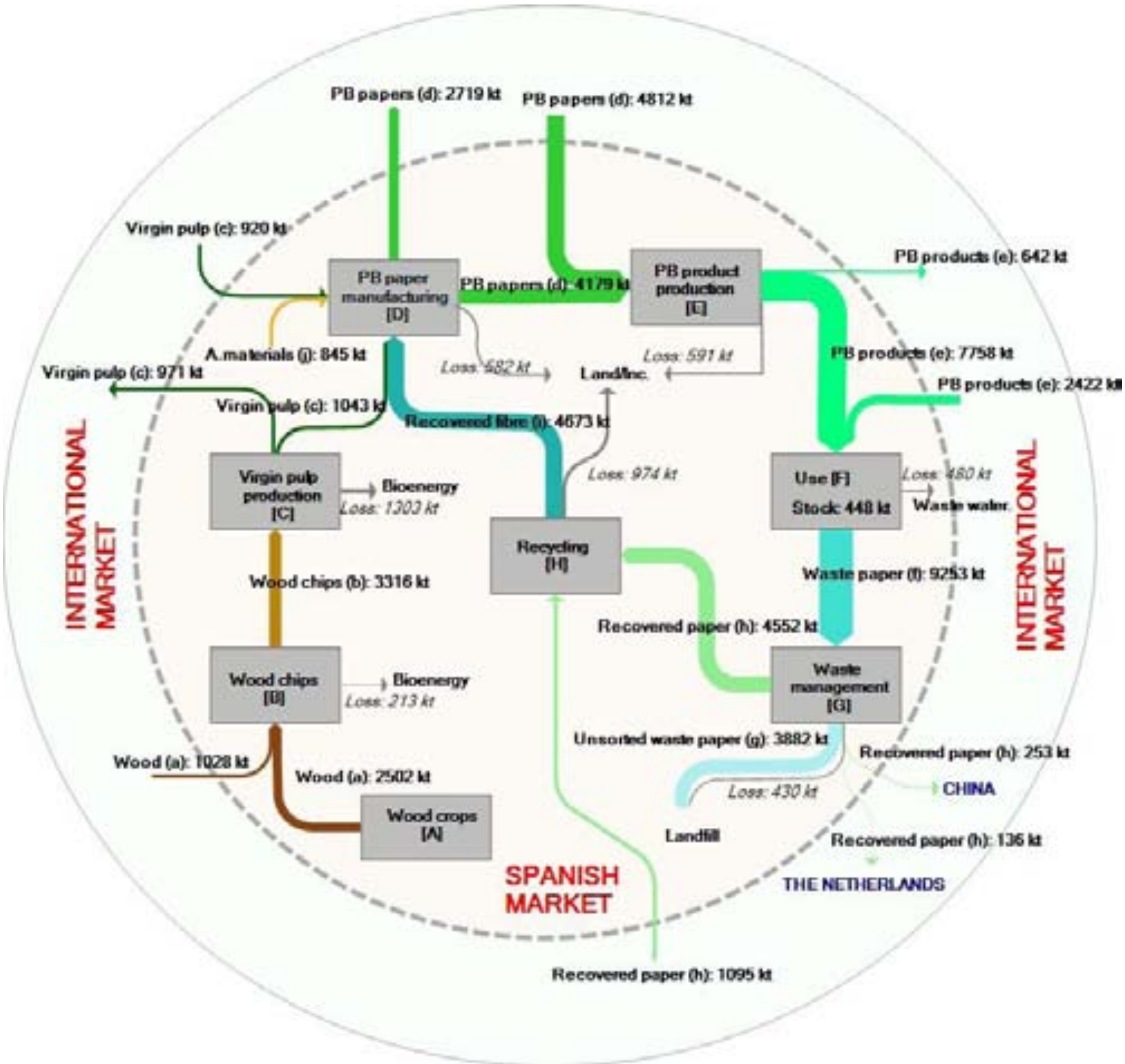

Figure 5.4: Spanish PB life cycle in 2006

5.3.1.2. From wood cultivation [A] to use of PB products [F]

Table 5.1 presents MFA results regarding the production, import and export of wood (a), virgin pulp (c), PB papers (d) and PB products (e) from 2006 to 2011. In the case of wood, from 2006 to 2009, production and import decreased and increased later in 2010 up to 2,231,061tonnes and 1,011,917 tonnes, respectively. Wood cultivation was affected by the large fires of 2006 in northwestern Spain and Portugal, which resulted in a shortage of wood that was supplemented by an increase in wood imports (ENCE, 2007), thus, between 2006 and 2011, around 30% of the consumed wood was imported. In addition, between 2006 and 2011, around 70% of cultivated wood was Eucalyptus while the rest wood pulp was produced from Pine. Regarding virgin pulp, between 2006 and 2009, production, import and export, decreased all following the same trends, and increased later from 2010 to 2011. Considering that more than 70% of the wood produced was Eucalyptus, around 90% of the virgin pulp produced was BEKP. Furthermore, between 2006 and 2011, Spain exported approximately 50% of the virgin pulp produced, ranking it as the second-largest exporter in Europe and the leading exporter in the European BEKP market (Hawkins Wright, 2011). In 2006, however, 47% of the virgin pulp consumed was imported. This percentage increased to 55% in 2010 up to 1,182,662 tonnes of virgin pulp imported.

Production of PB papers (d) decreased from 6,898,200 tonnes in 2006 to 5,700,009 tonnes in 2009, and increased later up to 6,491,508 tonnes in 2011. More than 50% of PB papers consumed in 2006 were imported, increased up to 60% next year and then decreased up to 3,042,121 tonnes in 2011 (45% of PB consumed). At the same time, 39% of PB papers produced were exported in 2006 and this percentage increased up to 50% in 2009 with 2,835,786 tonnes of PB paper exported. Detailed information grouped by PB paper type is presented in Table A2 in Appendix A showing that between 2006 and 2010, the production and import of corrugating materials, printing papers, and newsprint decreased, whereas the import and export of packaging paper, sanitary and household paper production increased up to 304,311 tonnes and 282,709 tonnes, respectively for import, and up to 403,129 tonnes and 45,433 tonnes, respectively for export.

Table 5.1: The production, import and export of wood, virgin pulp, PB papers and PB products in Spain for the years 2006-2011

	Wood (kt)		Virgin pulp (kt)			PB manufacture (kt)			PB products (kt)		
	Production	Import	Production	Import	Export	Production	Import	Export	Production	Import	Export
2006	2,502	1,028	2,013	920	971	6,898	4,812	2,719	8,400	2,422	642
2007	2,540	1,041	2,051	952	1,106	6,713	5,878	2,737	9,250	1,203	637
2008	2,440	1,002	1,978	976	873	6,414	3,997	2,860	6,967	3,351	620
2009	2,158	849	1,714	918	838	5,700	3,878	2,836	6,207	3,429	598
2010	2,231	1,012	1,846	1,183	862	6,193	3,397	2,952	6,066	3,640	756
2011	-	-	1,982	979	1,109	6,492	3,042	2,701	-	-	-

In the case of the PB products (e), production increased between 2006 and 2007 up to 9,250,430 tonnes, and later decreased constantly to 6,066,328 tonnes in 2010. Import of PB products followed opposite trend, decreased between 2006 and 2007, and later increased up to 3,640,457 tonnes in 2010, becoming more than 40% of PB products consumed. Table A3 provides more detailed information grouped by PB product type, showing that from 2006 to 2011, the production of packaging products also decreased from 4,446,900 tonnes to 3,150,609 tonnes. As a result, in 2006, the Spanish paper industry produced 8,400,080 tonnes of PB products, whereas the production decreased to 6,249,873 tonnes in 2011. However, from 2006 to 2010, the import of PB products increased significantly from 2,421,697 tonnes to 3,640,457 tonnes—an increase that was largely due to the import of packaging products, which represented approximately 90% of the total imports. However, these data have been estimated, and the results should be considered carefully.

5.3.1.3. Waste paper management and recovered paper recycling

From 2006 to 2010, the amount of waste paper (f), both refuse and selectively collected, decreased from 9,252,939 tonnes to 7,656,017 tonnes. Table A4 summarizes this information. However, as shown in Table 5.2, the RR increased by an average rate of 4.0%, indicating that although the total amount of waste paper selectively collected decreased, more waste paper was selectively collected compared to the total PB products produced, resulting in more waste paper. Nevertheless, the RUR decreased for the same period at an average rate of 2.2%, indicating that less recovered paper (h) was used for the production of PB papers, which resulted in an additional supply of recovered paper. Considering that the RER had increased at an annual average rate of more than 18% since 2006, one may conclude that the additional amount of recovered paper was exported. Table 5.2 presents these results for the years 2006 to 2011.

Table 5.2: The percentages of RR, RUR, RIR and RER for Spain from 2006-2011

	2006	2007	2008	2009	2010	2011
Recovery Rate (RR) (%)	59.7	49.9	66.2	68.6	69.9	69.1
Recovered Utilization Rate (RUR) (%)	88.1	84.8	84.8	80.6	82.8	78.5
Recovered Import Rate (RIR) (%)	27.6	30.5	25.4	22.8	29.6	28.6
Recovered Export Rate (RER) (%)	9.8	12.0	15.9	23.7	17.1	19.4

Waste paper selectively collected was classified in this work into three grades of recovered paper as Old Newsprint (ONP)-Old Magazine (OMG), Old Corrugated Container (OCC)-Kraft, and mixed grades. Though the Spanish paper industry imported recovered paper from 2006 to 2011, the results presented in Table 5.3 indicate that low-quality-grade paper (i.e., OCC-Kraft) had been exported, whereas better-quality-grade paper had been imported from 2006 to 2011. Lower grade fiber is largely used for the production of packaging papers and boards, which traditionally includes increased input levels of recovered paper, whereas higher grade fiber is used in the production of printing papers and sanitary papers, which traditionally includes increased input levels of virgin pulp (McDougall et al., 2001). During 2006 and 2011, the import of packaging products that contained increased proportions of low-quality fibers increased from 123,674 tonnes to 195,795 tonnes, respectively, which increased the proportion of low-quality fibers in the waste paper fraction. In addition, during this period of time, the production of packaging products decreased from 4,446,900 tonnes in 2006 to 3,150,609 tonnes in 201, such that the demand for low-quality fibers also decreased resulting in the export of extra-low-quality waste paper. In contrast, the Spanish industry increased the production of sanitary papers with higher virgin fiber content from 480,390 tonnes in 2006 to 786,440 tonnes in 2010, which increased the demand for these types of fibers (which are lacking in the waste paper fraction). Table 5.3 presents the import and export of recovered paper grouped by grade. The main destinations of recovered paper were China and the Netherlands, whereas France and Portugal were the origins of the high-quality-grade fibers (DataComex, 2014).

Table 5.3: **The percentages of import and export of recovered paper grouped by type from 2006 to 2011**

	Import (%)			Export (%)		
	ONP-OMG	**OCC-Kraft**	**Mixed grades**	**ONP-OMG**	**OCC-Kraft**	**Mixed grades**
2006	26.6	34.7	38.7	25.2	40.4	34.4
2007	30.1	34.7	35.3	27.9	42.1	30.3
2008	35.8	28.0	36.2	22.0	48.9	29.2
2009	36.2	29.0	34.8	24.4	51.9	23.6
2010	24.5	30.9	44.6	15.4	65.8	18.8
2011	21.8	28.0	50.2	11.6	75.1	13.4

5.3.2.GHG quantifications of waste paper recycling through CLCA

As stated above, the results of the MFA have allowed observing the supply and demand of recovered paper to and from Spain. We have observed that, in recent years, the increase in Spanish waste paper collection belied an increase in the export of recovered paper because the internal demand for recovered paper in Spain decreased, thus, we may conclude that if the trend is not reversed, an increase of waste paper selective collected will increase export flows in the future. Therefore, we have calculated the GHG emissions for different RER with the intention of evaluating how GHG quantifications vary depending on the amount of waste paper that is recycled either locally or internationally, considering that 65% would go to China and 35% to the Netherlands, as of 2011. Table 5.4 presents the results for the GHG quantifications for the Baseline scenario, in which the waste paper collected in Spain would avoid the BEKP in Brazil when it is recycled in Spain, China and the Netherlands.

Table 5.4: GHG quantifications in CO_2 eq. (kg) by ton of collected waste paper in Spain for the Baseline scenario for RERs of 5%, 15%, 25% and 50%

	kg CO_2 eq. t^{-1}			
BASELINE SCENARIO	5%	15%	25%	50%
Waste management [G]	**127**	**145**	**162**	**205**
Collection	71	71	71	71
Sorting	3	3	3	3
National transport to all facilities in Spain	46	48	50	54
International transport to China and The Netherlands	8	23	38	77
Recycling [H]	**86**	**110**	**134**	**195**
Recycling plants in Spain	70	63	55	37
Recycling plants in China	16	48	80	160
Recycling plants in The Netherlands	-0.2	-0.6	-1.1	-2
Avoided Pulp production (Brazil) [B-C]	**-425**	**-433**	**-440**	**-459**
Electricity	-268	-268	-268	-268
Fuel	-17	-17	-17	-17
Chemicals	-65	-65	-65	-65
Others	-2	-2	-2	-2
National transport	0	-1	-2	-4
International transport	-72	-79	-86	-102
Avoided Eucalyptus cultivation (Brazil) [A]	**-104**	**-104**	**-104**	**-104**
TOTAL (kg CO_2 eq. t^{-1})	**-317**	**-283**	**-249**	**-83**

For an RER of 5%, which indicates that 95% of the waste paper collected in Spain is recycled in Spain while 5% is sent abroad, the GHG saving is -317 kg of CO_2 eq. per ton. However, when the RER increases, which means that more waste paper collected in Spain is recycled abroad, the GHG saving decreases up to -83 kg of CO_2 eq. per ton of waste paper collected when 50% is exported. The analysis of the environmental impact of the LC reflects that the increased waste management emissions are due to national transport and particularly due to international transport to China and the Netherlands. The contribution of this international transport varies between 6% for a RER of 5% and 38% for a RER of 50%. However, the predominant variations in total emissions between RERs are due to the recycling process; GHG emissions increase with export rates due to increased recycling emissions in China from 26 kg of CO_2 eq. to 160 kg of CO_2 eq. Although same recycling inventory data were considered for all countries, there are differences in marginal electricity production between China, which has a higher contribution of coal, and Spain and the Netherlands, which have higher contributions of natural gas and cogeneration. Furthermore, the process losses are sent to incineration treatment plants in the Netherlands, such that the production of electricity and energy results in overall negative emissions. In pulp production, similar results are obtained for each stage, except for national and international transport. This similarity is attributed to the substitution of all options with the same values, causing all related impacts to be the same and differences to arise in the locations at which the recycling has occurred and the corresponding transport from Brazil to each country.

Table 5.5 presents the GHG quantification results for the Spanish scenario, in which the waste paper collected in Spain would avoid Spanish BEKP when it is recycled in Spain and avoid Brazilian BEKP when is recycled in China and the Netherlands. In this context, the system expansion reveals lower GHG savings compared to the baseline scenario. The same results were obtained for the waste management and recycling stages in the baseline scenario because the same considerations were accounted for in each scenario. The main differences in the baseline scenario are attributed to the avoided virgin pulp production in Spain, in which the energy used comes from biomass, resulting in reduced avoided emissions. If, for an RER of 5%, the baseline scenario avoids -425 kg CO_2 eq. from virgin pulp production, the Spanish scenario avoids -174 kg CO_2 eq. for virgin pulp production in Spain and -24 kg CO_2 eq. for virgin pulp production in Brazil. When increasing the export flow, the substitution with BEKP from Brazil and the avoided emissions increase.

Table 5.5: GHG quantifications in CO_2 eq. (kg) by ton of collected waste paper in Spain for the Spanish scenario for RERs of 5%, 15%, 25% and 50%

	kg CO_2 eq. t^{-1}			
SPANISH SCENARIO	5%	15%	25%	50%
Waste management [G]	**127**	**145**	**162**	**205**
Recycling [H]	**86**	**110**	**134**	**195**
Avoided Pulp production (Spain) [B-C]	**-174**	**-156**	**-138**	**-92**
Electricity	-88	-78	-69	-46
Fuels	-13	-12	-10	-7
Chemicals	-71	-64	-56	-38
Others	-2	-2	-2	-1
Avoided Eucalyptus cultivation (Spain) [A]	**-83**	**-74**	**-66**	**-44**
Avoided Pulp production (Brazil) [B-C]	**-24**	**-73**	**-122**	**-244**
Avoided Eucalyptus cultivation (Brazil) [A]	**-6**	**-17**	**-28**	**-56**
TOTAL (kg CO_2 eq. t^{-1})	**-74**	**-66**	**-57**	**-36**

Though the paper is focused in GHG emissions, the indicator of cumulative energy demand (CED) was also calculated in order to evaluate total energy consumed and saved. This indicator takes into account the energy requirements in units of mega Joule (MJ) through the life cycle, including direct and indirect uses of energy. Results for all scenarios and all RERs are presented in Tables A7 and A8 in Appendix A while Figure A2 and A3, also in Appendix A, evaluate direct and indirect energy for a RER of 15%. Finally, Table A5 presents the total energy comparison between the Spanish and the Brazilian BEKP production by source of energy. SimaPro 7.3.3 software was used for the environmental evaluation, together with the "Cumulative Energy Demand" method. In both scenarios and all RER evaluated, negative values were obtained what indicates that recycling recovered paper saves energy. Same trends as for GHG emissions were obtained and for both scenarios, an increase of the export of recovered paper implied a decrease in the energy saved. In the baseline scenario for a RER of 5%, total energy avoided per ton of waste paper collected was 6,863 mega Joule (MJ) per ton of waste paper collected and decreased up to 5,114 MJ per ton collected when 50% of recovered paper is exported. For the Spanish scenario, when 5% of recovered paper is exported 3,420 MJ per ton are avoided and decreased up to 3,302 MJ avoided per ton of waste paper collected. For both scenarios, the contribution of direct and indirect energy consumed is the same, 75% and 25%, respectively. For the avoided energy, however, in Baseline scenario 79% of energy avoided is direct energy while in the Spanish scenario this contribution decreased up to 63% because less energy is used in Spain for the virgin pulp production.

5.4.Discussion

5.4.1.Recovered paper flows

For the case study of PB in Spain, the MFA has revealed that in recent years, there has been an elevated supply of low-quality fibers due to both high import rates of packaging products and more waste paper collected. Additionally, there has been a decline in the demand for a domestic quantity of low-quality fibers due to a decrease in the consumption and production of corrugating products (which have a higher content of low-quality fibers). Consequently, there has been an oversupply of recovered low-quality paper that has been exported. The same trend was observed for Europe (FAO, 2012); between 2005 and 2010, the consumption of recovered paper decreased by 1.6%, while the exports of recovered paper increased by up to 27% during the same period. In addition, from 2000 to 2010, printing paper and sanitary & household paper production with high virgin pulp content increased by up to 3.1% and 2.8%, respectively, and newsprint and packaging production decreased to 1.0% and 0.9%, respectively (Hawkins Wright, 2011). In fact, corrugated material production has shifted to China, as has the demand for waste paper (IVEX, 2010). However, far from being a temporary adjustment, this shift seems to be a long-term trend. It has been predicted that China's demand for recovered paper is likely to increase in the future by 50% to approximately 39 Mt (BIR, 2011). The world trade of recovered paper is forecasted to reach 77 Mt by 2015, accounting for 27% of total worldwide collections, with a surplus in Western Europe, North America and Japan and a deficit in China and other Asian countries (WRAP, 2007). Thus, the volume of the trade of recovered paper is expected to increase in the future with enhanced collections and more waste going to international markets, and it will likely affect the market behavior and dynamics of recovered waste as it has affected previous years.

Moreover, we have observed through the MFA that the consumption of imported pulp in Spain has increased while the Spanish production decreased in the same period. This led to an increase in the imports of virgin pulp of up to 55% in 2010. The import flows came from Europe and South America and had high variability over the years. However, virgin pulp flow from Latin America (NEP, 2009), particularly BEKP, is beginning to enter the market at a greater rate. In this sense, although the marginal identification may imply some uncertainty and can be discussed, the increase in BEKP imports is in line with the identified marginal supply. Thus, the most competitive virgin pulp is displacing higher-cost fibers in the market (Hawkins Wright, 2011).

5.4.2.GHG quantifications with CLCA

Comparing the GHG emissions results that we obtained in this study with the results from other studies is difficult due to the lack of data regarding the GHG quantifications for Spain (Sevigné et al., 2013), although some studies evaluate various collection or waste treatments options within national boundaries (Bovea and Powell, 2006; Iriarte et al., 2009; Bovea et al., 2010; Font et al., 2012). A review of the relevant literature found that the GHG quantifications, in the sources which were found transparent enough, range from a saving of 3,1 to 0.3 kg CO_2 eq. per ton of paper recovered (Smith et al., 2001; US EPA, 2006; BIR, 2008; Prognos, 2008) depending on the inventory and geographic and temporal boundaries. Most of the credits revised are calculated based on the comparison of 100% primary material production with 100% recycled material production (Smith et al., 2001; Prognos 2008, BIR, 2008). However, none of sources have considered the systemic effects that may result from changes in the dynamic of supply and demand of recovered paper, and they have not considered other market mechanisms that can determine the source of marginal virgin pulp and may be avoided when switching to recycled fiber. However, as this study reflects, reality is much more complex, and to consider that neither the virgin pulp and recovered paper nor the resulting GHG estimates are affected by the market dynamics is incorrect and incomplete. In addition, the importance of including market dynamics is highlighted when export flows are taken into account. The GHG quantifications for Spain have revealed significant differences when recycling is performed within the country compared to abroad, with the highest GHG savings occurring when the waste paper is collected and recycled locally and the BEKP from Brazil is avoided. When 50% of the waste paper collected is recycled abroad, the variations are near 225 kg of CO_2 eq. per ton. Because the Spanish industry collects approximately 5 Mt of waste paper, the waste paper export flows reduce the benefits of recycling by approximately 1 Mt of CO_2 eq. Thus, the global consequences can be significant. Therefore, it is necessary that the method for accounting for the GHG impact of recycling reflect the market mechanisms, especially if the GHG estimates are to inform waste management policies and strategies.

Other studies (Schmidt et al., 2007; Merrild et al., 2008; Merrild et al., 2009; Laurijsen et al., 2010) have evaluated system expansions to include the avoided impacts of wood in the sense that the recycling of paper implies a reduced demand for virgin paper, which reduces the demand for land or wood (Schmidt et al., 2007). Thus, the wood that is not used for virgin paper production is used to produce energy to substitute for fossil fuels. This shift results in savings from 1,100 to 4,400 kg CO_2 eq. per ton (Merrild et al., 2009; Laurijsen et al., 2010) and shows that the methodology and assumptions followed for the GHG quantification of recycling may indicate important variations within the results. Although this study does not consider this second system expansion, the use of woody biomass as an energy source is expected to increase in Europe in the future (Lauri et al., 2012), and it should be studied if, as a consequence of less virgin pulp production in Spain (or in Europe), more bioenergy production from wood resources is used. In this sense, more research is needed to consider other aspects of the GHG quantifications, such as the bioenergy production or the GHG emissions derived from changes in land use (James, 2012).

The results of the CED showed similar trends as the GHG emissions because most of the CO_2 eq. emissions are due to the consumption of fossil fuels and the differences between both scenarios are due to the energy consumption such as the marginal electricity mix. Similar conclusions were obtained in a recent study in which it was highlighted the well correlation between both indicators for the comparison of 498 commodities and in fact, fossil energy use was identified as the most important driver of environmental burden of the majority of the commodities included (Huijbregts et al., 2010).

5.4.3.Supporting decision-making with MFA and CLCA

The increasing trend in exports of paper waste for recycling and in imports of virgin pulp should be considered for decision makers. The approach presented in this study enables the assessment of the consequences of different strategies on the material flows and GHG emissions as the whole life cycle is taken into account. For example regarding the waste exports, if no decision is taken in order to avoid the increasing trend of recovered paper sent to Asia, given that restrictions would affect prices or limit the free trade, consequently, the benefits of GHG would decrease as well as the efforts of waste paper collection. Otherwise, if an alternative policy is considered by taken into account the split between local recycling and exports, this strategy should promote the use of recovered paper in national industries. Further, other aspects can also be assessed, such as the import of virgin pulp related to the national virgin pulp production and its influence on the GHG emissions avoided.

5.5.Conclusion

LCA studies on waste management have been promoted because they can help policy makers choose the best environmental options (JRC, 2011). However, the impacts of imports and export have historically been excluded from LCAs (James, 2012), but as this study reflects, the effect of the markets may affect the GHG quantifications. The consequences are reflected not only in terms of increasing waste export flows, which increase the environmental impacts attributed to transport, but also in terms of the indirect effects associated with primary raw production, that they may eventually elicit significant environmental impacts. Recent reports highlighted that ever more waste is crossing EU borders, moving between Member States and to and from non-EU countries. At the same time, Europe is becoming dependent on imports of natural resources (Giljum, 2008, EEA, 2012; EAA, 2012b; Plastic Europe, 2013). Thus, the increasing globalization of raw materials and waste makes the optimization of waste management strategies and policies quite challenging justifying the need of new approaches to identify the consequences. Changes in those trends potentially may bring significant environmental, social and economic opportunities. In the future, GHG quantifications should be based on flows described by MFA analysis and quantified by CLCA as these quantifications vary significantly and CO_2 eq. has economic value. It is necessary to have methodologies that map properly all material flows as a first step in determining the potential environmental impacts to facilitate the accurate accounting of GHG for decision-making.

Chapter 6-

Environmental consequences of recycling aluminium old scrap in a global market

6- Environmental consequences of recycling aluminium old scrap in a global market

based on the following paper: Eva Sevigné Itoiz, Carles M. Gasol, Joan Rieradevall and Xavier Gabarrell. Environmental consequences of recycling aluminium old scrap in a global market. Resources, Conservation & Recycling (2014)

Abstract

Nowadays, aluminium scrap is traded globally. This has increased the need to analyze the flows of aluminium scrap, as well as to determine the environmental consequences from aluminium recycling. The objective of this work is to determine the GHG emissions of the old scrap collected and sorted for recycling, considering the market interactions. The study focused on Spain as a representative country for Europe. We integrate MFA with CLCA in order to determine the most likely destination for the old scrap and the most likely corresponding process affected. Based on this analysis, it is possible to project some scenarios and to quantify the GHG emissions (generated and avoided) associated with old scrap recycling within a global market. From the MFA results, we projected that the Spanish demand for aluminium products will be met mainly with an increase in primary aluminium imports, and the excess of old scrap not used in Spain will be exported in future years, mainly to Asia. Depending on the scenario and on the marginal source of primary aluminium considered, the GHG emission estimates varied between -18,140 kg of CO_2 eq. t^{-1} to -8,427 of CO_2 eq. t^{-1} of old scrap collected. More GHG emissions are avoided with an increase in export flows, but the export of old scrap should be considered as the loss of a key resource, and in the long term, it will also affect the semifinished products industry. Mapping the flows of raw materials and waste, as well as quantifying the GHG impacts derived from recycling, has become an essential prerequisite to consistent development from a linear towards a CE.

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Chapter 7-

The contribution of plastic waste recovery to resource efficiency and greenhouse gases (GHG) savings in Spain: quality, technological and market constraints

7- The contribution of plastic waste recovery to resource efficiency and greenhouse gas (GHG) savings in Spain: quality, technological and market constraints

based on a manuscript by Eva Sevigné Itoiz, Carles M. Gasol, Joan Rieradevall and Xavier Gabarrell.

Abstract

One baseline scenario and seven alternative scenarios were projected based on results from a MFA of Spanish plastic life cycle (1999-2011). The scenarios were defined regarding plastic waste management (recycling or energy recovery), plastic waste quality (high or low), recycled plastic applications (virgin plastic substitution or non plastic substitution) and markets (regional or global) of recovered plastic. The aim was to quantify the environmental consequences of the different alternatives in order to evaluate opportunities and limitations. Quantification is conducted with a CLCA and is limited to GHG emissions. Result showed that considering the improvement in plastic waste management and current in-use stock achieving its EOL, in coming years, an increase of plastic waste collection and supply is expected. Besides, in order to improve resource efficiency and avoid more GHG emissions, the options for plastic waste management are with or against the waste hierarchy depending on the quality of the recovered plastic: mechanical recycling for quality recovered plastic (-620 kg CO_2 eq. t^{-1}), export of recovered plastic (-138 kg CO_2 eq. t^{-1}), energy recovery (-27 kg CO_2 eq. t^{-1}) and mechanical recycling for low quality recovered plastic (54 kg CO_2 eq. t^{-1}). Thus, focus should be on increase recycling rates rather than on energy recovery rates, on increase the quality of the recovered plastic waste through strategies to facilitate sorting steps and a new focus should be introduced to take into account the split between local recycling and exports.

7.1.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 Mt in 1950 to 288 Mt in 2012. Global plastic production could triple by 2050 (European Commission, 2013; Plastic Europe, 2013). Regarding European production alone (EU-27+2), it has increased from 0.35 Mt in 1950 to 57 Mt in 2012 accounted for 20.4% of the world's total production (288 Mt) (Plastic Europe, 2013). Around 66.5 Mt of plastic will be placed on the EU market in 2020 (European Commission, 2013). However, plastics, as materials, are generating environmental problems along its whole life cycle. On one hand, in order to produce plastic products GHG emissions are produced. On the other hand, the characteristics that make plastic so useful also makes waste management problematic (European Commission, 2013) and combined with the throwaway culture that has grown up around plastic products (European Commission, 2011a), there is a considerable accumulation of plastic wastes in the environment. Once in the environment, particularly in the marine environment, plastic waste can persist for hundreds of years (Kaps, 2008). In fact, 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, 2011b; European Commission, 2013).

Hence, considerable concern has been focused on plastic waste management. At European level, for example, since 1994 several objectives for plastic waste recycling and recovery have been set (European Commission, 1994; European Commission, 2000a; European Commission, 2004). Moreover, 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 on 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. This is due to demand from fast-growing Asian economies driving higher prices. For example, in 2012 between 32% and 55% of plastic waste collected for recycling in the EU (2.0-3.5 Mt) was exported, 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 CE (European Commission, 2012b) where waste is regarded as a valuable resource within Europe (European Commission, 2011c; European Commission, 2012b).

Furthermore, last Directive 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) (Lazarevic et al., 2010; JRC, 2011; Laurent et al., 2014). In this regard, one important limitation to fulfill with the waste hierarchy is that although plastic is a fully recyclable material, only a small fraction of plastic waste is at present recycled due to contamination and technical limitations (Hong, 2012; Briassoulis et al., 2013). The options for use of recycled plastic depend on the quality and polymer homogeneity of material (JRC, 2012). 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, 2012). In this case, 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. It is not clear which are the best options to improve plastic waste management while reducing GHG emissions. For instance, if the efforts should be done to increase reuse, energy recovery, to improve technology of sorting and recycling or to promote the use of cheaper and less demanding applications of recycled plastics. Addressing these questions can be conducted with MFA, which evaluate the flows and stocks of materials within a system defined in space and time (Brunner and Rechberger 2004), in combination with CLCA, which quantify and describes how environmental impacts will change in response to possible decisions (i.e., increase collection or energy recovery) (Reinhard and Zah, 2009).

The aim of this study is to evaluate different options in order to identify limitations and opportunities of plastic waste recovery with the objective of decrease. GHG emissions. Spain was selected as a case study. It accounted for 7% of European market demand in 2012, positioning itself as the fifth European plastic consumer only behind Germany (25%), Italy (14%), France (9%) and United Kingdom (8%) (Plastic Europe, 2013). However, considering its plastic waste management, Spain is in the middle range of European averages regarding for instance, collection, recycling or export of plastic waste (ANARPLA, 2013; Plastic Europe, 2013). In addition, currently, there is an open debate between supporters of current management system for light packaging (i.e., paper, plastic or aluminium) 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 increase the collection and recycling rates similarly to other European countries (i.e., Germany) because collection would be derived by economic incentives (Inèdit, 2011; GRC, 2013). Thus, Spain may be a representative European country of the current context and it is also possible to evaluate consequences of following the trends and strategies of other European countries. The study conducted firstly, a dynamic MFA of plastic life cycle to evaluate trends over the years and secondly, based on the MFA results alternative scenarios were projected and integrated into the consequential LCI to quantify the GHG emissions associated and to identify both important and negligible influences on the GHG balance. The CLCA is limited to the estimation of GHG emissions on account of their current high priority in EU policies.

7.2.Methodology

In the following sections, the methodologies used for the quantifications of flows and stocks (7.2.1) and for the quantification of GHG emissions (7.2.2) are explained. More detail is available in Appendix C.

7.2.1.Dynamic MFA

7.2.1.1. 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.

MFA is focused on post-consumer plastic waste and on the current EOL options in Spain: 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, in which a certain degree of polymeric breakdown takes place (JRC, 2012). 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 MFA were defined as years 1999-2011 and Spain, respectively. Along its life cycle, we considered the following 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 7.1 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 the abovementioned GDS as well as through others selective ways (i.e., agriculture plastic waste). However, there is also an important fraction that is collected within the refuse fraction, latter namely plastic waste refuse fraction. 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, recycled plastic substitutes for virgin plastic, but if the plastic wastes are contaminated and/or are a mix of different plastic types, 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 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 7.1 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)).

7.2.1.2. 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, total flows entering the LC should equal to total flows leaving it, with flows detailed in Figure 7.1. 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 C.

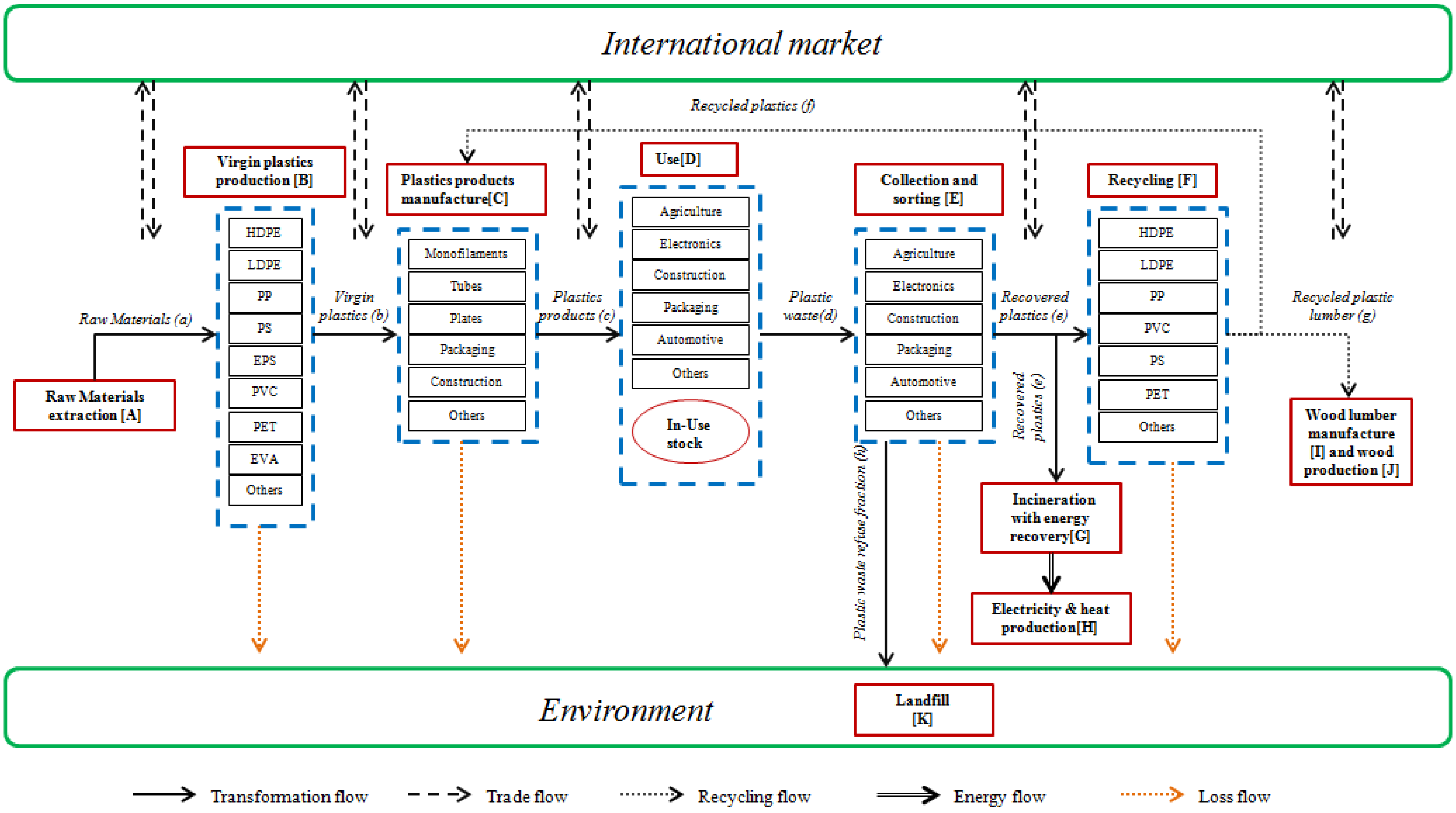

Figure 7.1: Spanish plastic cycle system boundaries

7.2.2. CLCA

7.2.2.1. Scope and system boundaries

Even though MFA takes into account the whole life cycle of plastics as shown in Figure 7.1, for the GHG quantifications we focus only into GHG emissions from 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 et al., 2012). This study concluded 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 et al., 2012). Nevertheless, as explain in section 7.2.1.1, 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 to 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 7.1 as loss flow).

7.2.2.2. FU, LCI and LCIA

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 C7-C8 in Appendix C 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.

7.2.2.3. Scenarios and sensitivity assessment

Eight scenarios were evaluated and discussed in order to identify both important and negligible influences in GHG quantifications; one Baseline scenario and seven alternative scenarios. MFA results of 2011 were the base for the Baseline scenario. MFA trends, the European trends and main parameters that were highlighted in literature as constraints of plastic waste recycling were the base for the alternative scenarios. Figure 7.2 summarizes the scenarios considered and indicates in brackets the name of the alternative scenario.

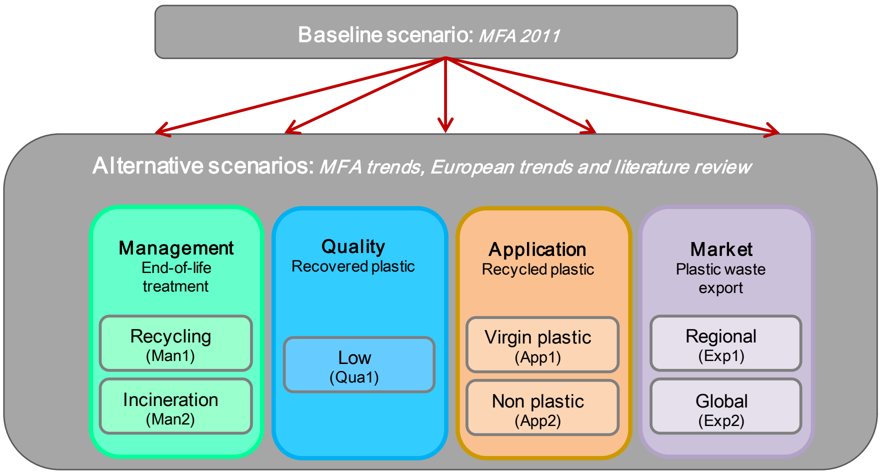

Figure 7.2: Baseline and alternative scenarios schema

7.3.Results

Following sections present the results for dynamic MFA (sections 7.3.1.1 to 7.3.1.5), the Baseline scenario and the alternative scenarios which are defined from the MFA results, trends projected for Europe and literature review (section 7.3.2) and results for the GHG quantifications (sections 7.3.3.1 and 7.3.3.2).

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

7.3.1.1. From virgin plastics to plastics products

Figure 7.3 presents total production of virgin plastic and of recycled plastics within the total plastic consumption per capita in Spain from 1999 to 2011. Total consumption of virgin plastic grew steadily from 1999 to 2007 exceeding 6,000,0000 tonnes and then decreased substantially until 2009. Consumption stabilized around 4,700,000 tonnes from 2009 probably to economic crisis which have affected all industries and sectors, especially the construction sector. Total recycled plastic consumption has increased year by year from 200,000 tonnes in 1999 to almost 600,000 in 2011. In fact, the contribution of recycled plastic over the primary form has grown from 5% in 1999 to 12% in 2011. Regarding the overall plastic consumption per capita, we observe that in 1999 it was 108 kg hab^{-1}, increased constantly until 2007 up to 147 kg hab^{-1} but decreased in subsequent years up to values close to those of 1999.

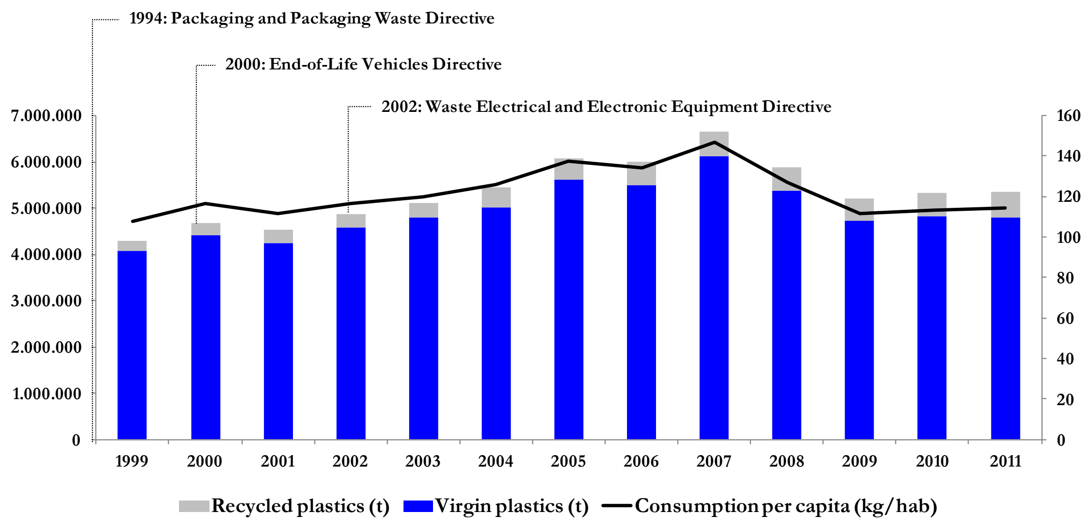

Figure 7.3: Consumption of virgin plastic and recycled plastics in tonnes from 1999 to 2011 in Spain in the left axis and overall consumption in kg per capita per year in the right axis from 1999 to 2011 in Spain

Table C1 and Table C2 in Appendix C presents production and consumption of virgin plastic by type of virgin plastic. These tables show that for 2011, polyethylene (PE) accounting for HDPE and LDPE has the highest share of total consumption of any polymer type (30%). It is followed by PP which accounts for 15% of plastic resin capacity and PET which accounts for 13%. These polymers account for about 59% of the total plastics consumption, followed by PVC (8%) and polystyrene (PS) (3%). The category other which considered different types of polymers accounts for 27% of consumption.

The main application of PE, PP and PET is packaging (Plastic Europe, 2012) and as it is shown in Figure 7.4, which presents consumption of plastic product by application sector in Spain, it is clear that in packaging is the largest single sector for plastics (46% in 2011), what justifies the higher share of PE, PP and PET consumption; 22%, 18% and 14%, respectively in 2011. Packaging products increased constantly until 2007 and then decreased up to 46% in 2011. The packaging sector is followed by the category others, which include sectors such as household (toys, leisure and sports goods), furniture or medical devices. Then it is followed by automotive, electronics and agriculture, which consumption has remained more or less constant over the period studied.

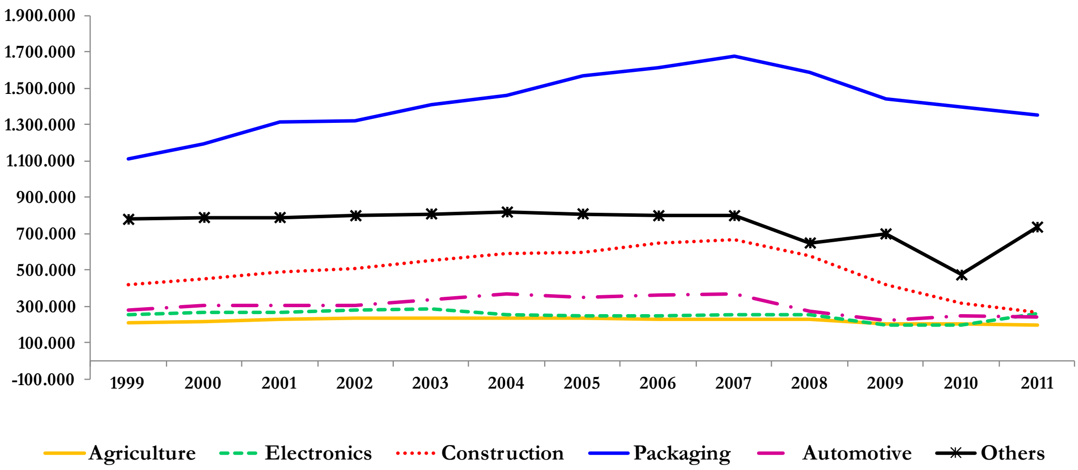

Figure 7.4: Consumption of plastics products in tonnes from 1999 to 2011 in Spain by main application sector

In the case of PVC, this type of polymer is mainly used in the construction sector (Plastic Europe, 2012) and its consumption follows the trend for this sector; in 1999 PVC consumption was more than 550,000 tonnes, increased up to 717,000 tonnes in 2006 and then decreased constantly up to 382,000 tonnes in 2011.

7.3.1.2. Plastic waste management

Table 7.1 shows total plastic waste generated and its treatments, packaging plastic waste generated and its treatment and 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 C 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 have considered that total packaging products consumed are 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, main consideration regarding this fraction is its waste management which has changed over the years. In 1999 more than 67% of packaging plastic waste selective collected was sent to landfill but the percentage has decreased up to 49% in 2011. 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 C3 and Table C4 in Appendix C. Complementary, Figure 7.5 presents recycling in percentage by type of sector in 1999 and 2011 (Figure 7.5a and 7.5b); recycling by type of polymer in 2011 (Figure 7.5c) and markets for recycled polymers (Figure 7.5d). In 1999, the recycling of packaging plastic waste constituted the highest fraction recycled followed by the agriculture sector. The recycling of plastic waste from automotive, electronics or construction sectors was negligible in 1999 but in 2011, the plastic recycling from these sectors increased up to 9,946 tonnes, 17,492 tonnes and 21,386 tonnes, respectively.

In 2011, the 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, 2012). 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, 2012). 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 (see Figure 7.5c). In this study we consider that this mix of polymers is used to produce the RPL in substitution with wood. In this regard, in Figure 7.5d for 2009 (last available data) the market referred as “others” includes the use of recycled polymers as hangers or footwear as well as street furniture or garden furniture.

Table 7.1: 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[1]	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

[1] There is only data for packaging plastic waste incinerated with energy recovery. We assumed same data for plastic waste incinerated with energy

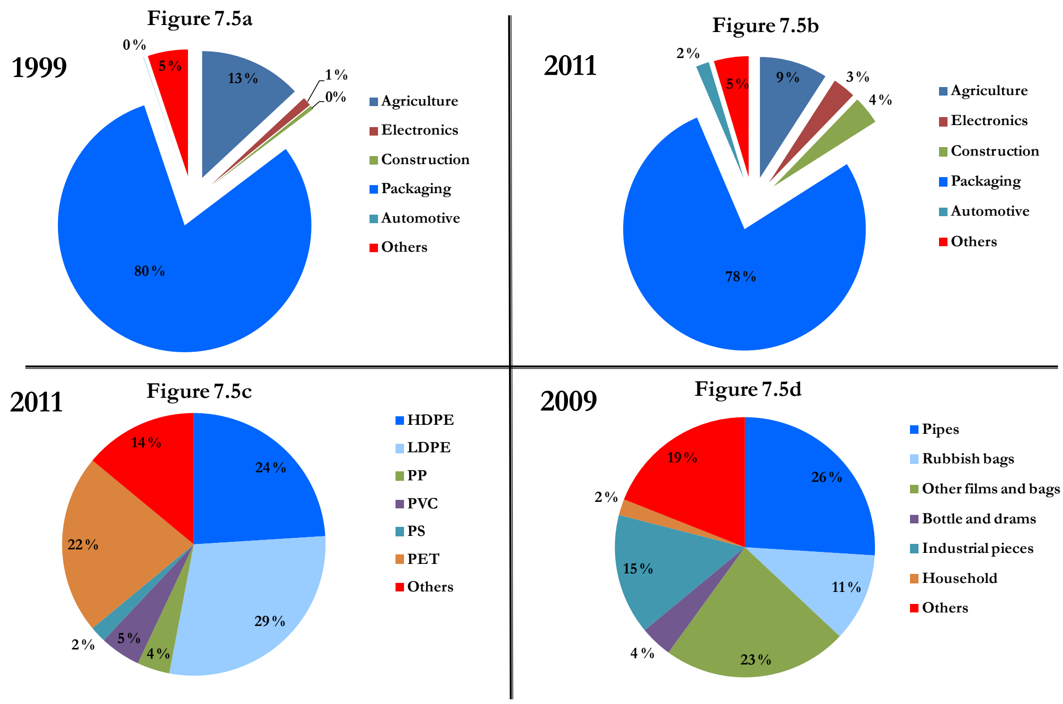

Figure 7.5: Plastic waste recycling in Spain by sector in 1999 and 2011 (Figure 7.5a and 7.5b, respectively), plastic waste recycling by type of polymer in 2011 (Figure 7.5c) and markets for recycled polymers in 2009 (Figure 7.5d) (Cicloplast, 2009; ANARPLA, 2013)

7.3.1.3. International trade

Figure 7.6 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, 2014). During the same period there is also a lack of plastics products, which were imported mainly from Germany and France (DataComex, 2014), 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, 2014). In fact, Spain was the fifth European exporter (ANARPLA, 2013).

Table C5 in Appendix C 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.

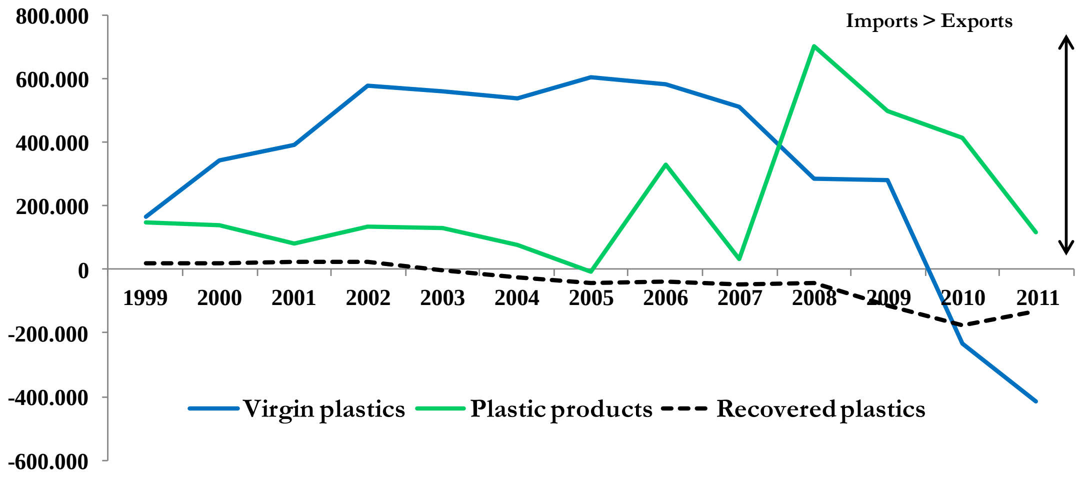

Figure 7.6: Commercial balances (imports minus exports) for virgin plastic, plastic products and recovered plastics (packaging plastic waste) in Spain from 1999 to 2011 in tonnes

7.3.1.4. 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.

Appendix D presents the methodology followed for the future plastic waste generation which is discussed in **Chapter 8**.

7.3.1.5. Spanish plastic life cycle in 2011

Figure 7.7 presents the Spanish plastic life cycle in 2011, as it is the most representative year of the current situation. It can first be observed that there was no raw materials production in Spain and total raw materials used for virgin plastics production were imported. In addition, around 50% of virgin plastics produced were exported. More than 30% of plastic products consumed were stocked and approximately, 50% of plastic waste generated was collected within the refuse fraction and sent to landfill. Around 12% of plastic waste selectively collected was exported, 21% was sent to energy recovery and 40% was sent to recycled from which 86% was used as substitutes of recycled polymers. There is no information regarding further treatments for the materials losses from virgin plastic production and plastic products manufacture but we have assumed that are sent to landfill but these amounts are not taken into account for the plastic waste accumulated in landfill presented in section 7.3.1.4 (30,542,493 tonnes).

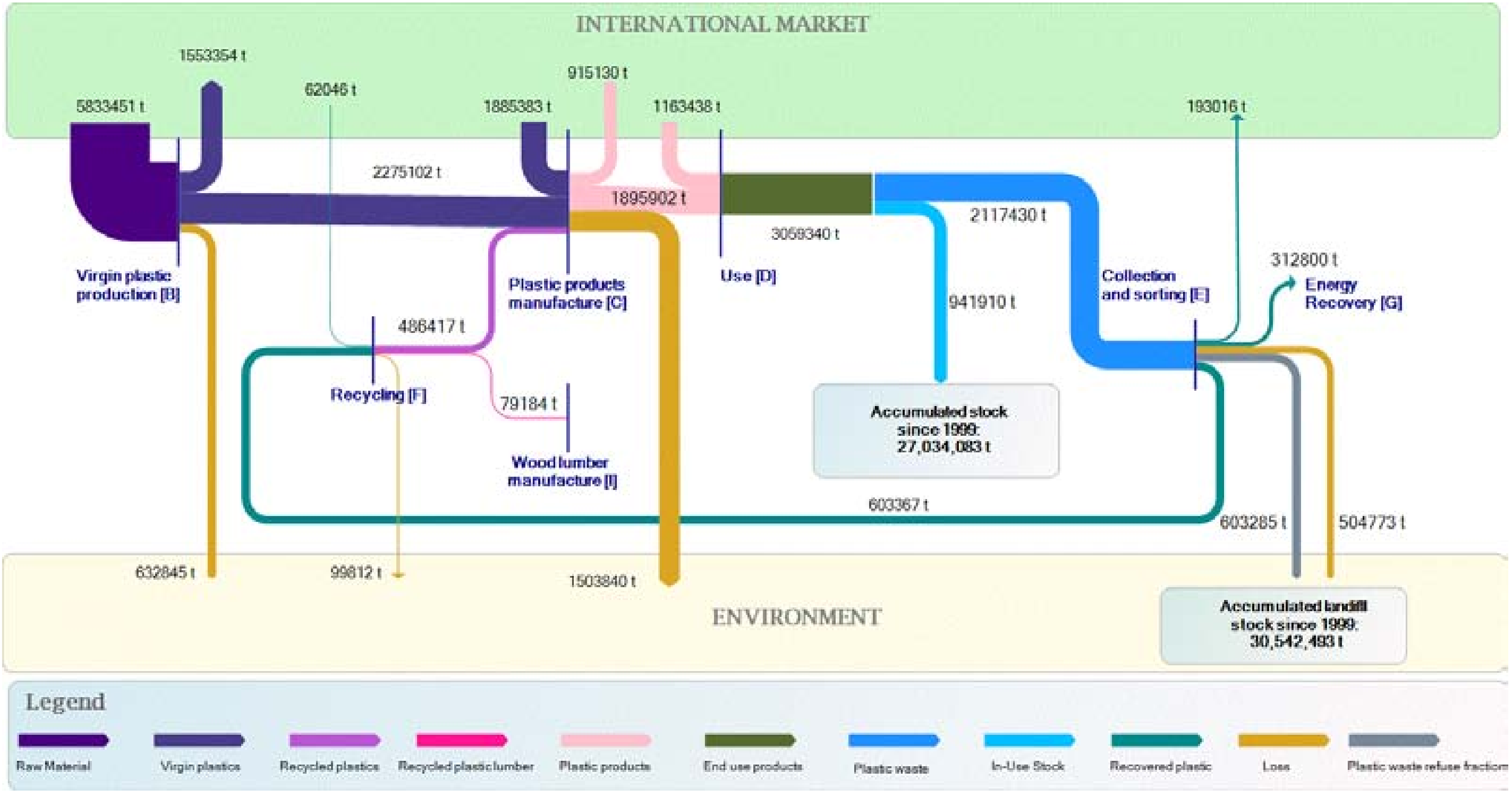

Figure 7.7: Spanish plastic life cycle for 2011

7.3.2. 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). This has contributed to an increase in the collection of 66% since 1999 in parallel with the improvement of material efficiency in sorting plants which has contributed to increase plastic waste recovery from 37% in 1999 to 73% in 2011. 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, in 2011 for every tonne of recycled plastic in Spain, 9.8 tonnes of virgin plastic were consumed, 4.4 tonnes were generated as plastic waste but only 3.1 tonnes were selective collected for recovery and 1.2 tonnes were sent directly landfilled. Therefore, it is clear that there is a huge potential for higher collection rates in Spain. Furthermore, the huge 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 27,034,083 tonnes. Besides, in 2011, 193.016 tonnes of Spanish recovered plastic were exported, and around 90% were exported to China and Hong Kong (ANARPLA, 2013). Therefore, if trends remain the same in following years we can expect an increase of plastic waste collection due to better plastic waste management and due to the in-use stock achieving its EOL, mainly from construction and electronic sectors. In addition, considering that the construction and electronic sectors have higher presence of PP and PVC more difficult to recycle (JRC, 2012; BIO Intelligence Service, 2013), we can also expect an increase of the energy recovery treatment and an increase of RPL production. It would also increase of recovered plastic export to Asian countries due to high demand and higher prices. Similar trends have been projected for Europe (European Commission, 2011b; Shonfield, 2008; WRAP, 2011; JRC, 2012, BIO Intelligence Service, 2013), thus, we evaluated this projections in some of the alternative scenarios.

Besides, we projected other alternative scenarios in order to evaluate contrary trends, such as an increase of recycling or the restriction to recovered plastic export, and the main parameters that were highlighted in literature as most important for GHG balance. Table 7.2 summarizes data for the Baseline scenario and the alternative scenarios.

Scenario Man1 evaluates an increase of the recovered plastic sent to recycling up to 90% and scenario Man2 evaluates an increase of the recovered plastic sent to energy recovery up to 50%

similar to Northern European countries. The recycled plastic waste was assumed to substitute virgin plastics and wood with a substitution ratio of 1:1. However, 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). Thus, the substitution rate is commonly less than 1, so different substitution ratios should be assessed. The literature review ranges the ratio substitution from a very low rate of 0.5 to 1 (Lazarevic et al., 2010; OECD, 2010; Hong, 2012) (scenario Qua1). Further, based on data from section 3.1.2, 86% of the mechanically recycled plastics are converted to recycled substitutes while the remaining 14% is used to produce RPL. Scenario App1evaluates that all recycled plastics substitute virgin plastic and Scenario App2 evaluates an increase of the application of recycled plastic to produce RPL up to 50%, which means that 50% of the recovered plastic is used to substitute virgin polymers and 50% to substitute other products (i.e., wood lumber). The MFA have revealed that in 2011, 29% of sorted plastic waste was exported to recycled globally with 10% to Intra-EU while 90% to outside EU (mainly to Hong Kong and China) (DataComex, 2014). Scenario Exp1 evaluates that all recovered plastic is recycling regionally in Spain and scenario Exp2 that all recovered plastic is recycling globally with 10% sent to Europe and 90% to China. We have used for both scenarios same classification of plastic waste by type of polymer than in 2011 (see section 7.3.1.2 and 7.3.1.3).

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 C.

Table 7.2: Data and assumptions for the Baseline scenario and the alternative scenarios used for the GHG emissions evaluation

	Management	Quality	Application	Market
	Recycling (%)[2]	Ratio substitution	Virgin plastic (%)	Global (%)
Baseline scenario	71	1:1	86	29
Alternative scenarios				
Scenario Man1 (*90% of recovered plastic to recycling*)	90	1:1	86	29
Scenario Man2 (*50% of recovered plastic to recycling*)	50	1:1	86	29
Scenario Qua1 (*0.5 tonne of recycled plastic= 1 tonne virgin plastic*)	71	1:0.5	86	29
Scenario App1 (*100% of recycled plastics substitute virgin plastics*)	71	1:1	100	29
Scenario App2 (*50% of recycled plastics substitutes virgin plastics*)	71	1:1	50	29
Scenario Mar1 (*All recovered plastic is recycled in Spain*)	71	1:1	86	0
Scenario Mar2 (*All recovered plastic is recycled outside of Spain*)	71	1:1	86	100

[2] Percentage over recovered plastic after sorting plants

7.4. GHG quantifications of recycling

7.4.1. Baseline scenario

Figure 7.8 presents the material flows for the Baseline scenario per the FU taken into account the plastic waste recycling by type of polymer presented in Figure 7.5c for the Spanish recycling and in the case of the recovered plastic export, we took into account data presented in section 7.3.1.3 by type of polymer.

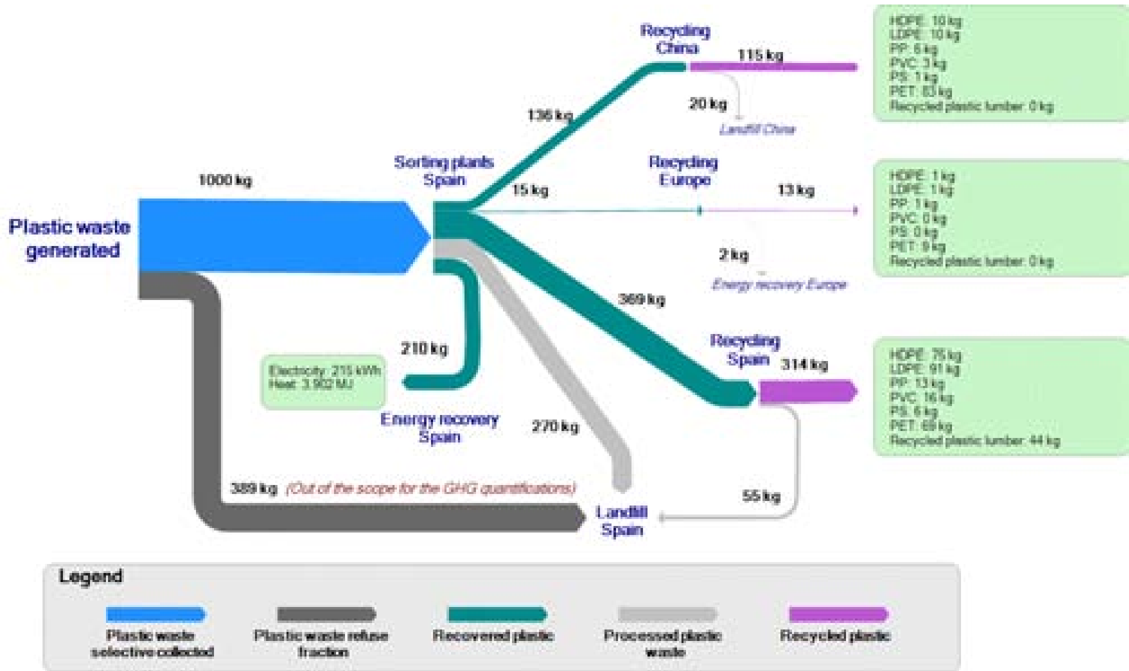

Figure 7.8: Material flows for the Baseline scenario

Table 7.3 present the GHG quantifications for the Baseline scenario taking into account the considerations and assumptions presented in Table 7.2 and the LCI data presented in Appendix C. 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. The GHG emissions generated due to recycling are considerable lower (141 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 7.3: 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		**-342**

Besides, the GHG comparison of recycling between the Spain, Europe and China 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.

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 7.3.1.2 and 7.3.1.4. 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}). This 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.

7.4.2. Alternative scenarios and sensitivity assessment

The assessment of the alternative scenarios leads to the GHG results presented in Figure 7.9 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}).

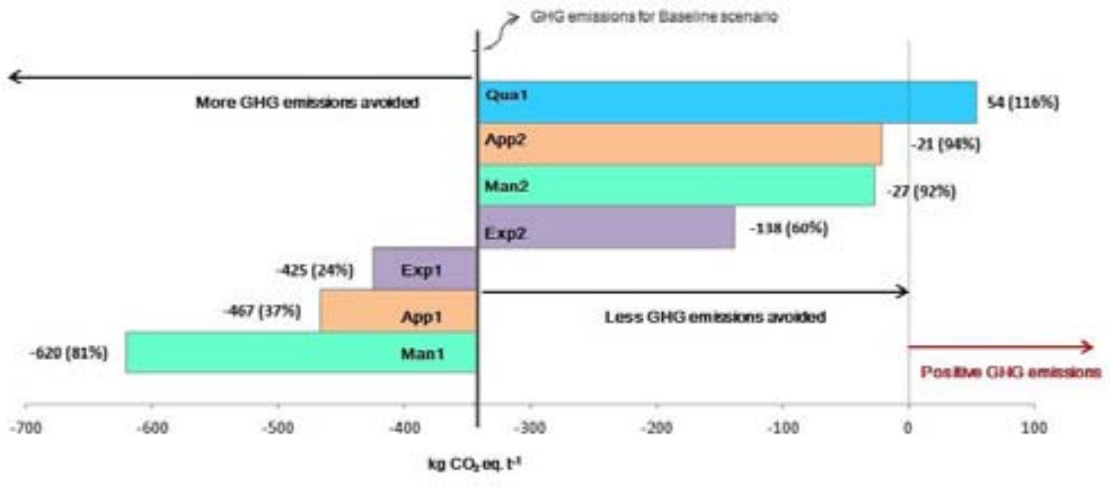

Figure 7.9: GHG emissions for the alternative scenarios (kg CO_2 eq. t^{-1})

We can observe that scenario Qua1, when the substitution ratio for recycled plastic for virgin plastics and wood is 0.5, 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 App2, when only 50% of recovered plastics were used for the substitution of virgin plastics, and scenario Man2, when 50% of the recovered plastic was sent to incineration with energy recovery, also showed an important reduction of the GHG benefits and 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 indicate that the mechanical recycling for virgin plastics substitution is better environmental option than for non plastic purposes and for energy recovery while between latter options, the results of this study suggest that the preference is not clear. Scenario Exp2 which evaluates total recovered plastic sent to global recycling (90% to China and 10% to Europe), show that this option avoids more GHG emissions than the energy recovery 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 Exp1, higher GHG benefits are obtained because the GHG emissions from the international stages are avoided. The best results are obtained for the scenario Man1 which evaluates an increase of the recovered plastic sent to recycling.

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 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 Exp1 (<1%), when all recovered plastic is recycled in Spain. Results are presented in Figure C1 in Appendix C.

7.5.Discussion

Generally in LCA studies, mechanical recycling of plastic waste is offered as the best option for EOL 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, 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 when 90% of recovered plastic was recycled (-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 scenarios Qua1, App1 and App2. Between these scenario, the best GHG quantification was obtained in scenario App1 when all recycled plastic substitute virgin plastics based on 1:1 (-467 kg CO_2 eq. t^{-1}). Contrary, the worst GHG quantification was obtained in scenario Qua1 when the substitution ratio is 0.5:1 (54 kg CO_2 eq. t^{-1}) following by the scenario App2 when the mix of plastic waste increased up to 50% and they were used for non plastic purposes (-21 kg CO_2 eq. t^{-1}). In fact, the evaluation of the GHG results per tonne of type of recovered plastic waste, also 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). Therefore, for a mixture of different plastic types, it has suggested that plastic waste should be used for energy utilization (Astrup et al., 2009; OECD, 2010). However, the results for scenario Man2 which increase the contribution of the energy recovery up to 50% suggested that the related GHG improvements are not clear (-27 kg CO_2 eq. t^{-1}). 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). In addition, for low efficiencies, landfill disposal is often preferable over energy recovery regarding GHG emissions. However, due to the fact that plastic have been commonly used for a few decades, their long-term behavior in landfill is little known but plastics in landfills decompose very slow (Kaps, 2008). The conditions within landfill may cause the chemicals contained within plastic to become more readily available to the environment (Barnes et al., 2009) and would generate other type of impacts. Therefore, plastic in landfills should not be proposed as an option due to the ignorance of their impacts in long term.

Though the best results were obtained when the recycling was increased up to 90%, we have seen that nowadays recycling appears to be restricted and to promote an increase of the recycling rate seems not realistic if other strategies and actions are not promote in parallel. Contamination was pointed out as one of the most important parameters limiting plastic waste recycling (JRC, 2012) what suggest that collection and sorting are very important processes. Currently, different strategies have been proposed at European level to increase collection and the improvement of sorting efficiencies (Plastic Recyclers Europe, 2012, BIO Intelligence Service, 2013). For instance, collection may be stimulated by the implementation of landfill bans which are currently implemented in some EU countries, such as Switzerland or Germany, and have resulted very effective to increase collection over 90%. However, in those countries with high plastic waste collection, they have high energy recovery rates (>60% for Switzerland) (Plastic Europe, 2012), so complementarily, sorting, recycling processes as well as the eco-design of plastic products must be improved (Plastic Recyclers Europe, 2012, BIO Intelligence Service, 2013). Otherwise, increase collection will increase plastic waste sent to energy recovery which is worse option regarding GHG emissions.

Nevertheless, other aspects could influence the decision between recycling and energy recovery, and 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, waste incinerators frequently recover materials for recycling (i.e., iron or aluminium) and also because plastic waste today are traded on a world market what could increase the environmental burden of the plastic waste recycling (Merrild et al,. 2012) which is an important aspect considering that it is expected an increase of plastic waste export to Asia in coming years. In this regard, the evaluation of scenarios Exp1 and Exp2 confirmed that 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). This lack of disaggregated data was also reflected in the results of the sensitivity assessment since it was impossible to modify the electricity mix for the virgin plastic production, thus the variation is less than 2% for the Baseline scenario.

7.6.Conclusions

This study demonstrates that for Spain the options for plastic waste management are with or against the waste hierarchy depending on the quality of the recovered plastic. In order to save more GHG emissions, the best options from high to low savings are mechanical recycling for quality recovered plastic (-620 kg CO_2 eq. t^{-1}), export of recovered plastic (-138 kg CO_2 eq. t^{-1}), energy recovery (-27 kg CO_2 eq. t^{-1}) and mechanical recycling for low quality recovered plastic (54 kg CO_2 eq. t^{-1}). Considering that in 2011 more than 50% or plastic waste generated was landfilled and the in-use stock, it is clear that there is a huge potential to increase collection and supply of plastic waste. Therefore, the focus should be put on increase the quality of the recovered plastics and for instance, more strategies should focus on promoting ecodesign, especially of packaging products, to ensure and facilitate the further processes of sorting and recycling. Complementarily, more research and ecodesign should be promoted to increase the applicability and demand of recycled mixes to develop more products and avoid different types of primary materials. Besides, though the transition to CDS system was outside the scope of this paper, it should be evaluated if the new model may increase the quality of the recovered plastic because the sorting process is done in origin. However, if the possibility to improve the plastic waste quality is not feasible, efforts should be put on improve the electricity efficiencies. Finally, regarding the export of plastic waste, the increasing trend of plastic waste traded internationally and the uncertainties encountered in modelling recycling of plastic waste in a global market are an important consideration for decision makers and plastic waste policies (Lazarevic et al., 2010). Thus, a new and more ambitious focus should be introduced to take into account the split between local recycling and exports.

SECTION IV

Conclusions and future research

Chapter 8-

Conclusions

8- General discussion and conclusions

A synthesis of this thesis is accomplished summarizing the main results obtained and main conclusions based on the objectives outlined in the introduction (**Chapter 2**) and **Chapters 4 to 7**. The general discussion is grouped in three areas: waste and climate change, CO2ZW ® tool, and GHG quantifications of recycling.

8.1.General discussion

8.1.1. The contribution of MSW management to climate change mitigation

To assess which is the best option to increase material savings and decrease GHG emissions from the Spanish MSW management, firstly we evaluated a sensitivity assessment to detect the most sensitive parameters for the calculation of GHG emissions. In this sensitivity assessment we evaluated the influence of:

- waste data;
- waste collection and transport;
- waste treatment;
- waste recycling credits referred as CO_2 credits.

Results of the assessment demonstrated that the CO_2 credits and the percentage of biogas capture in landfills are the most sensitive parameters for the GHG emissions. For both parameters, we used literature data to conduct the assessment and also for both parameters, the variations for the GHG emissions were over 30%.

Secondly we evaluated the GHG emissions of the Spanish MSW management in 2008, referred as Base scenario, and the GHG emissions of three alternative scenarios (Scenarios A, B and C) by assessing the influence of:

- material recycling due to the selective collections and material recovery in MBT plants;
- energy recovery due to the biogas capture in landfills;
- Different treatments for the mixed waste fraction.

Results of the evaluation showed that an increase in recycling rates along with the elimination of mixed waste disposed in landfill can contribute considerably to reduce GHG emissions from MSW management. Overall GHG emission reduction from Base scenario was 40%, 51% and 64% for scenarios A, B and C, respectively. **Figure 8.1** summarizes the scenarios evaluated and total GHG emissions obtained.

	Recycling rate (%)	Biogas capture (%)	Mixed waste MBT (%)	Mixed waste incineration (%)	Mixed waste landfill (%)
Base scenario	30	17	45.7	9.1	45.3
Scenario A	56	17	90.40	9.1	0
Scenario B	45	17	45.7	54.4	0
Scenario C	55	30	68.3	31.7	0

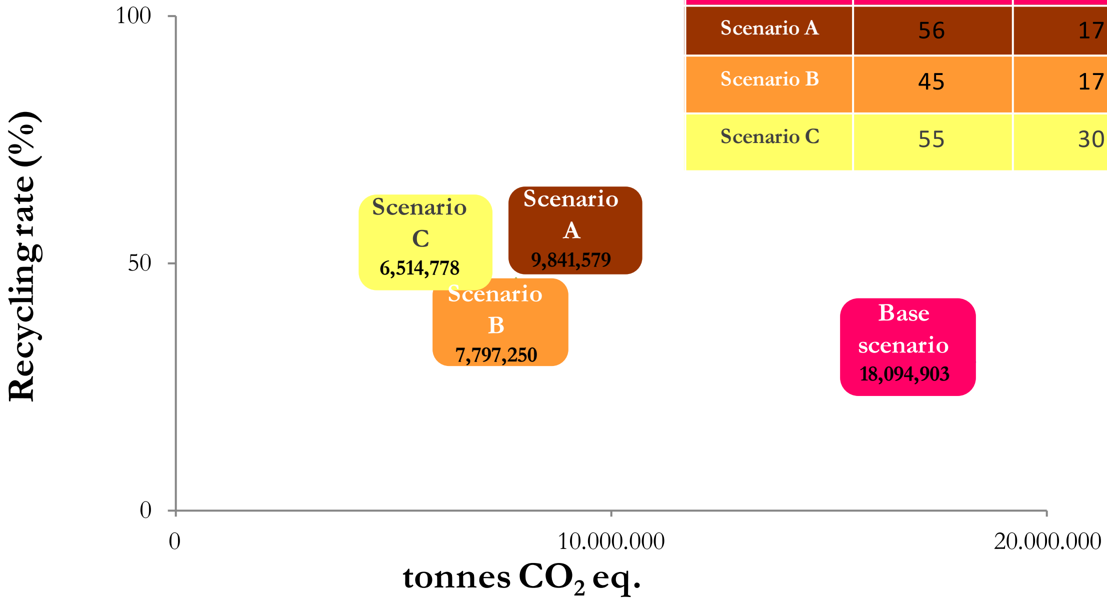

Figure 8.1: Total GHG emissions for the Base scenario and Scenarios A, B and C for the Spanish MSW management evaluation (2008)

As it is observed, although scenario A and scenario C have similar recycling rates (56% and 55%, respectively) GHG emissions in scenario C are lower. In fact, scenario A generated more GHG emissions than scenario B though the recycling rate was lower (45%). These results can be explained by evaluating the **direct GHG emissions** due to treatments themselves, the **indirect GHG emissions** due to the use of material and energy in the treatment plants and the **avoided GHG emissions** due to material recovery and energy recovery.

Regarding the direct GHG emissions from MSW management, **landfill was confirmed as the main LC stage affecting the direct GHG emissions due to the organic matter degradation of mixed waste sent without any previous treatment to landfill as well as due to the organic matter degradation of residual waste from MBT plants**. The incineration process generates more direct GHG emissions than the MBT plants but if the residual waste is sent to landfill, total direct GHG emissions from MBT plants are higher than those for the incineration process. Figure 8.2 summarizes graphically these results.

Figure 8.2: Direct GHG emissions per tonne of mixed waste treated in Spanish MSW treatment plants

Concerning the avoided GHG emissions, the sensitivity assessment has revealed that results of the GHG emissions are very sensitive to the CO_2 credits. Thus, **the potential to avoid GHG emissions is highly dependent on the values of CO_2 credits (also referred as GHG quantifications)**. In the evaluation of the Spanish MSW management we used same literature values for all scenarios, so we only observed the influence of the amount of material recovered. The literature review revealed that there are important CO_2 credits variations between the different sources consulted. Main differences between the CO_2 credits were due to the inventory data and different methodological assumptions. **Considering that none of the CO_2 was from Spanish data, we decided to calculate these values for Spain in order to be more accurate**. This was the purpose of **Chapters 5, 6 and 7** and section 8.1.3 discusses main results obtained.

Finally, the indirect GHG emissions have the lowest influence on total GHG emissions and they were equivalent to 2%-4% of direct GHG emissions.

Considering these results, in Scenario A, 90.9% of mixed waste was treated in MBT plants what contributed to increase the material recovery but also to increase the GHG emissions from residual waste disposed in landfills. Therefore, in comparison with scenarios B in which only 45.7% of mixed waste was treated in MBT plants, fewer materials were recovered but less GHG emissions were generated in landfill. Thus, total GHG emissions were lower than in Scenario A.

In Scenario C, 68.3% of mixed waste was disposed in landfill what contributed to generate more GHG emissions in landfill than in Scenario B but less than in Scenario A. However, the material recovery in MBT plants and the increase of biogas capture in landfill contributed to decrease the total GHG emissions. Consequently, this scenario represented the best situation to GHG emissions mitigation.

To eliminate landfill of mixed fraction by sending 68.3% to MBT plants and 30.7% to incineration plants; to increase the biogas capture in landfill from 17% to 30%; and to increase the recycling rates from 30% up to 56%, imply a 64% reduction of GHG emissions from Spanish waste management in 2008.

8.1.2. The accuracy of CO2ZW® tool for GHG emissions

The evaluation of Spanish waste management in 2008 and the alternative MSW schemas was conducted with the CO2ZW®. **It demonstrated to be very useful and suitable tool to quantify the GHG emissions and evaluate possible future strategies**. Though it is outside of the scope of this thesis, the CO2ZW® has been applied to quantify the GHG emissions from MSW management for all municipalities of Catalonia and also to other municipalities and regions outside of Spain. In addition, it has been recently updated to calculate the GHG emissions from urban waste at industrial areas.

In comparison with other available tools, **the CO2ZW® tool includes the key LC stages and current European MSW treatments of collection & transport, sorting facilities, biological treatments, incineration treatments and landfill deposition**. In addition, it incorporates and allows the modification of the principal parameters discussed in literature affecting the calculation of the GHG emissions; such as the differentiation of biogenic and fossil C content or the C sequestration in landfill. It has **also default values for Spain, Catalonia, Italy, Greece and Slovenia, average data for Europe as well as default data for the CO_2 credits.** This constitutes a huge advantage since it is possible to quantify the GHG emissions though there is a lack of information what improves the accessibility and versatility of the tool. For example, in the evaluation of Spain, we used the national average data for biogas capture and literature data for the CO_2 credits.

Besides, the CO2ZW® tool includes the **two approaches more used and proven in the international scientific framework of GHG calculations from landfills, the IPCC methodology and the LCA approach.** In this regard, the IPCC methodology is normally used to calculate national inventories. However, it requires inventory data from 50 previous years, what for most municipalities, regions or nations it is not feasible. Therefore, default data of GHG emissions in previous years were included depending on the climate and area of landfill. The LCA approach considered the future emissions and it is more suitable for evaluate alternative scenarios and waste management improvements, as the evaluation of 2008 and the alternative scenarios have showed.

However, there is still a lack of complementary assessment which are currently not include in the CO2ZW® such as uncertainty, economic or social which can be useful in the decision-support process. Besides, the **GHG emissions due to collection and transport should be also improved** as the GHG emissions due to this LC are based on emissions factors for urban collection and interurban transport from ecoinvent.

Quantifying GHG emissions from MSW has been crucial and tools have been promoted to help in the decision-support processes. However, tools with local data to calculate the GHG emissions from MSW management are necessary to have accurate results and reflect local characteristics. The CO2ZW® is the first tool which includes Spanish default data and it has demonstrated to be a powerful tool for the environmental comparison of different options for waste management.

8.1.3. GHG quantifications of recycling

To quantify the GHG quantifications of recycling considering market effects and the international trade, we proposed the integration of MFA and CLCA. **The studies conducted for paper, aluminium and plastic have demonstrated the potential of this integration. The methodological framework has offered a detailed description of the life cycle and the interrelationships between producers and consumers within a global economy (both raw material and waste). Consequently, the integration has allowed having more realistic and consistent data of waste, raw materials and markets which are the base of accurate GHG quantifications.**

8.1.3.1. MFA to evaluate past trends and forecast future trends in Spain

As the result of the application of dynamic MFA, we observed past trends of production, consumption and trade to and from Spain. In addition, we calculated the accumulation of paper, aluminium and plastics products as in-use stocks as well as the future waste generation of this in-use stock achieving its EOL. Finally, based on these results, we forecasted some scenarios for the GHG quantifications.

❖ **Past trends in Spain**

Figure 8.3 summarizes graphically the results obtained of production, consumption, import and export of raw materials and waste. Main results are that 1) **trade to and from Spain of virgin pulp, primary aluminium and virgin plastic increased** with more imports and exports; 2) **after 2007 there were a slight decrease in national consumption and production**; 3) and **trade to and from Spain of waste paper, aluminium old scrap and plastic waste also increased, specifically with more waste exported for international recycling**.

Between 2006 and 2011 waste paper exports increased over 104% mainly to China; between 1995 and 2010 old scrap exports over 800% with main destination shifted from Europe to China; and between 1999 and 2011 plastic waste exports increased up to 2,341%. Following the trends of waste paper and old scrap exports, the main destination of recovered plastic was China.

❖ **Accumulation of paper, aluminium and plastics products in Spain**

Through the dynamic perspective we quantified the in-use stock and **in the case of paper products, the accumulation is under 20% per year while in the case of aluminum and plastic, it is estimated that the annual accumulation is about 85% and 65%, respectively**.

The differences are due to the life time of products. The lowest lifetime is for packaging products which is under one year. In the case of aluminium, between 3% and 4% correspond to packaging products. However, in the case of plastics, packaging products correspond to 45% on average, thus, there is less plastic products accumulated each year. Between **2006 and 2011, 376,373 tonnes of paper products were accumulated; between 1995 and 2010, 9,688,798 tonnes of aluminium products; and between 1999 and 2011, 27,034,083 tonnes of plastic products.**

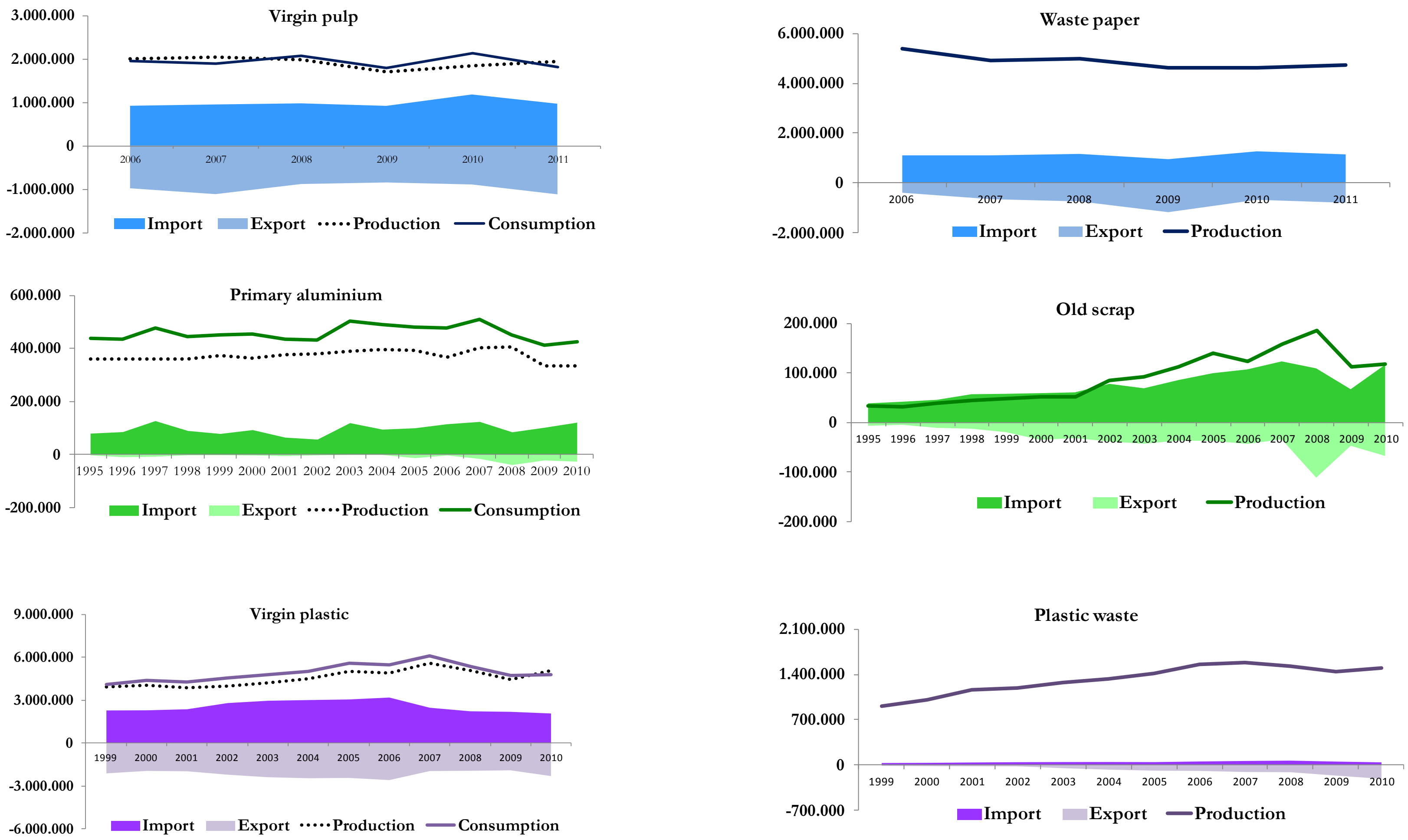

Figure 8.3: Spanish production, consumption, import and export of virgin pulp, primary aluminium, virgin plastics; and waste paper, old scrap and plastic waste

❖ Waste generation projections of aluminium old scrap and plastic waste

According to Melo (1999), future waste generation from the in-use stock could be calculated using different lifetime distribution models. **We calculated the future old scrap generation and post-consumer plastic generation up to 2020 with a normal lifetime distribution based on European data** (Ciacci et al., 2013, Kaps, 2008). The methodology followed is presented in Appendix D. Figure 8.4 and Figure 8.5 presents the potential waste generation obtained. In addition, it is also represented waste collection for a variation of recovery rate between 50% and 90%. For 2020, there is a potential **generation of aluminium old scrap and of plastic waste of 464,380 tonnes and 2,755,242 tonnes**, respectively.

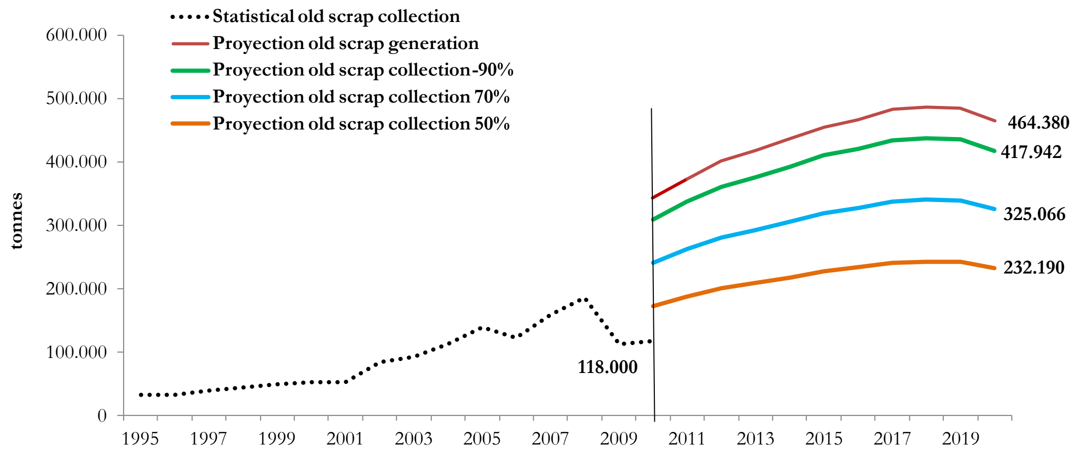

Figure 8.4: Aluminium old scrap collection between 1995 and 2010 and aluminium old scrap projections (2010-2020)

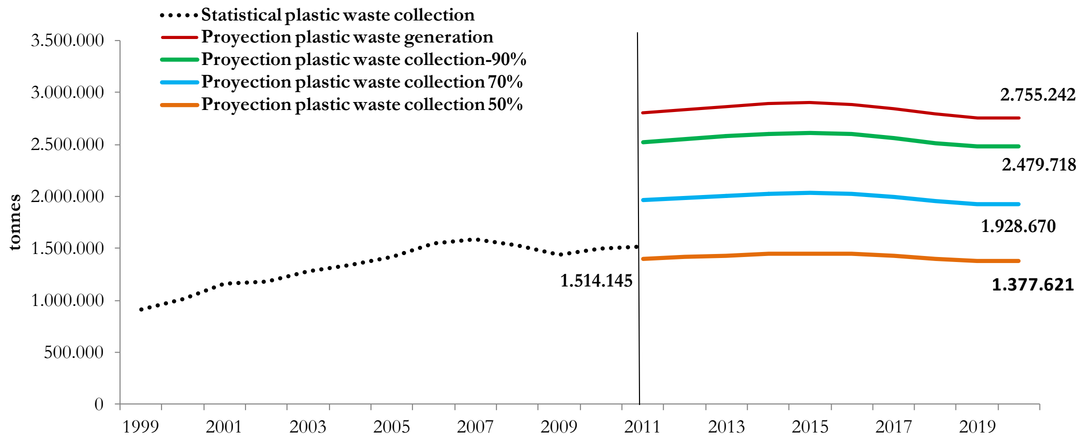

Figure 8.5: Plastic waste collection between 1999 and 2011 and plastic waste projections 2011-2020

❖ **Forecast trends for the GHG quantifications of recycling**

Based on all these trends and calculations, **we forecasted scenarios for the quantification of GHG emissions resulting from an increase in waste collection**. It is important to note that scenarios are not predictive, they describe futures that could be rather than what will be. Their role is to explore alternative futures and evaluate how different actions would play out in terms of GHG emissions (Barrett and Scott, 2012).

Corrugated material production has shifted to China as has the demand for waste paper (IVEX, 2010). However, far from being a temporary adjustment, this shift seems to be a long-term trend. It has been predicted that China's demand for recovered paper is likely to increase in the future (BIR, 2011). Consequently, the volume of the trade of recovered paper will increase in the future and it will likely affect the market behavior and dynamics of recovered waste as it has affected previous years. **We forecasted that an increase in waste paper supply will rely in an increase of waste paper exports of low quality to China**.

Demand for secondary aluminium will increase due to the increasing application of aluminium in light vehicles. It has also been predicted that the aluminium industry will be displaced to developing countries (Menzie et al., 2010; JRC, 2007). Thus, if the primary aluminium industry is affected and consequently the semifinished industry, but at the same time more old scrap is available due to more in-use stock at the end of its life, **the export of old scrap would increase in the future to countries with high demand, which are almost all derived from developing China**.

For **plastic waste, in short term, we forecasted similar effects of waste exports to China**. We evaluated also the option of increasing the energy recovery rates and we also evaluated several scenarios based on the effects of the quality and recycled plastic applications.

The use of MFA has been demonstrated to be useful for the evaluation of past trend and current trends as well as for the in-use stock calculations. The consideration of this accumulation is important for future waste scenarios. If the observed trends are not reversed, we can expect that an increase of waste supply will rely on an increase of waste exports, mainly to China. The increasing globalization of raw materials and waste makes the optimization of waste management strategies and policies quite challenging justifying the need of new approaches to identify the consequences.

8.1.3.2. CLCA to quantify the effects of markets and international trade

For the GHG quantifications, we assumed that the consequences of recycling are that raw material productions are avoided. However, avoided productions can be located locally or globally depending on the marginal identification. In addition, a percentage of waste were exported for its international recycling, and we have also forecasted an increase of this waste exports. As the result of the application of CLCA, we identified the marginal process. Moreover, we quantified the GHG emissions from recycling based on these marginal processes as well as GHG emissions for the forecast scenarios. Finally, based on these results, we evaluated the influence of markets and the international trade.

❖ **Marginal identifications**

The marginal identifications of raw material were based on reviewed studies which conducted market assessments and followed the step procedure of Weidema et al. (2009). They concluded that raw material markets of virgin pulp and primary aluminium are global and increasing. This means that:

- ✓ the marginal process should be identified in the global market;
- ✓ the most competitive processes on the global market are the marginal ones;
- ✓ the marginal processes will be the most affected by changes in supply and demand;
- ✓ these processes will be affected by recycling.

In the case of paper, the **marginal virgin pulp was the Brazilian BEKP production** identified by Reinhard et al., (2010). The marginal primary aluminium was based on the identification conducted by Schmidt and Thrane (2009), which consist of **a mix of primary aluminium production in China, Russia and Middle East**. Regarding **virgin plastics, the marginal identification was not available and thus the European average production was used as marginal process**. We referred these scenarios as base scenarios in which the increase in one tonne for recycling would substitute the global marginal processes.

Nevertheless, **Spain is a traditional producer of BEKP, primary aluminium and virgin plastic**. Thus, it would be logical to suggest that **waste recycling in Spain would avoid the Spanish raw material productions**. In addition, the Spanish BEKP production and the Spanish primary aluminium production were highlighted as ones of the least competitive on the market (James, 2012; EAA, 2012a; Reinhard et al., 2010). Thus, with the aim of evaluating the market effects between the global and most competitive processes and the local and least competitive processes, we also calculated and evaluated the GHG quantifications based on the least competitive processes. We referred these scenarios as alternative scenarios in which the increase in one tonne for recycling would substitute the Spanish marginal processes. Figure 8.6 summarizes graphically all this information. **We assumed in all scenarios that international recycling substitute global marginal process**.

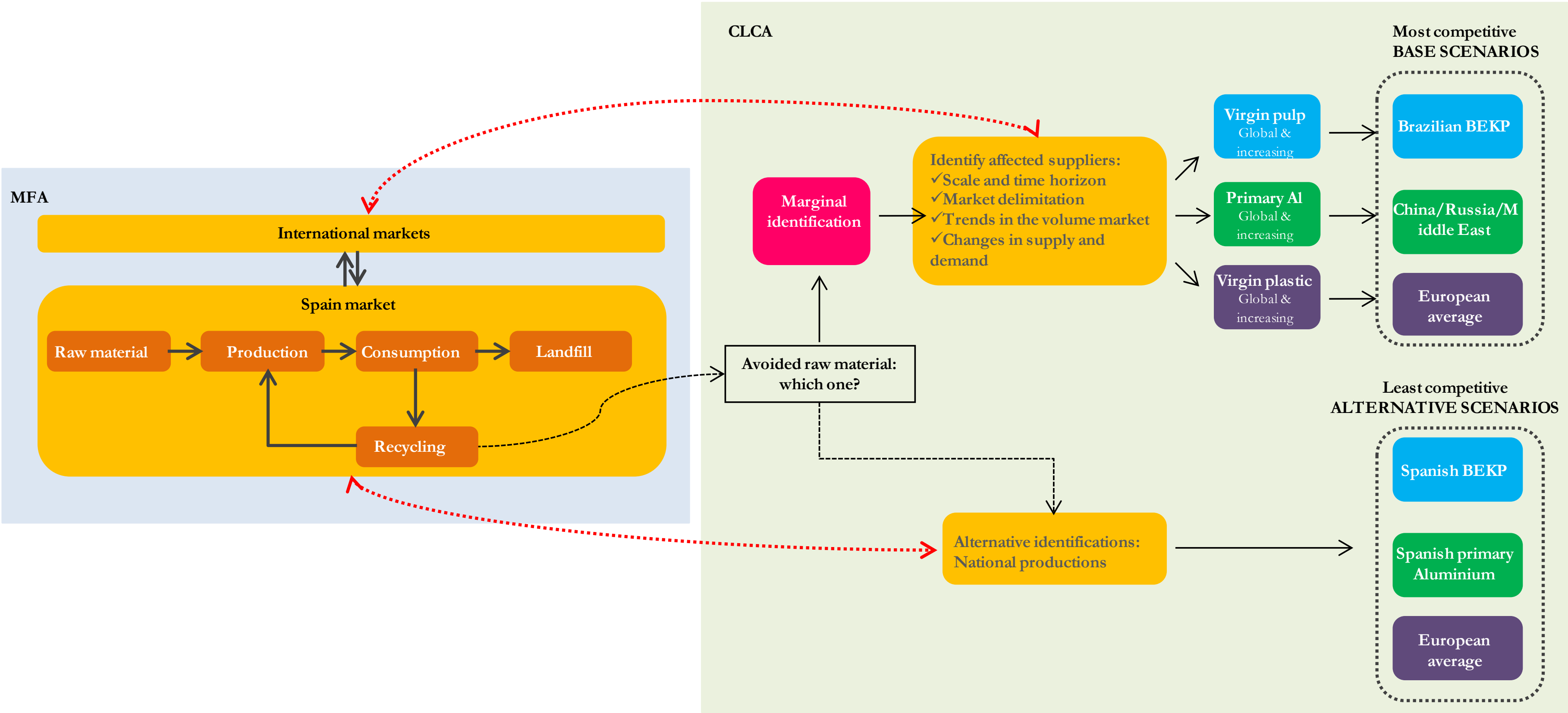

Figure 8.6: The interrelation of MFA and CLCA with the marginal identification procedure based on literature review for the base scenarios and assumptions for the alternative scenarios

❖ **The influence of markets on the GHG quantifications**

Results of the GHG quantifications showed that recycling carries significant GHG savings and the market considerations gave to significant differences between the base scenarios and the alternative scenarios. **The global marginal identifications, which are the most competitive processes, generated more GHG emissions. Thus, to avoid their production by recycling will lead to higher GHG savings**. In order to evaluate the influence of the marginal identifications, Table 8.1 presents the GHG results obtained for the Base scenarios and the alternative scenarios when the increase of one tonne of each waste is totally recycled in Spain.

The differences are justified because the Spanish BEKP production and Spanish primary aluminium production generate fewer GHG emissions as these processes have revealed to be more efficient than the most competitive on the global market. In fact, inventory data for the BEKP production and the primary aluminium production were obtained from literature (González-García, 2009a; González-García, 2009b; Schmidt and Thrane, 2009) and from the BAT technologies, which represent the best technologies currently available. Inventory data for the global marginal productions were also obtained from literature and in comparison; more energy and materials were consumed.

In the case of plastic waste, data is aggregated and the European average production is used as the virgin plastic production avoided in both cases. However, the European marginal electricity mix was used for the base scenario and the Spanish marginal electricity mix for the alternative scenario. The variation is less than 2%.

Table 8.1: GHG quantifications of recycling per tonne collected and recycling in Spain for the Base scenarios and alternative scenarios (0% of waste export)

	Base scenarios (Global marginal) kg CO_2 eq. t^{-1}	**Alternative scenarios (Spanish marginal) kg CO_2 eq. t^{-1}**	**Difference (%)**
Waste paper	-334	-78	77
Aluminium old scrap	-18,403	-5,183	72
Plastic waste	-425	-423	2

❖ **The influence of the marginal electricity mix on the GHG quantifications**

The marginal electricity mixes have increased these differences. Marginal electricity mixes with higher content of hard coal generate more GHG emissions per kWh generated. As reflected in Figure 8.7, the marginal electricity mix of Spain has the lowest content of hard coal. The marginal electricity mixes were calculated based on the guidelines of Schmidt et al. (2011). These marginal electricity mixes were calculated assumed that they represents the long-term marginal supply based on information on electricity capacity and generation for the years 2008-2010 (statistical data) and 2020 (outlook data).

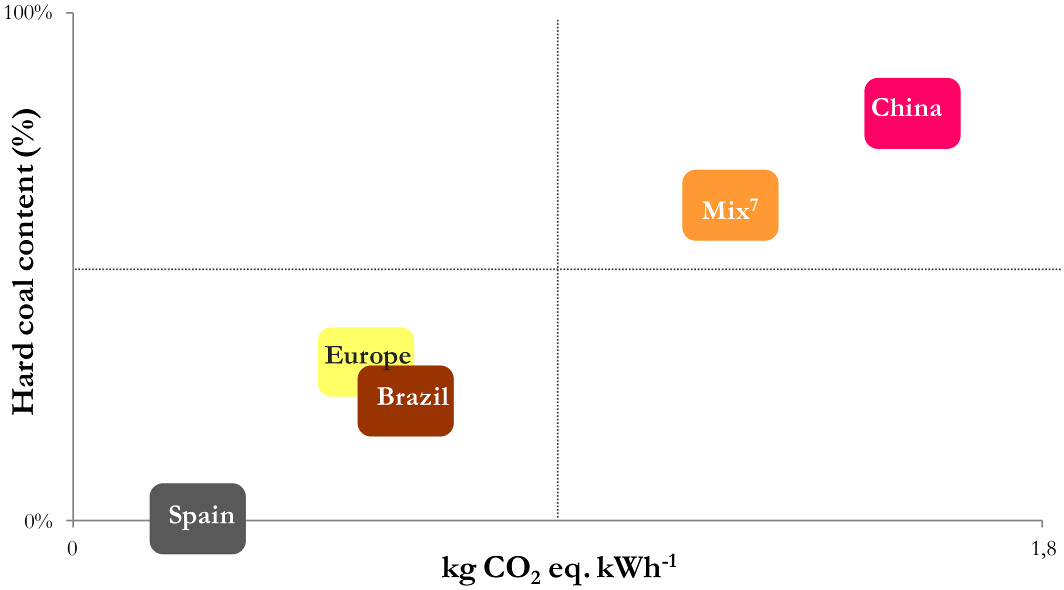

Figure 8.7: GHG emissions per kWh of long-term marginal supply (2020) of electricity depending on the hard coal content contribution[3]

❖ **The influence of the international trade on the GHG quantifications**

For all scenarios evaluated, except for the export of aluminium old scrap in the alternative scenario, savings of recycling decreases while increase the export of waste GHG. The consequences of increase the international transport are reflected in the GHG emissions generated due to international recycling processes.

Recycling processes in China generates more GHG emissions due to **lower efficiencies** but as reflected in Figure 8.7 also due to the **marginal electricity mix**. When recycling is conducted in Europe, although same inventory data is used as in Spain, higher GHG emissions are also generated due to the European marginal electricity mix. Therefore, more emissions are generated for the same quantity of waste treated.

[3] "Mix" represents the marginal electricity mix from the electricity mixes of China, Russia and Middle East

Furthermore, to increase the export of waste, decrease the benefits of the GHG emissions because the environmental impact of transport increases. However, we considered that the international recycling avoids the marginal global process. Thus, when increasing waste exports more or less GHG emissions were saved depending on the balance of the international transport, international recycling and the product substituted. Table 8.2 presents the GHG quantifications of recycling for the Base scenarios and the alternative scenarios when one tonne collected in Spain is recycled internationally.

Table 8.2: GHG quantifications of recycling per tonne collected in Spain and recycling internationally for the Base scenarios and alternative scenarios (100% of waste export)

	Base scenarios (Global marginal) kg CO_2 eq. t^{-1}	Alternative scenarios (Spanish marginal) kg CO_2 eq. t^{-1}	Difference (%)
Waste paper	6	6	0
Aluminium old scrap	-17,352	-18,160	5
Plastic waste	-138	-128	7

It is important to highlight that if all **waste paper** is sent to international recycling, 6 kg of CO_2 eq. are emitted in the Base scenario and the alternative scenario. As a consequence, **the GHG emissions created outweigh the GHG benefits of recycling**. In the case of aluminium, if the Spanish primary aluminium is the process avoided, to increase the international recycling generates more GHG savings as the GHG emissions created are much lower than avoided for the global primary aluminium production. In the case of plastic waste, due to the use of same inventory data, the variation is less over 7%.

Figure 8.8 summarizes graphically the GHG quantifications when all waste is collected and recycled in Spain (0% of waste export) and the recycling avoids the global marginal or the Spanish marginal, and when all waste is collected and recycled internationally (100% of waste export) and the recycling avoids the global marginal or the Spanish marginal.

- **The influence of the waste quality on the GHG quantifications**

Depending on the quality of the recycled fibers, the pulp should be mixed with virgin fibers to produce better grades (BIO Intelligence Service, 2011). We assumed that **1 ton of recovered fiber is not equivalent to 1 ton of virgin pulp and we assumed the equivalence ratio of 0.8:1** (virgin pulp: recovered fiber) (Ekvall, 2000). Moreover, although most metals can be recycled any number of time without loss of quality (BIO Intelligence Service, 2011), **aluminium old scrap** recycling requires the addition of 5% of primary aluminium to sweeten the melt and obtain the desired alloy mix (Cullen and Allwood, 2013). Therefore, we considered **the equivalence ratio of 0.95:1 (primary aluminium: old scrap)**. For both wastes, all GHG results were calculated based on these ratios.

In the case of plastic, **the options for use of recycled plastic depend on the quality and polymer homogeneity of the material** (JRC, 2012). **If the polymer is clean** and contaminant-free **it can be used to substitute virgin plastic**. We assumed that the recycled plastic waste substitute virgin plastics with a substitution ratio of 1:1. However, **if the polymer is mixed with other polymers**, the options for often involve down-cycling of plastics for cheaper and **less demanding applications** such as garden furniture (JRC, 2012) or **for energy recovery**. As a result, we also evaluated other scenarios to asses other substitution ratios, different recycled plastic applications, and plastic waste to energy recovery. **The results showed that the energy recovery avoids 27 kg CO_2 eq. t^{-1} while recycling of low quality recovered plastic emits 54 kg CO_2 eq. t^{-1}**.

The use of CLCA has demonstrated to be useful to quantify the GHG emissions of recycling by considering the current market trends and material flows. Results showed that there are significant differences in the GHG quantifications if the process substituted by recycling is identified in the global market or in the local market. In addition, results showed that there are also considerable variations if the exports of waste are taken into account. Forecast scenarios projected an increase of waste exports, thus, their consequences on the GHG savings and loss of valuable resources should be considered in waste management policies. Reality is complex, and to consider that neither the raw materials and waste nor the resulting GHG estimates are affected by the market dynamics is incorrect and incomplete. It is necessary that the method for accounting the GHG impact of recycling reflect the market mechanism.

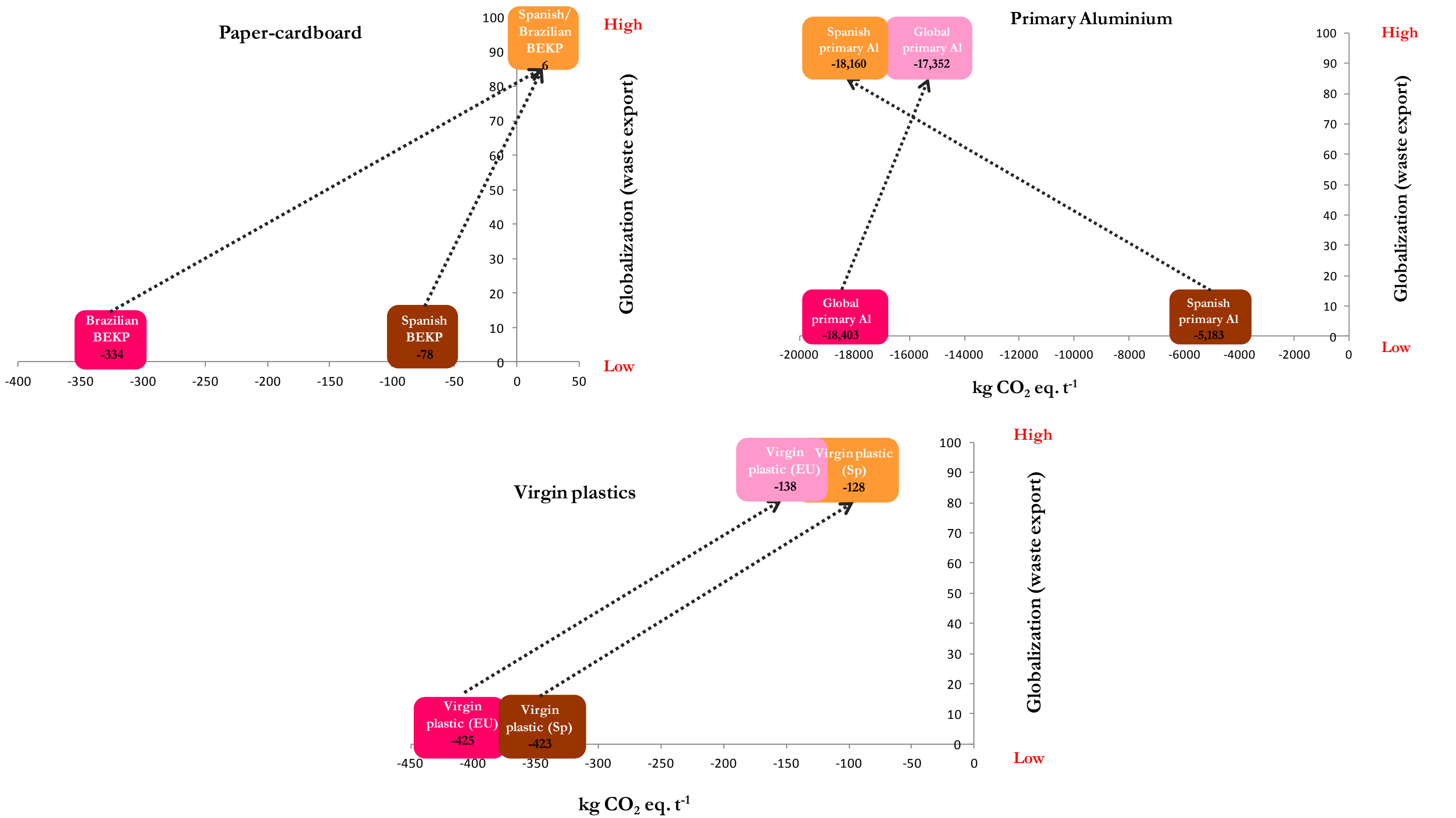

Figure 8.8: Summary of the GHG quantifications of recycling for the three materials studied if the marginal process avoided is global or local and everything is recycled in Spain (dark pink and dark brown, respectively), and if marginal process avoided is global or local and everything is recycled internationally (soft pink and orange, respectively)

8.1.3.3. *Potential material savings and GHG reductions in Spain*

Table 8.3 presents material recovered and GHG emissions that would be saved in 2020 due to the recycling. In the case of aluminium old scrap and plastic waste, based on calculations by the normal distribution, we projected one scenario in which waste recovered correspond to the accomplishment of EU waste targets, and other scenario in which all potential waste is recovered. For waste paper we considered an increment of 10% from data of collection of 2011. For the calculations we used the GHG quantifications obtained on the Base scenarios.

Table 8.3: Material recovered and GHG emissions avoided in 2020 due to the recycling of waste paper, aluminium old scrap and plastic waste by using the GHG quantifications of this thesis for the Base scenarios

		Directive targets		100%	
	kg CO_2 eq. t^{-1}	Tonnes	t CO_2 eq.	Tonnes	t CO_2 eq.
Waste paper	-317	5,195,300	-1,646,910	5,195,300	-1,646,910
Aluminium old scrap	-18,140	314,169	-5,699,026	464,380	-8,423,853
Plastic waste	-342	1,104,621	-377,780	2,755,242	-942,293
Total	**-**	**6,614,090**	**-7,723,716**	**8,414,922**	**-11,013,056**

It is clear that there is a huge potential of material recovery and GHG reductions through collection and recycling. At European level, it is projected that material savings due to collections of paper, aluminium and plastic will be equivalent to 50.3 Mt, 2.1 Mt and 16.1 Mt, respectively, if EU targets are achieved (BIO Intelligence Service, 2011). Tonnes projected in the Directive scenario are equivalent to 10% of those projections. Regarding GHG emissions, the avoided emissions of GHG from MSW management will be equivalent to 115 Mt of CO_2 in 2020 (EEA, 2008). The GHG savings obtained in this work are equivalent to 7% of the projected European savings. However, it should be taken into account that MSW only include mainly data of packaging waste. Thus, considering only packaging data from the forecast future waste projections, total material savings will be equivalent to 8% and GHG savings will be equivalent to 3% of total European reduction.

The contribution of material savings and GHG reduction through recovery and recycling of waste paper, aluminium old scrap and plastic waste is significant. However, it should be noted that the efforts made in recent years to increase collection rates and improve collection systems are not being rewarded because the benefits of recycling are occurring more and more in other countries. Moreover, the results suggest that if the market rules remain the same, potential material savings and efforts to reduce the impacts associated with primary productions will be lost as production moves to other countries with higher environmental impacts.

8.2.Conclusions

This thesis determined that there is a significant potential of material savings and GHG emissions reductions through elimination of mixed waste disposed on landfills along with increase recovery and recycling. However, tools with local data for the GHG emissions from MSW management as well as markets and international trade, need to be taken into account in order to have accurate projections and quantifications but also in order to select the best waste management scenario within current and future global economy.

This section presents the conclusions of the thesis within the framework of the objectives listed in **Chapter 2** of the thesis. Conclusions are grouped in CO2ZW ® tool, contribution of MSW management to climate change and GHG quantifications of recycling.

8.2.1.About the CO2ZW® tool (Objectives 1, 2 and 3)

- The CO2ZW® tool has been demonstrated **as a suitable tool for quantifying the GHG emissions from waste management, for guiding policies in waste management, as well as for developing by and for studies and policy makers** that need to evaluate the GHG emissions at the municipal, regional or national levels with a small amount of required input data.

- The **main advantage** in comparison with other tools is **the incorporation of default data** for the countries included while the **GHG calculation from collection and transport is the main disadvantage**. Besides, the **most sensitive parameters** for the GHG calculations **are biogas capture and the CO_2 credits**.

8.2.2. Contribution of MSW management to climate change (Objectives 4 and 12)

- In order to decrease the GHG emissions from MSW management, **the best strategy is to diminish deposition in landfill of mixed waste, to increase biogas capture in landfills and to increase the selective collections**.

- **For the mixed waste fraction, results showed that complementary solutions of MBT plants and incineration plants are the best way to reduce the GHG emissions and recovered materials.** The incineration process generates more GHG emissions than the MBT process itself; however, disposal in landfills of residual waste from MBT plants still generated 0.88 kg of CO_2 eq. per tonne.

- **Biogas capture should be increased on landfills**. It is one of the most sensitive parameters affecting the GHG emissions and considering data of 2008, the potential decrement of GHG emissions and the potential increment of avoided GHG emissions are 1.2% and 4.5%, respectively, for every 1% of biogas capture increase.

- However, the **potential of GHG reductions is higher if the future waste projections and the GHG quantifications obtained in this thesis are taken in account**. The potential GHG reduction is **7,723,716 tonnes of CO_2 eq.** equivalent to 7% of total expected European GHG reduction from MSW management in 2020.
    - Calculations were conducted for waste paper, aluminium old scrap and plastic waste. Thus, by considering all MSW fractions, the potential savings are even higher.

- The **potential material saving through recovery of waste paper, aluminium old scrap and plastic waste is 8,414,922 tonnes** which is equivalent to 12% of expected European recovered amounts.

8.2.3.About the GHG quantifications of recycling (Objectives 5, 6, 7, 8, 9, 10, 11 and 12)

- The **integration of the MFA and CLCA has been demonstrated as an effective method for evaluating waste flows and estimating GHG emissions of recycling process within a market context and international trade**.

- **MFA has been demonstrated to be useful for the evaluation of past trend flow and to calculate the accumulation of products through past consumption which is essential for future waste scenarios.** The calculation of the future waste generation in 2020 from this in-use stock achieving its EOL showed that **there is a potential waste generation of 464,381 tonnes** of old scrap, what it implies an increase of 294% from data of 2010, **and 2,755,242 tonnes of plastic waste** what it is an increase of 82% from data of 2011.
    - If trends are maintain, we can expect that an increase of waste supply will rely on **an increase of waste exports in coming years, mainly to China**.

- In relation to the marginal identifications, we observed that imports of virgin pulp and primary aluminium came from countries identified as marginal producers. Although the marginal identification may imply some uncertainty and can be discussed, the increase in imports is in line with the identified marginal suppliers. This suggests **that**

more competitive productions are displacing consumption from Spanish productions.

- Regarding the GHG quantifications of recycling, we have confirmed that **there are significant differences if market effects and the international trade are considered. The three more important factors affecting the GHG emissions are the marginal production avoided due to recycling, the marginal electricity mixes, and the export of waste**. All of them are related but not necessarily linked.

    - **Global marginal productions generate more GHG emissions than the Spanish marginal productions. The consequence is that to avoid the global marginal productions due to recycling, avoids more GHG emissions**. Global productions are less efficient and are produced in countries with higher carbon content in their electricity mixes. The GHG emissions savings are: -**334 kg CO_2 eq. t^{-1} for waste paper, -18,403 kg CO_2 eq. t^{-1} for old scrap and -425 kg CO_2 eq. t^{-1} for plastic waste**.

        - If the Spanish productions are the one avoided, the GHG emissions savings are: -**78 kg CO_2 eq. t^{-1} for waste paper, -5,183 kg CO_2 eq. t^{-1} for aluminium old scrap and -423 kg CO_2 eq. t^{-1} for plastic waste**.

        - However, as a result of more efficient productions with a cleaner marginal electricity mix compared to the marginal producers, **Spain is in a good position to prevent global GHG emissions if national production is maintained. Therefore, to promote a Spanish CE with more national productions and national recycling would be an opportunity to decrease the GHG emissions globally**.

    - **Higher hard coal content on the marginal electricity mixes generates more GHG emissions per kWh**. Thus, the consequence is that for same inventory data, higher GHG emissions are emitted in countries with higher coal content in its marginal electricity mix.

    - **Export of waste should be avoided as it represents a decrease of the GHG benefits. In addition, export flows are against the objectives of the CE, and from a material point of view**, it is essential to reverse the increasing trend in the export of waste because it allows importers in other regions to capture a key resource. **A new focus should be introduced to take into account the split between local recycling and exports.**

- If all **waste is exported and the global marginal production avoided**, the GHG results are **6 kg CO_2 eq. t^{-1} for waste paper, -17,352 kg CO_2 eq. t^{-1} for aluminium old scrap and - 138 kg CO_2 eq. t^{-1} for plastic waste**.

- If all **waste is exported and the Spanish marginal production avoided**, the GHG results are **6 kg CO_2 eq. t^{-1} for waste paper**, **-18,160 kg CO_2 eq. t^{-1} for aluminium old scrap** and **-128 kg CO_2 eq. t^{-1} for plastic waste**.

- **Regarding material and GHG trends, results obtained for consumption, production and trade**; except for the export of old scrap, were obtained. **However, depending on the source of raw material and on the intrinsic material differences, the potentialities to increase the GHG benefits of recycling varied.**

  - **Waste paper is a** resource coming from renewable sources and **their recycling avoids wood, which can be used for bio-energy purposes**. Thus, this second system expansion could increase the GHG balance of waste paper recycling.

  - **The smelting process to produce primary aluminium production is highly electricity consumer and the GHG results are very dependent on the marginal electricity mixes for this process.** In comparison, recycling contribute to generate lower GHG emissions. Thus, recycling of aluminium old scrap results in significant GHG savings.

  - Consumption and use of virgin plastic has diversified and increased sharply in recent decades due to the different material properties and their easy and cheap production. However, due to such versatility, **plastic waste as substitutes for virgin plastic is still very limited**. The use of plastic waste for energy recovery or as a substitute for other material results in GHG balance close to zero (**-27 kg CO_2 eq. t^{-1}**) or even positive (**54 kg CO_2 eq. t^{-1}**). Therefore, the focus should be put on increase the quality of recovered plastics and more strategies should focus on promoting ecodesign to ensure and facilitate the further processes of sorting and recycling.

- In the future, **GHG quantifications of recycling should be based on the flows described by MFA analysis and quantified by CLCA**. It is necessary to have a methodology that maps properly all the material flows as a first step in determining the potential environmental impacts to facilitate the accurate accounting of GHG for decision-making. The results not only help researchers to evaluate the GHG emissions from waste management but also can be used by producers, waste managers and waste politicians to evaluate and propose the best strategy to reduce the resource consumption and the GHG emissions. Figure 8.9 summarizes the proposal approach.

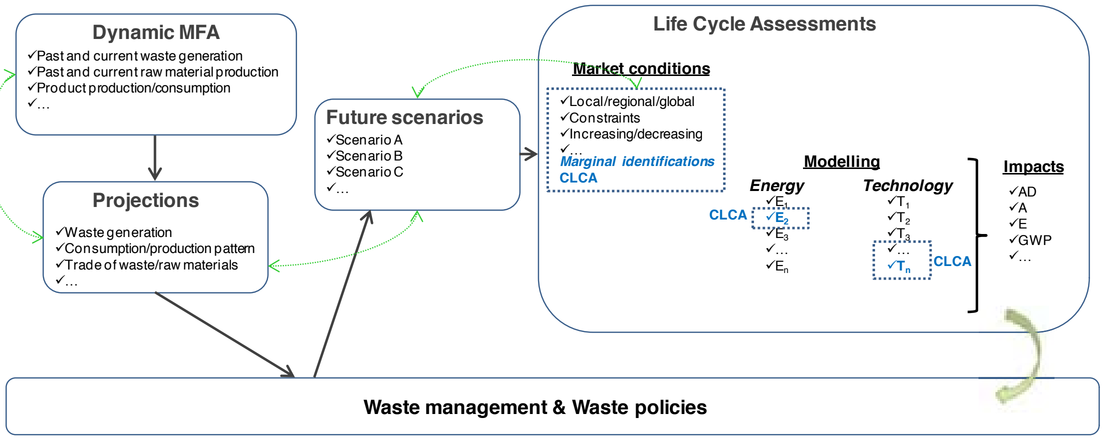

Figure 8.9: Schematic modelling for estimating environmental impacts for waste management

Chapter 9-

Future research

Photography by Eva Sevigné

9- Future research

Future lines of research that may follow this thesis are mentioned here. Future research and actions are focused in three areas: the CO2ZW ® tool, the contribution of MSW management to climate change and GHG benefits of recycling.

9.1. The CO2ZW® tool

- The **values of CO_2 credits obtained through this thesis should be incorporated** for future calculations in Spain. In addition, **same quantifications should be conducted for the other countries** included in the tool: Italy, Slovenia, Greece and European averages.
- **The COZW® should be updated to include new variables such as the percentage of waste exports, the application and quality of the recovered plastics or percentage of plastic energy recovery.** In addition, the GHG calculations from collection and transport should be improved. Within the inclusion of these variables more strategies and opportunities can be identified with the aim of reduce the GHG emissions and save more valuable materials.

9.2. Contribution of MSW management to climate change

- Biogas capture in landfill have revealed very important to decrease the GHG emissions generated in landfills. However, accurate data are in most cases not available and average estimations or default data should be used instead. In this regard, **more efforts should be put on quantify the biogas capture** in landfills which already have it or incorporate biogas capture in those landfills without it in order to reduce the uncertainty among this parameter.

9.3. GHG benefits of recycling

- The methodology framework proposed should be applied **to quantify the GHG emissions of other waste fractions, such as steel, glass or organic matter**.

- First assumption regarding the GHG quantification was that recycling avoids primary production. **More research should be conducted to evaluate this assumption** and determine if as a consequence of international recycling, other consequences are derived, such as if waste from other countries is not consumed and ends in landfills or is incinerated.

- Moreover, as a consequence of recycling, second, or other, consequences could be derived. For example, in the case of waste paper, it has suggested that the wood that is not used for virgin paper production is used to produce energy to substitute for fossil fuels. **It should be evaluated if there are other system expansions, and if so, which are the consequences of recycling in other sectors are or activities of the economy**.

- The in-use stock of aluminium products and especially plastic products is huge. The future waste generation conducted throughout a normal lifetime distribution has showed that important quantities of waste can be expected in coming years. **It should be evaluate different scenarios as a consequence of the in-use stock achieving its EOL**, such as the need of new waste management infrastructures or new waste legislation.

- This work could be complemented with a **social, economic and political assessment** in order to get a broader understanding of the potentialities of the GHG reductions and resource management.

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APPENDIX

Appendix.A. Methodology of supporting decision-making of waste management with MFA and CLCA: case study of waste paper recycling.

Based on the supplementary information of Chapter 5.

A.1. MFA

In this study, MFA was used in its simplest form, and only the material mass flows were studied. The temporal and spatial boundaries were the years 2006 to 2011 and Spain, respectively. The life cycle of PB (PB) is composed of eight LCs: wood crops [A], wood chip production [B], virgin pulp production [C], PB manufacturing [D], PB product production [E], use [F], waste management (collection and sorting) [G], and recycling [H]. Every LC produces products that are classified as follows: wood [a], wood chips [b], virgin pulp [c], PB papers [d], PB products [e], waste paper [f], refuse waste paper [g], recovered paper [h], and recovered fiber [i]. Additionally, we considered the auxiliary materials (a. materials) [j] necessary for PB paper production. Some of the products were classified into several subtypes. Table A1 describes these classifications and indicates the indices used in this study for LC and products.

Table A1: LC, LC indices, products, products indices and types of products of the life cycle of paper in this study

LC	LC indices	Paper-containing product (PCP)	PCP indices	Types of PCP
Wood crops	A	Wood	a	Eucalyptus Pine
Wood chips production	B	Wood chips	b	-
Virgin pulp production	C	Virgin pulp	c	Chemical Mechanical Semichemical
PB papers manufacture	D	PB papers Auxiliary materials*	d j	Newsprint Printing papers Sanitary & Household Packaging paper Carton board Corrugating materials Others -
PB products production	E	PB products	e	Newspaper Publication Sanitary Packaging Others
Use	F	Waste paper	f	-
Waste Management	G	Refuse waste paper Recovered paper	g h	- ONG-OMP OCC-Kraft Mixed grades
Recycling	H	Recovered fiber	i	-

A.1.1. Accounting method for flows and stocks

Several flows are associated with each LC. For each of them, the total input (consisting of flows from previous LCs) should equal the product production and loss according to equation [Eq. A1]. In addition, product production can be calculated based on the production yield (γ), as expressed in equation [Eq. A2]. Thus, loss can be calculated as fixed in equation [Eq. A3]. Consumption can be calculated using equations [Eq. A4] or [Eq. A5].

$$LC_{i,j}^{INPUT} = LC_{i,j}^{production} + LC_{i,j}^{stock} + LC_{i,j}^{loss} \qquad \text{[Eq. A1]}$$

$$LC_{i,j}^{production} = LC_{i,j}^{INPUT} \cdot \gamma \qquad \text{[Eq. A2]}$$

$$LC_{i,j}^{loss} = LC_{i,j}^{INPUT} \cdot (1-\gamma) \qquad \text{[Eq. A3]}$$

$$LC_{i,j}^{consumption} = LC_{i,j}^{production} + LC_{i,j}^{import} - LC_{i,j}^{export} - LC_{i,j}^{stock} \qquad \text{[Eq. A4]}$$

$$LC_{i,j}^{consumption} = LC_{i+1,j}^{INPUT} \qquad \text{[Eq. A5]}$$

where LC=LC; i=indicator for LC; j=indicator for the studied years; INPUTi,j=product demanded by LC i in year j; $production_{i,j}$=product produced in LC i in year j; $stock_{i,j}$= product stocked in LC i in year j; $Loss^{i,j}$=product discarded from LC i in year j; $Import_{i,j}$=product imported and generated from LC i in year j; $Export_{i,j}$=product exported and generated from LC i in year j; and $consumption_{i,j}$=product consumed from LC i in year j.

Regarding stocks, paper products are generally short-lived, and if the lifetime of a product is shorter than the temporal boundary (i.e., less than one year), the stock is equal to zero. However, if the lifetime of a product exceeds one year, some stock is likely to exist because the product would stay within the system boundary for a certain period of time (Hashimoto et al., 2004). In this study, the lifetimes of product are assumed to be one year or shorter, except for the publications. We used the model proposed by Hashimoto et al. (Hashimoto et al., 2004) to estimate publication stocks in year j based on equation [Eq. A6]:

$$\mathrm{Stock}_{pub}(j)=\mathrm{Stock}_{pub}(j-1)+(1-d_1(j-1))\times \mathrm{Consumption}_{pub}(j-1)-d_2(j-1)\times \mathrm{Stock}_{pub}(j-1) \quad \text{[Eq. A6]}$$

where stockpub(j)=the stock of the publication products on the year of study; stockpub(j-1)=the stock of the publication products from the previous year; d1=discard rate within a year, which in the study was defined as 0.8 (1); consumptionpub(j-1)=consumption of publication products in the previous year; and d2=discard rate of publication products, which was defined as 0.8 (Hashimoto et al., 2004).

A.1.2. Data sources and assumptions for accounting flows

A.1.2.1. Wood [A-B]

The data regarding national cultivation and wood chip production and the data concerning wood trade were obtained from the Spanish Association of Pulp and Paper Manufacturers (ASPAPEL) (ASPAPEL, 2012). The consumption of wood and wood chips and the losses were calculated using equations [Eq. A4] and [Eq. A1], respectively. The loss from the industrial processing of wood chips generates biomass that can be used as a fuel for bioenergy production (ASPAPEL, 2012).

A.1.2.2. Virgin pulp [C]

The data regarding virgin pulp production and trade were obtained from the Food and Agriculture Organization of the United Nations (FAO) (FAO, 2012). Losses generated during pulp production were taken from ASPAPEL (ASPAPEL, 2012), and our analysis considered that they are used for bioenergetic purposes during the production process (ASPAPEL, 2012). Consumption was calculated using equation [Eq. A4].

A.1.2.3. PB papers [D]

Data regarding production, import and export by paper type were obtained from FAO (FAO, 2012), and data regarding the auxiliary materials (a. materials) needed for their production is obtained from ASPAPEL (ASPAPEL, 2008). Equation [Eq. A3] was used to calculate the loss of each paper type manufacture with the production yields from the data of Hong et al. (Hong et al., 2011), and the total loss of this LC was calculated as the sum of all losses. Losses at this stage are generally sent to landfills and incineration plants.

A.1.2.4. PB products [E]

National production was calculated, considering that newspaper is produced from newsprint; publications from printing papers; sanitary products from sanitary and household paper; packaging from packaging paper, carton board, and corrugating materials; and other products from other materials. Data on the import and export of newspaper, publication, sanitary and other products were obtained from the Ministry of Economy and Competitiveness (DataComex, 2014). However, there were no trade data regarding packaging products. These data were calculated using data concerning the next use process as the difference in the total waste paper generated [f] and the PB products consumed and stocked. Losses of the LC are sent to landfills and incineration plants.

A.1.2.5. Use [F]

The following equation [Eq. A7] was used to calculate the consumption of this LC considering that production of the process is equal to the total waste paper generated, that publication products are stocked and that the sanitary and household products are lost in water waste streams:

$$LP_{F,j}^{INPUT} = LP_{F,j}^{production} + LP_{F,j}^{loss(wastewater)} + LP_{F,j}^{stock} \qquad \text{[Eq. A7]}$$

A.1.2.6. Waste management [G]

Waste paper is divided into groups that are either selectively collected or not, the latter of which is waste paper that is contained within the refuse fraction. Data for selectively collected waste paper were obtained from ASPAPEL and FAO (FAO, 2012; ASPAPEL, 2008). As observed in sorting plants that demonstrated a sorting efficiency of 92% (Doka, 2003a), waste paper selectively collected was classified into three grades of recovered paper: Old Newsprint (ONP)-Old Magazine (OMG), Old Corrugated Container (OCC)-Kraft, and mixed grades. Although there are no national data classified by the grade of the recovered paper, there are data regarding the import and export of recovered paper from the Ministry of Economy and

Competitiveness (DataComex, 2014). Refuse waste paper was calculated assuming that its fraction represented 19% of the total refuse waste generated (Magrama, 2011).

A.1.2.7. Recycling [H]

Recovered paper is sent to recycling, where it is converted into recovered fiber. The data regarding consumption and trade were obtained from ASPAPEL and FAO (FAO, 2012; ASPAPEL, 2008), while production and loss were calculated using equations [Eq. A4] and [Eq. A1], respectively.

Table A2: Production (P), imports (I) and exports (E) of PB papers by type from 2006 to 2011 in Spain (kt)

	Newsprint (kt)			Printing (kt)			Sanitary (kt)			Packaging (kt)			Board (kt)			Corrugating (kt)			Others (kt)		
	I	P	E	I	P	E	I	P	E	I	P	E	I	P	E	I	P	E	I	P	E
2006	421	380	315	1,895	1,593	928	79	607	66	297	169	362	952	400	146	1,036	3,044	734	132	705	168
2007	470	389	325	1,786	1,632	953	80	703	69	334	146	356	2,156	356	144	957	2,711	825	94	776	63
2008	386	380	299	1,520	1,594	984	93	728	95	241	180	367	414	330	126	1,222	2,518	824	119	683	166
2009	281	323	353	1,345	1,355	936	66	696	73	244	232	438	472	281	105	1,391	2,144	865	77	669	65
2010	270	314	288	1,289	1,318	884	283	713	45	304	192	403	417	272	341	790	2,592	944	43	791	46
2011	184	306	273	1,267	1,422	852	48	679	93	228	201	377	496	319	159	765	2,773	903	55	791	43

Table A3: **Production (P), imports (I) and exports (E) of PB products by type from 2006 to 2011 in Spain (kt)**

	Newspaper(kt)			Publication (kt)			Sanitary k)			Packaging (kt)			Others (kt)		
	I	P	E	I	P	E	I	P	E	I	P	E	I	P	E
2006	23	429	56	58	2,481	265	0	480	0	124	4,447	222	27	563	100
2007	24	475	146	64	2,383	156	0	552	0	163	5,149	235	27	691	100
2008	23	411	47	152	2,050	229	0	559	0	158	3,413	250	79	534	93
2009	18	202	35	60	1,696	219	0	529	0	215	3,199	253	24	581	91
2010	18	249	42	72	1,657	256	0	786	0	172	2,704	334	38	700	125
2011	13	171	34	67	1,765	253	0	479	0	196	3,151	369	32	684	104

Table A4: Selectively collected waste paper, refuse waste paper and total waste paper from 2006-2011

	Production (kt)		
	Waste paper selective collected	Refuse waste paper	Total waste paper
2006	5,371	3,882	9,253
2007	4,921	3,799	8,720
2008	4,999	3,773	8,772
2009	4,625	3,376	8,001
2010	4,637	3,019	7,656
2011	4,723	n.a	n.a

A.2. CLCA

A.2.1. FU

The FU has been defined as an increase of 1 ton of waste paper collected in Spain for recycling in Spain and internationally.

A.2.2. Scenarios

The following two scenarios have been studied: the Baseline scenario and the Spanish scenario. In the Baseline scenario, the increase of waste paper collected for recycling avoids BEKP production in Brazil when recycling occurs in Spain and internationally. In the Spanish scenario, the increase in waste paper collected for recycling avoids BEKP production in Spain when recycling occurs in Spain and avoids BEKP production in Brazil when recycling occurs in China and the Netherlands. For each scenario, the GHG quantifications are calculated as the sum of the positive (emitted) or negative (avoided) GHG emissions associated with each stage. The following LCs were included within the boundaries of Spain: waste management [G]; recycling [H]; virgin pulp production [C], wood chip production [B]; wood cultivation [A]; and transport to all facilities. The stages in China and the Netherlands included recycling and transport. The stages in Brazil included wood cultivation and wood chip and virgin pulp production with the corresponding transport. Figure A1 summarizes both scenarios, where the black arrows represent the product flows and their transport (i.e., wood, virgin pulp or recovered paper); the grey arrows represent the avoided production, which is the virgin pulp production from Eucalyptus wood and the transport avoided; and the dotted lines represent the country in which each stage has occurred.

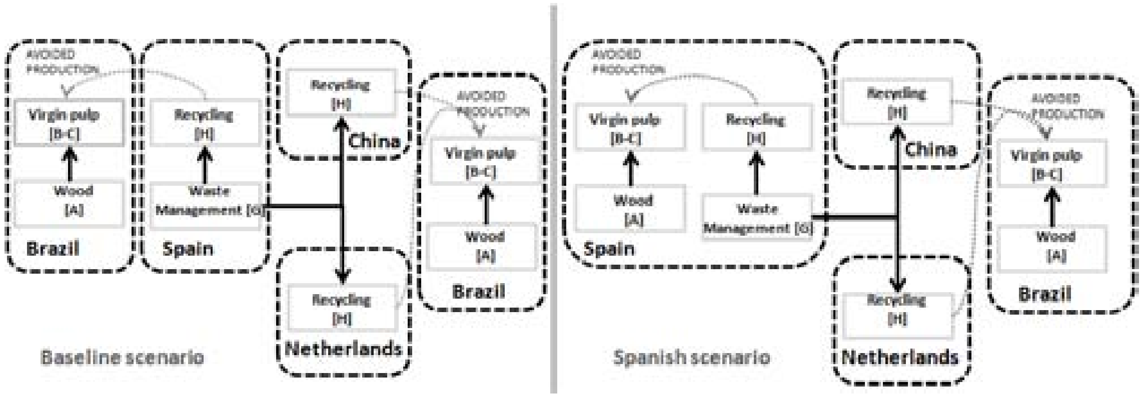

Figure A1: A schematic representation of the baseline scenario and the Spanish scenario where the black arrows represent the product flows and their transport (i.e., wood, virgin pulp or recovered paper); the grey arrows represent the avoided production, which is the virgin pulp production (i.e., BEKP) from Eucalyptus wood and the transport avoided; and the dotted lines represent the country in which each stage has occurred.

A.2.3. Inventory

The virgin production system included the cultivation and processing of Eucalyptus in Brazil and Spain for 1 ton of BEKP. The data are presented in Tables A5 and Table A6, and the explanations are provided in separate sections.

A.2.3.1. Wood cultivation and virgin pulp production [A-B-C]

Data for the Eucalyptus crop and bleached pulp in Spain were obtained from a previously published work (Gonzalez-García, 2009a, Gonzalez-García, 2009b). No other significant environmental studies have been conducted for Spain, though it is the top producer and exporter of Eucalyptus-based pulp products. However, these studies have examined representatives of Eucalyptus cultivation and pulp production in Spain. Energy consumption is particularly elevated in pulp mills, and in Spain, these energy requirements are satisfied by cogeneration units derived from biomass waste, black liquor and fossil fuels (ASPAPEL, 2012). Only 1% of total electricity requirements are purchased from the national grid (Gonzalez-García, 2009b). In addition, from the cogeneration units, a surplus of electricity is produced and sold to the national grid. Thus, the avoided electricity production has also been considered. Data regarding the cultivation of Eucalyptus in Brazil were obtained from a previous work (Alexandre-Kulay, 2006) and a sector-specific publication (Brazelpa, 2010). Brazilian plantations date back to the 1970s, and only recently has Brazil become a major contributor to the wood and pulp markets (3), ranking as the fourth largest producer worldwide in 2010 (Brazelpa, 2010). The high energy requirements of the production process are satisfied by the cogeneration units, as well as the Spanish system. However, in this system, the amount of electric energy generated in the recovery boiler corresponds to 35% of the total energetic demand, and additional energy is provided mainly by natural gas and fossil fuels (Brazelpa, 2010). Table A5 presents the relevant data for Spain and Brazil.

Table A5: Inventories for the production of 1 (air-dried) ton of bleached pulp in Spain and Brazil

	Spain	Brazil
WOOD [1]		
Fertilizers (NPK)-kg	10.7	8.2
Pesticides-kg	0.2	0.3
Diesel-kg	14.2*	14.2*
Transport to mill-km	60	210
VIRGIN PULP [2-3]		
Wood-t	1.5	1.8
Chemicals-kg	92.4	44
Fossil fuels-kg	53.4	10
Electricity-kWh	575	700
Steam-t	5.5	4.2
Water-m^3	32.7	18
Electricity production-kWh	32.3	-

* No data on the diesel consumption for agriculture in Brazil were found. Therefore, Spanish data were used in both scenarios.

Data regarding the artificial production of fertilizers were obtained from Patyk and Reinhard (Patyk and Reinhard, 1997) because these data were assumed to represent less competitive technology (Dalgaard et al., 2008). Because of the lack of data, the same assumptions were made for Eucalyptus cultivation in Brazil. N_2O emissions were calculated according to IPCC (IPCC, 2000) in both cases. Data regarding pesticide production were obtained from Nemecek and Kägi (Nemecek and Kägi, 2007). No data regarding the machinery used were available for Brazil. Therefore, this stage was not included in the evaluation of any scenarios. Electricity was modeled following previous recommendations by Schmidt et al., (2011). The marginal electricity was calculated differentially by country for all of the processes involved in this study. Steam was produced in both cases through the cogeneration of renewable sources (i.e., black liquor or wood residues), and the specific consequences related to the cogeneration process were considered. Data on the diesel used for machinery and transport and on fossil fuels were obtained from the ecoinvent database (Jungbluth, 2007). Data regarding chemical production were also obtained from the ecoinvent database (Althaus et al., 2007), excluding the allocation factors in the oxygen and NaOH production. The average data were assumed not to differ from marginal ones. In the case of NaOH recommendations, LCA 2.0 was used (Wesnæs and Weidema, 2006).

A.2.3.2. Waste Management [G]: collection and sorting

Data regarding the collection and transport to different facilities in Spain were estimated based on the data from previous Spanish works (Rives et al., 2010; Inèdit, 2011) and are shown in Table A6. The transport distances from Spain to China were calculated based on a previous work (ITENE, 2008). All of the 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 (not shown).

Table A6: Inventories for the collection and transport of 1 ton of recovered paper in Spain

	Road Transport (km)	**Sea transport (km)**
Collection	60	-
Transport to sorting plants	300	-
Transport to port	300	-
Transport to landfill	100	-
Transport to recycling plant China	225	16,500
Transport to recycling plant The Netherlands	1500	-
Transport from Brazil to Spain	300	7,300
Transport from Brazil to China	225	21,100
Transport from Brazil to The Netherlands	115	9,000

A.2.3.3. Recycling [H]

Data regarding the recycling process is based on the European Commission's Reference Document on the Best Available Techniques in the Pulp and Paper Industry (European Commission, 2001). The recycling processes in China and the Netherlands were also modeled and are included in the calculations. The same data were used for the three countries with electricity, steam and chemical consumption values of 150 kWh, 0.2 t and 20 kg per ton of recovered paper, respectively (European Commission, 2001). We considered that electricity and heat in Spain and the Netherlands are produced via cogeneration. In contrast, electricity and heat are not produced via cogeneration in China because the recycling sector there is still under development (IVEX, 2010). Notably, 1 kg of recovered fiber is not equivalent to 1 kg of virgin fiber; we assumed that the equivalence ratio is 0.8:1 (virgin pulp: recovered fiber) (Ekvall, 2000). In addition, based on the MFA results, a material recovery efficiency of 81% was used, implying that 19% of the starting material is lost as impurities. The following treatments were included: in China, such losses are sent to landfills; in Spain, 64% of such losses are sent to landfills, and the rest is incinerated (ASPAPEL, 2012); and in the Netherlands, such losses are incinerated. We considered that the incineration treatment plants also produce electricity and heat. Emissions from landfills are calculated according to IPCC (IPCC, 2000).

A.2.4. Cumulative Energy Demand (CED)

The following Tables and figures present the results for the CED indicator. Results for all scenarios and all RERs are presented in Tables A7 and A8 while Figure A2 and A3, also in Appendix B, evaluate direct and indirect energy for a RER of 15%. Finally, Table A9 presents the total energy comparison between the Spanish and the Brazilian BEKP production by source of energy.

Table A7: CED (MJ) by ton of collected waste paper in Spain for the Baseline scenario for RERs of 5%, 15%, 25% and 50%

	MJ t^{-1}			
BASELINE SCENARIO	5%	15%	25%	50%
Waste management [G]	**1,266**	**1,487**	**1,708**	**2,261**
Collection	439	439	439	439
Sorting	84	84	84	84
National transport to all facilities in Spain	646	671	697	761
International transport to China and The Netherlands	98	293	489	978
Recycling [H]	**1,050**	**1,336**	**1,573**	**2,227**
Recycling plants in Spain	873	781	689	459
Recycling plants in China	189	567	946	1,891
Recycling plants in The Netherlands	-12	-12	-62	-124
Avoided Pulp production (Brazil) [B-C]	**-8,028**	**-8,122**	**-8,216**	**-8,452**
Electricity	-3,694	-3,694	-3,694	-3,694
Fuel	-2,180	-2,180	-2,180	-2,180
Chemicals	-1,227	-1,227	-1,227	-1,227
Others	-2	-2	-2	-2
National transport	-6	-18	-29	-58
International transport	-920	1,002	-1,085	-1,291
Avoided Eucalyptus cultivation (Brazil) [A]	**-1,150**	**-1,150**	**-1,150**	**-1,150**
TOTAL (kg CO_2 eq. t^{-1})	**-6,863**	**-6,449**	**-6,086**	**-5,114**

Table A8: CED (MJ) by ton of collected waste paper in Spain for the Spanish scenario for RERs of 5%, 15%, 25% and 50%

	MJ t^{-1}			
BASELINE SCENARIO	5%	15%	25%	50%
Waste management [G]	**1,266**	**1,487**	**1,708**	**2,261**
Recycling [H]	**1,050**	**1,336**	**1,573**	**2,227**
Avoided Pulp production (Spain) [B-C]	**-4,467**	**-3,996**	**-3,526**	**-2,351**
Electricity	-2,247	-2,011	-1,774	-1,183
Fuel	-330	-295	-260	-174
Chemicals	-1,833	-1,640	-1,447	-965
Others	-56	-50	-44	-30
National transport	-0.1	-0.1	-0.1	0
International transport	0	0	0	0
Avoided Pulp production (Brazil) [B-C]	**-440**	**-1,321**	**-2,202**	**-4,403**
Avoided Eucalyptus cultivation (Spain) [A]	**-765**	**-684**	**-604**	**-403**
Avoided Eucalyptus cultivation (Brazil) [A]	**-63**	**-190**	**-317**	**-633**
TOTAL (kg CO_2 eq. t^{-1})	**-3,420**	**-3,369**	**-3-367**	**-3,302**

Table A9: CED (MJ) by ton of BEKP production in Brazil and in Spain by type of energy source

	MJ t^{-1}						
	Fossil	**Nuclear**	**Biomass (non rene.)**	**Biomass (rene.)**	**Wind, solar, geothe.**	**Water**	**Total**
Brazilian BEKP							
Electricity	1,012	247	0	1	0	833	2,095
Fuels	7,691	50	0	2	1	13	7.757
Chemicals	1,330	592	0	28	11	97	2,057
Others	3	1	0	0	0	0	3
Total	**10,037**	**889**	**0**	**31**	**12**	**943**	**11,913**
Spanish BEKP							
Electricity	-137	-1	0	0	0	0	-138
Fuels	5,561	38	0	2	1	9	5,610
Chemicals	1,784	522	0	29	9	66	2,411
Others	3	1	0	0	0	0	4
Total	**7,211**	**560**	**0**	**30**	**10**	**75**	**7,886**

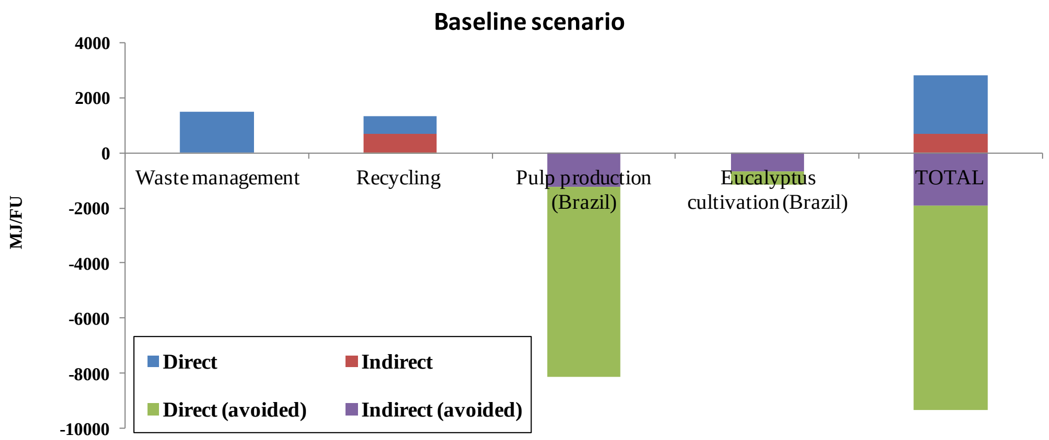

Figure A2: Direct and indirect energy in MJ for RER of 15% for the Baseline scenario

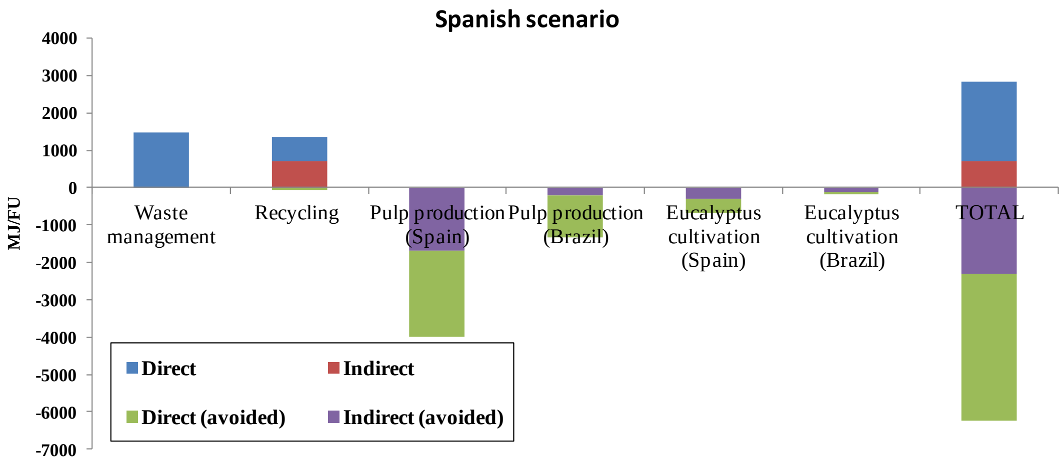

Figure A3: Direct and indirect energy in MJ for RER of 15% for the Spanish scenario

Appendix.B. Environmental consequences of recycling aluminium old scrap in a global market

Based on the supplementary information of Chapter 6.

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Reference link

Appendix.C. The contribution of plastic waste recovery to resource efficiency and greenhouse gases (GHG) savings in Spain

Based on the supplementary information of Chapter 7.

C.1. Dynamic MFA of plastics: data and assumptions

C.1.1. Accounting method of flows

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; thus, the total input consisting of flows from previous process should be equal to products production, stock and loss according to equation [Eq. C.1]. In addition, the products production can be calculated based on the production yield (γ) as expressed in equation [Eq. C.2]; thus, loss can be calculated as fixed in equation [Eq. C.3]. Consumption can be calculated by equations [Eq. C.4] or [Eq. C.5].

$$P_{i,j}^{INPUT} = P_{i,j}^{production} + P_{i,j}^{stock} + P_{i,j}^{loss} \quad \text{[Eq. C.1]}$$

$$P_{i,j}^{production} = P_{i,j}^{INPUT} \cdot \gamma \quad \text{[Eq. C.2]}$$

$$P_{i,j}^{loss} = P_{i,j}^{INPUT} \cdot (1-\gamma) \quad \text{[Eq. C.3]}$$

$$P_{i,j}^{consumption} = P_{i,j}^{production} + P_{i,j}^{import} - P_{i,j}^{export} \quad \text{[Eq. C.4]}$$

$$P_{i,j}^{consumption} = P_{i+1,j}^{INPUT} \quad \text{[Eq. C.5]}$$

where P= Processes; i= indicator for processes; j=indicator for the studied years; INPUT= products entering process i in year j; production=products produced in process i in year j; stock=products stocked from process i in year j; Loss=products discarded from process i in year j; Import= products imported to process i in year j; Export= products exported from process i in year j; consumption=products consumed from process i in year j.

Each flow is calculated in three ways; it is calculated directly based on statistics, calculated by combining statistics with coefficients and deduced using the mass balance.

C.1.1.1. Raw Materials for plastics [A]

A plastic material is an organic solid, essentially a polymer or combination of polymers of high molecular mass. The production of polymers begins with a distillation process in an oil refinery. The distillation process involves the separation of heavy crude oil into lighter groups called fractions (Plastic Europe, 2013). One of these fractions, naphtha, is the crucial element for the production of plastics which is passed to the next stage of monomer production. The monomer is then converted to the desired grade of polymer as determined by the application needs of the converted product (European Commission, 2007). Almost all plastics are currently derived from fossil sources, mainly oil and gas. Only 0.1-0.2% is derived from renewable organic sources such as starch, corn or sugar (JRC, 2012). Approximately, 4% of the world oil production goes to make plastics, the 6% goes to other industries and the remaining 90% is devoted to heating oil and locomotion (ANAIP, 2013). In this study we consider that all plastic is produced from imported oil, since there is no oil in Spain, and we assumed that 4% of the imported oil goes to make plastics. Data of imported oil was obtained from the Ministry of Economy and Competitiveness (DataComex, 2014).

C.1.1.2. Virgin plastics production [B]

There are many different types of synthetic polymers; with these being used in wide variety of applications. They can be classified as thermoplastics, polyurethanes, thermosets, adhesives, coatings and sealants (Salmons and Mocca, 2010). In addition, a classification is made between thermoplastics: standard thermoplastics and engineering thermoplastics. The former have limited stress and low temperature resistance, and are used mainly for inexpensive or disposable products and packaging; while the latter have higher strength and thermal resistance, and are used in applications requiring wear resistance, long life expectancy, flame resistance and / or the ability to endure cyclic stress loading (Salmons and Mocca, 2010). In this study we focus into the main standard thermoplastics which are low density polyethylene (LDPE), high density polyethylene (HDPE), polypropylene (PP), polystyrene (PS), expanded polystyrene (EPS), polyvinyl chloride (PVC), polyethylene terephthalate (PET) and ethyl vinyl acetate (EVA). Remaining plastics (i.e., engineering, polyurethanes, etc) are categorized as others in this study. Data of virgin plastics production was obtained from the Spanish National Statistics Institute (INE, 2013) and data of the trade from the Ministry of Economy and Competitiveness (DataComex, 2014).

C.1.1.3. Additives

In addition, more often than not, plastics contain a main polymer and a bespoke load of additives to improve specific properties (e.g. hardness, softness, UV resistance, flame formation resistance, etc). The content of additives in plastics varies widely, from less than 1% in PET bottles and up to 50-60% in hard PVC (JRC, 2012). In this study we assume an average content of additives of 20%.

C.1.1.4. Plastics products manufacture [C]

Plastic articles are produced from the synthetic polymer, usually in powder, granulate, pellet or flake form, by a range of different processes (JRC, 2012) or and/by recycled polymers.

However, the opportunity of using recycled polymers as substitutes of virgin polymers is very much influenced, and limited, by the end-use application (JRC, 2012). In section 1.6 more detail and explanations are given for the recycling life stage and recycled polymers. The following classification for plastics products was followed in this study: Monofilament rods, sticks and profiles products; rigid tubes, pipes and hoses products; plates, sheets, films, foils and strips products; packaging products; construction products; and other products. Data of plastic products production was obtained from the Spanish National Statistics Institute (INE, 2013) and data of the trade from the Ministry of Economy and Competitiveness (DataComex, 2014).

C.1.1.5. Use [D]

Plastic materials are used in a variety of end-use applications and in this study we consider the following classification: agriculture, electric and electronic, construction, packaging, automotive and others. Data of plastics products consumed were obtained through [Eq. 5] with data from Cicloplast (Cicloplast, 2009) and data plastic put in the market from (). According to Eurostat, this data represents total plastic packaging put in the market, considering the import of plastic packaging within other consumption products.

Besides, the use LC is different from other processes because most of the final products may serve in the use stage for a long time and will not be consumed, an in-use stock of plastics will gradually form and enlarge in a defined geographical area such as a city or a country. To calculate the in-use stock we have used the following equation [Eq. C.6]:

$$P_{j}^{stock} = P_{D,j}^{consumption} + P_{j-1}^{stock} - P_{E,j}^{production} \qquad \text{[Eq. C.6]}$$

C.1.1.6. Collection and sorting [E]

Plastic waste generated after consumption is collected after a consumer cycle, either separately (i.e., selective collection plastic waste) or mixed with the refuse fraction of municipal solid waste (i.e., refuse plastic waste), and in this stage we considered both collections. Data of plastic waste generated was obtained the National Association of Plastic Recyclers (ANARPLA, 2013) and Eurostat (Eurostat, 2013) and data of the trade from the Ministry of Economy and Competitiveness (DataComex, 2014). We considerer that the plastic selective collected 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). We calculated the material loss based on mass balance. After sorting, recovered plastic can be sent to recycling or energy recovery and losses are sent to landfill. The refuse plastic waste is collected within the refuse fraction.

C.1.1.7. Recycling [F]

Two main types of recycling can be distinguished, mechanical and chemical (also called feedstock recycling). Mechanical recycling involves the melting of the polymer, but not its chemical transformation. To a much smaller extent, recycling also takes place via chemical recycling, also called feedstock recycling, where a certain degree of polymeric breakdown takes place (JRC, 2012). Recycling plastic as chemical feedstock in industrial processes is negligible in Spain and is not considered in this paper. Data on quantity of plastic waste recycled were obtained from the Spanish Centre of Plastic (CEP, 2012) and data by type of plastic recycled were obtained from the National Association of Plastic Recyclers (ANARPLA, 2013). Beside, plastic waste can be recycled into a secondary raw material to form new products directly, or in combination with virgin plastic material. The material efficiency of recycling plants was assumed of 85% (JRC, 2012) and the options for use of recycled plastic depend on the quality and polymer homogeneity of the material (JRC, 2012). The following classification reflects the markets for the recycled polymer in Spain in 2009 (Cicloplast, 2009): pipes (26%), rubbish bags (11%), other films and bags (23%), bottles and drums (4%), industrial pieces (15%), household (2%) and others(19%). We use these percentages for all years studied.

C.1.1.8. Energy recovery [G] and landfill [K]

Data of plastic waste sent to energy recovery was obtained from Eurostat (Eurostat, 2013). Total plastic waste sent to landfill is the sum of the losses from sorting and recycling and the refuse plastic waste containing in the refuse fraction.

C.2. Results of MFA of plastics from 1999 to 2011 in Spain

Table C1: Virgin plastics production from 1999 to 2011 in Spain by type

	Virgin plastic (t)								
	HDPE	**LDPE**	**PP**	**PVC**	**PS**	**EPS**	**PET**	**EVA**	**Others**
1999	370,706	425,479	913,258	405,990	133,702	34,900	*430,464*[4]	54,028	1,154,265
2000	416,691	476,684	911,185	379,035	124,616	34,309	*475,206*[7]	52,386	1,192,350
2001	388,607	593,506	602,132	407,332	131,778	36,437	*475,928*[7]	65,489	1,152,240
2002	375,368	535,872	626,048	392,002	137,542	35,376	*507,680*[7]	71,559	1,313,234
2003	478,505	551,720	591,127	376,636	181,098	46,032	*527,887*[7]	70,068	1,407,427
2004	498,748	555,824	665,854	403,068	180,284	50,808	*564,691*[7]	77,367	1,478,941
2005	400,587	731,765	990,488	589,157	189,262	45,591	515,700	67,637	1,471,943
2006	443,978	609,172	888,156	631,174	212,607	55,362	515,700	78,007	1,474,554
2007	497,016	750,891	936,846	653,838	194,661	65,165	534,094	90,889	1,877,483
2008	440,926	600,092	909,439	577,951	176,174	59,961	554,812	79,215	1,682,836
2009	382,975	535,896	865,003	524,505	178,752	41,520	452,270	80,217	1,370,564
2010	383,330	608,002	934,514	613,420	140,006	27,179	711,350	85,018	1,541,921
2011	424,363	698,023	932,941	550,762	153,231	38,363	742,771	85,517	1,584,635

[4] estimated data

Table C2: Virgin plastics consumption from 1999 to 2011 in Spain by type

	Virgin plastic (t)								
	HDPE	**LDPE**	**PP**	**PVC**	**PS**	**EPS**	**PET**	**EVA**	**Others**
1999	569,233	539,424	743,250	557,238	164,701	54,133	407,976	27,876	1,023,110
2000	563,181	561,206	742,971	508,735	183,074	72,608	429,964	52,386	1,292,847
2001	636,435	756,112	443,949	549,910	156,116	61,121	430,173	65,489	1,145,834
2002	762,651	767,936	553,733	531,548	153,231	55,098	477,704	71,559	1,200,822
2003	856,711	804,972	500,267	471,983	183,039	75,626	499,561	70,068	1,348,221
2004	904,929	822,043	516,848	515,531	182,219	69,363	527,342	77,367	1,399,216
2005	833,290	952,471	890,175	690,537	197,622	93,369	422,086	67,637	1,458,706
2006	926,432	852,628	756,878	717,579	227,569	102,321	385,507	77,902	1,446,075
2007	966,087	923,801	818,290	708,731	236,026	82,349	516,595	67,066	1,794,826
2008	865,488	773,002	722,148	576,712	190,131	88,465	525,490	55,392	1,572,000
2009	734,462	641,400	812,654	490,786	192,709	71,897	495,496	80,202	1,194,490
2010	622,836	616,997	723,833	526,346	157,983	42,429	722,342	85,018	1,312,111
2011	736,494	733,632	735,238	382,675	135,692	42,091	632,731	85,517	1,301,499

Table C3: Plastic waste collection in Spain from 1999 to 2011

	Collection (t)				
	Agriculture	**Electronics**	**Construction**	**Automotive**	**Others**
1999	*138,875*[e]	*69,438*[e]	*34,719*[5]	*86,797*[e]	*295,109*[8]
2000	*149,163*[e]	*74,581*[e]	*37,291*[e]	*93,227*[8]	*316,970*[8]
2001	*164,625*[e]	*82,313*[e]	*41,156*[e]	*102,891*[8]	*349,828*[8]
2002	*164,875*[e]	*82,438*[e]	*41,219*[e]	*103,047*[8]	*350,359*[8]
2003	*175,913*[e]	*87,956*[e]	*43,978*[e]	*109,945*[8]	*373,814*[8]
2004	*182,897*[e]	*91,448*[e]	*45,724*[e]	*114,311*[8]	*388,656*[8]
2005	*195,663*[e]	*97,831*[e]	*48,916*[e]	*122,289*[8]	*415,783*[8]
2006	*201,875*[e]	*100,938*[e]	*50,469*[e]	*126,172*[8]	*428,984*[8]
2007	*209,875*[e]	*104,938*[e]	*52,469*[e]	*131,172*[8]	*445,984*[8]
2008	*198,125*[e]	*99,063*[e]	*49,531*[e]	*123,828*[8]	*421,016*[8]
2009	*180,365*[e]	*90,182*[e]	*45,091*[e]	*112,728*[8]	*383,275*[8]
2010	171,830	94,591	45,279	114,608	382,453
2011	171,346	104,256	41,302	93,215	359,963

[5] estimated data

Table C4: Plastic waste recycling in Spain from 1999 to 2011

	Recycling (t)				
	Agriculture	Electronics	Construction	Automotive	Others
1999	26,136	2,284	993	70	10,288
2000	47,076	4,758	1,055	988	10,491
2001	29,369	5,252	1,124	988	-
2002	29,430	6,282	680	783	8,528
2003	32,579	6,966	800	1,121	19,865
2004	47,133	8,450	1,189	1,566	68,667
2005	64,060	7,905	1,409	1,150	64,145
2006	69,405	11,184	1,978	3,482	50,407
2007	66,677	18,553	2,513	5,273	41,372
2008	56,069	10,104	2,813	10,203	34,077
2009	46,011	11,309	3,598	5,760	33,120
2010	48,336	10,946	1,245	9,512	38,069
2011	51,575	17,492	21,386	9,946	26,267

Table C5: Export trade by type of polymer (PE, PS, PVC, PP and others) from 1999 to 2011

	Export (t)				
	PE	**PS**	**PVC**	**PP**	**Others**
1999	2,482	812	2,717	7	-
2000	1,913	718	766	73	-
2001	2,097	186	945	343	-
2002	2,668	520	452	452	-
2003	9,995	1,285	1,246	1,154	-
2004	17,329	845	2,923	1,421	-
2005	20,307	763	2,642	1,931	-
2006	22,132	1,937	2,693	2,830	-
2007	31,552	2,451	3,555	3,096	-
2008	31,116	3,143	2,119	3,785	-
2009	35,897	3,508	1,702	4,658	-
2010	63,521	3,821	4,584	13,435	132,743
2011	34,684	2,340	6,518	9,563	139,911

C.3. CLCA of plastics recycling in Spain

C.3.1. LCI

C.3.1.1. Raw materials [A] and virgin plastics production [B]

Data for LCI of virgin plastic production are from Ecoinvent (Hischier, 2007) which are derived from datasets provided by Plastic Europe representing European Average production from 1999. Data is aggregated and take into account all materials inputs (i.e., electricity) needed from raw material extraction (i.e., oil) to virgin plastic production. Table C6 shows the virgin plastic considered and its respective LCI from ecoinvent.

Table C6: LCI of virgin plastic production from ecoinvent

Name	LCI data
Polyethylene, HDPE	Polyethylene, HDPE, granulate, at plant/RER
Polyethylene, LDPE,	Polyethylene, LDPE, granulate, at plant/RER
Polypropylene (PP)	Polypropylene, granulate, at plant/RER
Polyvinylchloride (PVC)	Polyvinylchloride, emulsion, polymerized, at plant / Polyvinylchloride suspension polymerized, at plant
Polystyrene (PS)	Polystyrene, general purpose, GPPS, at plant
Polyethylene terephthalate (PET)	Polyethylene terephthalate, granulate, amorphous, at plant/RER

C.3.1.2. Wood lumber manufacture [I] and wood production [J]

In the case of the LCI for wood lumber manufacture, we consider that the substituted would be wood. LCI data for the wood lumber production was review for (Astrup et al., 2009) and they used that data from Sathre (Sathre, 2007) for wood lumber production as substitute of virgin wood. Data for electricity consumption was, however, recalculated as only data on primary energy was available in Sathre (Sathre, 2007) as well as the amount of fuel oil (Astrup et al., 2009).

C.3.1.3. Recycling [F]

The mechanical recycling process requires the waste plastic to be shredded and extruded to form recycled granulate ready for use in new products (Shonfield, 2008). Electricity consumption for recycling of different plastic scrap types have been review in (Shonfield, 2008; Astrup et al., 2009; Chen et al., 2011 and Schmidt et al., 2012) ranging from 270 kWh t^{-1} to 330 kWh t^{-1} for Europe and an average of 229 kWh t^{-1} has been used in this study; and 575 kWh t^{-1} for China. LCI data of the recycling process was considered similar for the recycled plastic lumber production.

The recycling processes in China and Europe were also modeled and are included in the calculations also with the electricity marginal mixes considered for each of them. Detail explanation of the marginal electricity mixes are given in section *C.3.1.7*.

In addition, a material recovery efficiency of 85% was used, implying that 15% of the starting material is lost as impurities. The following treatments were included: in China, such losses are sent to landfills; in Spain, such losses are sent to landfills; and in Europe, such losses are incinerated. We considered that the incineration treatment plants also produce electricity and heat, and therefore, the avoided electricity and heat are also taken into account (see section 1.1.5) from the quantifications.

C.3.1.4. Collection, sorting and international transport [E]

Data regarding the collection and transport to different facilities in Spain were estimated based on the data from previous Spanish works (Rives et al., 2010; Inèdit, 2011) and are shown in Table C7. The transport distances from Spain to China were calculated based on a previous work (ITENE, 2008) while the transport distance to Europe was assumed of 5,000 km (average). All of the distances were calculated using the EcoTransIT tool (EcoTransIT, 2012) and LCI data for each type of transport was from ecoinvent (Spielmann et al., 2007). Data regarding the electricity and diesel consumption in sorting plants were provided by waste managers in Spain (not shown).

Table C7: Inventories for the collection and transport of 1 ton of waste plastic in Spain

	Road Transport (km)	Type of transport
Collection (SP)	60	Road
To sorting plants (SP)	300	Road
To port (SP)	300	Road
Waste from sorting plant to landfill (SP)	50	Road
Waste from recycling plant to final treatment (SP/CN/EU)	50/100/50	Road
To recycling plant China (CN)	16,500 (300)	Sea (Road)
To recycling plant Europe (average) (EU)	5000	Rail

C.3.1.5. Energy recovery [G]

The incineration with energy recovery of plastic waste as another EOL treatment is outside of the scope of the study but the processed waste from the sorting plants and recycling facilities undergoes to energy treatment or landfill. In this paper we assumed that the processed plastic waste in Europe undergo to energy recovery. LCI data was obtained from ecoinvent (Doka, 2003b). However, the ecoinvent activities do not include energy recovery. Therefore, this is added to the relevant ecoinvent activities. In Europe, it is considered that the energy recovery rates are 71% and 14% for electricity and heat, respectively, according to (Schmidt et al., 2012).

C.3.1.6. Landfill [K]

LCI data of waste from sorting and recycling facilities to landfill is based on ecoinvent data (Doka, 2003a).

C.3.1.7. Marginal electricity inventory

The marginal electricity production used in this study was modeled following previous recommendations by Schmidt et al. (2011) and (Schmidt and Thrane, 2009). The marginal electricity was calculated differentially by country (or region) for all of the processes involved in this study; Spain, China and Europe. For Spain, we used data and projections from the Ministry of Industry, Energy and Tourism (Minetur, 2011), in which it was established that the structure of electricity generation in Spain will continue to evolve over the forecast period in the same way as it has done in recent years, with a reduction in the weight of oil and coal in the generation mix, a slight increase in the natural gas weight and greater growth of renewable energy and hydroelectric pumping (Minetur, 2011). For Europe and China, we used data and projections from (Schmidt and Thrane, 2009).

Besides, for the sensitivity assessment we considered the electricity mixes from 2011 obtained from the Energy International Agency (EIA, 2013). Table C8 summarizes the electricity mixes for each country or region.

Table C8: Marginal electricity mixes considered for each country and the average electricity mixes from 2011 used for the sensitivity assessment

	Spain		Europe		China	
	Marginal electricity production (%)	Sensitivity assessment MIX 2011 (%)	Marginal electricity production (%)	Sensitivity assessment MIX 2011 (%)	Marginal electricity production (%)	Sensitivity assessment MIX 2011 (%)
Hard Coal	0	15	28	27	82	79
Oil	0	5	0	2	0	0
Natural Gas	94	29	46	21	2	2
Biomass	0	2	0	5	0	1
Nuclear	0	20	0	28	5	2
Hydropower	6	11	29	10	11	15
Wind	0	15	0	5	0	1
Solar PV	0	3	0	1	0	0

C.4. Results from sensitivity assessment

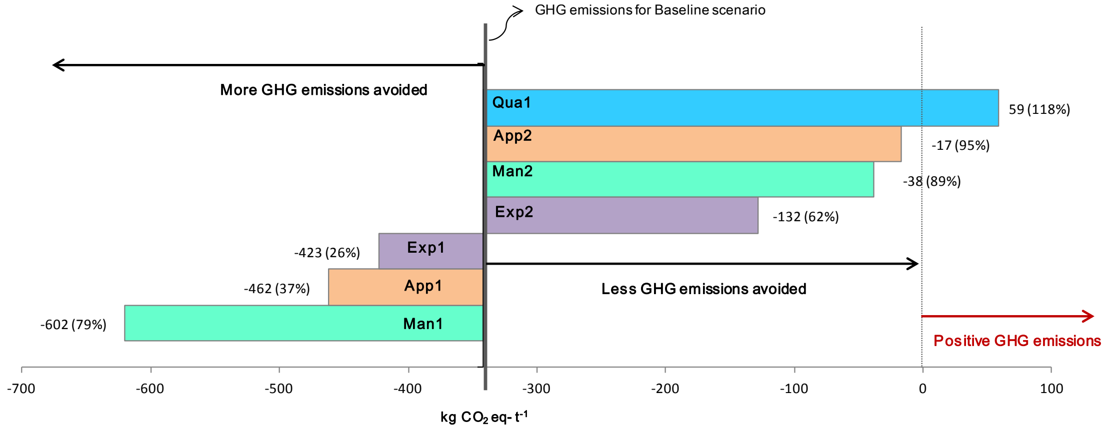

Figure C1: GHG emissions for the alternative scenarios (kg CO_2 eq. t^{-1}) for the sensitivity assessment

Appendix.D. The role of dynamic perspectives to model future scenarios: case of aluminium old scrap and plastic post-consumer

based on the following contribution Eva Sevigné Itoiz, Carles M. Gasol, Joan Rieradevall and Xavier Gabarrell. The role of dynamic perspectives to model future scenarios for attributional and consequential life cycle assessments: case of aluminium old scrap

D.1. Lifetime and in-use stock of aluminium and plastic products

Dynamic MFA has been intensively applied to quantify material cycles and in-use stocks, offering new perspectives on a variety of topics, such as in what pattern and proportion materials are used and how stocks affect future material demand, recycling potential, and associated environmental impacts (Liu and Müller, 2013b). Stocks of products and materials in-use are a major cause of disconnection between the system's inflow and its outflow in one year. Ignoring them may lead to erratic forecasts of future emissions and waste streams (Elshkaki et al., 2005). Thus, estimates of future waste flows are important for environmental policy.

In-use stock have been quantified in different studies using either a top-down approach based on estimation of annual consumption and product lifetime assumptions, or a bottom-up approach based on quantities of products in use and their concentrations (Liu and Müller., 2013b). In this thesis, we calculated the in-use stock of aluminium and plastic products in **Chapters 6** and **7** as the difference between consumption of year j minus waste generation on that year plus in-use stock of year j-1. With a top down approach it is possible to calculate the future waste generation from this in-use stock depending on the lifetime of the products. In this regard, the product lifetime can be assumed to be fixed. However, it is clear that this approach relies on considerable simplifications of reality since it neglects the uncertainty inherent to any ageing process. It can be assumed that the disposal of post-consumer articles occurs when their service life reaches t years with t ε [a, b], and a and b denoting the minimum and maximum average age of products upon disposal, respectively (Melo, 1999). Realistic intervals for the lifetime of aluminium applications are very difficult to obtain as a result of the large diversity of products and the lack of information on their age upon disposal. Therefore, in order to calculate future old scrap generation and post-consumer plastic generation, we used the lifetime intervals summarizes in Table D1 for aluminium products and plastic products classified by type as conducted in **Chapters 6** and **7** (Kaps, 2008; Ciacci et al., 2013).

Table D1: Lifetime intervals for aluminium products and plastic products by type of end use category

Aluminium products		
End use category (k)	**a**	**b**
Transport	7	20
Packaging	1	1
Construction & building	30	50
Engineering & electronics	20	30
Others	9	14

Plastic products		
End use category (k)	**a**	**b**
Agriculture	1	3
Electronics	5	50
Construction & building	30	50
Packaging	1	1
Automotive	7	20
Others	5	15

D.2. Lifetime distribution models

According to Melo (1999), the future waste generation could be calculated using different lifetime distribution models based on the idea that some kind of gradual ageing takes place yielding an increasing probability until a certain age after which the probability of scrapping a product gradually declines (Melo, 1999). Different distribution models can be used to calculate the future waste generation such as the normal distribution, the Weibull distribution or the beta distribution. In this study we have selected the normal distribution. Thus, the expected volume of waste that is theoretically generated in year j in the end use category k is calculated by [Eq. D.1] and within the normal distribution we can determine the probability p_t by [Eq. D.2] and [Eq. D.3].

$$S_{j,k} = \sum_{t-a}^{b-1} c_{(j-t)k} p_t \qquad \text{[Eq. D.1]}$$

$$p_t = \int_{t}^{t+1} \frac{1}{\sigma\sqrt{2\pi}} \exp\left\{-(x-\mu)^2/2\sigma^2\right\} dx, \qquad a \le t \le b \qquad \text{[Eq. D.2]}$$

$$f(x) = \frac{1}{\sigma\sqrt{2\pi}} \exp\left\{-(x-\mu)^2/2\sigma^2\right\} \qquad \text{[Eq. D.3]}$$

With $c_{(j-t)_k}$ denoting the amount of waste (i.e., aluminium or plastic) consumed in year j-t in the category k; σ is the standard deviation, μ is the mean and σ^2 is the variance of the distribution.

Other studies compared the aluminium old scrap results obtained from different lifetime distributions and they concluded that results were not very sensitive to differences in neither distribution models nor the standard deviations of lifespan but were sensitive to the mean values of lifespans (Melo, 1999; Müller et al., 2011; Chen and Shi, 2012).

D.3. End use consumption data before 1995

We conducted the MFA for the aluminium life cycle and plastic life cycle MFA between 1995-2010 and 1999-2011, respectively. In order to calculate the future waste generation based on previous consumption we used, in the case of aluminium life cycle, data provided by Liu (2014) from 1960 to 1994. In the case of plastic life cycle, we calculated the consumption by type of end use category considering the plastic consumption per capita of 1999 and assuming same consumption per capita between 1985 and 1999. In addition, for both calculations we assumed same packaging consumption from 2010 and 2011 until 2020 for aluminium and plastic, respectively.

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