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Life cycle model of waste to energy technologies in Spain and Portugal

Modelo de ciclo de vida de tecnologías para la valorización energética de residuos municipales en España y Portugal

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This thesis is a compendium of papers published or accepted for publication in international scientific journals, fulfilling the requirements of the Department of Chemical and Biomolecular Engineering for the elaboration of a thesis as a compendium of publications. During the elaboration of this thesis, a research stay of three moths (October-December 2011) was conducted at the *Universität Stuttgart* under the supervision of Matthias Fischer.

The papers included in this thesis are listed as follows:

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A mi familia

A la memoria de Pepe

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Contents

List of Figures

List of Tables

Resumen

La gran cantidad de residuos que se genera en la sociedad actual ha situado la gestión de residuos como una prioridad de las políticas europeas en materia de medioambiente. Las regulaciones ambientales desarrolladas con tal fin proponen como primera medida de gestión de residuos su prevención, seguido de la reutilización y el reciclaje, y por último, medidas como la incineración y el vertido. A pesar de que actualmente en Europa el vertido es la alternativa de gestión más utilizada, la incineración de residuos y el reciclaje están tomando cada día más protagonismo. En concreto, la incineración permite reducir el volumen y la masa de los residuos sólidos, así como recuperar la energía de los residuos con un poder calorífico significativo. Sin embargo, esta tecnología se ha ganado una mala reputación debido principalmente al impacto ambiental generado por la emisión de sustancias contaminantes como gases ácidos, dioxinas y furanos y gases de efecto invernadero. Para evaluar las ventajas y desventajas, así como los impactos ambientales asociados a esta tecnología, el uso de herramientas de gestión ambiental juega un papel importante. Una de las metodologías más utilizadas es el análisis de ciclo de vida (ACV), que permite evaluar los impactos ambientales de procesos, productos y servicios a lo largo de todo su ciclo de vida. En este contexto, la presente Tesis Doctoral aplica la metodología de ACV para analizar, desde una perspectiva ambiental, el proceso de incineración de residuos municipales en la península ibérica. En particular, se evaluó la tecnología aplicada en estos países, al existir una falta de estudios centrados en España y Portugal; países que debido a su proximidad geográfica, así como a que ambos están tutelados por la misma legislación europea (aunque después desarrollen su propia legislación estatal y local) aplican sistemas de gestión de residuos municipales similares.

En primer lugar, en esta tesis se realiza una revisión del estado del arte de la tecnología de incineración en Europa, así como un diagnóstico de la situación actual de la tecnología en España y Portugal. Una vez determinadas las principales técnicas aplicadas en estos países, se procede a la recogida de los datos de entrada y salida del proceso de incineración con el fin de elaborar un inventario representativo de la tecnología aplicada en la península ibérica.

En segundo lugar, partiendo de este inventario, se desarrolla un modelo de asignación de carga multi-entrada/multi-salida de 18 fracciones de residuos municipales. Dicho modelo, se aplica a diferentes plantas incineradoras españolas con el fin de comparar su comportamiento ambiental, así como determinar adicionalmente los puntos críticos del proceso. Para llevar a cabo este análisis se aplica un grupo de indicadores ambientales basados en el consumo de recursos naturales y en las cargas ambientales generadas. Así mismo, para reducir la complejidad de los resultados, se establece una metodología de normalización y ponderación que permite ayudar en el proceso de toma de decisiones.

Por último, se evalúa otro de los problemas asociados a la incineración, la gestión de los residuos sólidos generados en la combustión y el tratamiento de los gases. En concreto, se analiza y compara desde una perspectiva de ciclo de vida los tratamientos convencionales de gestión de cenizas con el reciclaje de las mismas en la fabricación de cemento Portland.

Abstract

The high rate of waste generation in the society today has brought waste management to be a priority in European Policies. The European environmental Regulation proposes waste prevention, reuse, recycling, and finally waste incineration and landfilling as fundamental principles. Despite landfilling remaining the most common practice, waste incineration and recycling have increased in recent years. In particular, waste incineration allows the reduction in waste mass and volume, and the energy recovery. However, incineration has gained a bad reputation because of its environmental impacts, specifically due to its emission of greenhouse gases, acid gases, and dioxins and furans. To assess the advantages, disadvantages, and environmental impacts of this technology, the use of environmental tools plays an important role. One of the most recognised methodologies is life cycle assessment (LCA), a powerful tool for assessing the environmental impacts of a product, process, or service along the whole life cycle. In this context, the present PhD Thesis, applies the LCA methodology to analyse from an environmental point of view, waste to energy (WtE) technologies in the Iberian Peninsula. The motivation to carry out this research work in these countries was the lack of studies focused in Spain and Portugal; countries with a similar WtE technologies due to the geographical proximity and the implementation of the European Regulations.

Initially in this thesis, the state of the art of WtE technologies in Europe, and the diagnosis of the current situation of the technology in Spain and Portugal are reviewed. Once the main techniques applied in these countries are identified, the input and output data of the WtE process are collected to compile a representative inventory of the technology in the Iberian Peninsula.

Secondly, on the basis of this inventory, a multi-input/multi-output allocation model for 18 municipal waste fractions. This model is applied to several WtE plants in Spain in order to assess its environmental performance, as well as the critical points of the process. This analysis is carried out by means of a set of environmental metrics based on the consumption of natural resources and the generation of environmental burdens. Moreover, to reduce the LCA complexity and assist the decision-making process, a normalisation and weighting procedure was established.

Finally, other important problem in waste incineration is assessed: the generation of solid waste in the combustion and flue gases cleaning stages. Specifically, it is analysed and compared from a life cycle approach, the conventional ash treatment and ash recycling in Portland cement manufacture.

Abbreviations and Glossary

ABBREVIATIONS

AA	Atmospheric acidification
AEVERSU	Spanish association of municipal solid waste valorisation
AOD	Aquatic oxygen demand
AqA	Aquatic acidification
BA	Bottom ash
BAT	Best available techniques
BFB	Bubbling fluidised bed
BREF	Best available techniques reference document
CFB	Circulating fluidised bed
Cr-Ga	Cradle to gate
Cr-Gr	Cradle to grave
EB	Environmental burden
EBS	Environmental burden sustainability
EMS	Environmental management system
E-PRTR	European Pollutant Release and Transfer Register
ESA	Environmental sustainability assessment
EU	Eutrophication

Eurostat	Statistical office of the European Communities
FA	Fly ash
FB	Fluidised bed incinerator
FGT	Flue gas treatment
FU	Functional unit
Ga-Ga	Gate to gate
Ga-Gr	Gate to grave
GDP	Gross domestic product
GI	Grate incinerator
GW	Global warming
HDPE	High density polyethylene
HHV	High heating value
HW	Hazardous waste
IChemE	Institution of Chemical Engineers
ILCD	International life cycle database
IPPC	Integrated Pollution and Prevention Control
LCA	Life cycle assessment
LCI	Life cycle inventory
LCIA	Life cycle impact assessment
LCT	Life cycle thinking
LDPE	Low density polyethylene
LHV	Low heating value
MEco	Ecotoxicity to aquatic life (metals)
MSW	Municipal solid waste
MSWI	Municipal solid waste incineration
NMEco	Ecotoxicity to aquatic life (others)

NMVOC	Non methane volatile organic compounds
nP	Non packaging
NR	Natural resources
NRS	Natural resources sustainability
P	Packaging
PAH	Polycyclic aromatic hydrocarbons
PC	Paper and cardboard
PCDD/F	Dioxins and furans
PET	Polyethylene terephthalate
POF	Photochemical ozone formation
RDF	Refused derived fuel
RK	Rotary kiln
SCR	Selective catalytic reduction
SFB	Stationary fluidised bed
SNCR	Selective non-catalytic reduction
SOD	Stratospheric ozone depletion
TOC	Total organic compounds
TSP	Total suspended particles
WFD	Waste framework directive
WtE	Waste to energy

GLOSSARY

Allocation - partitioning the input or output flows of a process or a product system between the product system under study and one or more other product systems.

Ancillary input - material input that is used by the unit process producing the product, but does not constitute part of the product.

Candidate countries - Croatia, the Former Yugoslav Republic of Macedonia, Turkey, and Bosnia and Herzegovina (potential candidate).

Co-product - any of two or more products coming from the same unit process or product system.

Cut-off criteria - specification of the amount of material or energy flow or the level of environmental impact that is considered the minimal significant contribution to the total results, to be used as criterion for exclusion from the study of the unit processes or product system that contribute less.

Data quality - characteristics of data that relate to their ability to satisfy stated requirements.

Ecoembes - Ecoembalajes Españoles S.A is a non-profit-making organisation in charge of the recovery and recycling management of plastic, paper and cardboard and beverage carton packaging.

EFTA Countries - The European Free Trade Association is a free trade organisation between four European countries that operates in parallel with – and is linked to – the European Union (EU). Today's EFTA members are Iceland, Liechtenstein, Norway, and Switzerland.

Elementary flow - material or energy entering the system being studied that has been drawn from the environment without previous human transformation, or material or energy leaving the system being studied that is released into the environment without subsequent human transformation.

Energy flow - input to or output from a unit process or product system, quantified in energy units.

Environmental burden - The environmental burden approach (developed by ICI) is a scientifically sound way to quantify environmental performance. It draws on developments in environmental science to estimate potential environmental impact, rather than merely stating quantities of material discharged.

EU-15 - Germany, Austria, Belgium, Denmark, Spain, Finland, France, Greece, Ireland, Italy, Luxembourg, the Netherlands, Portugal, United Kingdom, and Sweden.

EU-27 - Belgium, Bulgaria, Czech Republic, Denmark, Germany, Estonia, Ireland, Greece, Spain, France, Italy, Cyprus, Latvia, Lithuania, Luxembourg, Hungary, Malta, Netherlands, Austria, Poland, Portugal, Romania, Slovenia, Slovakia, Finland, Sweden, and United Kingdom.

Functional unit - quantified performance of a product system for use as a reference unit.

Incineration plant - many stationary or mobile technical unit and equipment dedicated to the thermal treatment of wastes with or without recovery of the combustion heat generated.

Input - product, material or energy flow that enters a unit process.

Integrated Pollution Prevention Control permit (IPPC permit) - a necessary document for those installations included in the IPPC Directive that allows them protecting the environmental and the human health, as well as the total or partial exploitation of the installation according to the IPPC Directive.

Intermediate flow - product, material or energy flow occurring between unit processes of the product system being studied.

Intermediate product - output from a unit process that is input to other unit processes that require further transformation within the system.

Life cycle - consecutive and interlinked stages of a product system, from raw material acquisition or generation from natural resources to final disposal.

Life cycle assessment (LCA) - compilation and evaluation of the inputs, outputs and the potential environmental impacts of a product system throughout its life cycle.

Life cycle impact assessment - phase of life cycle assessment in which the inputs and outputs data collected in the life cycle inventory are translated into an impact indicator results related to human health, natural environment, and resource depletion.

Life cycle inventory analysis (LCI) - phase of life cycle assessment involving the compilation and quantification of inputs and outputs for a product throughout its life cycle.

Life cycle inventory analysis result (LCI result) - outcome of a life cycle inventory analysis that catalogues the flows crossing the system boundary

and provides the starting point for life cycle impact assessment.

Municipal solid waste - materials we use and then throw away, such as product packaging, grass clippings, furniture, clothing, bottles, food scraps, newspapers, appliances, paint, and batteries. This comes from our homes, schools, hospitals, and businesses.

Output - product, material or energy flow that leaves a unit process.

Process - set of interrelated or interacting activities that transforms inputs into outputs.

Product - any good or service.

Product flow - products entering from or leaving to another product system.

Product system - collection of unit processes with elementary and product flows, performing one or more defined functions, and which models the life cycle of a product.

Raw material - primary or secondary material that is used to produce a product.

Reference flow - measure of the outputs from processes in a given product system required to fulfil the function expressed by the functional unit.

Residue - means any liquid or solid material (such as bottom ash and slag, fly ash and boiler dust) which is generated by the incineration or co-incineration process.

System boundary - set of criteria specifying which unit processes are part of a product system.

Unit process - smallest element considered in the life cycle inventory analysis for which input and output data are quantified.

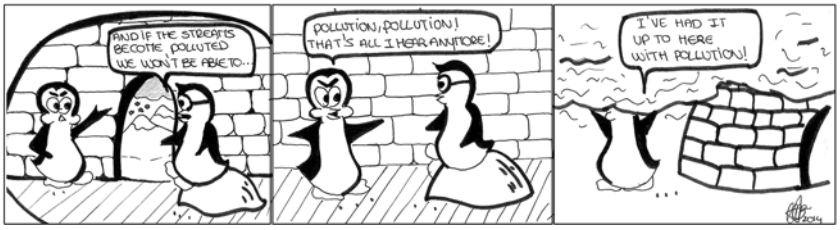
Waste - substances or objects which the holder intends or is required to dispose of.

Waste management - the collection, transport, recovery, and disposal of waste, including the supervision of such operations and the after-care of disposal sites, and including actions taken as a dealer or broker.

Waste to energy - the conversion of non-recyclable waste materials into useable heat, electricity, or fuel through a variety of processes, including combustion, gasification, pyrolization, anaerobic digestion, and landfill gas (LFG) recovery.

1. INTRODUCTION

Waste management and Life Cycle Assessment



Adapted from Johnny Hart

1.1 Waste management

1.1.1 Overview

Waste generation is not a new phenomenon. In the mankind beginnings, waste disposal was not a significant problem because of the low population density and the huge available land. As time went by, humans become sedentary, abandoned their life as hunters and gatherers, started to cultivate crops and larger and larger populations were made possible (Klang 2005). The consumption patterns in these population settlements were based on easy-decomposition food, generating waste easily assimilated by the environment.

However, the population growth and the non-existence of waste management systems started to cause problems. References of waste problems were already found in the ancient Rome. In this period, several measures to avoid waste disposal and water pollution were adopted in the cities. A sign of these measures is the marvelous sewer system of the city of Rome. During the Middle Age, food refuse in the cities entailed rats and bubonic flea reproduction. The lack of a waste management system may suppose the well-known Black Death. Nevertheless, it was in the 19th century, with the Industrial Revolution, when the negative side of the increased waste generation became obvious. Waste management systems were still poor, leading to significant illness, such as cholera or typhus. This was joined by the appearance of new types of waste as a result of the technological advances. This lead to the appearance in the late 19th and early 20th century of the first health control measures. The most common final treatment methods in this era were land and water dumping, use as animal food, reduction, and incineration (Tchobanoglous and Kreith 2002). In particular, incineration

appeared in the late 19th century as a revolutionary way of waste elimination due to the mass, volume, and hazard reduction. However, since the incineration technology was still at the beginnings, smoke and fire were associated with combustion processes. For this reason, although this technology was so extended in the United States of America, in 1909 more than 100 incineration plants were closed down (Colomer and Gallardo 2007). On the other side, until the 1900s, solid waste was dumped directly on the land. Around 1910, methods for creating controlled landfills or sanitary landfills were developed. Specifically, in the early 1930s in the United Kingdom, and in the 1940s in USA, the first sanitary landfills appeared. Until the 1950s open-pit dumping was standard practice and the wastes were often burned in situ. Spontaneous combustion sometimes occurred, and at other times controlled burning practices were followed for volume control. Considerable problems of odours, noise, seagulls, and smoke were obvious and immediate environmental impacts (McBean et al. 1995). As environmental awareness grew in the 1960s, much of these combustion practices were forbidden, and the amount of landfill rose. Nevertheless, in the 1970s it became more and more difficult to find suitable sites for new landfill. The Not-In-My-Backyard, or NIMBY, attitude has made it increasingly difficult to establish new landfill site, whereas consumption patterns have continued the trend towards an intensified generation of waste (Klang 2005).

In the last 30 years, waste has been at the centre of European Union environment policy and substantial progress has been made. Heavily polluting landfills and incinerators are being cleaned up. New techniques have been developed for the treatment of hazardous waste. With time, waste is increasingly seen as a valuable resource for industry. Approaches such as re-use, recycling, and energy recovery are starting to be applied to regulated wastes. However, despite these successes, waste remains a problem, since waste volumes continue to grow (COM 2005).

1.1.2 Waste generation and management in Europe

Waste generation in Europe has increased regularly in recent years, amounting to approximately 2.5 billion tons of waste in 2012. Municipal Solid Waste (MSW) makes up the 9.8 % of the total waste production in Europe, generating approximately 246.6 million tons in 2012, representing an annual per capita generation rate of 492 kg of MSW person^{-1} (Margallo et al. 2014a).

Since 1995, the generation of MSW in the European Union (EU-27) has shown a steady increase until 2002. In this period, MSW generation increased a 13 %, from 226.2 million tons (474 kg MSW person^{-1}) to 225.8 tons (527 MSW kg person^{-1}). In 2003 and 2004 this growth was interrupted, which can be attributed to the changes in the methodology and classifications that took place in many countries in this period. This reduction was followed by an increase from 2004 to 2007, and by a new declination from 2008 to 2012 by 13 million tons, corresponding to an annual decrease of 1.25 %. This diminish could be associated to a population drop, however, as Fig 1.1 shows, the number of inhabitants in the EU-27 has regularly increased from 1995 to 2012. Therefore, the population-related indicator on municipal waste generated (kg MSW person^{-1}) follows the same trend than the amount of waste generated (tons) (Eurostat 2011).

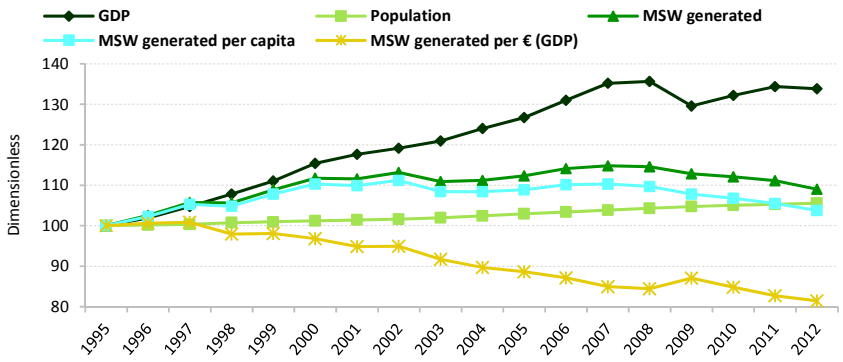


Fig 1.1 Evolution of GDP, population, and waste generation, considering year 1995=100.

Decoupling waste generation from gross domestic product (GDP) is essential to ensure sustainable use of the resources of the world (UNEP 2010). Decoupling can be relative if the MSW generation grows less than the GDP, or absolute if the MSW generation decreases although the GDP increases (OSE 2011). GDP shows, in the EU-27, an increasing trend with an annual growth rate of 2.2 % from 1995 to 2008, exceeding the MSW generation in the same period (1.1 %). Nevertheless, in 2009 the economic decline was even sharper than waste generation, leading to an increase of the indicator of MSW generation per €.

According to this, the waste generation decrease in 2009 reflects, at least in part, the downturn in economic activity as a result of the financial and

economic crisis (Eurostat 2011; Eurostat 2013). Nevertheless, since 2010 the GDP has been increasing, whereas MSW generation continued decreasing, noticing a more pronounced decoupling. However, more data are required to foresee if MSW generation will maintain this trend.

The total generation of MSW in 2012 varies considerably from country to country. Fig 1.2 groups the European countries by the annual per capita generation rate into 6 categories. In the first group (552-694 kg MSW person^{-1}) the countries with the highest generation ratio are included. In the lead, Switzerland (694 kg MSW person^{-1}) followed by Denmark (668 kg MSW person^{-1}), Luxembourg, and Cyprus (663 and 662 kg MSW person^{-1}). On the other side, Czech Republic, Poland, Latvia, Slovakia, Estonia, Iceland, and Bosnia and Herzegovina showed values below 346 kg MSW person^{-1}. The variation in the waste generation rate reflects differences in consumption patterns and economic wealth of the countries, but also depends greatly on the organisation of municipal waste collection and management (Eurostat 2011). In the second group are found Spain and Portugal, with a MSW generation ratio between 472 and 552 kg person^{-1}. In 2012, a generation rate of 464 and 453 kg MSW person^{-1} was reached in Spain and Portugal respectively, values below the EU-27 rate (492 kg MSW person^{-1}).

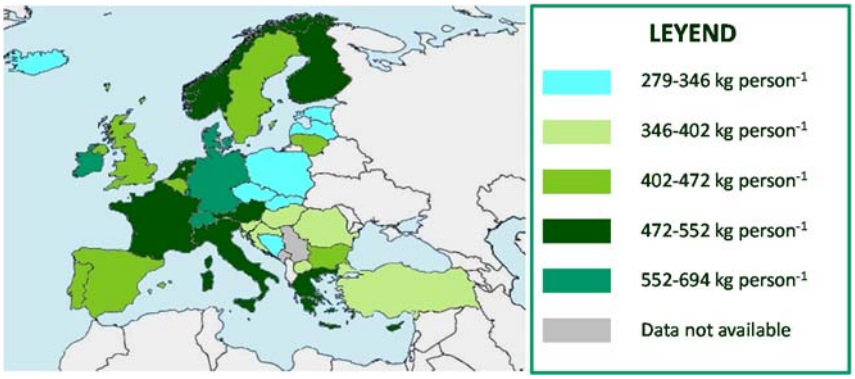


Fig 1.2 Map of MSW generation (kg MSW person^{-1}) in Europe.

However, as Fig 1.3 shows, Spain presented a higher generation rate than the EU-27 ratio from 1995 to 2011, year in which the situation was reversed. Portugal only exceeded the EU-27 quota in 2009 and 2010 (Eurostat 2013).

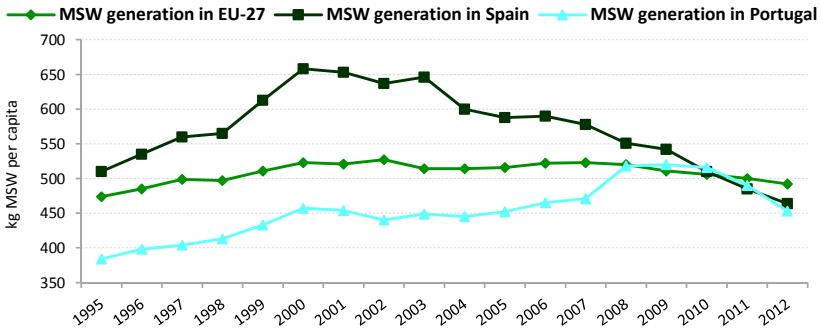


Fig 1.3 Evolution of MSW generation in Spain, Portugal, and EU-27.

Fig 1.4 illustrates the treatment method in units of kg per capita in the EU-27 for the period from 1995 to 2012.

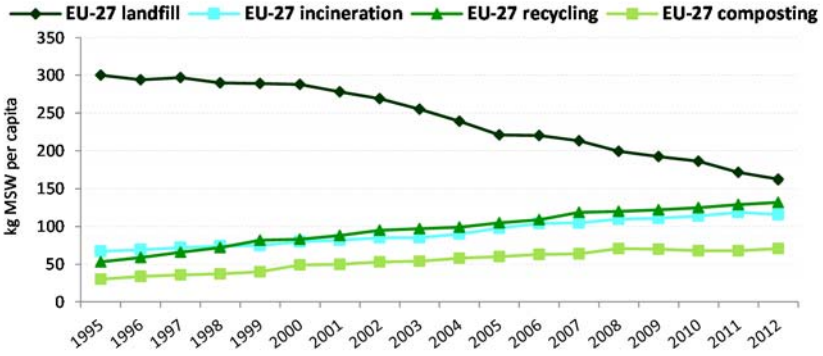


Fig 1.4 Evolution of waste management treatments in EU-27.

Although waste generation increased in this period, the amounts of MSW landfilled were reduced. The landfilled total in the EU-27 declined from 143 million tons in 1995 (300 kg MSW person^{-1}) to 81.3 million tons in 2012 (162 kg MSW person^{-1}). As a result, the share of landfilling in the EU-27 dropped from 68 % in 1995 to 34 % in 2012. This reduction can partly be attributed to the implementation of the Directive 94/62/EC on packaging and packaging waste that sets a recovery target of packaging of 60 % to be achieved by 2008, and the Directive 1999/31/EEC on the landfill which requires Member States to reduce the amount of biodegradable municipal waste landfilled to 35 % by 2016. In consequence, the amounts of waste recycled, incinerated, and composted increased. In particular, the rates of MSW recycled and composted increased in the period 1995-2012 from 11.7 % to 27.3 % (53 to

132 kg MSW person^{-1}) and from 6 % to 15 % (30 to 71 kg MSW person^{-1}) respectively.

Waste incineration has also grown in that period, although not to the extent of recycling and composting. In 1995, 32.2 million tons of MSW were incinerated in the EU-27 (67 kg MSW person^{-1}), whereas in 2012 this value increased to 58 million tons (116 kg MSW person^{-1}). The ratio of incineration increased from 15 % in 1995 to 24 % in 2012 (Eurostat 2013). A similar trend is followed in Spain and Portugal, however, a higher amount of waste was landfilled. In 1995 in Spain, 83.7 % of the MSW was landfilled, 6.5 % incinerated, and 9.8 % recycled. In 2012, the landfilling percentage decreased to 63 %, increasing the incineration quota to 9 %, and the recycling and composting to 17 % and 10 %, respectively. Composting was implemented in 1999 and has shaped as the second or third waste treatment alternative reaching the highest share in 2008 (24 %). In Portugal in 1995, composting (19 %) and landfill (75 %) were the only waste treatment alternatives. Incineration appeared in 2000 as an alternative waste treatment with a percentage of 21 %. From 1995 to 2012, the landfilling and composting rates dropped respectively from 75 % to 55 % and from 19 % to 15 %, becoming steady the incineration share and increasing the recycling one.

In order to give an overview of waste management in these countries at European level, Fig 1.5 presents the amounts of municipal waste landfilled, incinerated, recycled, and composted in 2012 as a percentage of the total amounts treated (sorted by the percentage of waste landfilled relative to the total amounts treated). The data were obtained from Eurostat; nevertheless, differences in the supply of waste information of each country were observed. All Member States supplied either data or estimates to Eurostat. For areas not covered by a municipal waste collection scheme, the amount of waste generated is estimated by the reporting country or by Eurostat.

Fig 1.5 illustrates the huge differences between countries with regard to the state of their waste management systems. Several countries are very advanced in diverting MSW from landfills, often due to the implementation of national measures to reduce landfilling of municipal waste, such as bans on organic waste or untreated municipal waste. Specifically, Switzerland (CH), Germany (DE), the Netherlands (NL), Sweden (SE), Austria (AT), Denmark (DK), and Belgium (BE) have reported landfill rates below 3 % due to the application of these measures. In the so-called old Member States (EU-15),

landfill ratios were below 40 % except for Greece (EL) (82 %), Spain (ES) (63 %), Portugal (PT) (55 %), and Italy (IT) (43 %).

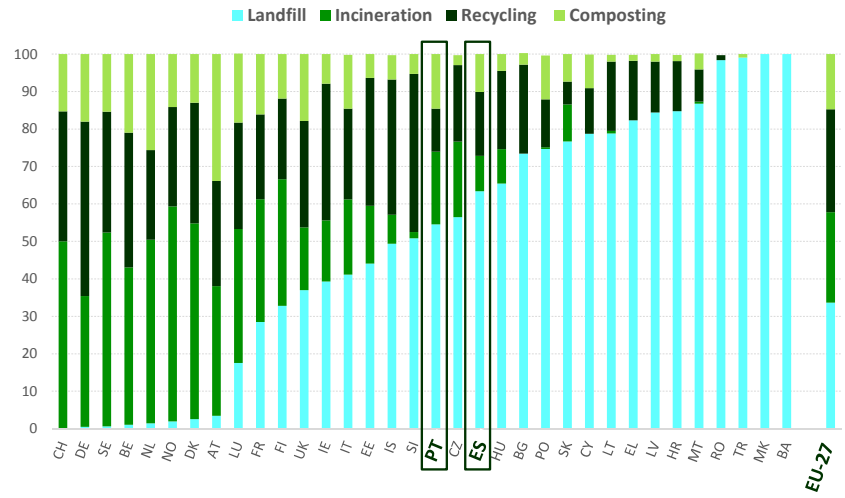


Fig 1.5 Municipal solid waste treated in 2012 by country (EU-27) and treatment category sorted by percentage of landfilling[1].

In the new Member States and the Candidate Countries, as well as in Iceland (IS), landfilling is still the predominant waste management option. Landfill rates in these countries range from 44 % in Estonia (EE) to 100 % in Bosnia and Herzegovina (BA) and the Former Yugoslav Republic of Macedonia (MK). The situation is further characterised by a low number of waste incineration facilities on the one hand, and collection and recycling schemes that are partly still in their infancy on the other hand. Mostly, the countries with low landfill rates have a larger combined share of recycling and composting than incineration. Among these countries, the highest shares for recycling were reported by Germany (47 %) and Belgium (36 %), whereas Austria (34 %) and The Netherlands (26 %) reported the largest rates of composting. Within the old members, Greece, Spain, and Portugal require to increase the quota of recycling and composting in order to reduce the landfill rate (Eurostat 2011).

[1] ***AT**: Austria, **BA**: Bosnia and Herzegovina, **BE**: Belgium, **BG**: Bulgaria, **CH**: Switzerland, **CY**: Cyprus, **CZ**: Czech Republic, **DE**: Germany, **DK**: Denmark, **EE**: Estonia, **EL**: Greece, **ES**: Spain, **FI**: Finland, **FR**: France, **HR**: Croatia, **HU**: Hungary, **IE**: Ireland, **IS**: Iceland, **IT**: Italy, **LT**: Lithuania, **LU**: Luxembourg, **LV**: Latvia, **MK**: The former Yugoslav Republic of Macedonia (fYR), **MT**: Malta, **NL**: The Netherlands, **NO**: Norway, **PL**: Poland, **PT**: Portugal, **SE**: Sweden, **SI**: Slovenia, **SK**: Slovakia, **TR**: Turkey, **UK**: United Kingdom.*

Regarding incineration, in 2012 in the EU-27, 116 kg MSW person^{-1} were incinerated in 453 incineration plants or Waste to Energy (WtE) plants (CEWEP 2012). In this year, 12 countries exceeded the European ratio.

As Fig 1.6 shows, the highest incineration values were observed in the old members, in particular Denmark with 29 incinerators (52 %, 349 kg person^{-1}), Switzerland with 30 plants (50 %, 347 kg person^{-1}), and the Netherlands with 12 plants (49 %, 270 kg person^{-1}), as well as in Norway (57 %, 268 kg person^{-1}) and Switzerland (50 %, 347 kg person^{-1}).

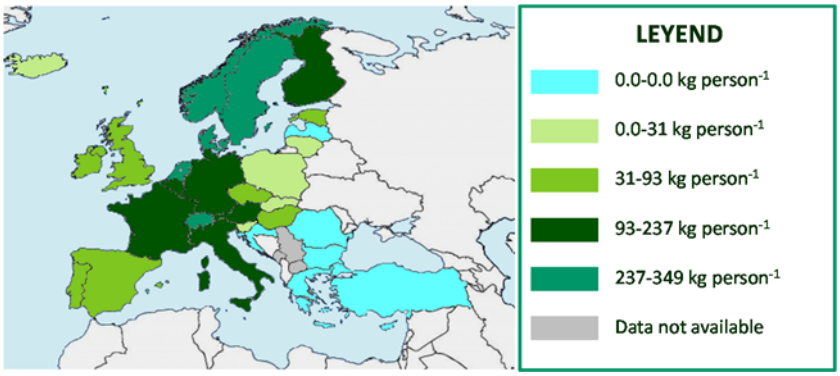


Fig 1.6 Map of incineration rates in Europe.

Particularly, nowadays, Switzerland is the European country with more plants per capita (one plant per 260,000 inhabitants) and km^2 (one plant each 1376 km^2). On the other hand, France (FR) is the European country with more incinerators (129 plants), since is the country with the greatest area and the second in number of inhabitants.

In other countries, such as Turkey (TK), Greece, Cyprus (CY), Latvia (LV), Croatia (HR), Bulgaria (BG), Bosnia and Herzegovina, the Former Yugoslav Republic of Macedonia, and Romania incineration is not an alternative to landfilling. Finally, Spain and Portugal with an annual incineration rate of 44 kg and 88 kg person^{-1} count with 10 and three WtE plants, respectively (Eurostat 2011).

1.1.3 Legal framework

Fig 1.7 shows the European waste legislation, which is divided in three sections: framework legislation, legislation on waste treatment, and waste stream legislation (EC 2014). On the other hand, in USA, the public law

that creates the framework for the proper management of hazardous and non-hazardous solid waste is the Resource Conservation and Recovery Act (RCRA). RCRA sets national goals for: protecting human health and the environment from the potential hazards of waste disposal, conserving energy and natural resources, reducing the amount of waste generated, and ensuring that wastes are managed in an environmentally-sound manner. To achieve these goals, RCRA established three distinct, yet interrelated, programs: the solid waste program, the hazardous waste program, and the underground storage tank (UST) program. Specifically, the solid waste program encourages states to develop comprehensive plans to manage non-hazardous industrial solid waste and municipal solid waste, sets criteria for municipal solid waste landfills and other solid waste disposal facilities, and prohibits the open dumping of solid waste (EPA 2014).

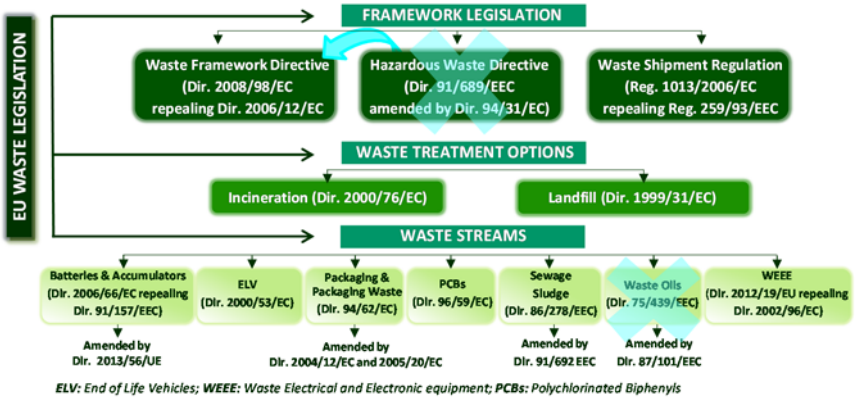


Fig 1.7 European waste framework legislation.

Framework legislation

In the 1970s and 1980s a number of problems related to the handling of waste alerted policy-makers to the potential impact that poorly managed waste could have upon the environment and human health. The Member States began taking national measures to control and manage waste, which then led to the first Waste Framework Directive (WFD) -Directive 75/442/EEC-, the Hazardous Waste Directive -Directive 91/689/EEC- and later, to the Waste Shipment Regulation -Regulation 259/93/EEC- (EC 2014). Directive 75/442/EEC was consequently modified by the Directive 91/156/EEC, the Directive 2006/12/EC, and the Directive 2008/98/EC (EC 2008). The latter Directive repealed the Directive 75/439/EEC on waste oils (amended by

Directive 87/101/EEC), the Directive 91/689/EEC on hazardous waste (amended by Directive 94/31/EC), and the previous Directive 2006/12/EEC on waste.

The first measures proposed in the WFD were end of pipe solutions once wastes were already generated. However, this approach is changing, focusing the current legislation more in prevention and minimisation measures than in treatment. Specifically, the current Directive 2008/98/EC (EC 2008) sets the basic concepts and definitions related to waste management and lays down some basic waste management principles. This Directive proposes the application of the waste hierarchy shown in Fig 1.8, which is based on three fundamentals principles: waste prevention, recycling and reuse, and improving final disposal and monitoring (Branchini 2012).

Fig 1.8 Priority order in waste prevention and management.

Directive 2008/98/EC incorporates provisions on hazardous waste and waste oils, and also includes two new recycling and recovery targets to be achieved by 2020: 50 % preparing for re-use and recycling of certain waste materials from households, and 70 % preparing for re-use, recycling and other recovery of construction and demolition waste (EC 2008). Finally, the Directive introduces the life cycle thinking (LCT) approach, clarifying when waste ceases to be waste (COM 2005). In Spain, the Directive 2008/98/EC was transposed by the Law 22/2011 on waste and contaminated land (BOE 181 2011), repealing the Law 10/1008 on waste. In Portugal, the Decree-Law 73/2011 transposed the WFD (Diario da República 116 2011).

Waste treatment options in Europe

The environmental impacts of waste treatment have been reduced in the recent years through legislative measures (EC 2014). This trend was reinforced through the Directive 1999/31/EC on the landfill of waste (EC 1999) and the Directive 2000/76/EC on waste incineration (EC 2000b). Implementation of the landfill directive is one of the major drivers for the development of waste management policies at national level, including efforts to promote the diversion of waste towards material recycling and biological treatment. Particularly important are the restrictions on landfilling introduced by this directive, specifically the reduction in the amount of biodegradable waste going to landfill and the prohibition of the landfilling of certain waste types, including liquid waste and tyres (EC 2014).

Waste incineration operations have to fit the legislation in order to prevent or to limit as far as practicable negative effects on the environment, in particular pollution by emissions into air, soil, surface water and groundwater, and the resulting risks to human health. In this sense, the Directive 2000/76/EC presents the legal framework to regulate waste incineration in Europe and sets emission limit values for waste incineration and co-incineration plants as well as the required measures and controls (EC 2000b). This Community Directive was transposed in Spain by the Royal Decree 653/2003 (BOE 142 2003) and in Portugal by the Decree-Law 85/2005 (Diario da Republica 82 2005).

According to the Directive requirements, incineration plants are obliged to operate to a temperature of 850 ºC (1,100 ºC in some hazardous wastes) for two seconds to achieve a good burn out of the gases. Each line of the incineration plant has to be equipped with at least one auxiliary burner. This burner must be switched on automatically when the temperature of the combustion gases falls below 850 ºC. It shall also be used during plant start-up and shut-down operations in order to ensure that the temperature of 850 ºC is maintained in the combustion chamber. The auxiliary burner shall not be fed with fuels which can cause higher emissions than those resulting from the burning of gasoil, liquefied gas or natural gas. Incineration plants must be designed, equipped, built, and operated in such a way that the emission limit values set out in legislation are not exceeded in the exhaust gas (Rodríguez and Irabien 2013).

Discharges to the aquatic environment of wastewater from the cleaning of exhaust gases shall be limited as far as practicable. In any case, the wastewater may be discharged to the aquatic environment after treatment provided that the mass concentrations of the polluting substances do not exceed the corresponding emission limit values.

Additionally, under no circumstances shall dilution of wastewater take place for the purpose of complying with these emission limit values. Finally, measurement equipment must be installed and techniques used in order to monitor the parameters, conditions, and mass concentrations relevant to the incineration process (EC 2000b).

Waste streams in Europe

Better management of certain problematic waste streams has been achieved through specific directives addressing certain hazardous waste such as waste oils , polychlorinated biphenyls (PCBs) -Directive 96/59/EC (EC 1996)- and batteries and accumulators - Directive 2006/66/EC (EC 2006) amended by Directive 2013/56/EU (EU 2013)-. Additionally, the use of some waste, such as sewage sludge in agriculture has been regulated by the Directive 86/278/EEC (EEC 1986) amended by the Directive 91/692/EEC (EEC 1991). In other cases, for some key complex waste flows, recycling and recovery targets have been set, i.e. packaging and packaging waste -Dir. 94/62/EC (EC 1994) amended by Directive 2004/12/EC and Directive 2005/20/EC (EC 2004; EC 2005)-, end of life vehicles (ELV) - Directive 2000/53/EC (EC 2000a)- and Waste Electrical and Electronic Equipment (WEEE) - Directive 2012/1/EU (EU 2012)- (EC 2014). Related to packaging waste, the Directive 94/62/EC was transposed in Spain by the Law 11/1997 of 24 April, on packaging and packaging waste (BOE 99 1997), developed by the Royal Decree 782/1998 (BOE 104 1998). In Portugal the Decree-Law 366-A/97 of 20th December (Diario da República 293 1997) with the modifications included in the Decree-Law 162/2000 of 27th July (Diario da República 172 2000) and the Decree-Law 92/2006 of 25th May (Diario da República 101 2006) transposed the Directive 94/62/EC.

1.2 Application of LCA in waste management

The high amount of generated waste produces a huge impact on the environment. Good waste management might significantly reduce these impacts, and life cycle thinking (LCT) and life cycle assessment (LCA) can help policy makers choose the best environmental options (JRC 2014). As Fig 1.9 illustrates, LCA is a powerful tool for assessing the environmental performance of a product, process, or activity from raw material extraction ("cradle"), to the product is converted in a waste the final ("grave"). It can be used to support decision-making in order to identify cleaner and more sustainable alternatives in the process design activity (Rebitzer et al. 2004).

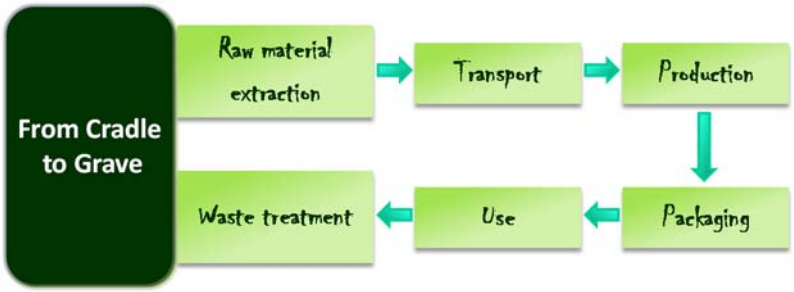


Fig 1.9 Life cycle assessment of a product: from "cradle" to "grave".

LCA is a standardised assessment and should be applied using the ISO 14040 (ISO 2006a) and 14044 (ISO 2006b) describing LCA as a four-phase process:

a) **Goal and scope definition**. This step defines the intended application of the study, the system description, the functional unit, the system boundaries, the allocation procedures and the assumptions.
b) **Life Cycle Inventory (LCI) analysis**. The relevant mass and energy input and output data are collected in a LCI.
c) **Life Cycle Impact Assessment (LCIA)**. LCIA quantifies the potential for

environmental impact over all of the stages involved in the delivery of a product or service.

d) **Interpretation**. In the last step, the interpretation of the LCI and LCIA results, the conclusions and recommendations are conducted.

All LCA studies, as much from a product as from a waste should follow the ISO requirements. Moreover, the LCA of a waste management system is divided in the same stages that the LCA of a product. The main difference between the LCA of a product and a waste resides in what it is meant by cradle and grave. Whilst it shares the same grave as individual products, the lifecycle of waste does not share the same cradle. Fig 1.10 summarises the key differences between a product and waste LCA (McDougall et al. 2001).

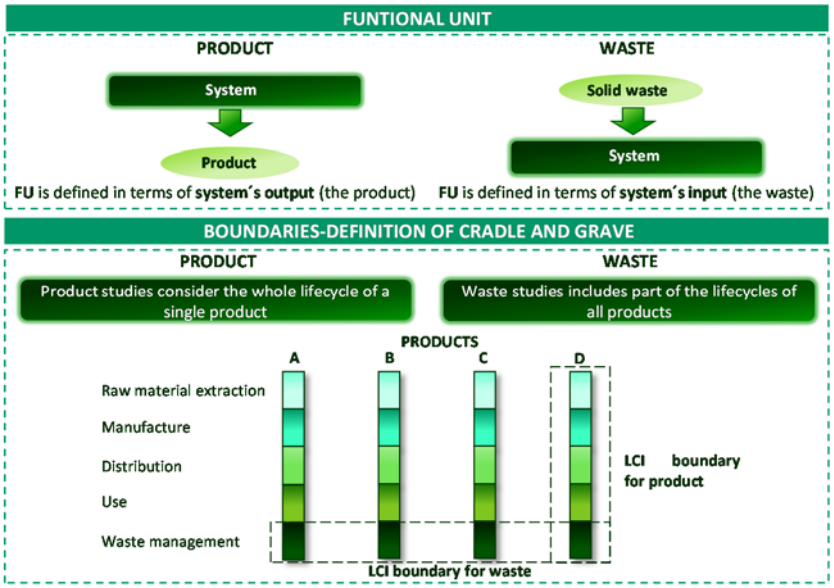


Fig 1.10 Differences between LCA studies for products and wastes.

Waste only becomes waste at the point at which it is thrown away; i.e. it ceases to have any value to the owner. Therefore, there is not impact associated to the extraction, production, and use, considering that waste starts with null impact. Thus, the cradle of household waste is usually the dustbin. This is another key difference between an LCA for a product and an LCA for waste. Every product spends part of its life cycle as waste. Conversely, a lifecycle study of waste includes part of the lifecycle of every product or package. Regarding the functional unit (FU), in products studies, the FU is

related to the product and the comparisons are made on the basis of per amount or per equivalent use of the product. In contrast, the function of a waste management system is not to produce anything, but to deal with the waste of a given area. Therefore, the FU in these studies is the waste of the geographical area under study. The key difference in approach is that in a product LCA the FU is defined by the output, i.e. the product of the system and in an LCA of a waste the FU is defined in terms of system's input, i.e. the waste (McDougall et al. 2001).

An increasing number of publications related to the LCA of waste were observed in recent years, starting from 1995, date of the first papers. The trend is likely to reflect the importance of LCA as an increasingly accepted approach to analyse the environmental performance of waste management (Laurent et al. 2014). Most of the LCAs were conducted in Europe, largely driven by Italy (Blengini et al. 2012, De Feo and Malvano 2009), Spain (Aranda Usón et al. 2013; Bovea et al. 2010; Bovea and Powel 2006; Muñoz et al. 2004; Rodríguez-Iglesias et al. 2003), Sweden (Bernstad et el. 2011; Klang et al. 2008), and Denmark (Kirkeby et al. 2006), and in less extend in the United Kingdom (Tunesi et 2011), Portugal (Ferrão et al. 2014), and Greece (Korone and Nanaki 2012), among others. In addition, nowadays, an important role is also played by the BRIC countries (Brazil, Russia, India, and China), nations that in the future will generate a large amount of MSW. In this regards, several works addressed waste management systems in China (Song et al. 2013; Zhao et al. 2011), India (Mondal et al. 2010), Russia (Tulokhonova and Ulanova, 2014), and Brazil (Leme et al. 2014).

Nevertheless, recently, LCA studies evaluating incineration processes have taken off. The aim of these works was to assess the environmental performance of waste incinerators (Beylot and Villeneuve 2013; Zhao et al. 2012; Scipioni et al. 2009; Riber et al. 2008; Morselli et al. 2008 and 2007; Ciroth et al. 2002), to compare different waste treatment options such as landfill and incineration (Dong et al. 2014; Cherubini et al. 2009 and 2008; Liamsanguan and Gheewala 2008; Mendes et al. 2004), as well as landfill with other combustion techniques, such as pyrolysis and gasification (Zaman 2010). Other studies compared several thermal treatment technologies (Chen and Christensen 2010), flue gas cleaning processes (Moller et al. 2011; Chevalier et al. 2003), management options for pollution control of residues from waste incineration (Fruergaard et al. 2010), bottom ash treatments

(Margallo et al. 2014b; Huntzinger and Eatmon 2009; Toller et al. 2009; Birgisdottir et al 2007; Olsson et al. 2006), and different energy recovery strategies (Guigliano et al. 2008; Consoni et al. 2005a and 2005b). LCA studies have had a dominant focus on household waste, in particular for mixed MSW. However, the studies of specific waste fractions, such as plastic packaging (Foolmaun and Ramjeeawon 2013 and 2012; Wollny et al. 2002), and paper waste (Arena et al. 2004) are playing an important role. Along the same lines, several waste LCA models have been developed in the last two decades. Fig 1.11 summarises the waste LCA models, the nationality and the development phases (Gentil et al. 2010). Particularly, more than 50 waste models are available in Europe, and more on a worldwide basis, with different applicability, functionality license restrictions, and costs. Among these models are noteworthy EASEWASTE (Environmental Assessment of Solid Waste Systems and Technologies), EPIC/CSR, IWM2, LCA–IWM, MSW-DST, ORWARE (Organic Solid Waste Research), SSWMSS, WIZARD, and WRATE (Waste and Resources Assessment Tool for the Environment). Other models less significant because the availability of the information is too scarce, the tool only consider a specific waste management technology, or the model is not an LCA tool have been developed: ARES, HOLIWAST, LCA-LAND, MIMES, MSWI and WAMPS. The models differ on the functional unit, time horizon, waste composition and waste treatment (Gentil et al. 2010).

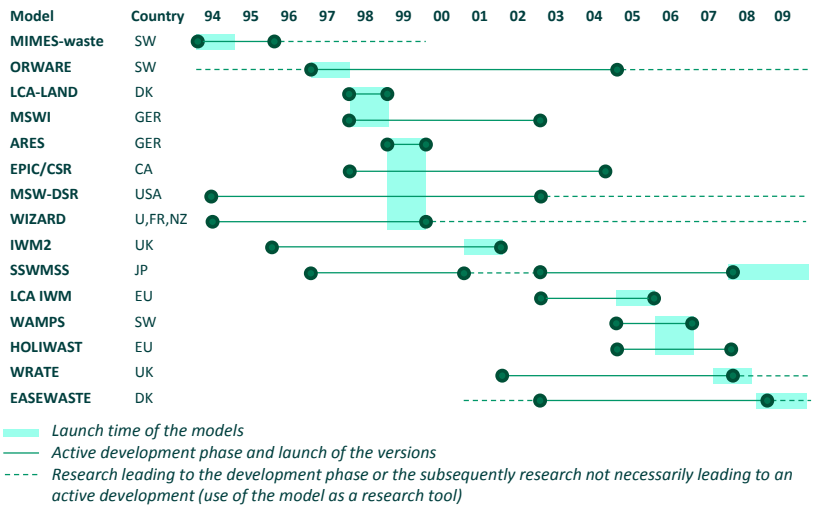


Fig 1.11 Timeline and nationality of waste LCA models.

1.3 Thesis scope and objectives

This thesis was conducted within the framework of the FENIX-Giving Packaging a New Life project. FENIX is a 3-year European LIFE+ funded project that started in January 2010. The aim of the project was to assist to municipalities and other organisations in Spain and Portugal to look for more ecoefficient and sustainable solutions for packaging waste management, according to the European policy principles on waste management. Within the framework of this project, it was developed a flexible and easy to-use software tool to obtain LCA results for packaging waste management. The tool allows the different users to introduce and modify parameters to adapt the models created in the tool to their real-life situations (FENIX 2010). FENIX was coordinated by the Environmental Management Research Group, GiGa (Escola Superior de Comerç Internacional-Universitat Pompeu Fabra, Barcelona) with the participation of three partners Ecoembes (Spanish non-profit organisation on packaging waste management), PE International (experts in sustainability, Stuttgart, Germany), and Sociedade Ponto Verde (Portuguese non-profit-making company on packaging waste management), and the cooperation of universities and other centres in Spain and Portugal (Fig 1.12).

In this project, the Development of Chemical Processes and Pollutant Control (DEPRO) research group of the University of Cantabria, has participated in the data collection and modelling of the incineration process in Spain and Portugal. Within this context, the primary goal of this thesis is to develop a life cycle model representative of the incineration process in the Iberian Peninsula, in order to assess and compare the environmental impacts of the waste fractions that compose the MSW and the WtE plants under study.

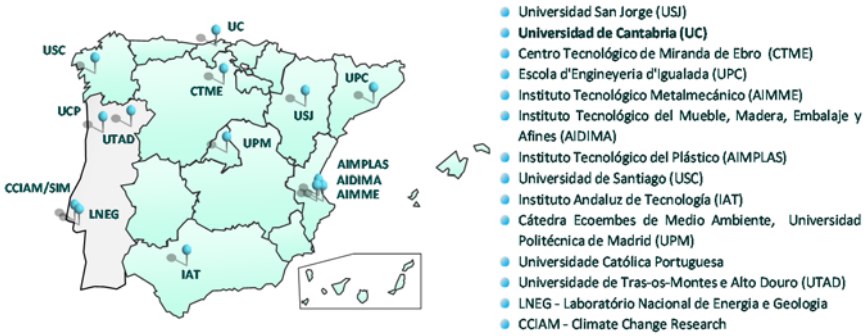


Fig 1.12 Collaborators of the FENIX project.

To reach the general aim of the thesis the following specific objectives were addressed:

- Review of the state of the art of MSW incineration in Spain and Portugal and collection of all the input –fuels, reagents, auxiliary materials- and output data –products, emissions, waste- of the Spanish and Portuguese WtE plants.
- Life cycle modelling of the incineration process using the LCA software Gabi 4. Particularly, the 18 waste fractions that compose the MSW were modelled: PET, HDPE packaging (P) and non-packaging (nP), LDPE (P and nP), plastic mix (P and nP), paper and cardboard (PC) (P and nP), beverage carton, steel (P and nP), aluminium (Al) (P and nP), glass (P and nP), organic matter, and remaining materials (wood, construction and demolition wastes, textiles, and others).
- Development of an environmental sustainability assessment (ESA) methodology based on natural resources sustainability (NRS) and environmental burdens sustainability (EBS) to be applied to waste management sector, in order to simplify decision-making process.
- Application of the model and the ESA methodology to carry out the life cycle impact assessment (LCIA) of the WtE plants under study and the alternative treatments of waste from the thermal treatment and flue gas cleaning.

According to the specific objectives, and taking into consideration the requirements of the University of Cantabria to the preparation of a PhD thesis as a compendium of scientific publications, the dissertation was organised in

four chapters:

- **Chapter 1** includes a general introduction about waste generation and management problems, as well as the application of LCA methodology to evaluate the associated environmental impacts.
- **Chapter 2** describes the state of the art and the life cycle model of waste incineration, as well as the developed ESA methodology. Additionally, this chapter shows the main results of the application of the model and the ESA methodology.
- **Chapter 3** summarises the general conclusions and the overview of the challenges and recommendations for further research.
- The dissertation ends with **Chapter 4**, the core of the thesis that includes the papers supporting this research, and finally, the annexes that provide extra information.

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2. METHODOLOGY AND RESULTS

Development and application of a life cycle model of waste to energy process

Adapted from Nina Paley

2.1 State of the art of waste to energy process

Waste prevention should be the first priority of waste management. However, once waste is produced, and reuse or recycling are not possible, end-of-pipe treatments such as landfilling or incineration are required. Although landfilling remains the most common practice, incineration has increased in the last 15 to 20 years, driven by the specific legislation to reduce emissions to air. The primary objective of municipal solid waste incineration (MSWI) is to treat waste by reducing the solid waste mass and allowing energy recovery. For this reason, the original designation of "incinerator" was dropped, and today it is talked about energy from waste (or waste to energy, WtE) (Margallo et al. 2014c). Nevertheless, regarding to operational conditions, the high combustion temperature makes necessary the employment of very specific materials, increasing the installation and maintenance costs. Likewise, additional fuel is required when waste does not reach the required low heating value (LHV), or when it has high water content (Rodríguez and Irabien 2013). Moreover, this technology had gained a bad reputation owing to the environmental impact specifically, due to the emissions of acid gases, dioxins and furans (PCDD/F), and greenhouse gases (Margallo et al. 2012). The emission of these pollutants depends basically on the composition of waste, the combustion and flue gas treatment, and the operation of the plant (Riber et al. 2008).

2.1.1 Description of waste to energy process

The basic linear structure of a WtE plant (Fig 2.1) may include the incoming storage and pre-treatment of waste, the thermal treatment with energy recovery and conversion, the flue gases and wastewater treatment,

and the management and treatment of ash and slag.

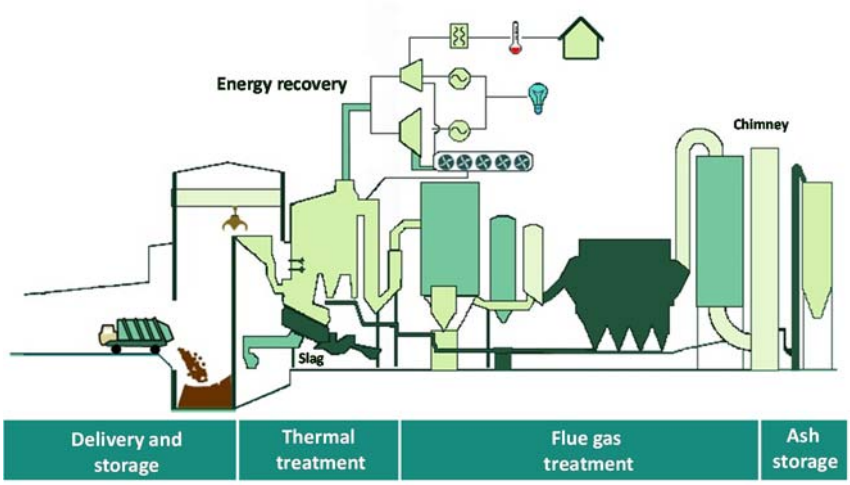


Fig 2.1 Scheme of a WtE plant structure.

WASTE PRE-TREATMENT AND THERMAL TREATMENT

Wastes arrive to the WtE plant and are discharged in the bunker where usually takes places the waste mixing. Commonly, the pre-treatment of MSW is limited to the shredding of pressed bales, bulky waste, etc., although sometimes more extensive shredding is used. Waste is discharged from the storage bunker into the feeding chute by an overhead crane, and then fed into the kiln by a hydraulic ramp or another conveying system (EC-IPPC 2006).

Different types of thermal treatments are applied to the different types of wastes; however, not all thermal treatments are suited to all wastes. The most common technologies are grate incinerator (GI), rotary kiln (RK), fluidised bed (FB), and pyrolysis and gasification systems. For MSW and refuse derived fuels (RDF) incineration grates are widely applied, FB and RK are also applied but to a lesser extent. Specifically, treatment of sewage sludge in the case of FB and hazardous and clinical waste in a RK is well known. On the other hand, pyrolisis and gasification are rarely applied to all the studies wastes (Margallo et al. 2012). These techniques are characterised by a great flexibility in terms of energy production and material recycling. Nevertheless, doubts about the less proved technology and unclear economic benefits hamper a larger market penetration (Saft 2007).

In Table 2.1 the main applied techniques in the thermal treatment stage are summarised (EC-IPPC 2006).

Table 2.1 Summary of the thermal treatment techniques applied to the main waste types.

Technique	Untreated MSW	Pre-treated MSW	Hazardous waste	Sewage sludge	Clinical waste
Grate	Widely applied	Widely applied	Rarely applied	Not normally applied	Applied
Rotary kiln	Not normally applied	Applied	Widely applied	Applied	Widely applied
FB	Rarely applied	Applied	Not normally applied	Widely Applied	Not normally applied
Pyrolisis	Rarely applied	Rarely applied	Rarely applied	Rarely applied	Rarely applied
Gasification	Rarely applied	Rarely applied	Rarely applied	Rarely applied	Rarely applied

GIs are widely applied for the incineration of mixed municipal wastes. In Europe, approximately 90 % of installations treating MSW use this technology. The main types of grates are summarised in rocking, reciprocating, travelling, roller and cooled grates.

An alternative to GIs are FBs; a technique which has been developed and implemented particularly in Japan and China due to several operational advantages (McDougall et al. 2001). Some of the benefits of this technology are that FB is adaptable to a wide variety of wastes, even this technique is suited to burn high moisture waste with low calorific value, such as sewage sludge (Margallo et al. 2012). On the other hand, FB has high contact allowing the operation to a lower temperature and residence time of solids. The relative low cost of design and construction simplicity, the low maintenance costs, and the few operational problems of the technology, has boosted the increasing implementation of FB (Rodriguez and Irabien 2013). However, FB usually requires a pre-treatment stage consisting of sorting out and crushing larger inert particles, and shredding to obtain a small particle size of good homogeneity. The relatively high cost of pre-treatment processes required for some wastes, has restricted the economic use of these systems to larger scale projects. Nevertheless, new developments reduce this pre-treatment,

as they accept coarser waste materials (Van Caneghem et al. 2012). Other disadvantages of FB are that it produces more fly ashes, and it is easily subject to heavier deposit and erosion of heat transfer surfaces (Chen and Christensen 2010). Three types of FBs are commercialised: stationary (SFB), bubbling (BFB), and circulating (CFB).

Regarding GI, the technology requires at start-up, auxiliary burners to heat up the furnace to a specified temperature and they are usually switched on automatically if the temperature falls below the specified value during operation. Primary air, generally taken from the waste bunker, is pumped through the small grate layer openings into the fuel layer. This air cools the grate bar and carries oxygen into the incineration bed. More air, so-called secondary air, is generally added above the waste bed to complete combustion, mix of flue gases, and prevention of the free passage of unburned gas streams (EC-IPPC 2006). During combustion the majority of the energy produced is transferred to the flue gases. Cooling of these gases allows the recovery of the energy and the cleaning of flue gases before they are released to the atmosphere. Conventional energy recovery involves passing these hot flue gases through a boiler. Water circulated through these tubes is turned to steam, which can be heated further, to increase its temperature and pressure to make electricity generation more efficient (McDougall et al. 2001). The principal uses of the energy are the production and supply of heat, the production and supply of electricity, or combinations of both (Margallo et al. 2012).

FLUE GAS TREATMENT (FGT) AND CONTROL SYSTEMS

Emissions at WtE plants are mainly influenced by waste composition, applied technology, and operational conditions. The main pollutants in flue gases from waste incinerators are nitrogen oxides (NO_X) and nitrogen compounds, heavy metals, acid gases (SO_X, HCl, HF, etc.), organic compounds (PAH, PCDD/F, etc.), dust, and greenhouse gases (CH_4, CO_2, CO, etc.). These pollutants require a cleaning by a combination of individual process units that together provide an overall treatment system (Margallo et al. 2012).

Techniques for reducing particulate emissions. Dust emissions from waste combustion mainly consist of the fine ashes from the process and other fine materials that travel with the flue gases. To remove these pollutants the selection of gas cleaning equipment is determined by the particle load as well

as the average size and particle distribution in the gas stream, the flow rate and temperature of gas, the compatibility with other components of the entire FGT system, and the required outlet concentrations (EC-IPPC 2006). Electrostatic precipitators, ionisation wet scrubbers, bag filters, and cyclones or multicyclones are the most common techniques to reduce particle emissions (Margallo et al. 2012).

Techniques for the reduction of acid gases (HCl, HF, and SO_X). Many wastes contain chlorinated organic compounds or chlorides. In MSW approximately 50 % of the chlorides come from PVC. The organic component of these compounds is destroyed in the combustion and the chlorine is converted to HCl. The formation and emission of Cl_2 is of minor importance under normal incineration conditions. The formation mechanism of HF corresponds to that of HCl. The main sources of HF emissions in WtE plants are probably fluorinated plastic or fluorinated textiles. Regarding SO_X, common sources of sulphur in some waste streams are waste paper, plaster board (calcium sulphate), and sewage sludge. Due to the acid character of these pollutants, a neutralisation is generally performed with lime, hydrated lime, or other basic reagents. In particular, three basic alternatives exist for removing acid gases: dry, semi-dry or semi-wet, and wet method (McDougall et al. 2001). With the latter technique a high degree of acid gas removal is reached and the waste products from the cleaning process may be re-usable. The disadvantages are that extensive equipment is necessary and wastewater is produced, which requires treatment prior to discharge (EC-IPPC 2006).

Techniques for the reduction of emissions of oxides of nitrogen (NO_X). There are two main sources of NO_X from MSW combustion, the thermal and fuel NO_X. Also NO_X may be formed via radical reaction (prompt NO_X) where the atmospheric nitrogen can also be oxidised by reaction with CH radicals and intermediate formation of HCN. However, this mechanism of formation is of relatively low importance in waste incineration (EC-IPPC 2006).

- Thermal NO_X: during combustion a part of the air nitrogen is oxidised to nitrogen oxides. This reaction only takes place significantly at temperatures above 1,300 °C, not common in MSW incineration, even though they may be found in localised hot spot where high concentrations of plastic exist momentarily. Further, they could be experienced in some incinerators using relatively low levels of excess of air.

- Fuel NO_X: during combustion a part of the nitrogen contained in the fuel is oxidised to nitrogen oxides. Formation of fuel NO_X is largely independent of temperature.

Formation of NO_X from fuel is the dominant mechanist due to the temperature regime associated with MSW combustion is more conducive to fuel formation than thermal NO_X formation (Tillman et al. 1989). NO is the only NO_X formed directly in the flame, NO_2 appears latter by means of slower reactions. The proportion of NO/NO_2 in the total NO_X stack emissions is usually approximately 95 % NO and 5 % NO_2.

Achieving effective gas mixing, prevent over supply of air, and the use of unnecessarily high furnace temperatures are important elements in NO_X emission control. However, as primary techniques for NO_X control are very limited, post-combustion or secondary techniques have emerged as the most cost-effective method. These technologies are based on the application of ammonia or derivatives of ammonia (e.g. urea) as agent, reducing the nitrogen oxides (NO and NO_2) to N_2 and water vapour. Particularly, post-combustion technologies are classified as selective non-catalytic reduction (SNCR) and selective catalytic reduction (SCR). The SCR process gives high NO_X reduction rates, typically over 90 % at close to stoichiometric additions of the reduction agent. Nevertheless, low-temperature SCR (180-450 ºC) requires catalyst regeneration due to salts formation, increasing the operating cost (EC-IPPC 2006). Likewise, other drawback of the SCR is the high capital cost associated with the catalyst and the construction of the reactor to house the catalyst bed (Pickens 1996).

Techniques for the reduction of heavy metals emissions. Metals and metalloids are converted mainly in non-volatile oxides and deposited with flue ash. Therefore, the main techniques are those applicable to dust removal. As a special case, Hg is emitted usually as Hg metallic or $HgCl_2$ depending on the amount of HCl contained in the flue gas. Techniques for preventing mercury emissions are those which prevent and control the inclusion in the waste. Also mercury can be removed before emission; e.g. by wet scrubbing or dry filtration of fly ashes in bag filters. If Hg is in metallic form transformation techniques in order to be converted in a more stable form are applied. Among these techniques highlight the transformation adding oxidants and deposition in a scrubber, or direct deposition on sulphur doped activated or zeolites. However, in the incineration of MSW and

hazardous waste (HW) incineration, the chlorine content in the average waste is usually high enough to ensure that Hg is present mainly in the ionic form (EC-IPPC 2006).

Techniques for the reduction of organic carbon compounds emissions. Effective combustion provides the most important means of reducing emissions to air of organic carbon compounds. Flue gas from WtE plants can contain trace quantities of organic species, such as halogenated aromatic hydrocarbons, polycyclic aromatic hydrocarbons (PAH), benzene, toluene and xylene (BTX) and dioxins and furans. (EC-IPPC 2006). PCDD/F are highly toxic micro-pollutants emitted from the combustion sources. The presence of this pollutant in MSWI plants was first detected in 1997 at the incinerator of Amsterdam (Fiedler 2003). PCDD/F can pass through the furnace without any destruction process due to presence of this pollutant in the input waste. It may also be formed during waste combustion or in the cooling gasses process in catalytic reactions of carbon or carbon compounds with inorganic chlorine compounds over metal oxide. These reactions will occur especially on fly ash or filter dust at temperatures between 200 and 450 °C. Specifically, high formation levels of PCDD/F are associated with poor combustion conditions, feeding of problematic materials, or dust collectors operating at high temperatures (PNUMA 2005). Optimum flue gas incineration largely destroys the precursor compounds, so the formation of PCDD/F is, therefore, suppressed. Adsorption on activated carbon processes, SCR systems used also for NO_X reduction, static bed filters, and oxidising catalysts are available, amongst others, to achieve the emission limit value of Directive 2000/76/EC (0.1 ng ITEQ) (Buekens and Huang 1998). From of all of then, the adsorption on activated carbon is most common alternative. This reagent with a high absorption efficiency for PCDD/F as well as for mercury, is injected into the gas flow and is filtered from the gas flow using bag filters.

WASTEWATER TREATMENT AND CONTROL TECHNIQUES

Wastewater generation in WtE plants is a minor problem compared with air emissions and solid waste generation. Nevertheless, it remarks evident the need of controlling some effluents (Rodriguez and Irabien 2013). Potential emissions to water from WtE plants are mainly process wastewater (it only arises to any significant degree from wet FGT systems), wastewater form the collection and treatment of bottom ash (BA), and used cooling water.

Specifically, process wastewater resulting from wet flue gas treatment contains a wide variety of polluting components, which depend on the composition of the waste and on the design of the wet flue gas system. The treatment of these wastewater is carried out by means of physic-chemical methods based on pH-correction and sedimentation, and evaporation processes (EC-IPPC 2006).

SOLID RESIDUES TREATMENT

Waste incineration results in various types of solid wastes, distinguishing mainly those wastes directly resulting from the incineration process and those resulting from the FGT system. The main wastes arising from the combustion stage are bottom ash, so-called slag, boiler ash and fly ash (FA). Primary measures for controlling residue outputs involve optimising control of the combustion process in order to guarantee an excellent burn-out of carbon compounds, promote the volatilisation of heavy metals, and fix lithophilic elements in the bottom ash, thus reducing their leachability (EC-IPPC 2006).

BAs present a large production volume, lower hazardous character, and leachability than FAs. BAs are cooled and subjected to a magnetic separation, obtaining the metallic fraction comprised by the metallic waste contained in MSW, and a non-metallic fraction comprised by ceramic and vitreous materials and particles not burned in the combustion process. The non-metallic fraction is considered a non-hazardous material that can be dumped in a MSW landfill whereas the metallic fraction, ferrous scrap, could be used to produce steel (Lopez-Delgado et al. 2007). On the other hand, fly ashes, due to the hazardous character, require a stabilization process to produce a material with physical and mechanical properties that promote a reduction in the release of contaminant from the residue matrix. Once waste is stabilised, it is usually sent to the landfill. In other cases ashes are sent directly to the landfill in big-bag.

In relation to the FGT wastes, these residues contain concentrated amounts of pollutants and therefore, normally are not considered appropriate for recycling purposes. In this group are included waste from dry, semi-dry or wet flue gas treatment, and waste from flue gas polishing. The former wastes are a mixture of calcium and/or sodium salts, mainly as chlorides and sulphites/sulphates but also contain some fluorides and unreacted reagent

chemicals, fly ash, heavy metals, and PCDD/F. The normal way of disposal is landfilling as hazardous waste (e.g. big-bags). The composition of wastes from final flue gas cleaning depend on the adsorbent used (activated carbon, cokes, lime, sodium bicarbonate, and zeolite). The residue of activated carbon is sometimes permitted to be incinerated in the WtE plant itself. If a mixture of other reagents and activated carbon is used, the residue is generally sent for external treatment or disposal, since there might be risks of corrosion.

2.1.2 Waste to energy plants in Spain and Portugal

There are 14 WtE plants with a capacity of more than three tonnes per hour in the Iberian Peninsula (considering the plant located in Andorra). In particular, Spain has 10 WtE plants, four incinerators located in the North-East (Catalonia), three in the North and North-West (Cantabria, Galicia and Basque Country), one in Madrid, one in the Balearic Islands, and one in Melilla. Finally, Portugal has three WtE plants located in Porto, Lisbon, and Madeira. Fig 2.2 illustrates the location of the Spanish and Portuguese WtE plants and the type of thermal treatment applied (Margallo et al. 2012).

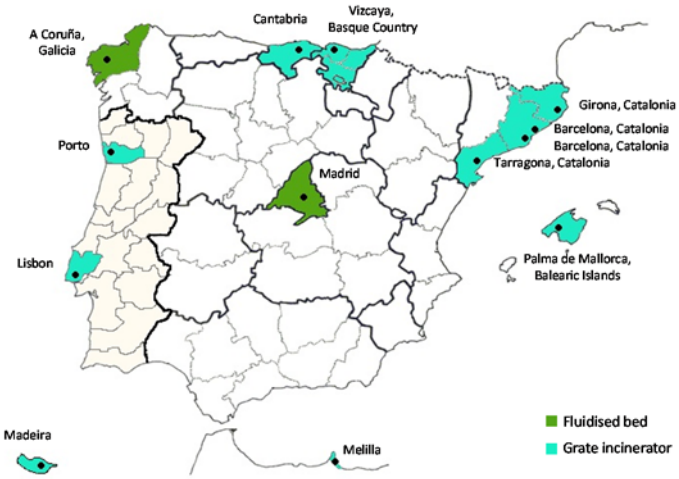


Fig 2.2 Location of the Spanish and Portuguese WtE plants.

Most WtE plants in Spain are located in the North of Spain due to the limited availability of land in this area, as well as the great amount of landfill leachates generation because of the intensive rainfall. Regarding the thermal treatment, in the Spanish and Portuguese WtE plants only GI and FB are

applied. Particularly, grate-based technology was present in 80 % of thermal treatment systems in Spain, whereas 100 % of the plants employed this technology in Portugal (Margallo et al. 2012). The operational temperature of the plants ranges from 900 to 1,100 ºC, and the LHV from 1,791 to 3,500 kcal kg^{-1}.

Related to the FGT, acid gases are treated by means of semi-dry scrubbers and in a lesser extend with dry systems; whereas, for PCDD/F and heavy metals cleaning and for dust reduction, activated carbon injection and bag filters are respectively used. For NO_X control, SNCR is the most common technology although the lower NO_X reduction rates are obtained, the lower operational cost are generated. Urea and ammonia are used equally as reagent; however, the application of urea instead of ammonia in SNCR leads to relatively higher N_2O emissions in comparison with ammonia reduction (EC-IPPC 2006). Table 2.2 and Table 2.3 depict the operation and technical data of the plants.

Table 2.2 Technical and operation data of the Portuguese WtE plants.

		I_{P1}	I_{P2}	I_{P3}
Star up year		1999	2000	2004
Incineration capacity (t h^{-1})		28.0	8.00	24.6
Nº lines		3	2	2
Thermal treatment		Reverse-Acting GI	Roller GI	GI
Combustion temperature (ºC)		N.A.[1]	N.A.	1,000-1,200
LHV (kcal kg^{-1})		1,870	1,791	1,840
FGT system	NO_X	SNCR	N.A.	SNCR
	Particles	Bag filters	Bag filters	Bag filters
	Acid gases	Semi-dry	Semi-dry	Semi-dry
FGT reagent	Activated C	✓	✓	✓
	Urea		✓	
	NH_3	✓		
	CaO			
	$Ca(OH)_2$	✓	✓	✓
Fuels	Diesel	✓	✓	✓
	Natural gas	✓		✓

[1] *N.A.: Non-available data.*

Table 2.3 Technical and operation data of the Spanish WtE plants.

		I_{S1}	I_{S2}	I_{S3}	I_{S4}	I_{S5}	I_{S6}	I_{S7}	I_{S8}	I_{S9}	I_{S10}
Star up year		1975	1994	1984	1991	1997	2004	2006	2002	1997	1996
Incineration capacity (t h^{-1})		14.5	10.0	2.50	9.60	27.0	30.0	12.0	23.5	9.71	4.50-6.00
Nº lines		3	2	2	2	2	1	1	2	3	1
Thermal treatment		Roller GI	Travelling GI	Reverse acting GI	Roller GI	Roller/ cooled GI	Reciprocating GI	Roller GI	Circulating FB	Rotating FB	Serrated GI
Temperature (ºC)		950	1,000-1,100	1,050	900-1,500	1,050	1,100	1,025	900	910	900
LHV (kcal kg^{-1})		1900-2,200	2,100	1,800	2,000	1,800	2,000	2,800	3,500	3,500	1,400-3,000
FGT system	NO_x	SNCR	SNCR	SNCR	SNCR	SCR	SNRC	SNRC		SCR	SNRC
	Particles	Electro-filter	Bag filter	Electrofilter / bag filter	Bag filter	Bag filter	Bag filter	Bag filter	Bag filter	Bag filter /cyclons	Bag filter
	Acid gases	Semi-dry	Semi-dry/dry	Dry	Semi-dry	Semidry	Semidry/dry	Semi-dry	Semidry	Semidry	Semidry
FGT reagent	Activated C	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
	Urea	✓	✓	✓				✓			✓
	NH_3				✓	✓	✓			✓	
	CaO	✓			✓	✓					✓
	$Ca(OH)_2$		✓	✓		✓	✓	✓	✓	✓	
Fuels	Diesel	✓		✓	✓	✓			✓	✓	✓
	Natural gas	✓	✓				✓	✓	✓		

2.2 Life cycle model of a waste to energy process

LCA methodology has been applied to develop a model to better understand incineration processes by investigating the average data from 13 WtE plants in Spain and Portugal. In particular, 18 waste fractions of MSW were modelled. Therefore, the functional unit selected was one ton of each waste fraction incinerated in Spain and Portugal: PET, HDPE packaging (P) and non-packaging (nP), LDPE (P and nP), plastic mix (P and nP), paper and cardboard (PC) (P and nP), beverage carton, steel (P and nP), aluminium (Al) (P and nP), glass (P and nP), organic matter, and remaining materials (wood, construction and demolition wastes, textiles, and others). This reference unit was selected taken into account the literature and the fact that the main function of incineration is to treat and reduce the volume and hazard of waste. In relation to cut-offs, all material and energy inputs that have a cumulative total of at least 98 % of the total mass and energy inputs have been included. However, those flows that do not meet this criterion, but are thought to potentially have a significant environmental impact have also been included. Any material, no matter, how small its mass or energy contribution, that has a significant effects in its extraction, manufacture, use, or disposal, is highly toxic, or is classified as hazardous waste (environmental significance) was included.

The process under study comprises thermal treatment with energy recovery, flue gases cleaning, and solid waste treatment. Construction of major capital equipment and, the maintenance and operation of support equipment were excluded from the study. Considering the system boundaries, the incineration process was modelled as a black box including four subsystems (Fig 2.3) (Margallo et al. 2014a).

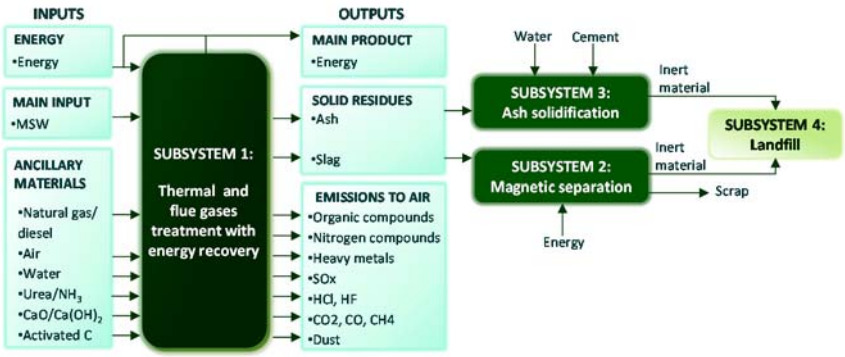


Fig 2.3 Flow diagram of the incineration process.

- **Subsystem 1:** Thermal and flue gases treatment. These processes were considered as a single subsystem because the composition and flow of flue gases before treatment is not measured. Regarding the thermal treatment, FB was modelled together with the grate based technology. This means that FB was modelled as if it were a GI. The main reasons to apply this assumption were the lack of information of FB, and that they are only present in two WtE plants in Spain. Moreover, the input and output data of FB and GI are similar; and therefore, there are no significant differences in the inventory of both technologies, as well as in the model results. The inputs of this system are MSW, fuels, ancillary materials and reagents; the outputs are energy production, waste and air emissions. Because of the lack of significance, the emissions of several pollutants were excluded. Only one plant registered hexachlorobenzene, endrin, heptachlor, naphthalene, pentachlorobenzene, tetrachloroethylene, tetrachloromethane, trichlorobenzene, benzene ethylene oxide, vinyl oxide, and hydrogen cyanide emissions; therefore and according to the cut-off criteria, these pollutants were not included in the inventory. Additionally, the emission of these pollutants has a low influences in the results of the model. Heavy metals are often significant in toxicity assessment. Directive on waste incineration sets the emissions limit value for the following metals Cd+Tl (0.05 mg $(Nm^3)^{-1}$), Hg (0.05 mg $(Nm^3)^{-1}$), and Sb+As+Pb+Cr+ Co+Cu+Mn+Ni+V (0.5 mg $(Nm^3)^{-1}$) (EC 2000). However, not all of these metals have a threshold value for release to air in the European Pollutant Release and Transfer Register regulation (E-PRTR), and even some of them are not included in the sector specific sub-list of air pollutants of the E-PRTR. For this reason, only were included in the model the metal emissions of

arsenic (As), cadmiun (Cd), chromium (Cr), cooper (Cu), manganese (Mn), mercury (Hg), nickel (Ni), lead (Pb) and zinc (Zn). Water emissions from waste incineration are only related to plants with an exhaust gas cleaning system (Bjarnadóttir et al. 2002). In these plants, wet scrubbers are not applied; therefore water emissions were not considered.

- **Subsystem 2:** Magnetic separation of slag. The separation entails an energy demand of 0.042 kWhel kg^{-1} of iron removed (Doka 2003). Approximately 10 % of slag is recovered as scrap and the remainder as inert slag. Outside of the system boundaries, steel is produced with scrap, being a recycled material.
- **Subsystem 3**: Ash solidification with a mixture of water (30 %) cement (20 %) and ash (50 %) to produ ce an inert ash that is landfilled.
- **Subsystem 4**: Final disposal. Inert slag is sent to a MSW landfill next to the incinerator. Ash is a hazardous material, but once stabilised, it is sent to a specific landfill.

2.2.1 Life cycle inventory

In the life cycle inventory (LCI), all relevant inputs and outputs for the process in a specified year are collected. The properties of the waste mixture are also determined (Seyler et al. 2005). Data collection in this study was performed from July 2010 to September 2011. The collaboration of the WtE plants as well as other entities is essential for data collection. For this reason, a cover letter was sent to the Spanish and Portuguese WtE plants in order to encourage them to participate in the model development, providing high quality data. Nevertheless, a poor amount of data was received from the plants. Therefore, alternative data sources were used in the inventory.

- The composition and amount of MSW treated in 2008 and 2009, and the amount of slag and ashes generated in the Spanish WtE plants was provided by Ecoembes, which is responsible for the collection and recovery of packaging waste (Ecoembes 2014). In Portugal this information was supplied by the Portuguese plants.
- The consumption of fuels, reagents, and ancillary materials was collected from the Integrated Pollutant and Prevention Control permit (IPPC permit) of the plants. However, as this information was in the coverage time-

frame of 2004 to 2006, it was required to check the data representativeness. According to different experts in the incineration field, this technology has not undergone many changes, and the data included in the IPPC permit are fairly representative.

- To determine the energy production of the WtE plants, the high heating value (HHV) and moisture content are essential. In this sense, AEVERSU - the Spanish association of MSW valorisation- provided technical information about the energy production in the Spanish plants (AEVERSU 2013). Likewise, this association supplied the HHV, moisture, and ashes content of each waste fraction. These data are based on Tchobanoglous and Kreith 2002. For Portugal, this information was collected from the Website of the plants.
- The amount of slag, ashes, and scrap generated in waste combustion was provided by AEVERSU and the Website of the Portuguese plants. Regarding waste treatment, the information about magnetic separation of slag, and ash solidification was based on Doka 2003.
- In order to determine the emissions of each input waste fraction, two types of data are basic. The total emissions of the WtE plants, and the C, S, Cl_2, F_2, and metalloids and heavy metals content of the input waste fraction. The emissions data were collected from the Spanish Pollutant Release and Transfer Register (PRTR) (PRTR 2012) and the Portuguese PRTR (PRTR 2014). Regarding waste composition, AEVERSU supplied the C, H, N, O_2, and S content of each waste fraction, based on Tchobanoglous and Kreith 2002. Metalloids and heavy metals, Cl_2, and F_2 content are based on data published by Riber et al. 2009, whereas the total and fossil carbon content of each fraction was collected from the data reported by Bjarnadóttir et al. 2002.

Additionally, background data were taken from the PE database (PE International 2011). Table 2.4 displays a summary of the data source, time-frame, and geography of the data collected. Table 2.5 displays the LCI of thermal and flue gases treatment of the Spanish and Portuguese WtE plants, whereas

Table 2.6 shows the LCI for magnetic separation of slag and ash solidification. These inventories consist of annual material and energy inputs and outputs of Spanish and Portuguese plants in 2009. Specifically, the LCI described the

average data of the 13 WtE plants under study.

Table 2.4 Data source, time-frame, and geography of the LCI.

Data Source		Time-frame	Geography
Ecoembes	Tons of MSW incinerated	2008-2009	Spain
Portuguese WtE plants	Tons of MSW incinerated	N.A.	Portugal
AEVERSU	HHV, S content, and moisture	N.A.	Europe
	Energy production, generation of ashes, slag, and scrap	2008-2009	Spain
IPPC permit	Consumption of energy, fuels, ancillary materials, and reagents	2004-2006	Spain/Portugal
PRTR	Emissions to air	2008-2009	Spain/Portugal
Riber et. al 2009	Content of heavy metals, Cl_2, and F_2	2009	Europe
Bjarnadóttir et al. 2002	Total C and fossil C content	2000	Europe
Doka 2003	Consumption of energy in magnetic separation	2000-2001	Switzerland
	Cement and water consumption in ash solidification	2000-2001	Switzerland

Table 2.5 Average inputs and outputs of thermal treatment and flue gases cleaning at Spanish and Portuguese WtE plants.

THERMAL AND FLUE GASES TREATMENT				
INPUTS	***Fuels /ancillary materials (kg t^{-1}MSW)***: *AEVERSU 2013; WtE plants 2009*			
	Natural gas	$7.12\ 10^{-1}$	Urea	3.46
	Diesel	$2.06\ 10^{-1}$	Ammonia	1.70
	Water	492	CaO	9.20
	Air	4,282	$Ca(OH)_2$	4.18
	Activated Carbon	$4.80\ 10^{-1}$		
OUTPUTS	***Waste (t t^{-1}MSW)***: *AEVERSU 2013*		***Products (MJ t^{-1} MSW)***: *AEVERSU 2013*	
	Slag	$2.12\ 10^{-1}$	Energy production	3,386
	Ashes	$5.48\ 10^{-2}$	Self-consumption	382
	Scrap	$2.02\ 10^{-2}$	Energy sales	3,004
	Air emissions (kg t^{-1}MSW): *PRTR 2012*			
	Arsenic (As)	$1.83\ 10^{-5}$	Dioxins and furans (PCDD/F)	$3.91\ 10^{-10}$
	Cadmium (Cd)	$1.08\ 10^{-5}$	Carbon dioxide (CO_2)	446
	Chromium (Cr)	$9.95\ 10^{-5}$	Carbon monoxide (CO)	$1.26\ 10^{-1}$
	Copper (Cu)	$9.97\ 10^{-4}$	Methane (CH_4)	$5.80\ 10^{-1}$
	Lead (Pb)	$9.02\ 10^{-5}$	Total organic compounds (TOC)	$5.63\ 10^{-3}$
	Manganese (Mn)	$1.08\ 10^{-4}$	Sulphur oxides (SO_X)	$5.07\ 10^{-2}$
	Mercury (Hg)	$1.70\ 10^{-5}$	Nitrogen oxides (NO_X)	$9.05\ 10^{-1}$
	Nickel (Ni)	$4.58\ 10^{-5}$	Nitrous oxide (N_2O)	$1.58\ 10^{-1}$
	Zinc (Zn)	$7.70\ 10^{-4}$	Ammonia (NH_3)	$1.26\ 10^{-2}$
	Hydrogen chloride (HCl)	$2.93\ 10^{-2}$	Total suspended particles (TSP)	$8.98\ 10^{-3}$
	Hydrogen fluoride (HF)	$1.20\ 10^{-3}$	Particulate matter (PM_{10})	$4.85\ 10^{-3}$
	Polycyclic aromatic hydrocarbons (PAHs)	$1.26\ 10^{-4}$	Non-methane volatile organic compounds (NMVOC)	$1.57\ 10^{-2}$

Table 2.6 Average inputs and outputs of magnetic separation of slag and ash solidification at Spanish and Portuguese WtE plants.

MAGNETIC SEPARATION OF SLAG		ASH SOLIDIFICATION	
INPUTS		**INPUTS**	
Electric energy	3.05 MJ t^{-1} MSW	Ashes	5.48 10^{-2} t t^{-1} MSW
Slag	2.12 10^{-1} t t^{-1} MSW	Cement	2.19 10^{-2} t t^{-1} MSW
		Water	3.28 10^{-2} t t^{-1} MSW
OUTPUTS		**OUTPUTS**	
Slag	1.92 10^{-1} t t^{-1} MSW	Stabilised ashes	1.09 10^{-1} t t^{-1} MSW
Scrap	2.02 10^{-2} t t^{-1} MSW		

Additionally, the details and quality of this study were analysed by a critical review according to ISO 14040 (ISO 2006a). A summary of the critical review is detailed in Chapter 2.2.3. The inventory and waste properties of each WtE plant were provided in Annex A1. The LCI form part of a specific database on waste treatment process and recycling technologies. The database is compatible with the International Life Cycle Database (ILCD), which is a common format to exchange inventory data among different software tools and practitioners, promoted by the European Joint Research Center (Italy).

2.2.2 Model description

Waste incineration is defined as a multi-functional process. In these systems, the environmental burdens (EB) associated with a particular process must be partitioned over the various functional flows of that process (Huijungs and Guinée 2007). There are three basic types of allocations problems: multi-output (co-product systems such as a refinery), multi-input (waste treatment), and open-loop recycling (when a recycled product is transformed into another product) (Finnveden et al. 2009). In addition, some processes could be defined as multi-input/multi-output. Incineration is a clear example, in which several inputs -(i.e., waste fractions)- and outputs -(e.g., energy recovery, waste generation, and emissions)- coexist. The procedure of establishing a multi-functional model is illustrated in Fig 2.4 and is described in three steps (Seyler et al. 2005).

1. Compilation of an LCI for the waste mixture.
2. Classification of the input and output parameters according to its product or process dependence. Process dependent parameters do not rely on the

properties of the product but only on the process conditions. Product dependent parameters rely on the properties of the production, such as the chemical composition or heating value. Likewise, some parameters can depend on both the process and product (Seyler et al. 2005).

3. Calculation of consumptions and emissions from each waste fraction after applying allocations rules. ISO 14044 (ISO 2006b) proposes as a first solution for allocation problems to expand the system boundaries or divide the process into sub-processes. When this solution is not possible, the allocation should be assigned based on physical causation, which reflects the underlying relation among different flows. The final allocation should be based on other criterion, such as economic-value, mass or energy.

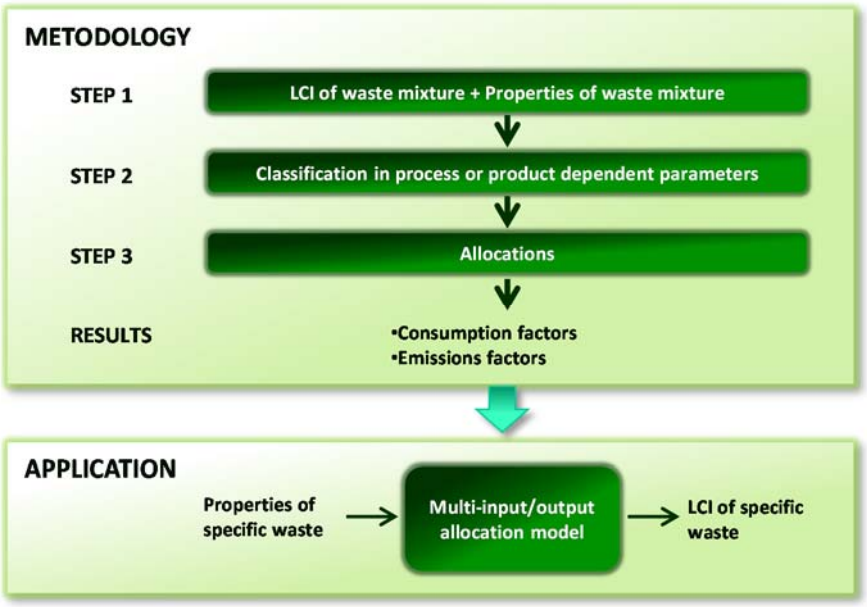


Fig 2.4 Description of the methodology to develop a multi-functional allocation model.

In this work, the allocation was based on the first option with physical causation. Additionally, when that allocation was not possible, a mass or energy allocation was applied. Table 2.7 summarises the allocation rules employed in the model.

Table 2.7 Summary of allocations applied.

Type of allocation	Parameter
Mass	-Fuels, ancillary materials, reagents -Ash and slag -Emissions of PCDD/F, NO_X, N_2O, NH_3, dust
Heating value	Energy production
Metals content	Metals emission
Fossil C content	CO_2 emission
C content	CO, TOC, CH_4, NMVOC, PAHs emission
Cl content	HCl emission
F content	HF emission
S content	SO_X emission

This allocation methodology was applied in this study using GaBi 4 software (PE International 2011) by dividing the model into three sections illustrated in Fig 2.5: thermal treatment with energy recovery, flue gases cleaning, and solid waste generation.

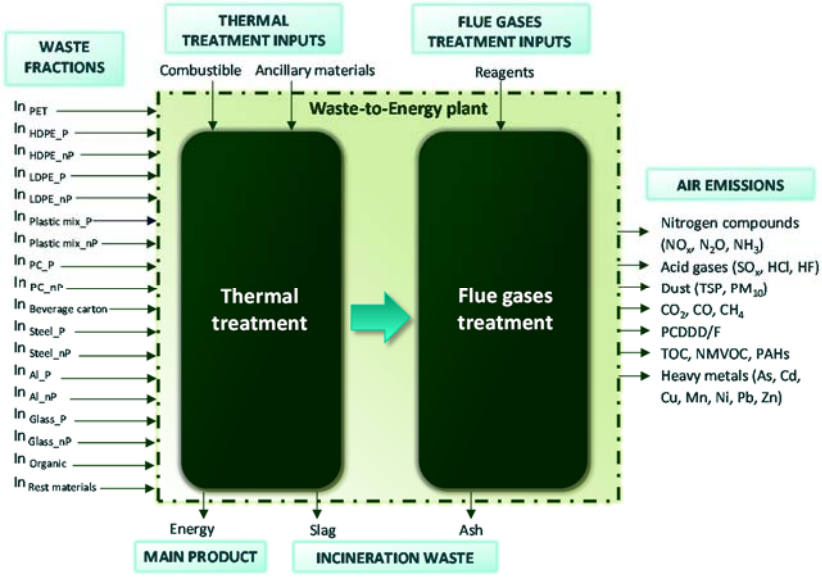


Fig 2.5 Description of the allocation model for WtE process in Spain and Portugal.

A) THERMAL TREATMENT

Waste composition. Based on the percentage of each waste fraction, the total amount of MSW, and the moisture content, the wet and dry weight of

each fraction was calculated according to Eqs 2.1 and 2.2. Additionally, Fig 2.6 displays the waste composition of the studied WtE plants and Annex 1 includes the waste composition and moisture content of each WtE plant.

$$\mathrm{In}_{\mathrm{fraction_i}} = \mathrm{In}_{\mathrm{wet\ MSW}} * \frac{\mathrm{Fraction_i}}{100} \quad [2.1]$$

$$\mathrm{In}_{\mathrm{dry\ fraction_i}} = \mathrm{In}_{\mathrm{fraction_i}} * \left(1 - \frac{\mathrm{Moisture_i}}{100}\right) \quad [2.2]$$

where i is the waste fractions; $\mathrm{Fraction_i}$ the percentage of each waste fraction [%]; $\mathrm{In}_{\mathrm{wet\ MSW}}$ the wet weight of MSW [t]; $\mathrm{In}_{\mathrm{fraction_i}}$ the wet weight of each waste fraction [t]; $\mathrm{In}_{\mathrm{dry\ fraction_i}}$ the dry weight of each waste fraction [t] and $\mathrm{Moisture_i}$ the moisture of each waste fraction [%].

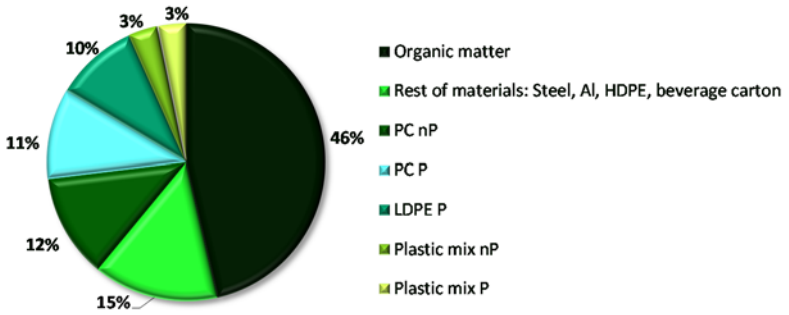


Fig 2.6 Waste composition of the Spanish and Portuguese WtE plants.

Combustibles and ancillary materials consumption. At start up, auxiliary burners that consume diesel or natural gas are used to heat the furnace to a specific temperature. The burners are also switched on if the temperature decreases. Other consumptions include water and air to cool the furnace and to ensure complete combustion. These consumptions are not related to the waste composition; therefore, mass allocation procedure was used.

$$\mathrm{Consumption}_{\mathrm{fraction_i}} = \frac{\mathrm{Consumption}}{\mathrm{In}_{\mathrm{wet\ MSW}}} \quad [2.3]$$

where, Consumption is the water, air, diesel, and natural gas consumed [kg year^{-1}] and $\mathrm{Consumption}_{\mathrm{fraction_i}}$ the consumption of water, air, diesel, and natural gas per ton of waste fraction [kg t^{-1} wet waste].

Energy production. The majority of the energy produced in combustion is transferred to flue gases. Cooling of these gases allows energy recovery, which could be used in the production and supply of heat, electricity or both.

In the Iberian Peninsula, approximately 80 % of the energy produced is sold to the public grid, with the rest used for self-consumption. The energy produced must be calculated via the energy content of the waste (Riber et al. 2008). In this work, based on the High Heating Value (HHV) of each waste fraction reported in Anexx 1 and the amount of waste incinerated, the theoretical energy produced is calculated according to Eq 2.4. Subsequently, the real energy produced must be allocated (energy allocation) to each waste fraction according to Eq 2.6.

$$E_{theo_{fraction_i}} = In_{fraction_i} * HHV_{fraction_i} * 1000 \quad [2.4]$$

$$E_{theo_{MSW}} = \sum_i^n E_{theo_{fraction_i}} \quad [2.5]$$

$$E_{fraction_i} = \frac{\frac{E_{theo_{fraction_i}} * E_{production}}{E_{theo_{MSW}}}}{In_{fraction_i}} \quad [2.6]$$

where $HHV_{fraction_i}$ is the HHV of each waste fraction [MJ kg^{-1}]; $E_{production}$ the energy production [MJ year^{-1}]; $E_{theo_{fraction_i}}$ the theoretical energy production of each waste fraction based on the HHV [MJ year^{-1}]; $E_{theo_{MSW}}$ the theoretical energy production of MSW based on the HHV [MJ year^{-1}]; $E_{fraction_i}$ the energy production of each waste fraction, that in theory, can be obtained from $E_{theo_{MSW}}$ assuming an average efficiency of the incinerator in converting the HHV to energy output [MJ t^{-1} wet waste].

B) FLUE GAS TREATMENT

Flue gas is cleaned using a combination of individual process units that together provide overall treatment. The following reagents are consumed: CaO or $Ca(OH)_2$ for acid gases reduction in dry, semi-dry, or wet scrubbers; NH_3 or urea for NO_X in a SCR or Selective SNCR; and activated carbon for organic compounds. The consumptions of these reagents were allocated by mass to each fraction according to Eq 2.7.

$$Reagents_{fraction_i} = \frac{Reagents}{In_{wet\ MSW}} \quad [2.7]$$

where Reagents is the consumption of urea, NH_3, CaO, $Ca(OH)_2$, and activated carbon [kg year^{-1}] and $Reagents_{fraction_i}$ the consumption per ton waste fraction [kg t^{-1} waste].

Although flue gas cleaning reduces pollutant concentration, releases to air are

produced requiring control and monitoring measures. These emissions are generated during the combustion of different waste fractions. Therefore, it is essential to determine the emissions of different pollutants associated with each waste fraction using allocation rules.

Nitrogen compounds (NO_X, NH_3, and N_2O), dioxins and furans (PCDD/F), and dust (TSP and PM_{10}). Emissions of NO_X, NH_3, N_2O, and dust depend on applied technology rather than waste composition. A special case is PCDD/F emissions. Often, dioxins are suggested to be allocated to different waste components in relation to the chlorine content of the waste. However, other authors advocate that the emissions of these pollutants are more related to the operating conditions; therefore, PCDD/F should be allocated to the waste component in relation to the mass or energy content of the waste (Finnveden 1999). In this study, the latter proposal was applied. PCDD/F emissions are thought to depend more on operational conditions and treatment technologies (of both combustion and flue gases) than on the Cl content of the input waste. Moreover, MSW has a vast surplus of chlorine; the mere fact that the MSW is incinerated under combustion conditions indicates that the process can form PCDD/Fs. Mass allocation was presented as the best option for these pollutants according to Eq 2.8.

$$\mathrm{P}_{\mathrm{fraction_i}} = \frac{\mathrm{P}}{\mathrm{In}_{\mathrm{wet\ MSW}}} \qquad [2.8]$$

where, P is the emission of dust, nitrogen compounds, and PCDD/F [kg $year^{-1}$] and $\mathrm{P}_{\mathrm{fraction_i}}$ the emission per ton waste fraction [kg t^{-1} wet waste].

Metals (Cd, Cr, Cu, Hg, Ni, Pb, Zn, and Mn) and metalloids (As). Emissions of these compounds depend on the input waste composition; therefore, they were allocated based on the content of the respective elements in the input waste fractions. The metal and metalloid content showed in Annex A1 is based on the data published by Riber et al. (2009).

$$\mathrm{M}_{\mathrm{content\ fraction_i}} = \frac{\mathrm{M}_{\mathrm{fraction_i}} * \mathrm{In}_{\mathrm{dry\ fraction_i}}}{1000} \qquad [2.9]$$

$$\mathrm{M}_{\mathrm{emission_i}} = \frac{\mathrm{M} * \frac{\mathrm{M}_{\mathrm{content\ fraction_i}}}{\sum_1^i \mathrm{M}_{\mathrm{content\ fraction_i}}}}{\mathrm{In}_{\mathrm{fraction_i}}} \qquad [2.10]$$

where, M is the emission of As, Cd, Cr, Cu, Hg, Ni, Pb, Zn, and Mn [kg $year^{-1}$]; $\mathrm{M}_{\mathrm{fraction_i}}$ the metal and metalloid content in each waste fraction [mg metals

kg^{-1} dry waste]; $M_{\text{content fraction}_i}$ the metal and metalloid content in each waste fraction [kg metals]; and M_{emission_i} the emission of metals per ton waste fraction incinerated [kg metals t^{-1} wet waste].

Acid gases (SO_X, HCl, and HF). These emissions were allocated to the input waste based on the S, Cl, and F content of the waste displayed in Annex A1. Therefore, the calculation shown in Eq 2.11 is identical to those proposed for metals in Eqs 2.9 and 2.10.

$$X_{\text{emission}_i} = \frac{X * \frac{X_{\text{content fraction}_i}}{\sum_1^i X_{\text{content fraction}_i}}}{\text{In}_{\text{fraction}_i}} \quad [2.11]$$

where X is the emission of SO_X, HF, and HCl [kg $year^{-1}$]; $X_{\text{content fraction}_i}$ the S, F, and Cl content in each waste fraction [mg S, F, and Cl]; and X_{emission_i} the emission of SO_X, HF, and HCl per ton waste fraction incinerated [kg t^{-1} waste].

Carbon compounds (CO_2, CO, CH_4, NMVOC, TOC, and PAHs). The allocation of these pollutants must be performed according to the total or fossil carbon content of the input waste reported in Annex 1. Carbon dioxide emissions are related to the C content of the waste. Nevertheless, the climate-relevant CO_2 emissions from waste incineration are determined by the proportion of waste carbon compounds that are of fossil origin. However, the contribution of CH_4, CO, NMVOC, TOC, and PAHs to climate change is only partially dependent on (for CO and CH_4) or completely independent of the fossil C content. Therefore, these compounds were allocated based on the total C content.

$$CO_{2\ \text{fraction}_i} = \frac{\frac{\text{Fossil C}_{\text{fraction}_i} * \text{In}_{\text{dry fraction}_i} * 1000}{\sum_i^n \left(\text{Fossil C}_{\text{fraction}_i} * \text{In}_{\text{dry fraction}_i} * 1000\right)}}{\text{In}_{\text{fraction}_i}} * CO_{2\ \text{fossil}} \quad [2.12]$$

$$\text{Ccomp.}_{\text{fraction}_i} = \frac{\frac{C_{\text{fraction}_i} * \text{In}_{\text{dry fraction}_i} * 1000}{\sum_i^n \left(C_{\text{fraction}_i} * \text{In}_{\text{dry fraction}_i} * 1000\right)}}{\text{In}_{\text{fraction}_i}} * \text{Ccomp.} \quad [2.13]$$

where C_{fraction_i} is the total C content in each waste fraction [g C kg^{-1} dry waste fraction]; $\text{Fossil C}_{\text{fraction}_i}$ the fossil C content in each waste fraction [g fossil C kg^{-1} dry waste fraction]; $CO_{2\ \text{fossil}}$ the fossil CO_2 emission per year [kg CO_2 $year^{-1}$]; $CO_{2\ \text{fraction}_i}$ the fossil CO_2 emission per ton waste fraction incinerated [kg CO2 t^{-1} wet waste]; Ccomp. the emission of CO, CH_4, NMVOC, TOC, and PAHs per year [kg $year^{-1}$] and $\text{Ccomp.}_{\text{fraction}_i}$ the emission per ton waste fraction incinerated [kg t^{-1} wet waste].

C) SOLID WASTE

Slag and ashes are the main waste generated in combustion. Among all incinerated fractions, only inert materials (steel, Al, glass, and construction and demolition waste) are completely transferred to slag. The remaining materials were allocated by mass according to Eq 2.14.

$$\mathrm{Slag}_{\mathrm{fraction\ (non-inert)}} = \frac{\mathrm{Slag}_{\mathrm{non-inert}}}{\mathrm{In}_{\mathrm{dry\ non-inert\ MSW}}} \quad [2.14]$$

$$\mathrm{In}_{\mathrm{dry\ non-inert\ MSW}} = \mathrm{In}_{\mathrm{dry\ MSW}} - \mathrm{In}_{\mathrm{dry\ inert\ MSW}} \quad [2.15]$$

$$\mathrm{Slag}_{\mathrm{non-inert}} = \mathrm{Slag} - \mathrm{In}_{\mathrm{dry\ inert\ MSW}} \quad [2.16]$$

where, $\mathrm{In}_{\mathrm{dry\ MSW}}$ is the dry weight of MSW [t]; $\mathrm{In}_{\mathrm{dry\ inert\ MSW}}$ the dry weight of inert waste [t]; $\mathrm{In}_{\mathrm{dry\ non-inert\ MSW}}$ the dry weight of non-inert waste [t]; Slag the amount of slag generated [t year^{-1}]; $\mathrm{Slag}_{\mathrm{non-inert}}$ the amount of non-inert slag [t year^{-1}] and $\mathrm{Slag}_{\mathrm{fraction\ (non-inert)}}$ the amount of non-inert slag per ton waste [kg t^{-1} waste fraction].

In relation to ash, inert materials are not incinerated; therefore, they are not transferred to ash, whereas, non-inert materials are allocated by mass according to Eq 2.17.

$$\mathrm{Ash}_{\mathrm{fraction(non-inert)}} = \frac{\mathrm{Ash}}{\mathrm{In}_{\mathrm{dry\ fraction_i}}} \quad [2.17]$$

where, Ash is the amount of ashes generated [t year^{-1}] and $\mathrm{Ash}_{\mathrm{fraction(non-inert)}}$ the amount of ashes generated in incineration of non-inert material fraction per ton waste [kg t^{-1} waste fraction].

2.2.3 Critical review report

The proposed LCI and model are part of a software tool that will assist to look for more eco-efficient and sustainable solutions for packaging waste management. LCA intended to be disclosed to the public; therefore, a review of the study by an external expert is required. Specifically, a professional with expertise in LCA and waste management was selected in order to assure a complete coverage of the relevant elements of the study. This critical review was carried out from May to October 2012. The review was performed by the external reviewer Michael Zwicky Hauschild, Chemical Engineer (1988) and PhD (1992) by the Technical University of Denmark

(DTU). He is the Head of Section of Quantitative Sustainability Assessment, responsible for research and education in quantitative sustainability assessment and decision support tools at DTU. Main focus on scientific methodologies and principles on life cycle impact assessment in quantification of the sustainability dimensions. He has managed and participated in several projects and has supervised 10 PhD thesis. Dr. Hauschild is receiver of several prices and awards. such as the Nordic Council's Great Nature and Environment Award (1997) and the European Better Awards for Industry (1998) for the results of the EDIP-programme. He form part of the editorial board of several scientific journals, such as the International Journal of LCA. He is reviewer for more than 10 international journals, and he has published more than 100 papers. Additionally, Dr. Hauschild is Chairman of the UNEP-SETAC task force on LCIA of toxic impacts, and member among others, of the management board of the Danish LCA Center, the board of the Danish topic centre for waste, and the Danish delegation to the ISO standardisation of LCIA.

The study and the critical review were conducted following the ISO 14040 and ISO 14044 requirements. The critical review report was focused on the most critical inconsistences found during the review. Thus, the report did not include the assumptions and models that were considered correct. The name and affiliation of the reviewers, as well as the report and the answers of the authors to the suggestions of the reviewers must be included in the report of the LCA study. According to the clause 6.1 of the ISO 14044, *"the critical review process shall ensure that:*

- *The methods used to carry out the LCA are consistent with this International Standard;*
- *The methods used to carry out the LCA are scientifically and technically valid;*
- *The data used are appropriate and reasonable in relation to the goal of the study;*
- *The interpretations reflect the limitations identified and the goal of the study;*
- *The study report is transparent and consistent".*

Particularly, the reviewer commented a far advanced draft of the study, and the authors replied and reviewed the work according to these comments. Moreover, the reviewer had the opportunity to reply the final draft and the answers of the authors. The reviewer has commented general aspects of the study, the definition of the goal, the assumptions and allocations applied, the LCI, and the model developed with the LCA software GaBi 4. In particular, the reviewer made more than 60 comments classified in three groups: editorial, general and technical. From all of the comments, near 50 comments were of technical nature. Summarising, the reviewer considers that in general the LCA report is an objective and balanced study of the WtE technologies in Spain and Portugal.

2.2.4 Application of the model

The results of the model are the natural resources consumption and emissions values and the waste and products of the incineration process associated with each input waste fraction. These factors can be used to calculate the waste fraction LCIs.

To display the application of the model, Table 2.8 details the results obtained for five waste fractions. According to this method, those input and output data parameters, which are process dependent, were allocated by mass. This group included; the consumption of air, water, combustibles, and reagents; the generation of ashes and slag; and the emissions of dust, nitrogen compounds, and dioxins and furans. PCDD/F is a special case because the distinction between process and product dependence in not clear. The emission of this pollutant are suggested to be allocated based on the chlorine content; however, the operating conditions have a high influence on the emissions as well. In this case, the choice of allocations methods will have a strong influence on the emissions of each waste fraction. For all of the process-dependent parameters, identical results were obtained for the different waste fractions, reasserting its independence with the input waste composition (Seyler et al. 2005). Nevertheless, the results of the product dependent parameters, such as energy production, emissions of carbon compounds, heavy metals and acid gases were different for each fraction. This difference is because process dependent parameters depend on waste composition and heating value.

Table 2.8 Results of the input and output data associated with each waste fraction.

	HDPE	LDPE	Steel	Al	PC	Organic
Consumptions (kg t^{-1} waste)						
Air	3,105	3,105	3,105	3,105	3,105	3,105
Water	415	415	415	415	415	415
Diesel	$1.74\ 10^{-1}$	$1.74\ 10^{-1}$	$1.74\ 10^{-1}$	$1.74\ 10^{-1}$	$1.74\ 10^{-1}$	$1.74\ 10^{-1}$
NH_3	2.92	2.92	2.92	2.92	2.92	2.92
Waste (kg t^{-1} waste)						
Slag	153	153	1,000	1,000	153	153
Ash	78.78	78.78	0.00	0.00	78.78	78.78
Products (MJ t^{-1} waste)						
Energy	8,844	8,844	0.00	0.00	1,890	850
Air emissions (kg t^{-1} waste)						
NOx	$7.64\ 10^{-1}$	$7.64\ 10^{-1}$	$7.64\ 10^{-1}$	$7.64\ 10^{-1}$	$7.64\ 10^{-1}$	$7.64\ 10^{-1}$
PCDD/F	$3.30\ 10^{-10}$	$3.30\ 10^{-10}$	$3.30\ 10^{-10}$	$3.30\ 10^{-10}$	$3.30\ 10^{-10}$	$3.30\ 10^{-10}$
As	$8.38\ 10^{-6}$	$3.51\ 10^{-6}$	$4.31\ 10^{-4}$	$1.57\ 10^{-4}$	$4.85\ 10^{-6}$	$9.99\ 10^{-7}$
HCl	$1.01\ 10^{-2}$	$7.10\ 10^{-3}$	-	-	$2.86\ 10^{-3}$	$1.24\ 10^{-2}$
HF	$1.33\ 10^{-3}$	$1.33\ 10^{-3}$	-	-	$4.99\ 10^{-3}$	$2.88\ 10^{-4}$
SOx	$3.37\ 10^{-2}$	$3.37\ 10^{-2}$	-	-	$1.20\ 10^{-1}$	$1.17\ 10^{-2}$
CO_2	2,237	2,236	-	-	-	-
CO	$2.88\ 10^{-1}$	2.8810^{-1}	-	-	$1.37\ 10^{-1}$	$3.17\ 10^{-2}$

The energy production assessment shows that, the fractions that generate more energy during combustion are PET, HDPE, and LDPE because of the high energy content (43.47 MJ kg^{-1}), while steel and aluminium fractions do not generate energy as their energy content is null. The following was found in relation to the emissions:

- Steel and aluminium are the fractions with the highest heavy metals emissions because of the high metal content in the input waste (2.00 mg As kg^{-1} waste), with lower contributions from the organic matter and plastic mix (0.26 and 0.20 mg As kg^{-1} waste). The combustion of Al and steel do not generate other types of pollutants because Cl, F, C, and S are absent from the input waste.

- Carbon compound emissions are mainly associated with HDPE and LDPE combustion because of the high fossil and biological carbon content. However, some differences are observed in the CO and CO_2 emissions. Al and steel combustion does not generate emissions of CO and CO_2 because this waste lacks fossil and biological C. Nevertheless, PC and organic matter combustion generate only CO emissions because the entire C content is of biological origin. Regarding acid gases, the

combustion of PC and the incineration of PET, HDPE, and LDPE contribute to HF emissions. The high Cl content in the plastic mix and organic matter generates the highest emissions of HCl. SO_X emissions are primarily associated with paper and plastic combustion. To complete the analysis of air emissions, the EB of global warming (GW) and atmospheric acidification (AA) proposed by the Institution of Chemical Engineering (IChemE) (ICheme 2002) were studied for the described waste fractions (Fig 2.7). The greatest AA burden was observed in paper and cardboard combustion due to emissions of acid gases, whereas the great carbon content of HDPE and LDPE had the highest contribution to GW.

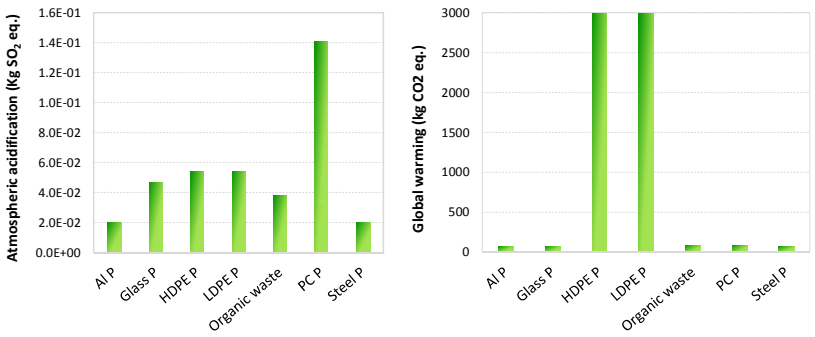


Fig 2.7 Global warming and atmospheric acidification of the studied waste fractions.

Finally, the results for waste generation displayed, on one hand, that identical amounts of slag is generated for non-inert fractions because a mass allocation was applied. For inert fractions, such as Al and steel, the entire waste is expected to be completely transferred to slag, achieving a value of 1,000 kg of slag t^{-1} waste. On the other hand, in ash generation, inert materials are not converted to ash, whereas for non-inert materials, a mass allocation was applied, producing identical results for all fractions.

2.2.5 Inclusion of the model in the FENIX tool

The developed model was included in an ad-hoc software tool for evaluating alternative scenarios for the management of post-consumer packaging waste, enabling the selection of more sustainable waste management practices, in accordance with the current European policy principles. The tool is intended to be used directly by the local municipalities

and waste management entities in Spain and Portugal, and therefore, a special effort was devoted to developing a user-friendly on-line interface whereby the user is allowed to build their own scenario by simply typing in the waste quantities and selecting the appropriate collection, sorting, and treatment (recycling, landfilling, and/or incineration) processes. Up to three different and independent waste management scenarios can be defined. Per each of them, up to three ambits (urban, semi-urban, and rural) can also be defined. A large number of parameters may be adjusted, such as: initial waste composition, types of collection systems and relative distances, collection frequency, types of transference plants, selection plants efficiencies, and end-of-life treatment options (recycling, incineration, landfill). Access to the tool is web-based, free of charge in the Website of the FENIX project (FENIX 2010). The waste management processes that have been included at FENIX´s model are divided into three stages:

- **Collection**. This stage includes collection, transport to the waste until transference or treatment plants and operation of the transference plant, and the transport of waste from them to treatment plants.
- **Treatment**. The sorting plants (for light packaging waste and unsorted municipal waste), and also the direct incineration or landfilling of the unsorted waste, are included in this stage. Final treatment of the residual flows of sorting plants to incineration, landfilling or cement kilns, and their respective transport to those treatments, are also included. Credits for treatment associated with energy recovery are considered as well.
- **Valorisation.** This phase includes transport to the recycling plant – including preconditioning operations, if necessary (for paper and cardboard and glass) – operation thereof, and management of all generated residues. In the case of high-energy-content plastic waste flows (LDPE and plastic mix recovered from the sorting plant), the option of their transport and incineration with energy recovery (either in municipal waste incinerators or cement kilns) apart from its mechanical recycling is also considered. Credits for recycling and energy recovery of all these valorisation processes are included as well.

The developed model was included in the treatment step, assisting to decide the best method of handling MSW in Spain and Portugal.

2.3 Environmental assessment of waste to energy plants in Spain

LCA is recognized as the best tool for assessing the environmental impacts of products, process, and services (Finnveden et al. 2009). Among the numerous LCA applications, the following ones are highlighted (Iribarren 2010; Muñoz 2006):

- To identify major environmental impacts and the life cycle stages or "hot spots" contributing to these impacts. This identification allows to detect opportunities to improve the environmental performance of products throughout the different stages of their life cycle.
- To compare environmental impacts of alternative products or processes.
- To provide a picture as complete as possible of the interactions of any activity with the environment.
- To provide decision maker with information on the environmental effects of the activities.
- Marketing. For instance, by implementing an environmental labelling scheme, making an environmental claim, or supporting a product's environmental statement.

Historically, most of the LCA applications have been product-oriented, encompassing a broad range of products, from beverage packaging and washing machines in the earlier studies, to products from several industrial sector, such as energy, metals, polymers, food, agriculture, and also chemicals, in the last works. Apart from direct product application, LCA can also be used in a wider sense. Rather than dealing with physical goods, LCA

can be applied to assess process, business strategies, or government policies, like comparing different waste management strategies (Muñoz 2006). Nevertheless, there are still barriers that inhibit the broader implementation of LCA and life cycle thinking (LCT). Several of the limitations of this tool are:

- LCA is a very data intensive, and lack of data can restrict the conclusions that can be draw from a specific study.
- LCA aims at providing a comprehensive view of environmental impacts. However, not all types of impact (such as land use) are equally well covered in a typical LCA.
- An LCA can include several methodological choices which are uncertain and may potentially influence the results. Examples include allocation methods, definition of system boundaries, methodological choices, and choices of characterization methods.

On the other hand, the LCA methodological development has been strong over the last decades. The methodology has been harmonized and unified by means of development of the ISO regulations. According to the ISO 14040 (ISO 2006a) the methodology of LCA is composed of the four stages illustrated in Fig 2.8: a) the definition of the goal and scope; b) life cycle inventory analysis; c) life cycle impact assessment; and d) interpretation.

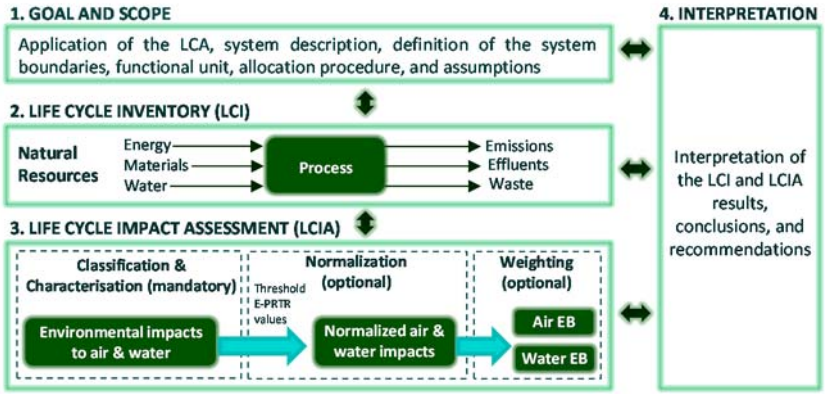


Fig 2.8 Life cycle assessment methodology.

As a result of the standardization of the methodology, the confidence in LCA and life cycle thinking has grown. This is illustrated in the increased use of this tool in different parts of society, and enhanced through the development of

recommendations by authoritative bodies. Another trend is the increased use of LCA and promotion LCT on a policy level. Particularly, life cycle thinking is an important element of European environmental policy (Finnvenden et al. 2009). LCT has been introduced in several policies, such as the Integrated Product Policy (IPP) (EC 2003), the Waste Framework Directive (EC 2008), the thematic strategic on sustainable use of natural resources (EC 2005b), and the thematic strategic on the prevention and recycling of waste (EC 2005a).

Related to LCIA, the objective of this step is to provide further information to evaluate the results of the LCI in order to better understand the environmental performance of a product system (Iribarren 2010). In the LCIA, the inputs and outputs data collected in the LCI are translated into an impact indicator results related to human health, natural environment, and resource depletion (EC JRC 2010b). LCIA is composed of two mandatory steps and two optional stages:

- **Selection of impact categories and classification**: It includes the selection of the impact categories and the characterisation models of the study (Bare 2010). This selection should include an exhaustive set of environmental issues related to the system under study, and should be in agreement with the defined goal and scope (Iribarren 2010). Fig 2.9 shows the classification of the impact categories into midpoints and endpoints.

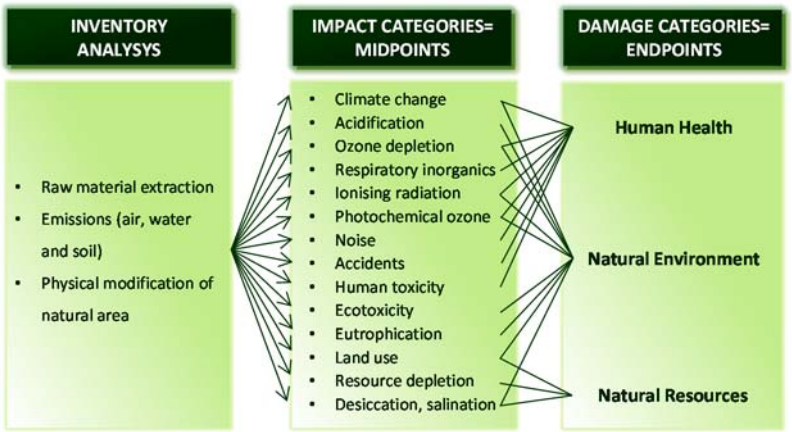


Fig 2.9 Framework of impact categories at midpoint and endpoint.

Midpoint categories also denominated environmental impact assessment methods, result in the definition of an environmental profile by means of

the quantification of the environmental effect of the product on several categories (global warming, acidification, etc.). Midpoint methods evaluate the indirect/intermediate effects on the human being. On the other hand, endpoint or damage assessment methods, evaluate the final effect of the environmental impact by identifying and determining the damage caused to the human being and the natural systems (Iribarren 2010). Endpoint are less comprehensive and have much higher levels of uncertainly than better-defined midpoint categories. Midpoint categories, on the other hand, are harder to interpret because they do not deal with and endpoint associated with an area of protection, which may be more relevant for decision making (Reap et al. 2008). Table 2.9 shows the main midpoint and endpoint LCIA methods (Rack et al. 2013; IHOBE 2009).

- **Characterisation:** The impact of each emission or resource consumption is modelled quantitatively using a characterisation factor or potency factor (PF). That factor expresses how much that flow contributes to the impact category indicator (EC JCR 2010b).

- **Normalisation (optional):** It related the magnitude of impacts in different impact categories to reference values (Bare 2010). The aim of normalisation is two-folded: a) to place the LCIA indicator results into a broader context; and b) to adjust the results to have common dimensions (Bojarski 2010). In particular, the characterised impact scores associated with a common reference facilitate comparisons across impact categories. Additionally, normalisation can be used to check inconsistences of LCI and LCIA results, provide and communicate information of the relative significance of the indicator results, and prepare for additional procedures such as grouping, weighting, or life cycle interpretation (Bare et al 2006).

- **Weighting (optional):** The different environmental impact categories are ranked according to their relative importance. Weighting may be necessary when trade-off situations occur in LCAs which are being used for comparing alternative products (EC JCR 2010a). The advantage of this stage is that different impact categories are converted to a numerical score of environmental impact; therefore, making it easier to make decisions. However, a lot of information is lost, and reality is simplified (Muñoz 2006). Moreover, weighting remains a controversial element of LCA, mainly because weighting involves social, political, and ethical values choices (Pennington et al. 2004).

Table 2.9 Review of the LCIA methods.

Scope of application		Method	Impact assessment level	Creator	Reference
Asia	Japan	LIME	Combined	-Tokyo City University / Kogakuin University	Itsubo and Inaba 2012
	Singapore	Singapore IMPact ASSessment (SIMPASS)	Midpoint	-National University of Singapore/Institute of Chemical and Engineering Sciences of Singapore	Chan et al. 2012
Europe	Switzerland	IMPACT 2002+	Combined	-École polytechnique fédérale de Lausanne (EPFL)	Jollie et al. 2003
		Ecological Scarcity	Midpoint	-Swiss Ministry of Environment (BUWAL)	Frischknecht et al. 2006
	The Netherlands	ReCiPe	Combined	-Pré Consultans	Goedkoop et al. 2012
		Eco-Indicator 99	Endpoint	-Pré Consultans	Goedkoop et al. 2000
		CML 2001	Midpoint	-Centre of Environmental Sciences (CML)	Guinée et al. 2001
		LC-IMPACT	Combined	-Radboud University	LC-IMPACT 2009
	Denmark	EDIP 2003	Midpoint	-Technical University of Denmark (DTU)	Hauschild and Potting 2005
America	USA	TRACI	Midpoint	-Environmental Protection Agency (EPA)	Bare et al. 2003
	Canada	LUCAS	Midpoint	-CIRAIG	Toffoletto et al. 2007
Global		IMPACT World+	Combined	-CIRAIG/ DTU / Quantis international / University of Michigan / EPFL / Cycleco	IMPACT World+ 2012

2.3.1 Description of the environmental sustainability assessment methodology

Most existing LCA studies of waste management applied the conventional impact assessment methods showed in Table 2.9, such as CML 2001 (Guinée et al. 2001), EDIP 97 (Wenzel et al. 1997), or Eco-indicator 99 (Goedkoop et al. 2000). These methods use a set of metrics, which in some cases could be difficult to understand and thus confuse the process comparisons (Margallo et al. 2014b). In this sense, a reduction of complexity of the LCA would improve the comprehension of the results and thus assist the decision making process. For this reason, this work propose a technical procedure to carry out the environmental sustainability assessment (ESA) of waste incineration. ESA is composed of the two mandatory and the two optional steps included in the stage of LCIA. The advantage of this procedure based on an LCA approach regarding the conventional methodologies, is that it allows to evaluate the use of natural resources (NR) (i.e., depletion/exhaustion) and the release of pollutants to specific environmental compartments (i.e., air, water, and land), providing a complete overview of the environmental performance of the process. Moreover, the normalisation factors of conventional LCIA are calculated with the substance emissions and characterised factors per substance. In this methodology, the normalisation was conducted by means of the threshold values of the E-PRTR regulation (EC, 2006). This way, the relevance of each EB at a policy and regulatory level was included because the European Commission sets these threshold values for each specific pollutant. The normalisation and weighting procedure supplies a framework to compare all the European installations included in the industrial sectors of the Integrated Pollutant Prevention and Control Directive (IPPC) using a European policy weighting.

A) CLASSIFICATION AND CHARACTERISATION

Fig 2.10 displays the ESA methodology based on two main variables: natural resources sustainability (NRS) and environmental burdens sustainability (EBS). NRS includes the consumption of the final useful resources such as energy ($X_{1,1}$) [MJ], materials ($X_{1,2}$) [kg], and water ($X_{1,3}$) [kg] for the considered process and/or product, thus, it can be described by a NRS dimensionless index X_1. Land is currently excluded as an NR because WTE is not considered

an intensive-land use process (Margallo et al. 2014d). On the other hand, EBS includes the primary burdens to air, water, and land due to the release of pollutant (i.e., gas, liquid, and solid). EBS is given by the environmental sustainability metrics developed by IChemE. This set of indicators can be used to measure the environmental sustainability performance of an operating unit, providing a balanced view of the environmental impact of inputs (resource usage), and outputs (emissions, effluents, and waste) (IChemE 2002). In relation to the outputs, a set of environmental impacts on the atmosphere, aquatic media, and land was chosen. The EB approach was used to estimate and quantify the potential environmental impacts (García et al. 2013). In particular, the environmental impacts were classified in 12 variables grouped into the release to each environmental compartment: air ($X_{2,1}$), water ($X_{2,2}$), and land ($X_{2,3}$). These environmental impact categories are a sub-set of those used internationally in environmental management, selected to focus on areas where the activities of process industry are most significant.

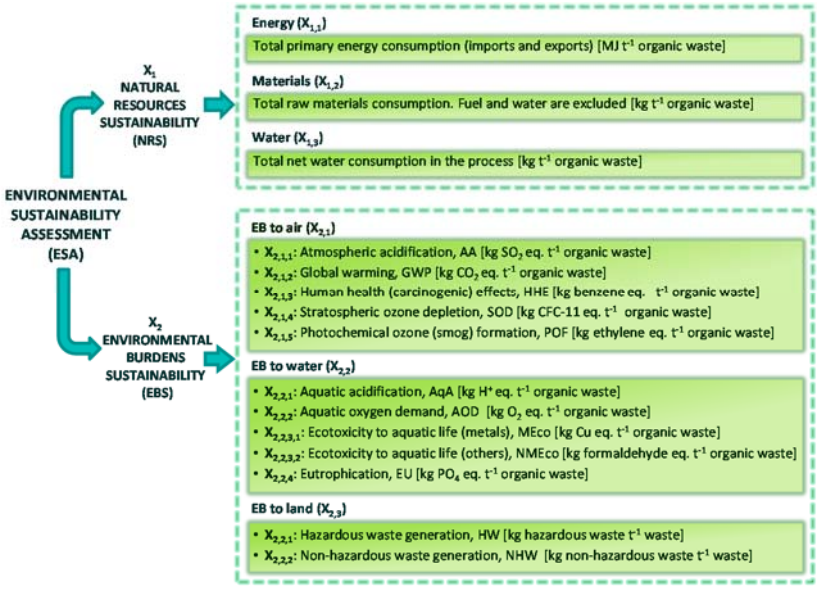


Fig 2.10 LCIA methodology based on natural resources and environmental burdens.

B) NORMALISATION AND WEIGHTING

NR and EB are rarely normalised; however, the consumption of NR (X_1) varies from plant to plant; therefore, to understand whether the consumption of a given plant is acceptable and to compare each plant, a reference should be

used. Particularly, the average consumption of each i NR in the 10 existing WtE plants located in Spain was selected as the reference value ($X_{1,i}^{ref}$). On the other hand, the variables for EBS were compared using the threshold values taken from the E-PRTR regulation (E-PRTR Regulation 2006) (Table 2.10), leading to normalised variables ($X_{2,j,k}^{*}$) (Irabien et al. 2009). The E-PRTR regulation establishes the contaminants for which the European installations must provide notification to the authorities along with the threshold values of those pollutants. The threshold values can be used as an important aid in the normalisation process because they provide an overview of the environmental performance of the installation at a European level (Margallo et al. 2014c).

Table 2.10 Threshold values from the E-PRTR regulation for normalisation and nº of substances included in each impact category. The units are given in kg equivalents (kg eq.).

Environmental burdens (EB)	**Threshold value (kg y^{-1})**	**Nº substances**
EB to air		
Atmospheric acidification (AA) [kg SO_2 eq.]	150,000	6
Global warming (GW) [kg CO_2 eq.]	100,000,000	23
Human health effects (HHE) [kg benzene eq.]	1,000	52
Photochemical ozone formation (POF) [kg ethylene eq.]	1,000	100
Stratospheric ozone depletion (SOF) [kg CFC-11eq.]	1	60
EB to water		
Aquatic oxygen demand (AOD) [kg O_2 eq.]	50,000	14
Aquatic acidification (AqA) [kg H^+ eq.]	100	4
Ecotoxicity to aquatic life (organics) (MEco) [kg Cu eq.]	50	11
Ecotoxicity to aquatic life (metals) (NMEco) [kg formaldehyde eq.]	50	18
Eutrophication (EU) [kg phosphate eq.]	5,000	8
EB to land		
Hazardous waste (HW) (kg hazardous waste)	2,000	-
Non-Hazardous waste (NHW) (kg non-hazardous waste)	2,000,000	-

Eqs. 2.18 and 2.19 show the basic calculations that were used for the NRS and EBS normalisation.

$$X_{1,i}^{*} = \frac{X_{1,i}}{X_{1,i}^{ref}} \quad [2.18]$$

$$X_{2,j,k}^{*} = \frac{X_{2,j,k}}{X_{2,j,k}^{ref}} \quad [2.19]$$

In Eqs. 2.18 and 2.19, i represents different NR (energy, materials, and water);

j represents different environmental compartments (air, water, and land); k represents the environmental impacts to air, water, and land described in Fig 2.9; $X_{1,i}$ is the consumption of each i NR; $X^{*}_{1,i}$ is the normalised value of $X_{1,i}$; $X^{ref}_{1,i}$ is the NR taken as reference value; $X_{2,j,k}$ are the EB to air, water, and land; $X^{*}_{2,j,k}$ is the normalised value of $X_{2,j,k}$; and $X^{ref}_{2,j,k}$ is the reference value used for EBS normalisation.

The three NRS normalised variables ($X^{*}_{1,i}$) that represent energy, materials, and water consumption and the 12 EBS normalised variables ($X^{*}_{2,j,k}$) are subjected to direct summation. Therefore, the NRS index (X_1) can be assessed according to Eq 2.20, whereas the calculations of the EBS index to air ($X_{2,1}$), water ($X_{2,2}$), and land ($X_{2,3}$) are based on Eq 2.21.

$$X_1 = \gamma\, \alpha_{1,1} X^{*}_{1,1} + \sum_{i=2}^{i=n} \alpha_{1,i} X^{*}_{1,i} \qquad n \in [2,3] \qquad [2.20]$$

$$X_{2,j} = \sum_{j=1}^{j=n} \beta_{2,j,k} X^{*}_{2,j,k} \qquad n \in [1,2] \qquad [2.21]$$

In Eqs. 2.20 and 2.21, X_1 is the NRS index that includes energy, materials and water consumption; $\alpha_{1,i}$ is the weighting factor for the materials and water variables; $X_{2,j}$ are the EBS indexes for air, water, and land; $\alpha_{1,1}$ is the weighting factor for the energy variable; $\beta_{2,j,k}$ is the weighting factor for EBS; and γ is the factor accounting for the energy net importer or exporter character of the plant. The factor γ has a value of -1 when the plant exports energy and +1 when plant imports energy. Consequently, the NRS index depends on the weight assigned to each final resources variable. When the three final resources are equally relevant, $\alpha_{1,i} = 1/3$ for each i. This was assumed because it is the clearest way to obtain a single index that allows a comparison across several plants. The application of a weighting factor of 1/3 to all the components of NR to obtain a single index, yields an overview of the performance of the plant requiring the evaluation of the individual NR to determine the critical points of the process. Other sets are also possible, but different weighting procedures must be discussed.

2.3.2 System description

The developed life cycle model and the ESA methodology were applied to several WtE plants located in Spain, in order to assess and compare the environmental performance of the incinerators. From all of the input waste fractions, only organic waste was evaluated using the variables NRS and

EBS. Four Spanish incinerators, namely I_{S1}, I_{S2}, I_{S3}, and I_{S4}, all of which are equipped with GI, were selected as a case study. Specifically, these incinerators were selected because the data available were the most representative of all of the Spanish plants and because the chosen incinerators share similar geographical locations. Nevertheless, they differ in age, pollutant abatement technologies, and consumption of reagents and combustible. Table 2.11 displays some technical data of the plants (AEVERSU 2013), whereas Table 2.12 shows the waste composition of the incinerators, as well as the average composition of Spanish MSW (Ecoembes 2014).

Table 2.11 Technical characteristics of the Spanish WtE plants under study.

		I_{S1}	I_{S2}	I_{S3}	I_{S4}
Star up year		1975	1994	1984	1991
Incineration capacity (t h^{-1})		14.5	10.0	2.50	9.60
Nº lines		3	2	2	2
Thermal treatment		Roller GI	Travelling GI	Reverse acting GI	Roller GI
Combustion temperature (ºC)		950	1,000-1,100	1,050	900-1,500
FGT treatment	NO_x	SNCR	SNCR	SNCR	SNCR
	Particles	Electro-filter	Bag filter	Electrofilter/ bag filter	Bag filter
	Acid gases	Semidry	Semidry/ dry	Dry	Semidry
FGT reagents	Activated carbon	✓	✓	✓	✓
	Urea	✓	✓	✓	
	NH_3				✓
	CaO	✓			✓
	$Ca(OH)_2$		✓	✓	

Table 2.12 Waste composition of the plants and the Spanish average as percentage (%).

	Type of C origin	I_{S1}	I_{S2}	I_{S3}	I_{S4}	**Spanish average**
PET	Fossil	1.96	1.17	1.89	2.07	2.23
HDPE	Fossil	1.13	0.95	1.34	1.36	1.38
LDPE	Fossil	5.55	4.29	7.15	7.01	7.56
Plastic mix	Fossil	5.31	4.89	4.29	4.91	4.53
Steel	-	3.02	2.72	3.09	4.11	2.81
Aluminium	-	0.46	0.25	0.45	0.30	0.39
Beverage carton	Fossil and biological	1.04	0.57	1.23	0.93	1.40
Glass	-	4.37	5.38	4.94	4.08	3.55
Paper and cardboard	Biological	13.3	9.28	9.74	12.5	14.9
Organic matter	Biological	43.8	53.3	47.9	45.3	40.5
Remaining materials	-	16.4	13.6	14.3	13.6	16.3
Moisture		3.70	3.46	3.76	3.81	4.50

According to the goal of the study, one ton of organic waste at the gate of the

WtE plant was selected as functional unit. The system includes the thermal treatment and the cleaning of flue gases, ash solidification, and final disposal of ash and slag. Fig 2.11 depicts the system under study considering the life cycle stages.

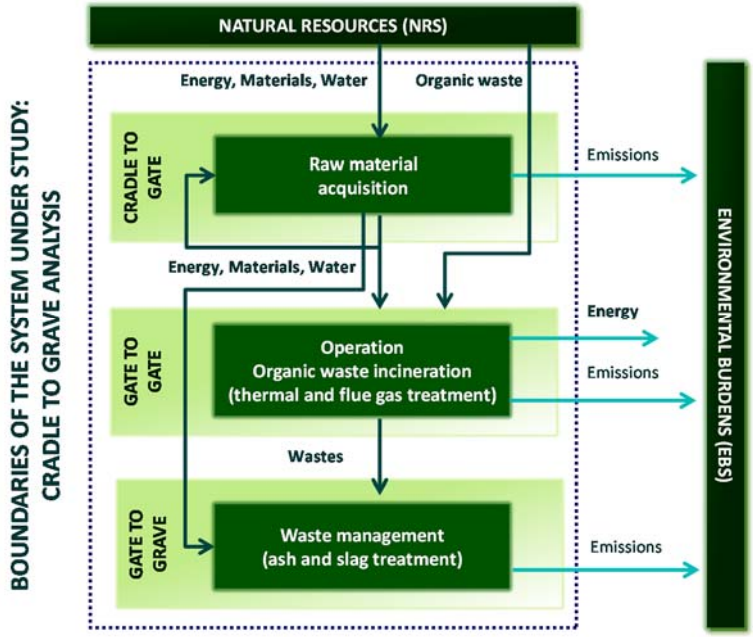


Fig 2.11 System description of the cradle to grave analysis of organic waste incineration.

The analysis was conducting during the Cradle-to-Gate (Cr-Ga), Gate-to Gate (Ga-Ga), and Gate-to-Grave (Ga-Gr) steps.

- **Cr-Ga:** This step includes the final resources of thermal and flue gas treatment: reagents for flue gas treatment, electricity, combustibles, auxiliary materials, and water.
- **Ga-Ga:** In this study, this step refers to the incineration of organic waste. The EB originated from the emissions to air of greenhouse gases, acid gases, organic compounds, dust, and heavy metals. Water emissions from waste incineration are only related to plants with an exhaust gas cleaning system (Bjarnadóttir et al. 2002). In all of the studied plants, wet scrubbers are not used; therefore, water emissions were not considered. Although a high degree of acid gas removal is reached (near

90%) and the waste products from may be re-usable, wet scrubbers have not been applied in these plants because an extensive equipment is necessary and the wastewater produced would require treatment prior to discharge, translating into higher capital costs. Moreover, the land emissions generated were allocated to the Gate-to-Grave step related to the final waste treatment.

- **Ga-Gr:** This analysis considers the burdens from the consumption of final resources of ash and slag treatment. Ash treatment included ash solidification with a mixture of water (30 %), cement (20 %), and ashes (50 %), a process that generates a non-hazardous waste (NHW) that is later landfilled. Slag from MSWI required a magnetic separation to recover ferrous materials. In this study only the organic matter was considered; therefore, slag has a null content of ferrous materials and is sent to a non-hazardous landfill.

The total emissions and consumptions associated with MSWI must allocated to the organic fraction. Specifically, the model developed for organic waste in Chapter 2.2 was applied. Additionally, waste incineration involves waste treatment and energy production, providing to the system an additional function. This situation was handled through system expansion by subtracting the function of alternative system (energy production) to the system under study. In this study the electric power mix of Spain included in the ELCD-PE GaBi database was selected as the technology replaced in the system expansion (PE International 2011).

2.3.3 Life cycle inventory

Table 2.13 shows the inventory for the four Spanish WtE plants. The LCI does consist of annual material and energy inputs and outputs of plants in 2009. All data are given in reference to one ton of organic waste. Data source, geography, and timeframe, are those provided in Chapter 2.2. A zero value for emissions indicates that the plant has not notified that it releases that pollutant; however, the substance could potentially still have been emitted. As previously described, water emissions were not generated. Emissions of carbon dioxide are generated in the MSW incineration; nevertheless, in this analysis, CO_2 was not considered because the organic matter has a null content of fossil carbon (Table 2.12).

Table 2.13 LCI for the selected WtE plants (values per one ton of organic waste as functional unit).

		I_{S1}	I_{S2}	I_{S3}	I_{S4}	Spain average	Units
	THERMAL AND FLUE GAS TREATMENT						
Inputs	**Organic waste**	1.00	1.00	1.00	1.00	1.00	t organic waste
	Energy and combustibles						
	Diesel	2.75	-	1,224	424	205	10^{-3} kg t^{-1}organic waste
	Natural gas	7.46	573	-	-	709	10^{-3} kg t^{-1}organic waste
	Electricity	158	222	241	68.9	312	MJ t^{-1} organic waste
	Reagents and auxiliary materials						
	Urea	3.59	11.3	1.68	0.00	3.44	kg t^{-1} organic waste
	Ammonia (NH_3)	-	-	-	0.97	1.69	kg t^{-1} organic waste
	$Ca(OH)_2$	0.00	13.1	9.54	0.00	4.16	kg t^{-1} organic waste
	Lime (CaO)	8.38	0.00	0.00	6.52	9.15	kg t^{-1} organic waste
	Activated carbon	0.37	0.80	1.68	0.12	0.48	kg t^{-1} organic waste
	Water	313	.49	156	545	344	kg t^{-1}organic waste
	Air	6.70	9.35	6.71	6.71	4.58	10^{3} kg t^{-1}organic waste
Outputs	**Products**						
	Electricity	1,164	1,442	488	574	1,713	MJ t^{-1}organic waste
	Emissions to air						
	Carbon monoxide (CO)	8.30	6.92	8.85	3.88	6.97	10^{-2} kg t^{-1}organic waste
	Carbon dioxide (CO_2)	-	-	-	-	-	
	Methane (CH_4)	0.00	0.00	0.00	0.00	1.26	10^{-3} kg t^{-1}organic waste
	Non-methane volatile organic compounds (NMVOC)	2.48	0.00	0.00	0.00	8.71	10^{-3} kg t^{-1}organic waste
	Polycyclic aromatic hydrocarbons (PAH)	0.00	0.00	10.5	0.00	6.99	10^{-5} kg t^{-1}organic waste
	Total organic compounds (TOC)	2.61	0.00	17.3	3.67	3.12	10^{-3} kg t^{-1} organic waste
	Arsenic (As)	1.34	4.80	19.5	2.08	2.35	10^{-6} kg t^{-1} organic waste

Table 2.13 (cont.) LCI for the selected WtE plants (values per one ton of organic waste as functional unit).

		I_{S1}	I_{S2}	I_{S3}	I_{S4}	Spain average	Units
THERMAL AND FLUE GAS TREATMENT							
Outputs	**Emissions to air**						
	Cadmium (Cd)	52.4	96.8	9.44	12.5	34.3	10^{-7} kg t^{-1} organic waste
	Chromium (Cr)	6.09	4.19	8.76	3.08	26.9	10^{-7} kg t^{-1} organic waste
	Copper (Cu)	38.2	4.47	3.49	4.52	247	10^{-6} kg t^{-1} organic waste
	Manganese (Mn)	36.3	4.15	1.13	0.00	18.8	10^{-6} kg t^{-1} organic waste
	Mercury (Hg)	4.52	6.09	5.45	3.76	3.92	10^{-6} kg t^{-1} organic waste
	Nickel (Ni)	75.71	5.10	7.06	5.15	18.3	10^{-7} kg t^{-1} organic waste
	Lead (Pb)	9.25	25.7	5.55	3.55	8.20	10^{-6} kg t^{-1} organic waste
	Zinc (Zn)	0.00	0.00	0.00	0.00	2.46	10^{-4} kg t^{-1} organic waste
	Dioxins and furans (PCDD/F)	5.65	1.32	1.61	1.22	3.46	10^{-11} kg t^{-1} organic waste
	Hydrogen chloride (HCl)	11.5	5.52	11.7	8.85	8.52	10^{-3} kg t^{-1} organic waste
	Hydrogen fluoride (HF)	13.9	77.1	213	2.54	58.6	10^{-5} kg t^{-1} organic waste
	Ammonia (NH_3)	1.15	0.00	9.23	6.07	1.26	10^{-2} kg t^{-1}organic waste
	Nitrogen oxides (NOx)	7.08	11.2	17.8	6.57	7.50	10^{-1} kg t^{-1} organic waste
	Nitrous oxide (N_2O)	0.00	0.00	0.00	0.00	6.37	10^{-2} kg t^{-1} organic waste
	Sulphur oxides (SO_X)	1.48	8.52	6.63	4.37	2.38	10^{-2} kg t^{-1} organic waste
	Particles (PM_{10})	4.61	0.00	29.7	6.31	4.82	10^{-3} kg t^{-1}organic waste
	Total suspended particles (TSP)	11.5	189	24	7.32	8.94	10^{-3} kg t^{-1} organic waste
	Waste generation						
	Bottom ashes	95.0	236	161	180	140	kg t^{-1} organic waste
	Fly ashes	65.9	90.5	42.4	45.9	83.8	kg t^{-1} organic waste
ASH SOLIDIFICATION							
Inputs	Cement	26.4	36.2	17.0	18.4	33.5	kg t^{-1} organic waste
	Water	39.5	54.3	25.4	27.5	50.3	kg t^{-1} organic waste
	Ash	65.9	90.5	42.4	45.9	83.8	kg t^{-1} organic waste
Outputs	Inert ash	132	181	84.8	91.8	168	kg t^{-1} organic waste

2.3.4 Life cycle impact assessment

The LCIA was conducted following ISO 14040 (ISO 2006a) and ISO 14044 (ISO 2006b) requirements using the LCA software GaBi 4 (PE International 2011). The results were divided into NRS and EBS.

Particularly, the EB proposed by IChemE (IChemE 2002) for air emission were divided into atmospheric acidification (AA), global warming (GW), human health (carcinogenic) effects (HHE), stratospheric ozone depletion (SOD), and photochemical ozone (smog) formation (POF). The EBs for water emissions were defined by the aquatic acidification (AqA), aquatic oxygen demand (AOD), ecotoxicity to aquatic life (metals to seawater) (MEco), ecotoxicity to aquatic life (other substances) (NMEco), and eutrophication (EU). The EB to land was given by the amount of generated hazardous and non-hazardous waste and its management (García et al. 2013).

A) NATURAL RESOURCES SUSTAINABILITY

NRS supports a benchmark comparison in terms of final useful resources including energy, materials, and water. These values were obtained by considering the consumption of energy ($X_{1,1}$), materials ($X_{1,2}$), and water ($X_{1,3}$) in the thermal treatment and flue gases cleaning and during the treatment of slag and ash. Subsequently, the consumption of NR was normalised taking as reference the average consumption of each NR in the 10 existing WtE plants located in Spain. Table 2.14 displays the NRS results normalised respect to this reference value.

Table 2.14 Comparison of dimensionless NRS variables for the selected WtE plants and the Spanish reference.

		Normalised NRS				
Dimensionless NRS variables		**I_{S1}**	**I_{S2}**	**I_{S3}**	**I_{S4}**	**Spanish average**
Energy	$\mathbf{X^*_{1,1}}$	-0.74	-0.88	-0.14	-0.36	-1.00
Materials	$\mathbf{X^*_{1,2}}$	0.83	1.48	0.72	0.58	1.00
Water	$\mathbf{X^*_{1,3}}$	0.89	1.27	0.46	1.45	1.00
TOTAL ($\mathbf{X_1 = \frac{\gamma X^*_{1,1}}{3} + \frac{X^*_{1,2}}{3} + \frac{X^*_{1,3}}{3}}$)	$\mathbf{X_1}$	**0.33**	**0.62**	**0.34**	**0.56**	**0.33**

- **Energy ($X_{1,1}$)**. It includes the consumption of electricity, diesel, and natural gas and the export of electricity. Negative values are associated with net energy export behaviour because plants are able to export much more energy than the amount obtained from diesel or natural gas; the net calorific value of the waste is not considered as it was considered an elementary input flow. All of the studied plants showed negative values; nevertheless, greater energy export was observed in I_{S1} and I_{S2}. Although no plant presented a superior energy performance above the Spanish reference, the $X_{1,1}$ variable for I_{S2} is very similar to the Spanish value; the plant consumed less combustibles but was able to produce less energy. Conversely, I_{S3} and I_{S4} presented values quite different from the Spanish average, mainly due to the lower energy production of these plants.

- **Materials**. The consumption of activated carbon, lime (CaO), calcium hydroxide ($Ca(OH)_2$), urea, and ammonia (NH_3) for flue gases cleaning and the consumption of cement for ash solidification were included in the materials variable ($X_{1,2}$). The consumption of reagents varied in these plants from 0.12 to 1.68 kg t^{-1} organic waste for activated carbon, from 9.54 to 13.1 kg t^{-1} organic waste for $Ca(OH)_2$, from 6.52 to 8.40 kg t^{-1} organic waste for CaO, and from 3.60 to 11.3 kg t^{-1} organic waste for urea. On the other hand, the consumption of cement ranged from 17.0 to 26.4 kg t^{-1} organic waste depending on the generation of ashes. I_{S3} and I_{S4} had the lowest consumption of materials; in particular, the material consumptions of $I_{S1,}$ I_{S3}, and I_{S4} awere all below the Spanish average. The greatest consumption of materials was observed in I_{S2}. Nevertheless, this value should be assessed regarding the emissions to air of the plant so as to determine the efficiency of the flue gas treatment. In this sense, I_{S3} presented a higher rate of emissions of acid gases and NO_X per kg of reagent consumed than I_{S2}, whereas I_{S1} produced the greatest emission of PCDD/F per kg of reagent consumed. Therefore, plant I_{S3} displays the lowest efficiency of flue gas treatment.

- **Water**. The water variable ($X_{1,2}$) comprises the consumption of water in the thermal treatment and flue gases cleaning and in ash solidification. For the selected plants, water consumption varied from 155 to 545 kg t^{-1} organic waste in the incineration process and from 25.4 to 39.5 kg of water t^{-1} organic waste in the ash solidification process. Plant I_{S4} showed the greatest water consumption linked to the incineration process and the

slag cooling. Of all of the incineration plants, only plant I_{S3} presented a lower value than the Spanish average.

To obtain a complete overview of incineration consumption in Spain, the Spanish average of NRS was compared with the survey of MSWI facilities included in the document on the best available techniques for waste incineration (BREF Document), as shown in Fig 2.12 (EC-IPPC 2006). The references values proposed in the BREF are -7,760 MJ of energy t^{-1} waste, 15 kg materials t^{-1} waste, and 250 kg of water t^{-1} waste.

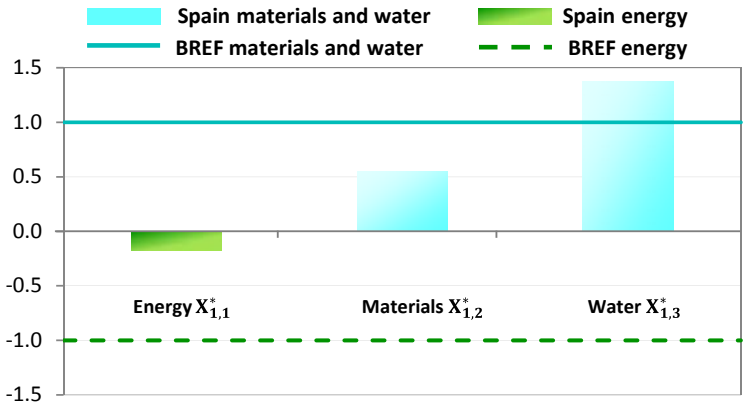


Fig 2.12 Comparison of dimensionless NRS variables of Spain and the BREF reference.

When the Spanish average was compared with the values included in the BREF document, it was observed that only the variable related to materials ($X_{1,2}$) presented lower consumption, primary due to the lower consumption of hydrated lime in the Spanish plants. The plants showed slightly higher water consumption ($X_{1,3}$); however, these values are close to the European data. The largest difference was observed in the energy variable ($X_{1,1}$); this was due the fact that the reference plants included in the BREF document are able to export much more energy than the Spanish plants, and the latter plants do not recover generated heat.

B) ENVIRONMENTAL BURDENS SUSTAINABILITY (EBS)

The values of EB and avoided burdens (AB) to air, water, and land are summarised in Table 2.15 and Table 2.16.The results are divided into Cradle to Gate (Cr-Ga), Gate to Gate (Ga-Ga), and Gate to Grave (Ga-Gr).

Table 2.15 Environmental burdens of the WtE plants I_{S1}, I_{S2}, I_{S3}, and I_{S4}.

			ENVIRONMENTAL BURDENS															
			I_{S1}				**I_{S2}**				**I_{S3}**				**I_{S4}**			
			Cr-Ga	**Ga-Ga**	**Ga-Gr**		**Cr-Ga**	**Ga-Ga**	**Ga-Gr**		**Cr-Ga**	**Ga-Ga**	**Ga-Gr**		**Cr-Ga**	**Ga-Ga**	**Ga-Gr**	
			Manu-facturing of raw materials and supplies	Thermal treat-ment and FGT	Ash treat-ment	Slag land-filling	Manu-facturing of raw materials and supplies	Thermal treat-ment and FGT	Ash treat-ment	Slag land-filling	Manu-facturing of raw materials and supplies	Thermal treat-ment and FGT	Ash treat-ment	Slag land-filling	Manu-facturing of raw materials and supplies	Thermal treat-ment and FGT	Ash treat-ment	Slag land-filling
Air EB $X_{2,1}$	AA $X_{2,1,1}$	10^{-3} kg SO_2 eq. t^{-1} org. waste	8.61	36.4	4.23	-	31.5	85.17	5.81	-	7.41	240	2.72	-	3.90	158	2.95	-
	GWP $X_{2,1,2}$	10^{1} kg CO_2 eq. t^{-1} org. waste	1.80	2.86	0.84	-	3.83	4.51	1.16	-	1.45	7.13	0.54	-	1.12	2.64	0.59	-
	HHE $X_{2,1,3}$	10^{-3} kg benzene eq. t^{-1} org. waste	2.50	39.7	6.71	-	5.06	314	9.22	-	2.92	47.0	4.32	-	2.57	16.6	4.68	-
	POF $X_{2,1,4}$	10^{-4} kg CFC-11 eq. t^{-1} org. waste	19.1	29.5	8.49	-	84.4	59.6	11.7	-	18.1	55.7	5.46	-	7.88	31.47	5.91	-
	SOD $X_{2,1,5}$	10^{-7} kg ethylene eq. t^{-1} org. waste	5.33	-	2.78	-	18.1	0.00	3.81	-	28.4	0.00	17.9	-	9.03	0.00	19.3	-
Water EB $X_{2,2}$	AOD $X_{2,2,1}$	10^{-5} kg H^+ eq. t^{-1} org. waste	1,333	-	0.92	-	4,198	-	1.27	-	624	-	0.59	-	262	-	0.64	-
	AqA $X_{2,22}$	10^{-8} kg O_2 eq. t^{-1} org. waste	94.9	-	67.0	-	114	-	92.0	-	876	-	431	-	741	-	467	-
	MEco $X_{2,2,3,1}$	10^{-8} kg Cu eq. t^{-1} org. waste	28.3	-	11.5	-	113	-	15.7	-	120	-	7.37	-	23.5	-	8.0	-
	NMEco $X_{2,2,3,2}$	10^{-8} kg formal. eq. t^{-1} org. waste	18.7	-	7.36	-	62.2	-	10.1	-	73.7	-	4.74	-	13.1	-	5.13	-
	EU $X_{2,2,4}$	10^{-5} kg PO4 eq. t^{-1} org. waste	16.4	-	2.01	-	40.4	-	2.76	-	8.75	-	1.29	-	13.5	-	1.40	-
Land EB $X_{2,3}$	HW $X_{2,3,1}$	kg HW t^{-1} org. waste	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	NHW $X_{2,3,2}$	kg NHW t^{-1} org. waste	-	-	132	95.0	-	-	181	236	-	-	84.8	161	-	-	91.8	180

Table 2.16 Avoided burdens of the WtE plants I_{S1}, I_{S2}, I_{S3}, and I_{S4}.

			AVOIDED BURDENS			
			I_{S1}	**I_{S2}**	**I_{S3}**	**I_{S4}**
			Ga-Ga Thermal treatment and FGT	**Ga-Ga** Thermal treatment and FGT	**Ga-Ga** Thermal treatment and FGT	**Ga-Ga** Thermal treatment and FGT
Air EB $X_{2,1}$	AA $X_{2,1,1}$	10^{-3} kg SO_2 eq. t^{-1} org. waste	-1,038	-1,259	-255	-521
	GWP $X_{2,1,2}$	10^{1} kg CO_2 eq. t^{-1} org. waste	-20.3	-24.6	-5.00	-10.2
	HHE $X_{2,1,3}$	10^{-3} kg benzene eq. t^{-1} org. waste	-39.1	-47.4	-9.62	-19.6
	POF $X_{2,1,4}$	10^{-4} kg CFC-11 eq. t^{-1} org. waste	-746	-906	-184	-375
	SOD $X_{2,1,5}$	10^{-7} kg ethylene eq. t^{-1} org. waste	-363	-440	-893	-1,824
Water EB $X_{2,2}$	AOD $X_{2,2,1}$	10^{-5} kg H^+ eq. t^{-1} org. waste	-19.1	-23	-4.71	-9.62
	AqA $X_{2,22}$	10^{-8} kg O_2 eq. t^{-1} org. waste	-2.93	-3.56	-7.22	-14.7
	MEco $X_{2,2,3,1}$	10^{-8} kg Cu eq. t^{-1} org. waste	-1,086	-1,318	-267	-546
	NMEco $X_{2,2,3,2}$	10^{-8} kg formal. eq. t^{-1} org. waste	-394	-478	-96.9	-198
	EU $X_{2,2,4}$	10^{-5} kg PO4 eq. t^{-1} org. waste	-109	-132	-26.8	-54.7
Land EB $X_{2,3}$	HW $X_{2,3,1}$	kg HW t^{-1} org. waste	-	-	-	-
	NHW $X_{2,3,2}$	kg NHW t^{-1} org. waste	-	-	-	-

The results of Table 2.16 displayed negative values, which come from the avoided electricity production allocated to the Ga-Ga stage. The characterisation factor from the TRACI method (3.13 10^8 kg benzene eq. kg^{-1} PCDD/F) (Bare et al. 2003) was used to incorporate the effect due to the release of dioxins and furans (PCDD/F) in the human health effects (HHE) impact category because it was not originally defined in the reference metrics (IChemE).

In most of the metrics of the air and water compartments, the Cr-Ga step presented the highest total burdens (EB+AB); however, the Ga-Gr stage also contributed significantly; specifically, the burdens were of the same order of magnitude in several categories. In the land compartment only, the Ga-Gr stage generated non-hazardous waste, while the Cr-Ga and Ga-Ga steps were associated with a null contribution.

- **Cradle to Gate**. The production and consumption of reagents for flue gas cleaning and fuels and ancillary materials for the thermal treatment are the most significant contributors to the Cr-Ga stage. Overall, the production and consumption of lime and slaked lime for treatment of acid gases generated high air emissions air of CO_2 and CO which contribute to global warming (GW), whereas the release of H_2SO_4 to water contributed to aquatic acidification (AqA).

  The air emissions of acid gases such as SO_X, H_2SO_4, HCl, and HF, CFCs, and other organic compounds during the manufacture of urea and ammonia contributed to atmospheric acidification (AA), stratospheric ozone depletion (SOD) and photochemical ozone formation (POF). In HHE, although urea production caused high emissions of most of the pollutants contributing to HHE, the production of treated water emitted a larger amount of dust to air, presenting a greater burden to HHE.

  In the water categories, the emission of methanol in the production and consumption of ammonia and urea for NO_X cleaning contributed to aquatic oxygen demand (AOD). Moreover, the release of pollutants such as ammonia, nitrogen, phosphorus, and chemical oxygen demand (COD) to water in ammonia production gave the highest contribution to eutrophication (EU). With respect to ecotoxicity to aquatic life by metals (MEco) and ecotoxicity to aquatic life by non-metal substances (NMEco), the consumption of diesel, lime or slaked lime, and urea contributed the

most due to emissions of Zn and Cu, benzene and xylene to the aquatic media.

- **Gate to Gate.** Although the Ga-Ga analysis linked with waste combustion process had a high impact in the air categories, the avoided burden that was associated with electricity production compensated for these impacts in most of those categories. From an environmental point of view, without any energy recovery, the Ga-Ga would be the worst stage. In the water categories, waste combustion has a null influence because water emissions were not generated in these plants.
- **Gate to Grave.** Ga-Gr burdens are associated with the treatment of ash. Among ash solidification processes, cement production and consumption made the largest contribution. Cement manufacturing is an industry that consumes a lot energy, particularly in the decarbonation and clinkering of raw materials (Margallo et al. 2014b).

For all the WtE plants, the Cr-Ga step displayed the highest total burdens in most of the air categories and in all the water categories. Nevertheless, the Ga-Gr stage showed burdens that were 2.7 and 1.8 times higher than those obtained in the Cr-Ga step in the category of HHE for I_{S1} and I_{S4}, respectively. Moreover, for the latter plant, in the Ga-Gr step a burden 2 times higher than in the Cr-Ga step was observed in SOD. The HHE and SOD burdens in the Ga-Gr analysis are the result of the emissions to air of CFCs, HCFCs, heavy metals, dust, and PCDD/F in the consumption of energy and coke in the clinkering process for cement production.

In contrast to the Ga-Gr analysis, the Ga-Ga analysis showed a burden 53 times higher than in the Cr-Ga step in the HHE category for I_{S2}, whereas for I_{S3}, burdens 1.5 and 13 times higher were obtained in the categories of GW and HHE, respectively. The high impact in the Ga-Ga step in these air categories for I_{S2} and I_{S3} is associated with the emission of greenhouse gases; in particular, NO_X and CO, that contributed to GW and the emissions of PCDD/F, dust, and heavy metals, specifically As and Cd that contributed to HHE. CO_2 emissions were not generated in the organic waste incineration because this waste flow has a null content of fossil carbon.

Emissions of NO_X depend on the applied technology rather than on the waste composition. Therefore, the high emission of nitrogen oxides by these plants can be associated with poor combustion processes and poor NO_X cleaning.

Nitrogen oxides are usually formed during combustion in which part of the nitrogen contained in the MSW is oxidised to NO_X, but it can also be formed during combustion in which a part of the nitrogen in air is oxidised to NO_X (Tillman et al. 1989). NO_X can be reduced using furnace control measures that prevent oversupply of air and prevent the use of unnecessarily high furnace temperatures (Pickens 1996). Moreover, the use of SCR, a more advanced technology than SNCR for reducing NO_X to N_2, could be useful in reducing the environmental impact. However, in this case, it would be necessary to assess whether the production and the periodic maintenance of the catalyst could be environmentally advantageous compared to the non-catalytic system (Morselli et al. 2007). Similarly, emission of dioxins and furans are thought to depend more on operational conditions and treatment technologies related to combustion and flue gases than on the chlorine content of the input waste (Margallo et al. 2014a). High levels of PCDD/F formation are associated with poor combustion conditions, feeding of problematic materials, or the operation of dust collectors at high temperatures (PNUMA 2005). To minimise PCDD/F formation, both good combustion and reduction of the time during which flue gases are subjected to temperatures in the range of 400ºC to 200ºC are required. Particularly, incineration plants should be operated at a temperature of 850ºC for two seconds to achieve good burnout of the gases (EC 2000). Heavy metals and carbon monoxide emissions depend on the input waste composition (i.e., on the heavy metal and carbon content of the waste). Therefore, the release of large amounts of CO and heavy metals to the atmosphere is related to the presence of high levels of carbon and heavy metal in the input waste. These results are consistent with the NRS values and demonstrate that I_{S3} was the incineration plant with the poorest flue gas cleaning system.

To compare the EB to air, water, and land, the threshold values stated in the Regulation of the European Pollutant Release and Transfer Register (E-PRTR) were considered as weighting factors to obtain normalised burdens (Margallo et al. 2014b). The average EB of Spain was calculated with inputs and outputs data collected from the 10 Spanish WtE plants Table 2.17 displays the normalised results and the threshold values proposed in the E-PRTR. The burdens were divided into AB, EB, and total burden. It was observed than in all the WtE plants, land and air were the dominant compartments due to the high influence of the air emissions and solid residues generation in waste combustion.

Table 2.17 Normalised results of the WtE plants and threshold values from the E-PRTR regulation.

			Normalised results (FU^{-1})														
	Theshold value ($kg\ y^{-1}$)	**Factor[2]**	**I_{S1}**			**I_{S2}**			**I_{S3}**			**I_{S4}**			**Spain average**		
			EB	**AB**	**Total burden**	**EB**	**AB**	**Total burden**	**EB**	**AB**	**Total burden**	**EB**	**AB**	**Total burden**	**EB**	**AB**	**Total burden**
Air EB																	
AA $X^*_{2,1,1}$	150,000	10^{-7}	3.29	-69.2	-65.9	8.17	-83.9	-75.8	16.7	-17.0	-0.36	11.0	-34.8	-23.8	7.43	-96.4	-89.0
GW $X^*_{2,1,2}$	100,000,000	10^{-7}	5.50	-20.3	-14.8	9.51	-24.6	-15.1	9.13	-5.00	4.13	4.35	-10.2	-5.85	8.85	-28.3	-19.5
HHE $X^*_{2,1,3}$	1,000	10^{-6}	48.9	-39.1	9.86	328	-47.4	281	54.3	-9.62	44.7	23.9	-19.6	4.26	39.5	-54.5	-15.0
POF $X^*_{2,1,4}$	1,000	10^{-5}	0.57	-7.46	-6.89	1.56	-9.06	-7.50	0.79	-1.84	-1.04	0.45	-3.75	-3.30	0.73	-10.4	-9.68
SOD $X^*_{2,1,5}$	1.00	10^{-6}	0.81	-36.3	-35.5	2.19	-44.0	-41.8	0.46	-8.93	-8.47	0.28	-18.2	-18.0	1.01	-50.6	-49.6
Total EB to air ($X_{2,1}$)		**10^{-6}**			**-103**			**155**			**26.1**			**-49.6**			**-172**
Water EB																	
AOD $X^*_{2,2,1}$	50,000	10^{-8}	26.7	-0.38	26.3	84.0	-0.46	83.5	12,5	-0.09	12.4	5.26	-0.19	5.07	34.7	-0.53	34.2
AqA $X^*_{2,2,2}$	100	10^{-8}	1.62	-0.03	1.59	2.06	-0.04	2.02	1.31	-0.01	1.30	1.21	-0.01	1.19	2.26	-0.04	2.21
MEco $X^*_{2,2,3,1}$	50	10^{-8}	0.80	-21.7	-20.9	2.58	-26.4	-23.8	2.55	-5.35	-2.79	0.63	-10.9	-10.3	1.61	-30.3	-28.7
NMEco $X^*_{2,2,3,2}$	50	10^{-8}	0.52	-7.88	-7.36	1.45	-9.56	-8.11	1.57	-1.94	-0.37	0.36	-3.96	-3.59	0.96	-11.0	-10.0
EU $X^*_{2,2,4}$	5,000	10^{-8}	3.68	-21.8	-18.1	8.63	-26.4	-17.8	2.01	-5.35	-3.35	2.99	-10.9	-7.94	4.25	-30.3	-26.1
Total EB to water ($X_{2,2}$)		**10^{-6}**			**-0.18**			**0.36**			**0.07**			**-0.16**			**-0.28**
Land EB																	
HW $X^*_{2,3,1}$	2,000		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
NHW $X^*_{2,3,2}$	2,000,000	10^{-6}	113	-	113	208	-	208	123	-	123	136	-	136	154	-	154
Total EB to land ($X_{2,3}$)		**10^{-6}**			**113**			**208**			**123**			**136**			**154**
X_2		**10^{-6}**			**10.6**			**364**			**149**			**86.2**			**-18.5**

[2] The "factor" is the multiplied value of normalised results. For instance, the total burden to air of I_{S1} with a factor of 10^{-6} gives as a result a value of -1.03 10^{-4}.

In the air compartment, only HHE showed a positive value in all the plants, indicating that this category had the greatest influence due to the emissions of PCDD/F, heavy metals, and dust that occur during waste combustion.

The analysis of this compartment depicted the highest total burden to air, with positive values for I_2 and I_3, whereas I_4 and I_1 displayed negative values. This was due to the larger amount of energy produced by I_1 and therefore also the higher avoided burden generated, but also to the lower amount of materials consumed by I_4. In particular, in all air categories except HHE, in which a greater value was observed in I_2, I_3 showed the highest impact. This was due to the high emissions of PCDD/F and heavy metals generated by I_2. Nevertheless, the total impact to air ($X_{2,1}$) of I_2 was 6.0 times higher than the burden produced by I_3; thus, I_2 emerged as the plant with the worst air performance.

In the water categories, only AqA and AOD displayed positive values. For I_2 and I_3, AOD contributed the most to the total water burden. Nevertheless, for I_1 and I_4, MEco had the highest influence on the total EB to water ($X_{2,2}$). Specifically, I_1 and I_4 showed negative values in the total EB to water because a lower burden was obtained in several categories in the Cr-Ga and Ga-Gr steps, and in the Ga-Ga analysis thermal treatment did not generate water emissions, thereby contributing only the avoided burden to this step. The largest total water burden was produced by I_2; particularly, this plant presented the highest burdens in AOD and AqA. This was due to the fact that plant I_2 showed the greatest consumption of urea and lime; as previously described, these reagents have a strong influence in these categories. In the remaining categories, I_3 produced the greatest burden; however, the high influence of AOD in $X_{2,2}$ made up for these burdens. Specifically, the total water burden from I_2 was 9 times higher than that of I_3.

Land burden ($X_{2,3}$) had a positive impact in all of the WtE plants because there were no avoided burdens and because wastes were considered as final flows. The highest burden to land was generated by I_2, mainly due to the larger amount of slag and ash generated. In contrast, I_1 showed the best land performance with a burden 1.8 times lower than that produced by I_2.

The EB and NR are classified in Fig 2.13 according to their intensities. Values near the symbol "+" indicate the highest burden, whereas "-" represents the lowest burden. A similar performance in all the environmental compartments

was observed in all the studied plants. In particular, I_{S2} presented the highest burden to air, water, and land, whereas I_{S1} displayed the best environmental performance in all compartments. The comparison of EB and NR emphasises the great interdependence of these variables because the plant with the greatest consumption of NR produced the highest EB.

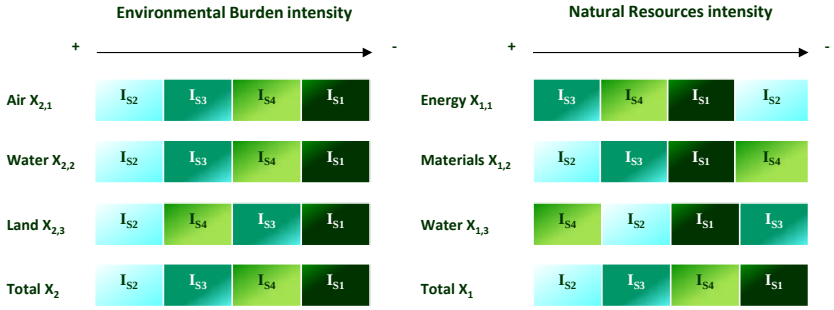


Fig 2.13 Intensity of the environmental burdens and natural resources.

When the normalised values are multiplied by the threshold value of NHW proposed in the E-PRTR regulation -2,000 ton y^{-1}- (Table 2.18), the burdens to air ($X_{2,1}$), water ($X_{2,2}$), and land ($X_{2,3}$), as well as the total burden (X_2), follow the same trend that the results provided in Table 2.17 and Figure 2.13. Regarding land impact, it was observed that all of the WtE plants presented burdens bellow these threshold value. That is to say, that all of the plants had an efficient waste management system. In particular, the comparison of the EB of the WtE plants with the NHW threshold displayed a reduction of 77 %, 59 %, 74 %, and 73 % in the land EB of I_{S1}, I_{S2}, I_{S3}, and I_{S4} respectively.

Table 2.18 Normalised total burdens (dimensionless)

	Normalised total EB (dimensionless)				
	Factor	**I_{S1}**	**I_{S2}**	**I_{S3}**	**I_{S4}**
Total EB to air ($X_{2,1}$)	10^{-3}	-205	310	52.3	-99.3
Total EB to water ($X_{2,2}$)	10^{-3}	-0.37	0.72	0.14	-0.31
Total EB to land ($X_{2,3}$)	10^{-3}	227	416	246	272
Total EB X_2	10^{-3}	21	727	298	172

Finally, to analyse the overall environmental performance of the WtE plants under study, the EB to air ($X_{2,1}$), EB to water ($X_{2,2}$), and EB to land ($X_{2,3}$) are compared to the average EB of Spain in Fig 2.14.

All the plants present a higher EB to air and water than the Spanish average; this is even more notable for I_{S2} and I_{S3}. In the land compartment, only I_{S2} exceeds the burden of Spain, whereas the rest of the plants have values

similar to the Spanish average. Generally, the plants under study show worse environmental performance in the air and water compartments and a lower land impact

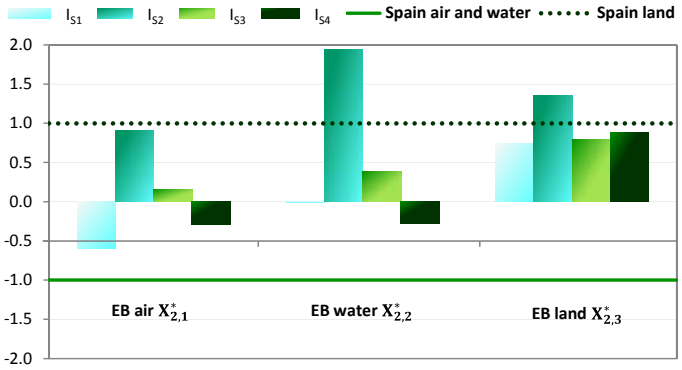


Fig 2.14 Air, water, and land EB of the incineration plants taking as reference the Spanish average EB.

Finally, it is remarkable that corporations with an environmental management system (EMS) could use the EB and NR results to define its environmental objectives and targets, and to evaluate the degree of compliance of these objectives. The proposed indicators facilitate the dissemination of the performance and progress of industrial plants from an environmental point of view. Moreover, on the basis of these metrics, corporations could develop its environmental policy, a report required in EMS that includes commitments to continually improve the environmental performance, to comply with environmental legislation, and to educate and train employees to enable them to work within the policy.

2.4 Environmental assessment of ash treatment

Despite of the mass and volume reduction in waste incineration, this process results in various types of solid wastes, primarily including bottom, boiler, and fly ashes. In municipal waste incinerators, bottom ash (BA) is approximately 10 % by volume and approximately 20 to 30 % by weight of the solid waste input. Fly ash (FA) quantities are much lower, generally only a few percent of input. The proportions of solid residue vary greatly according to the waste type and detailed process design (EC-IPCC 2006).

FAs are fine and are normally characterised by a high content of chlorides and significant amounts of dangerous substances, such as heavy metals or organic compounds. BAs have coarser dimensions, and the amount of chlorides and hazardous chemicals is typically significantly lower than in FA. BAs are commonly subjected to a stabilisation process producing a material with physical and mechanical properties that promote a reduction in the release of contaminants from the residue matrix. These methods use inorganic binder reagents, such as cement, lime, and other pozzolanic materials. However, despite the heavy metal content, the use of BA as a natural aggregate has become increasingly common (Margallo et al. 2013). In particular, in the cement production it was studied the use of BA to replace clinker raw materials (Lam et al. 2011) and clay (Pan et al. 2008), as a supplementary cementitious material mixed with coal fly ash (Li et al. 2012), and as a fine aggregate in the cement mortar manufacture (Saikia et al. 2008). Other studies analysed the production of concrete production with BA and exhausted sand from a FB (Abbà et al. 2014) and with washed BA from a GI (Kuo et al. 2013). Bertolini et al. (2004) assessed the use of bottom and fly ash to replace part of Portland cement to produce concrete. Due to the high mineral content of BA, additional possible management options include frit

production (Barbeiro et al. 2010), the utilisation of BA as a landfill cover material (Puma et al. 2013), and as a solid substitute in embankments (Ma et al. 2007). Likewise, BA is now increasingly used for construction. A practice now commonly observed in Denmark, Belgium and the Netherland is to use BA to repair roads and produce asphalt concrete and permeable pavement (Shih and Ma 2011). The importance of this application is reflected in several studies published over recent years. Specifically, Forteza et al. (2004) characterised BA for its use in a road base in Spain, Toraldo et al. (2013) studied the use of BA in the production of asphalt concrete for road pavement, De Windt et al. (2011) analysed the reuse of BA in a basement of a pilot-scale road, as well as Del Valle-Zermeño et al. (2014) that mixed BA with air pollution control (APC) ashes. Recycling processes have a material and energy consumption associated with the manufacture of the product. Nevertheless, BA recycling avoids waste disposal and the associated impacts, and replaces non-renewable resources (Margallo et al. 2013). To compare the environmental advantages and disadvantages, a life cycle approach is required. Several LCA studies have been conducted to evaluate the environmental performance of BA recycling in road and pavement construction, highlighting the studies developed in Denmark by Birgisdottir et al. (2006 and 2007), in Sweden by Olsson et al. (2006) and in Taiwan by Geng et al. (2010). Other LCAs have compared the environmental impacts of traditional Portland cement and blended cement production with the addition of BA (Huntzinger and Eatmon 2009) and the utilisation of BA as landfill cover (Toller et al. 2009). However, a comparison based on a process-approach between the traditional stabilisation process and the BA recycling process has not been developed. In this sense, this study applies the LCA methodology with a process perspective to evaluate and compare the environmental impacts of ash solidification with cement and ash recycling in Portland cement production. The methodology employed is that proposed in Chapter 2.3, which is based on the use the variables NRS and EBS.

To carry out the LCA study, 57.5 kg of BA were selected as functional unit. The motivation is that from the incineration of one ton of MSW, 57.5 kg of BA were generated. Because the function of the system is to treat ashes, all of the data are realated to this FU. As case study a WtE plant located in the north of Spain was selected. In 2009, 113,338 tons of MSW with a LHV of 2,800 kcal kg^{-1} were incinerated in a roller GI generating 82,800 MWh (AEVERSU 2013).

The system is composed of ash transport and ash treatment. Thermal and flue gas treatment were excluded from the study. These processes are identical in both scenarios (Sc.); thus, the associated impacts are identical and can be neglected in a comparative analysis. Internal transport in the complex, the construction of major capital equipment, and the maintenance and operation of support equipment were excluded from the study. According to Fig 2.15 a) and b) in which the system boundaries are shown, three scenarios were analysed.

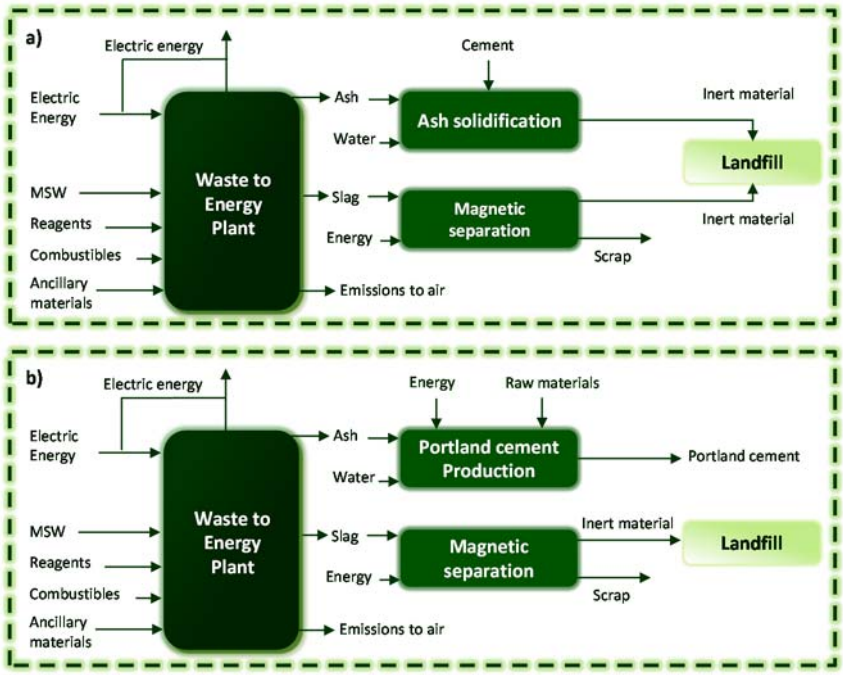


Fig 2.15 System description of a) ash solidification and b) ash recycling in Portland cement production.

Scenario 1: Ash solidification. The BA solidification includes ash transport to a waste manager located 81 km from the WtE plant, the solidification process and ash landfilling. The solidification process employs a mixture of water (30 %), cement (20 %), and ashes (50 %) (Margallo et al. 2013).

Scenario 2: Ash recycling in Portland cement production as a clinker substitute. Portland cement is composed primarily of calcium silicate materials, such as limestone and sand. The raw materials are quarried,

crushed, and milled into a fine powder that is fed into a rotary kiln. The clinker from the kiln is cooled, and gypsum is added (typically 20 %) to regulate the setting time (Margallo et al. 2013). The amount of clinker is reduced using certain materials, such as coal fly ash, slag, and natural pozzolans. The addition of these materials not only reduces the amount of material landfilled but also reduces the amount of clinker required per ton of cement produced. The strength, durability, and life of blended cement using ashes are equivalent to traditional Portland cements with a substitution range of 25-60 %. To ensure an equivalent cement product in this study, a substitution percentage of 25 % by mass was assumed (Huntzinger and Eatmon 2009). This system encompasses ash transport to the cement plant (118 km) and the Portland cement production, which is composed of the burning of raw materials and the grinding of clinker with gypsum and ashes. Additionally, the clinker substitution percentages of 10 % in Sc. 2.1 and 2.5 % in Sc. 2.2 were studied. These systems are identical to Sc. 2, except for the amount of clinker substituted.

Scenario 3: Ash recycling in Portland cement production as a gypsum substitute. The amount of gypsum required to produce Portland cement has changed in recent years because of the replacement of gypsum with natural or industrial pozzolans (Margallo et al. 2013). In this scenario, ashes substituted 25 % of the gypsum.

A system can have multiple functions, and to assign the environmental burden associated with each function, allocations are applied. In Sc. 2, 2.1, 2.2, and 3, ashes are recycled to produce blended cement. Therefore, the system has the following functions: ash treatment and cement production. These multifunctional processes are typically handled through system expansion. To expand the system and subtract the environmental impacts associated with the recovery of recycled materials, the determination of the type of material replacing the recycled material and its equivalence to the virgin material is required. BA replaces clinker in Sc. 2, 2.1, and 2.2, whereas in Sc. 3, BA replaces gypsum in the Portland cement production. Because the properties of traditional Portland cement are equivalent to those of the blended cement (with BA), a substitution factor of one was applied (Margallo et al. 2013).

2.4.1 Life cycle inventory

Table 2.19 gives the LCI of ash solidification (Sc. 1), ash recycling with a clinker substitution of 25 % (Sc. 2), 10 % (Sc. 2.1), and 2.5 % (Sc. 2.2), and with a gypsum substitution of 25 % (Sc. 3). The negative values are associated with an environmental benefit, whereas the positive values indicate a detriment to the environment.

Table 2.19 LCI of Sc. 1 Ash solidification, Sc. 2 Ash recycling (25 % clinker substitution), Sc. 2.1 Ash recycling (10 % clinker substitution), Sc. 2.2 Ash recycling (2.5 % clinker substitution), and Sc. 3 Ash recycling (25 % gypsum substitution).

	INPUT/OUTPUT DATA	Sc. 1	Sc. 2	Sc. 2.1	Sc. 2.2	Sc. 3
	Energy resources (MJ t^{-1} MSW)	**113**	**-177**	**-105**	**-31.2**	**6.01**
INPUTS	**Material resources (kg t^{-1} MSW)**	**117**	**-143**	**-86.9**	**-29.2**	**-11.6**
INPUTS	Gypsum	1.15	-1.04 10^{-4}	-5.49 10^{-5}	1.76 10^{-3}	-11.5
INPUTS	Inert rock	36.7	-65.4	-39.8	-13.4	-9.41 10^{-2}
INPUTS	Limestone	25.8	-46.0	-28.0	-9.43	-9.49 10^{-2}
INPUTS	Sand	3.45	-6.16	-3.75	-1.26	-1.29 10^{-2}
INPUTS	Water	39.0	7.68	-4.66	-1.54	3.13 10^{-2}
INPUTS	Air	10.8	-17.7[1]	-10.7	-3.57	2.36 10^{-2}
OUTPUTS	**Emissions to air (kg t^{-1} MSW)**	**34.6**	**-59.6**	**-36.1**	**-11.8**	**3.78 10^{-1}**
OUTPUTS	Vanadium (+III)	1.91 10^{-6}	-1.73 10^{-6}	-8.85 10^{-7}	-8.9210^{-9}	4.31 10^{-7}
OUTPUTS	Carbon dioxide	20.9	-35.7	-21.6	-7.00	3.39 10^{-1}
OUTPUTS	Carbon monoxide	5.05 10^{-2}	-8.68 10^{-2}	-5.26 10^{-2}	-1.74 10^{-2}	3.19 10^{-4}
OUTPUTS	Nitrogen (N-compounds)	5.89 10^{-2}	-1.05 10^{-1}	-6.40 10^{-2}	-2.16 10^{-2}	-2.19 10^{-4}
OUTPUTS	Nitrogen oxides	1.42 10^{-2}	-1.38 10^{-2}	-7.55 10^{-3}	-1.09 10^{-3}	2.16 10^{-3}
OUTPUTS	Steam	4.95	-8.66	-5.26	-1.76	3.57 10^{-3}
OUTPUTS	Sulphur dioxide	7.55 10^{-2}	-1.34 10^{-1}	-8.14 10^{-2}	-2.73 10^{-2}	-6.78 10^{-5}
OUTPUTS	Dioxins	1.10 10^{-5}	-1.97 10^{-5}	-1.20 10^{-5}	-4.04 10^{-6}	-4.12 10^{-8}
OUTPUTS	NMVOC	1.00 10^{-3}	-7.03 10^{-4}	-3.57 10^{-4}	-5.08 10^{-7}	-1.79 10^{-4}
OUTPUTS	Exhaust	8.47	-1.47 10^{1}	-8.96	-2.99	1.26 10^{-2}
OUTPUTS	Dust (PM_{10})	4.06 10^{-3}	-6.84 10^{-3}	-4.17 10^{-3}	-1.40 10^{-3}	-1.19 10^{-5}
OUTPUTS	**Emissions to fresh water (kg t^{-1} MSW)**	**2.74 10^{-2}**	**-4.10 10^{-2}**	**-2.42 10^{-2}**	**-6.93 10^{-3}**	**1.76 10^{-3}**
OUTPUTS	Chemical oxygen demand	8.98 10^{-4}	-8.74 10^{-4}	-5.23 10^{-4}	-1.60 10^{-4}	2.17 10^{-5}
OUTPUTS	Iron	3.84 10^{-4}	-6.59 10^{-4}	-3.99 10^{-4}	-1.30 10^{-4}	5.03 10^{-6}
OUTPUTS	Chloride	1.79 10^{-2}	-3.00 10^{-2}	-1.81 10^{-2}	-5.79 10^{-3}	3.96 10^{-4}
OUTPUTS	Fluoride	5.54 10^{-4}	-9.63 10^{-4}	-5.84 10^{-4}	-1.92 10^{-4}	4.39 10^{-6}
OUTPUTS	Sulphate	2.62 10^{-3}	-4.53 10^{-3}	-2.74 10^{-3}	-8.98 10^{-4}	2.87 10^{-5}
OUTPUTS	Solids (suspended)	3.79 10^{-3}	-1.95 10^{-3}	-6.96 10^{-4}	5.99 10^{-4}	1.25 10^{-3}
OUTPUTS	**Emissions to sea water (kg t^{-1} MSW)**	**7.46 10^{-3}**	**-6.48 10^{-3}**	**-3.25 10^{-3}**	**8.74 10^{-5}**	**1.76 10^{-3}**
OUTPUTS	Chemical oxygen demand	1.22 10^{-5}	-6.98 10^{-6}	-2.74 10^{-6}	1.63 10^{-6}	3.82 10^{-6}
OUTPUTS	Strontium	8.86 10^{-7}	2.94 10^{-7}	3.71 10^{-7}	4.50 10^{-7}	4.90 10^{-7}
OUTPUTS	Zinc (+II)	3.45 10^{-7}	-2.87 10^{-7}	-1.41 10^{-7}	9.16 10^{-9}	8.47 10^{-8}
OUTPUTS	Chloride	6.84 10^{-3}	-6.27 10^{-3}	-3.21 10^{-3}	-5.63 10^{-5}	1.53 10^{-3}
OUTPUTS	Solids (suspended)	4.39 10^{-4}	-6.92 10^{-5}	3.10 10^{-5}	1.34 10^{-4}	1.86 10^{-4}
OUTPUTS	**Emissions to industrial soil (kg t^{-1} MSW)**	**4.80 10^{-5}**	**-6.33 10^{-5}**	**-3.63 10^{-5}**	**-8.39 10^{-6}**	**5.62 10^{-6}**

The WtE plant data were provided by Ecoembes (Ecoembes 2014), AEVERSU (AEVERSU 2013), the Spanish Pollutant Release Transfer Register PRTR (PRTR 2012), the IPPC permit of the plant, the WtE plant, and bibliographic data (Margallo et al. 2013). The ash solidification data were collected from Doka (2003), and the Portland cement production was based on the BREF for the production of cement, lime and magnesium oxide manufacturing industries (EC-IPPC 2010), and the Spanish Handbook on the Best Available Techniques (BAT) for cement manufacturing (BAT 2004).

2.4.2 Life cycle impact assessment

The LCIA was conducted following the ISO 14040 (ISO 2006a) and ISO 14044 requirements (ISO 2006b), with the LCA software GaBi 4 (PE International 2011) and the environmental sustainability metrics developed by IChemE (IChemE 2002). The results were divided according to the developed ESA methodology, into the NRS and EBS.

A) NATURAL RESOURCES SUSTAINABILITY

NR include energy, material and water consumption. Fig 2.16 shows that a higher NR consumption was obtained in the solidification process (Sc. 1) than in all of the recycling scenarios (Sc. 2, 2.1, 2.2, and 3). This was due to the significant NR consumption during the solidification process in Sc. 1, and the reduction in the amount of required clinker in the cement production in Sc. 2, 2.1, and 2.2. The reduction in the amount of clinker lowers the NR consumption, thus lowering the emissions. Negative NR consumption values were obtained in cement manufacturing (avoided burden), displaying only positive values for ash transport. For the consumption of NR in Sc. 1, 95 % of the energy was consumed in the production of cement, which is used in the solidification process. Cement manufacturing is a high energy-consuming industry, focusing the energy consumption in the decarbonation and clinkering of raw materials (BAT 2004, EC-IPPC 2010). The comparison of Sc. 1 and 2 (clinker substitution of 25 %) displayed a relative change of 373 %, 380 %, and 120 % in energy, material and water consumption, respectively. The reduction in the percentage of clinker substitution from 25 % (Sc. 2) to 10% (Sc. 2.1) produced a 40 % increase in the NR consumption. This increase was even more noticeable (80 %) when the clinker substitution decreased to 2.5 % (Sc. 2.2). Therefore, a higher clinker substitution indicates a lower NR

consumption. Sc. 3 (25 % gypsum substitution) also showed a lower NR consumption than Sc. 1, with a reduction of 95 %, 85 % and 99 % in the consumption of energy, material and water, respectively. Nevertheless, higher consumptions resulted in Sc. 3 than in Sc. 2, 2.1, and 2.2 because the amount of clinker in this blended cement is identical to that required in the conventional cement; therefore, there is no consumption savings associated with clinker production. The NR savings were only afforded by the amount of gypsum substituted.

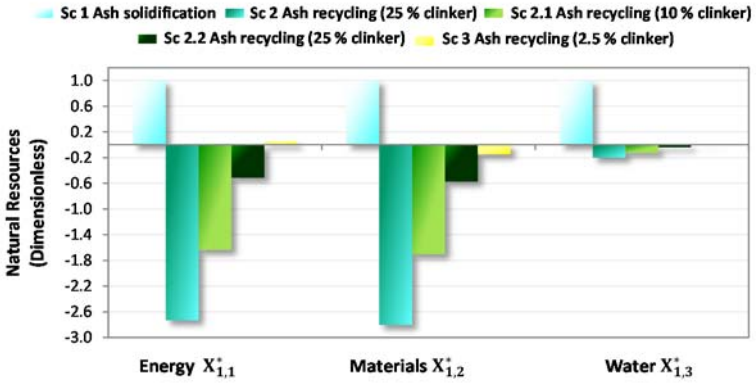


Fig 2.16 Natural resources sustainability (NRS) of the studied scenarios.

B) ENVIRONMENTAL BURDENS SUSTAINABILITY

Table 2.20 shows the EB values and the normalised results of Sc. 1, 2, and 3. Before the normalisation, Global Warming (GW) displayed the highest air impact in all of the scenarios due to the emission of greenhouse gases in the clinker production (CO_2, CO, VOC), the consumption of coke and energy in the clinkering of raw materials (CH_4, CO, CO_2, NO_X, N_2O), the diesel consumption and the landfill emissions (NO_X, N_2O), and the BA transport (NO_X, N_2O). Ecotoxicity to aquatic life (organics) (NMEco) presented the highest contribution to the water impact in all of the scenarios due to the seawater emissions of ammonia, chloride, benzene, toluene and xylenes in the clinker production, the coke and energy consumption in the clinkering of raw materials and the diesel consumption in the transport and landfilling of BA. After the normalisation, Photochemical Ozone Formation (POF) had the highest contribution in the air categories, whereas NMEco continued as the highest contributor to the water impact. The primary reason for this result

was that although GW had the highest air impact, when the EB was referenced to the threshold value (100,000,000), the normalised results were reduced by 5 orders of magnitude. Moreover, although POF had a lower EB than GW, a lower threshold value (1,000) was used as the reference value. For the water impacts, there were no significant differences after normalisation because the threshold values are lower than in the air categories.

Table 2.20 Environmental burdens and the normalised results of Sc. 1 ash solidification, Sc. 2 ash recycling (25 % clinker substitution), and Sc. 3 ash recycling (25 % gypsum substitution).

	Environmental Burdens (EB)			**Normalised results (FU^{-1})**		
	Sc. 1	**Sc. 2**	**Sc. 3**	**Sc. 1**	**Sc. 2**	**Sc. 3**
EB to air						
AA (kg SO_2 eq.)	$7.57\ 10^{-2}$	$-1.34\ 10^{-1}$	$-6.18\ 10^{-5}$	$5.04\ 10^{-7}$	$-8.95\ 10^{-7}$	$3.63\ 10^{-16}$
GW (kg CO_2 eq.)	22.0	-37.1	$4.38\ 10^{-1}$	$2.20\ 10^{-7}$	$-3.71\ 10^{-7}$	$4.38\ 10^{-9}$
SOD (kg CFC11 eq.)	$3.44\ 10^{-7}$	$-6.10\ 10^{-7}$	$-2.36\ 10^{-10}$	$3.44\ 10^{-7}$	$-6.10\ 10^{-7}$	$-2.36\ 10^{-10}$
HHE (kg benzene eq.)	$2.10\ 10^{-3}$	$-3.50\ 10^{-3}$	$9.75\ 10^{-7}$	$2.10\ 10^{-6}$	$-3.50\ 10^{-6}$	$9.75\ 10^{-10}$
POF (kg ethylene eq.)	$5.51\ 10^{-3}$	$-9.52\ 10^{-3}$	$5.10\ 10^{-5}$	$5.51\ 10^{-6}$	$-9.52\ 10^{-6}$	$5.10\ 10^{-8}$
EB to water						
AOD (kg O_2 eq.)	$8.95\ 10^{-6}$	$-1.41\ 10^{-5}$	$3.41\ 10^{-7}$	$1.79\ 10^{-10}$	$-2.82\ 10^{-10}$	$6.83\ 10^{-12}$
AqA (kg H^+ eq.)	$1.02\ 10^{-9}$	$-1.47\ 10^{-9}$	$5.44\ 10^{-11}$	$1.02\ 10^{-11}$	$-1.47\ 10^{-11}$	$-6.18\ 10^{-7}$
NMEco (kg formal. eq)	$3.42\ 10^{-3}$	$-3.14\ 10^{-3}$	$7.65\ 10^{-4}$	$6.84\ 10^{-5}$	$-6.27\ 10^{-5}$	$1.53\ 10^{-5}$
MEco (kg Cu eq.)	$4.16\ 10^{-7}$	$-4.35\ 10^{-7}$	$7.94\ 10^{-8}$	$8.32\ 10^{-9}$	$-8.69\ 10^{-9}$	$1.59\ 10^{-9}$
EU kg (PO_4 eq.)	$2.04\ 10^{-5}$	$-1.97\ 10^{-5}$	$6.61\ 10^{-7}$	$4.08\ 10^{-9}$	$-3.94\ 10^{-9}$	$1.32\ 10^{-10}$

A comparison of the scenarios showed that the ash recycling scenarios (Sc. 2 and 3) had a lower impact than the ash solidification scenario (Sc. 1) in all of the environmental categories. This result was consistent with the NR results because lower NR consumption and emissions result in a lower EB. The primary reason for these results is that although the BA recycling (Sc. 2 and 3) obtained a high impact in the blended cement manufacturing process and a higher transport impact than ash solidification (Sc. 1), the avoided burden associated with BA recycling compensated for these impacts. However, in Sc. 1, cement production for BA solidification had the highest influence in all of the categories. In particular, solidification contributed to the EB to air over a range of 94 % to 99 %. For the water EB, although cement manufacturing is the process with the highest impact, the transport impact reached values of 31 % and 36 % in NMEco and MEco, respectively, and water consumption in the solidification process contributed 39 % to the eutrophication. The normalised results are grouped in Fig 2.17 into two impacts, EB to air and EB to water.

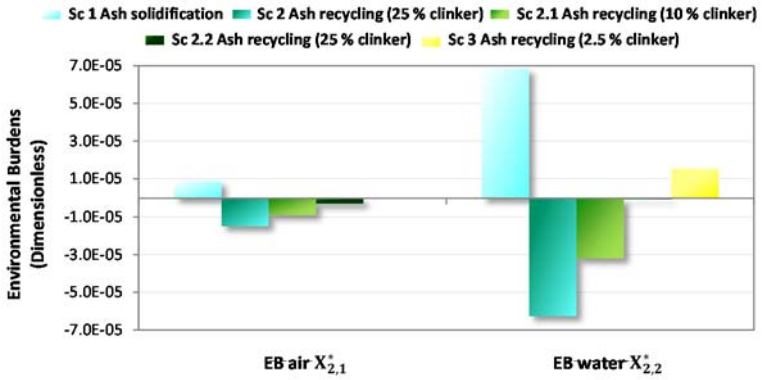


Fig 2.17 Air and water EBs of Sc. 1 ash solidification, Sc. 2 ash recycling (25 % clinker substitution), Sc. 2.1 ash recycling (10 % clinker substitution), Sc. 2.2 ash recycling (2.5 % clinker substitution), and Sc. 3 ash recycling (25 % gypsum substitution).

The highest air and water EBs were observed in Sc. 1. Of the recycling scenarios, Sc. 3 presented the highest EBs to air and water, whereas Sc. 2 displayed the lowest air and water impacts. Finally, the reduction in the amount of substituted clinker reduced the air and water EBs.

2.4.3 Semi-quantitative economic analysis

The comparison of the different ash treatment options was carried out in terms of the associated environmental impacts. However, it is also recommended from a sustainable development point of view, to include a discussion of the different scenarios in economic terms. In this sense, Table 2.21 displays a semi-quantitative economic analysis performed with the Gabi 4 LCA software (PE International 2011). The symbols "+", "-", and "0" indicate a positive influence (i.e., an economic cost), a negative influence (i.e., an economic benefit), and a low or null influence, respectively. The following three cost variables were evaluated: transport (X_1), raw materials (X_2), and landfill taxes (X_3). The variable X_1 represents the cost of the ash transport to the landfill (Sc. 1) or to the cement plant (Sc. 2, 2.1, 2.2, and 3). Therefore, the distance from the WtE plant to the cement plant or to the landfill has a significant influence on the transport cost, requiring further evaluation. The variable X_2 provides the cost of the raw materials, primarily water, cement, and diesel, whereas X_3 displays the landfill taxes (around 30 € t^{-1} stabilised waste). In Sc. 1, the landfill taxes (X_3) and the raw materials (X_2) presented the

highest costs. For the variable X_2, cement consumption in the solidification process had the highest influence. In Sc. 2, 2.1, and 2.2, the cost was associated with the ash transport from the WtE plant to the cement plant (X_1). Nevertheless, in these scenarios, the variable X_2 assumed an economic benefit because ash recycling reduced the amount of required raw materials, thus reducing the costs. A comparison of Sc. 1 with Sc. 2, 2.1, and 2.2 displayed an economic benefit of 125 %, 114 %, and 102 %, respectively. Percentages higher than 100 % indicate that the process generates an economic benefit instead of a cost. In particular, the results showed that the higher amount of clinker substituted, the higher the economic benefit. Similar to Sc. 2, 2.1, and 2.2, ash transport (X_1) was the primary cost in Sc. 3, with the raw material cost (X_2) exerting a very low influence because in the recycling process, ash is only replacing the gypsum. A cost reduction of 97 % was obtained in Sc. 3.

Table 2.21 Semi-quantitative economic analysis of Sc. 1, Sc. 2, Sc. 2.2, and Sc. 3.

Cost variables	Sc. 1	Sc. 2	Sc. 2.1	Sc. 2.2	Sc. 3
X_1 (transport)	0	+	+	+	+
X_2 (raw material)	+	-	-	-	-
X_3 (landfill taxes)	+	0	0	0	0
Economic benefit related to Sc. 1		125 %	114 %	102 %	97 %

2.4.4 Distance evaluation

The distance from the WtE plant to the cement facility is an important parameter in BA recycling. To evaluate the influence of this distance, the EBs of ash solidification (Sc. 1), BA recycling with a 2.5 % clinker substitution (Sc. 2.2), and with a 25 % gypsum substitution (Sc. 3) were compared, considering several distances (Fig 2.18).

The negative values in Fig 2.18 are associated with an environmental benefit. The EB to air in Sc. 1 was higher than that in Sc. 2.2 and Sc. 3, even though the distance reached 9,000 km. With regard to the individual air impacts, for atmospheric acidification (AA), stratospheric ozone depletion (SOD), human health effects (HHE) and POF, although the distance increased, the impact of Sc.1 remained higher than in the ash recycling scenarios. Only for GW, ash recycling reached a higher air impact than Sc. 1 for distances up to 6,800 km in Sc. 2.2 and up to 5,050 km in Sc. 3. In the base case (118 km), the water impacts were higher in all of the categories for Sc. 1. However, the distance

variation reversed this situation. When the distance exceeded 651 km in Sc. 2.2 and 523 km in Sc. 3, ash solidification presented a lower EB to water. In particular, for AqA, when the distance surpassed 2,096 km in Sc. 3 and 3,000 km in Sc. 2.2, Sc. 1 displayed reduced impacts. The same results were obtained for distances up to 2,851(Sc. 3) and 3,790 km (Sc. 2.2) for AOD; 588 (Sc. 3) and 762 km (Sc. 2.2) for MEco; 520 (Sc. 3) and 644 km (Sc. 2.2) for NMECO, and 3,430 (Sc. 3) and 4,123 km (Sc. 2.2) for EU.

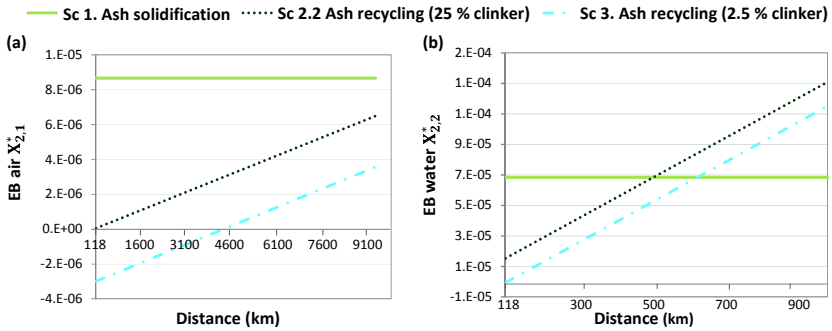
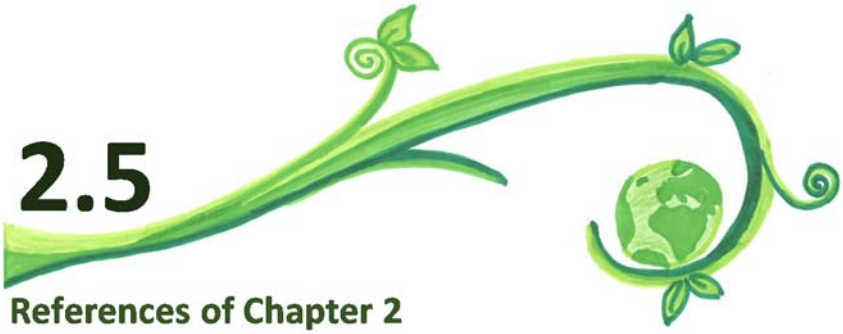


Fig 2.18 Evolution of the a) EB to air and b) EB to water of Sc. 1, Sc. 2.2, and Sc. 3 as a function of distance.



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3. CONCLUSIONES

Conclusions

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Conclusiones

Esta tesis aborda la evaluación cuantitativa de la sostenibilidad ambiental de la incineración de residuos municipales en España y Portugal. En concreto, se ha analizado la situación actual de la tecnología en estos países con el fin de desarrollar y aplicar posteriormente un modelo de ciclo de vida representativo. Las conclusiones más significativas derivadas de este trabajo se resumen en:

i. **Se ha recogido el inventario de ciclo de vida (ICV) de la de la tecnología de incineración de residuos municipales en la península ibérica**. El ICV desarrollado ha sido incluido en una base de datos específica de procesos de tratamiento y reciclaje de residuos dentro del marco del programa LIFE de la Comisión Europea, la cual es compatible con la base de datos internacional de ciclo de vida (ILCD).

ii. **Se desarrollado un modelo multifuncional que describe el proceso actual de incineración de residuos municipales en España y Portugal.** La entrada principal del modelo es la masa total de residuos municipales, así como su composición a través de sus 18 fracciones, lo que permite analizar la influencia de la composición de los residuos en el ICV. En dicho modelo se ha determinado que **los factores críticos de los que depende el ICV y el tipo de asignación de carga son la masa, la composición y el poder calorífico de cada fracción de residuo**. Por lo tanto, las cargas ambientales asociadas a cada una de estas fracciones dependerán de estos parámetros. **El modelo desarrollado se ha implementado en un software, financiado y apoyado por la Comisión Europea, para evaluar diferentes escenarios de gestión de residuos de envase, permitiendo determinar y seleccionar el sistema de gestión de residuos**

ambientalmente más sostenible.

iii. **Se ha desarrollado un procedimiento para evaluar cuantitativamente la sostenibilidad ambiental de la incineración de residuos municipales, el cual permite reducir la complejidad de los resultados del análisis de ciclo vida (ACV) y simplificar la toma de decisiones**. Esta metodología se basa en la cuantificación del uso de los recursos naturales (energía, materiales y agua) y en las cargas ambientales generadas en los diferentes compartimentos ambientales (impactos al aire, agua y suelo). Con el fin de reducir el número de indicadores, **se ha propuesto un procedimiento de normalización y ponderación** basado en los valores umbrales del E-PRTR, **reduciéndose así los tres indicadores de recursos naturales y los 12 de cargas ambientales a dos únicos índices: sostenibilidad de recursos naturales (NRS) y sostenibilidad de cargas ambientales (EBS)**. Estas dos variables pueden ser utilizadas para establecer una función multi-objetivo en la que la sostenibilidad ambiental queda definida mediante dos índices.

iv. **El modelo de ciclo de vida y el procedimiento de evaluación de la sostenibilidad ambiental desarrollados fueron aplicados al estudio de diferentes plantas de incineración de residuos municipales en España**. En términos de uso de recursos naturales, **las plantas presentaron un indicador global de consumo de recursos materiales (NRS) que varía entre 0,33 y 0,62; superando tres de las plantas el valor medio de consumo de materiales de España (0,33). De los resultados obtenidos se concluye que, es importante controlar el consumo de agua y mejorar la eficiencia de la producción de energía en las plantas españolas con el fin de aproximarse a las mejores técnicas disponibles**. En términos de cargas ambientales, las plantas estudiadas presentaron un peor comportamiento ambiental al aire y al agua respecto al valor medio de España, mientras que las cargas al suelo fueron similares a la media española. En particular, respecto al suelo, la comparación con el valor umbral del E-PRTR para residuos no peligrosos (2.000 t año^{-1}) muestra una reducción del 77 %, 59 %, 74 % y 73 % en el impacto al suelo de las plantas estudiadas respectivamente. **En relación al índice global de cargas ambientales**, **las plantas de incineración de residuos municipales en España mostraron un valor comprendido entre 21 y 172, superando la carga ambiental media de España.** Por lo tanto, se puede concluir por un lado, que **las plantas tienen un sistema de gestión de residuos eficaz,** y por otro lado, que **para**

minimizar las cargas ambientales, se deben optimizar las condiciones de operación, así como mejorar la eficiencia del sistema de tratamiento de gases.

v. **Se ha comparado y evaluado la sostenibilidad ambiental de la solidificación de las cenizas generadas en la combustión de residuos municipales con el reciclaje de las mismas en la producción de cemento Portland**. **La solidificación se presenta como la opción de gestión de cenizas menos favorable desde un punto de vista ambiental y económico** debido al consumo de cemento y la tasas de vertido. De las alternativas de reciclaje, **el escenario con un mayor porcentaje de sustitución de clínker (25 %) es el más beneficioso ambientalmente** ya que implica un mayor ahorro de recursos y menor carga ambiental, mientras que el escenario que presenta peores resultados es aquel en el que las cenizas sustituyeron al yeso. En concreto, **cuando las cenizas sustituyeron al clinker y yeso se observaron disminuciones del indicador NRS superiores al 100 % respecto a la solidificación, mientras que para el índice EBS las reducciones fueron superiores al 100 % en la sustitución de clinker, y del 80 % en el caso del yeso.**

Progreso de la investigación

Esta Tesis pretende contribuir al desarrollo de un modelo representativo de la incineración de residuos municipales en la península ibérica, así como, reducir la complejidad de los resultados de un estudio de ACV mediante una metodología de evaluación de la sostenibilidad ambiental que ayude al proceso de la toma de decisiones. Sin embargo, a pesar de los avances logrados, hay nuevos retos que deben superarse para mejorar la presente investigación:

i. Inclusión en la metodología de evaluación de la sostenibilidad ambiental de factores sociales y económicos, con el fin de poder evaluar los procesos utilizando los tres pilares fundamentales de la sostenibilidad. Por otro lado, sería interesante aplicar un análisis multi-criterio para definir nuevos factores de ponderación de manera que se pueda determinar la influencia de las diferentes cargas ambientales.

ii. Adaptación del modelo de ciclo de vida a fin de comparar las diferentes tecnologías de tratamiento térmico y de gases de acuerdo a las mejores técnicas disponibles recogidas en el correspondiente documento BREF.

iii. Estudio de alternativas para la gestión ambiental de cenizas generadas en las plantas de incineración de residuos municipales. Contribución técnica para el desarrollo e implementación de políticas reguladoras del reciclaje de estos residuos.

iv. Estudio de la posible utilización del material férrico separado magnéticamente de las escorias. En concreto, de la utilización de este material en la producción de acero.

Conclusions

This Thesis deals with the quantitative environmental sustainability assessment (ESA) of WtE technologies in Spain and Portugal. Particularly, the current diagnosis of the technology in these countries was analysed to develop a representative life cycle model, which was further applied. The main conclusions derived from this work are reported below:

i. **The life cycle inventory (LCI) of municipal solid waste incineration (MSWI) in the Iberian Peninsula was compiled**. This LCI is part of a specific database on waste treatment processes and recycling technologies within the framework of the LIFE program of the European Commission. This database is compatible with the International Life Cycle Database (ILCD).

ii. **A multifunctional model that describes WtE process in Spain and Portugal was developed.** The main inputs of the model are the total mass of MSW, as well as their composition through the 18 waste fractions, which allow to analyse the influence of waste composition in the LCI. **In this model, the mass, composition, moisture, and heating value of the input waste are critical factors in determining the LCI and the type of allocation applied**. Therefore, the environmental burdens associated with each waste fraction will depend on these factors. **The developed model was included in an ad-hoc software tool, financed and supported by the European Commission, for evaluating alternative scenarios for the management of post-consumer packaging waste, enabling the selection of more environmental sustainable waste management practices.**

iii. **A quantitative environmental sustainability assessment (ESA) procedure was developed to evaluate MSWI. This procedure allows to reduce the**

complexity of LCA and simplify decision-making process. The methodology is based on the quantification of the use natural resources (energy, materials, and water) and the environmental burdens to specific environmental compartments (impact to air, water, and land). In order to minimise the number of variables, a **normalisation and weighting procedure** based on the threshold values of the E-PRTR **was proposed, reducing the three NRS and 12 EBS into two indexes: natural resources sustainability (NRS) and environmental burdens sustainability (EBS)**. These two variables can feed a multi-objective function in which environmental sustainability was described by means of the two indexes.

iv. **The life cycle model and the ESA procedure were applied to several WtE plants in Spain to assess and compare the environmental performance of the incinerators**. Regarding NR, **the plants under study presented a global index of natural resources (NRS) that ranges from 0.33 to 0.62, exceeding three plants the Spanish average consumption.** Therefore, to **improve the environmental performance of the Spanish plants, the control of water consumption and the improvement of energy production is essential to be as close as possible to the best available techniques (BAT).** Regarding the Spanish EB, a worst environmental performance in air and water compartments was observed in the studied plants, whereas land burden in most of the plants was similar to the average Spanish EB. The comparison of the EB to land of the WtE plants with the threshold value of non-hazardous waste (NHW) proposed in the E-PRTR regulation (2,000 ton $y^{-1)}$) displayed a reduction of 77 %, 59 %, 74 %, and 73 % in the land burden of the incinerators respectively. Specifically, **the plants showed a global index of EBS that ranged from 21 to 172, surpassing the average EBS of Spain.** Consequently, it can be concluded that on one hand, **all of the plants had an efficient waste management system,** and on the other hand, that **to minimise air and water EB, it is necessary to optimise the operational conditions and efficiency of flue gases treatment**.

v. **The ESA of BA solidification and recycling in Portland cement production were analysed and compared.** In this regard, **ash solidification shapes up as the least favourable treatment from an environmental and economic point of view** due to the landfill taxes and the great costs associated to cement consumption. From all of the recycling alternatives, **the higher**

clinker substitution (25 %) is presented as the best option because the higher the NR savings, the lower EB. The worst results from all of the recycling options, were obtained in the scenario in which ashes substituted gypsum. **In particular, reductions higher than 100 % were obtained in the EBS index when BA solidification is compared with ash reclining substituting clinker and gypsum. For EBS reductions greater than 100 % were observed for clinker substitution, whereas for gypsum substitution the values were close to 80 %**.

This Thesis intents to contribute to the development of a representative life cycle model of waste to energy technologies in Spain and Portugal. Likewise, this research expects to reduce the complexity of LCA results by means of an environmental sustainability assessment methodology that help the decision making process. However, despite the achievements described in the Thesis, there are still new challenges ahead that must be overcome to improve the present research:

i. Inclusion of social and economic metrics in the ESA methodology to provide the decision-making process with the three pillars of sustainable development. On the other hand, it would be interesting to improve the methodology by considering several weighting factors so as to determine the influence of the different environmental burdens.

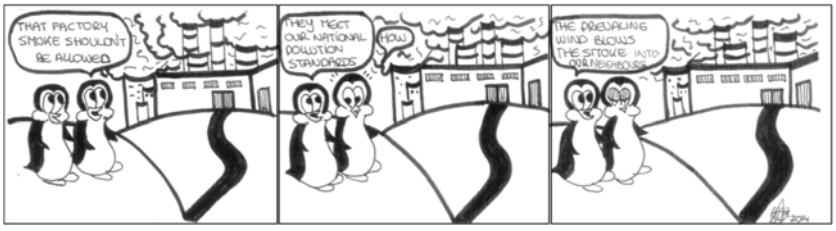
ii. Adjustment of the life cycle model allowing the comparison across different technologies of thermal treatment and flue gases cleaning according to the BAT contained in the BREF document.

iii. Study of several environmental management options of MSWI ash. Technical contribution to the development and implementation of regulatory policies of ash recycling.

iv. Study of second-life use of scrap. In particular, environmental assessment of the use of this ferric material in steel production.

4. ARTÍCULOS CIENTÍFICOS

Scientific articles



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4.1 Margallo M, Aldaco R, Bala A, Fullana P, Irabien A (2012) Best available techniques in municipal Solid waste incineration: State of the art in Spain and Portugal. Chem Eng Trans 29, 1345-50.

Resumen

En el año 2010 se generaron en España y Portugal más de 24 Mt y 5 Mt de residuos municipales respectivamente. Los tratamientos más comunes para este tipo de residuos son el vertido, la incineración y el reciclaje. En 2010, en la península ibérica, entre el 58 y 62 % de los residuos municipales fueron enviados a vertedero, entre 9 y 19 % fueron incinerados y el resto de residuos fueron reciclados o tratados mediante compostaje. A pesar de que el vertido es actualmente la práctica más habitual, cada vez es más común el tratamiento de residuos municipales mediante incineración. Las principales ventajas de esta tecnología son la reducción de la masa y el volumen de residuos y la recuperación de energía. Sin embargo, la incineración se ha ganado una mala reputación, principalmente debido al impacto ambiental generado por las emisiones de gases ácidos, dioxinas y furanos y gases de efecto invernadero. Para evaluar las ventajas y desventajas asociadas a esta tecnología, así como sus potenciales impactos ambientales, se requiere utilizar una perspectiva de ciclo de vida. Dentro de este marco, se ha desarrollado el proyecto FENIX-Giving Packaging a New Life, un proyecto europeo LIFE+ con una duración de tres años. Este trabajo presenta la primera etapa de este proyecto, en el cual se desarrollará en el futuro una base de datos y un modelo basado en el análisis de ciclo de vida (ACV) para evaluar los impactos ambientales del proceso de incineración en España y Portugal. En particular, el objetivo de este trabajo es revisar las diferentes tecnologías de incineración de residuos municipales y determinar la diagnosis actual de la tecnología en España y Portugal, así como recoger todos los datos necesarios para construir el inventario de ciclo de vida (ICV).

Original abstract

In the year 2010 more than 24 Mt and 5 Mt of municipal solid waste (MSW) were generated in Spain and Portugal. Landfilling, incineration and recycling are the most common treatments. In 2010, in the Iberian Peninsula between 58-62 % of the MSW generated was sent to the landfill, 9-19 % was incinerated and the rest was recycled and composting. Despite landfilling is still the most common practice, waste treatment by means of an incineration process has increased. The main advantages of this type of waste treatment are the reduction of mass and volume of residues and the energy recovery. Nevertheless, incineration had gained a bad reputation owing to the environmental impact, in particular, due to the emissions of acid gases, dioxins and furans (PCDD/F) and greenhouse gases. To assess the environmental advantages and disadvantages as well as the potential environmental impacts of waste incineration a life cycle perspective is required. Within this framework is the development of FENIX-

Giving Packaging a New Life, a 3-year European LIFE+ funded project. This work is just the first step within this project where a database and a model based on life cycle assessment (LCA) to assess the environmental impacts of waste incineration in Spain and Portugal will be developed. Particularly, the aim of this paper is to review the different technologies applied to MSW solid waste incineration and to carry out both the diagnosis of the current situation at the incineration plants in Spain and Portugal and to collect data to develop the life cycle inventory (LCI).

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Best Available Techniques in Municipal Solid Waste Incineration: State of the Art in Spain and Portugal

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In the year 2010 more than 24 Mt and 5 Mt of Municipal Solid Waste (MSW) were generated in Spain and Portugal. Landfilling, incineration and recycling are the most common treatments. In 2010, in the Iberian Peninsula between 58-6 2 % of the MSW generated was sent to the landfill, 9-19 % was incinerated and the rest was recycled and composting (EUROSTAT, 2010). Despite landfilling is still the most common practice, waste treatment by means of an incineration process has increased. The main advantages of this type of waste treatment are the reduction of mass and volume of residues and the energy recovery. Nevertheless, incineration had gained a bad reputation owing to the environmental impact, in particular, due to the emissions of acid gases, dioxins and furans (PCDD/F) and greenhouse gases. To assess the environmental advantages and disadvantages as well as the potential environmental impacts of waste incineration a life cycle perspective is required. Within this framework is the development of FENIX-Giving Packaging a New Life, a 3-year European LIFE+ funded project. This work is just the first step within this project where a database and a model based on Life Cycle Assessment (LCA) to assess the environmental impacts of waste incineration in Spain and Portugal will be developed. Particularly, the aim of this paper is to review the different technologies applied to MSW solid waste incineration and to carry out both the diagnosis of the current situation at the incineration plants in Spain and Portugal and to collect data to develop the Life Cycle Inventory (LCI).

1. Introduction

Municipal Solid Waste (MSW) generation in Europe has increased regularly in recent years, amounting in 2010 to more than 2 billion tons of waste, that is to say a waste generation rate of 502 kg MSW/person. The same growth trend can be seen in Spain and Portugal, where in 2010 a waste generation of 535 and 514 kg of MSW/per capita was reached respectively. In 2010 in Spain more than 24 Mt of MSW were generated, 58 % being sent to the landfill, 9 % incinerated and the rest being recycling and used produce compost. For the same year, in Portugal more than 5 Mt of municipal solid waste were generated. 62 % of this waste was sent to the landfill, 19% was incinerated and the rest was recycled and composting (EUROSTAT, 2008). According to the available data, from 1998 to 2010 an increase in the incineration share of 68 % in Spain and 100 % in Portugal has come about. This rapid development of the sector has taken place over the last 10 to 15 years driven by the legislation specific to industry that has reached reducing emissions to air (European Commission, 2006). The main advantages of this type of waste treatment are the reduction of mass and volume of residues and the recovery of energy content in that waste with a significant heating value. However, incineration had

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gained a bad reputation owing to the environmental impact specifically due to the emissions of acid gases, dioxins and furans (PCCD/F) and greenhouse gases (Chevalier et al., 2003, Morselli et al., 2007). To assess the advantages and disadvantages and the environmental impacts of the incineration process a life cycle perspective is required. Life Cycle Assessment (LCA) is a powerful tool for assessing the environmental performance of a product, process or activity from “cradle to grave” (ISO 2006a and 2006b). Within this framework is the development of FENIX-Giving Packaging a New Life, a 3-year European LIFE+ funded project that started in January 2010. The aim of this project is to develop a flexible and user-friendly software tool to be used by Spanish and Portuguese municipalities and other territorial organizations, to obtain LCA results for packaging waste management, integrating environmental, economic and social aspects.

This work is just the first step in the development of a database and a model based on LCA to assess the environmental impacts of waste incineration in Spain and Portugal. Specifically, the aim of this paper is to review the different technologies applied to MSW solid waste incineration and to carry out both the diagnosis of the current situation at the incineration plants in Spain and Portugal and to collect data to develop the Life Cycle Inventory (LCI).

2. The incineration process

The main objective of Municipal Solid Waste (MSW) incineration is to treat waste so as to reduce its volume and hazard, while capturing or destroying potentially harmful substances. Incineration processes can also provide a means to enable recovery of the energy, mineral and/or chemical content from waste. Incineration is used as a treatment for a very wide range of waste types such as MSW, Hazardous Waste (HW) or sewage sludge (European Commission, 2006).

2.1 Thermal treatment

Different types of thermal treatments are applied to the different types of waste, however not all treatments are suited to all waste. The most common technologies applied are grate incinerators, rotary kilns, fluidised beds (FB) and pyrolysis and gasification systems. For MSW and Refuse Derived Fuel (RDF) incineration grates are widely applied, FB and rotary kilns are also applied but to a lesser extent. On the other hand, pyrolisis and gasification are rarely applied because they are still considered as emerging technologiesthis. In poarticular it is a less proven technology and the unclear economic benefits hamper a larger market penetration (Van Caneghem et al., 2012). In particular, in Spain and Portugal only grate incinerators and fluidised beds are applied at MSW incineration plants. In Spain 80 % of thermal treatment systems are grates while in Portugal this goes up to 100 %.

Grate incinerators (GI): in Europe approximately 90% of the installations treating MSW use grates. Grate incinerators usually comprise the waste feeder, incineration grate, bottom ash discharge, the incineration air duct system to ensure complete combustion, incineration chamber and auxiliary burners to heat up the furnace to a specified temperature. The optimum incineration conditions in which to achieve a good burn out of the gases are a minimum gas phase combustion temperature of 850 ºC (1,100 ºC in some hazardous wastes) and a minimum residence time of the flue-gases, above this temperature, of 2 seconds after the last incineration air supply (European Commission, 2000).The main types of grates are rocking, reciprocating, travelling and cooled grates (European Commission, 2006).

Fluidised beds (FB): they are widely applied to the incineration of finely divided waste such as RDF and sewage sludge. The FB incinerator is a lined combustion chamber in the form of a vertical cylinder. In the lower section a bed of sand, combustion ash, or other sand-like material is suspended in an upward flowing airstream. Normally this type of incineration requires a preparatory process step which makes raise the process costs. The main types of FB are stationary or bubbling fluidised bed, spreader-stoker furnace and rotating FB (European Commission, 2006; Van Caneghem et al., 2012).

2.2 Energy recovery

The majority of the energy produced during combustion is transferred to the flue-gases. Cooling of these gases allows energy recovery and the cleaning of flue-gases before they are released into the atmosphere. Conventional recovery involves passing the flue gases through a boiler, thereby obtaining steam, which can be turned into energy by means of an engine (White et al., 1995). The principal uses

of the energy transferred to the boiler are the production and supply of heat and the production and supply of electricity. Specifically, in Spain and Portugal is carried out but the energy recovered is used for the self consumption at the plant and sold to the public grid.

2.3 Flue-gas treatment (FGT) systems

Before the emission to air, flue gases must be cleaned by a combination of individual process units that together provide an overall treatment. The number of different treatment processes used varies widely from plant to plant, reflecting the emission standard required (Chevalier et al., 2003). Different techniques are applied to clean different pollutants such as acid gases, organic compounds or NOx.

Particulates: the main types of techniques are electrostatic precipitator (EP), ionisation wet scrubbers (IWS), fabric filters or bag filters and cyclones and multi-cyclones.

Acid gases (HCl,HF and SO_x): these gases are cleaned using alkaline reagents (CaO and $Ca(OH)_2$) by means of dry, semidry or wet processes. The main different between them is that in the wet process the reaction product is aqueous requiring a treatment prior to discharge (White et al., 1995).

Nitrogen oxides (NO_x): NO_X are reduced to N_2 and water vapour by the reduction agent (NH_3 or urea) applying Selective Non-Catalytic Reduction (SNCR) process or Selective Catalytic Reduction (SCR) process where the flue-gas pass over a catalyst (European Commission, 2006).

Dioxins and furans (PCCD/F): most usual treatment is adsorption on activated carbon but also bag filters and SCR could be applied.

2.4 **Solid residue treatment**

The main waste types arising from the combustion stage are bottom ash and boiler and fly ashes that are usually treated together. They are generally disposed of, often after a solidification process with water and cement, but could be used as a filling material in civil construction. Slag is usually subjected to magnetic separation, from which a metallic fraction is obtained made up of metallic waste contained in MSW and non-metallic fraction comprising ceramic and vitreous materials and particles not burned in the combustion process. The metallic fraction, ferrous scrap, is used to produce steel in an electric arc furnace (Lopez-Delgado et al., 2003). The inert material is sent to landfill.

3. The incineration of MSW in Spain and Portugal

According to the European Pollutant Release and Transfer Register E-PRTR and Directive 2008/1/EC, the so-called IPPC Directive (that replaced Directive 96/61/EC), dated September 2010, 10 Spanish facilities and 3 Portuguese plants are included in group 5.b; installations for the incineration of non-hazardous waste with a capacity of 3 tonnes per hour (European Parliament and Council, 2006). Figure 1 shows a map of the plants location.

Figure 1. Location of the Spanish and Portuguese incinerators (Source: own elaboration based on E-PRTR information).

In relation to the Spanish incineration plants, there are four plants in Catalonia (Spain). Two of them are located in Barcelona -Planta de Valortització Energética Sant Adrià de Besòs (TERSA) and Tractament i Revalorització de Residus del Maresme, S.A. (UTETEM)- one in Girona -TRARGISA area de residus- and the last one in Tarragona -Incineradora de Tarragona (SIRUSA)-. Table 1 shows technical and operational data for the four incineration plants sited in Catalonia (AEVERSU, 2010).

Table 1: Technical and operational data of TERSA, UTETEM, TRARGISA, SIRUSA.

	Tersa	**Utetem**	**Trargisa**	**Sirusa**
Type furnace	Von Roll grate	Martin travelling grate	Martin reverse-acting grate	Reciprocating grate
Tons MSW incinerated	321,728	170,274	35,053.46	151,849
LHV (kcal/kg)	1,900-2,200	2,100	1,800	2,000
Energy production (MWh)	167,504	86,105		44,552
Energy sales (MWh)	144,761	72,809		N.A
Slag (t)	55,642	41,973	6,338	30,921
Ashes (t)	12,039	7,237	650	3,508
Flue gases treatment	SNCR, bag filters, scrubbers, electro filters, activated carbon	SNCR, semidry and dry scrubber, bag filter	Bag filter, activated carbon	Semidry system, bag filter, activated carbon

Three incinerators are sites in the North of Spain. In particular in Cantabria -Planta de Tratamiento Integral de RSU de Cantabria (URBASER)-, Vicaya (Basque Country) -Zabalgarbi, S.A.- and in A Coruña (Galicia) -Complejo medioambiental de Cerceda (SOGAMA)-. The rest of incineators are located in Madrid -TIR Madrid-, Melilla –PIR Melilla, REMESA- and in Mallorca (Balearic Islands)-TIRME S.A.-. In Table 2 and 3 are given the data of these plants and in Table 4 the emissions to air of all the Spanish incinerators are shown (AEVERSU, 2010) .

Table 2: Technical and operational data of Zabalgarbi, SOGAMA and URBASER.

	Zabalgarbi	**SOGAMA**	**URBASER**
Type furnace	Reciprocating grate	Circulating FB	Roller grate
Tons MSW incinerated	223,933	550,000	113,338
Energy production (MWh)	661,160	335,078	82,800
Energy sales (MWh)	632.000	332,761	N.A
Slag (t)	42,547	69,038	14,972
Ashes (t)	8,375	33,240	4,536
Flue gases treatment	SNCR, semidry system, bag filter, activated carbon	Semidry system, bag filters, activated carbon	Scrubber, bag filter, activated carbon

Table 3: Technical and operational data of Zabalgarbi, SOGAMA and URBASER.

	TIR Madrid	**PIR Melilla**	**TIRME**
Type furnace	bubbling FB	serrated grate	roller and cooled grates
Tons MSW incinerated	418,905	40,986.7	294,185
LHV (kcal/kg)	3,500	1,400-3,000	1,800
Energy production (MWh)	234,841	N.A	152,389
Energy sales (MWh)	170,014	11,298	119,759
Slag (t)	N.A	9,397	69,133
Ashes (t)	N.A	1,043	28,242
Flue gases treatment	Cyclones, bag filters, SCR, scrubber, activated carbon	Semidry system, bag filter, activated carbon	Semidry scrubber, SCR, bag filter, activated carbon

Table 4: Emissions to air of the Spanish incinerators in 2009.

Pollutants mg/Nm^3	Incineration plants									
	Tersa	Utetem	Trargisa	TIR Madrid	Zabal-Garbi	Sirusa	Tirme	Sogama	PIR Melilla	Urbaser
HCl	5.9	2.11	3.8	4.2	3.2	7.0	0.0	<2.5	8.7	5.3
Particles	3.24	2.73	3.0	9E-1	8.7	7.0	<1.1	<6	22.6	3.68
CO	35.9	24.9	4.1	14.1	4.47	8.35	<4.0	<27	<4.6	11.6
HF	6E-2	1.9E-1	0.0	6E-2	2.8E-2	1.0	<4E-2	<1.5	<3.2E-2	3.3E-1
SO_2	14.4	20.15	0.0	3.2E-1	11.37	33	<7.5	<14	<13.6	1.36
Metals [(1)]	3.9E-2	3.8E-2	5E-3	5.3E-2	6.1E-2	6E-3	<2.9E-2	7E-2	<9.8E-2	1.4E-1
Cd+Tl	7.2E-3	5.4E-3	2.5E-3	<5E-4	2E-3	4.4E-3	<2E-3	<3E-2	<7.2E-3	1.4E-2
NH_3	6.1E-3	2.3E-3	N.A	N.A	N.A	N.A	N.A	N.A	N.A	N.A
TOC	1.59	3.63	N.A	N.A	N.A	N.A	N.A	N.A	N.A	N.A
Hg	N.A[(3)]	N.A	3E-3	6E-4	2E-3	5E-2	<2.5E-3	<1E-2	<1.6E-4	2.5E-3
VOC	N.A	N.A	3.4	1.4	1.4	8E-1	<5	<7	<1.4	2.8
PCDD/F[(2)]	1.4E-2	4.6E-3	6E-3	9E-3	7E-3	3E-3	5.6E-3	1.4E-2	2.5E-2	1.7E-2
NO_x	135 ppm	155	225	123	157	161	51.3	116	195	158

[1)] Sb+As+Pb+Cr+Co+Cu+Mn+Ni+V; [2)] ng ITEQ/Nm^3; [3)] N.A: Data Not Available.

In Portugal the 3 incinerators are located in Lisboa -VALORSUL, Valorização e Tratamento de Resíduos da Área Metropolitana de Lisboa (Norte), S.A S.-, Madeira -Valor Ambiente Gestão e Administração de Resíduos da Madeira- and Porto –LIPOR, Serviço Intermunicipalizado de Gestão de Resíduos do Grande Porto-. In Table 5 the main data and emissions of the Portuguese incinerators are given (VALORSUL, 2010, Valor Ambiente, 2010, LIPOR, 2010).

Table 5: Technical and operational data and emissions to air of the Portuguese incinerators in 2009.

	VALORSUL	Valor Ambiente	LIPOR
Incineration capacity (ton/year)	662,000	126,000	400,000
Type furnace	Reverse-Acting grate	Roller grates	Grate
Energy production (MWh)	N.A	N.A	200,000
Slag (kg/ton MSW)	200	160	N.A
Ashes kg/ton MSW)	30	N.A	N.A
Emissions to air (mg/Nm^3)			
CH_4	N.A	515,000	N.A
PCDD/F	N.A	1E-4	N.A
CO_2	502,000,000	N.A	357,000,000
Hg	N.A	N.A	N.A
HCl	17,600	N.A	N.A
NO_x	502,000	N.A	265,000
N_2O	59,500	N.A	N.A
NH_3	N.A	N.A	11,700

4. Conclusions

In this work the most relevant technologies applied in MSW incineration in Spain and Portugal have been determined and will be included in the future database and model based on LCA. The main data are collected from the European Pollutant and Transfer Register (E-PRTR), the Business Association of MSW valorisation (AEVERSU) and websites of different incineration plants.Regarding the thermal stage, grate incinerators, rotary kilns and FB could be applied to a range of waste types. In the case of MSW treatment, only grate incinerators and FB are used. Fluidised bed are applied to a smaller extent than grate incinerators because a preparatory stage is required when heterogeneous waste is treated. Specifically, in Spain the application of grate incinerators makes up 80% of incinerators, rising to 100% in Portugal. The majority of the energy produced during combustion is used for the self consumption at the plant and sold to the public grid.The amount of energy produced differs from one plant to another

and depends on the amount of waste incinerated and the heating value. Before being emitted the flue gases need treatment. Different systems are applied depending on the pollutants contained in the gases. For reducing particulate emissions in Spain and Portugal, electrofilters and bag filter are the technologies that are most often applied, and to a less extent cyclones and multicyclones. Acid gases such are treated through dry and semi-dry processes using an alkaline reagent such as CaO and $Ca(OH)_2$. To remove NOx Selective Non Catalytic Reduction (SNCR) and the Selective Catalytic Reduction (SCR) are applied. In both cases NH_3 or urea is the reagent used to reduce the NOx to N_2. Other important pollutants generated during the combustion are organic compounds like PCDDD/F. These substances are usually treated by absorption on activated However, SCR systems, catalytic bag filters, and static bed filters are also available. With regard to waste, slag and ashes (bottom, fly and boiler ash) are generated during the combustion process. Ashes are usually disposed of at a landfill, sometimes following a stabilization process with cement and water. Slag is usually subjected to magnetic separation, with the metallic fraction, ferrous scrap used to produce steel in an electric arc furnace and the inert material sent to landfill.

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4.2 García V, Margallo M, Aldaco R, Urtiaga A, Irabien A (2013) Environmental sustainability assessment of an innovative Cr (III) passivation process. ACS Sustain Chem Eng 1, 481-7.

Resumen

En este trabajo se ha llevado a cabo el análisis de ciclo de vida de productos procedentes del zincado electroquímico que son posteriormente pasivados en pequeñas y medianas empresas mediante diferentes procesos. El objetivo de este trabajo ha sido evaluar y comparar los impactos ambientales asociados al proceso de convencional de pasivación y a un proceso de pasivación alternativo mediante un análisis desde la "cuna a la tumba". El análisis de ha dividido en las etapas de la "cuna a la puerta", de la "puerta a la puerta" y de la "puerta a la tumba", utilizando para ello como indicadores el uso de recursos y las cargas ambientales. El proceso innovador de pasivación está basado en la integración de la tecnología de pertracción por emulsión con el baño de pasivado, alargando así el tiempo de vida del baño. Los resultados mostraron que la transferencia de los residuos peligrosos hasta el vertedero es el proceso que generó un mayor impacto ambiental tanto en el proceso convencional como en el innovador. La producción del hidróxido de calcio requerido para el tratamiento del efluente generado juega un papel muy importante en la etapa de la "cuna a la puerta". Este trabajo concluye que el proceso innovador disminuye la mayor parte de los residuos generados (92 %) durante el proceso de pasivación como consecuencia de la extensión del tiempo de vida del baño de pasivado. Se ha obtenido una reducción de las cargas ambientales al aire y al agua, así como del uso de recursos naturales durante toda la etapa de fabricación del producto. En concreto, las principales cargas ambientales al aire y al agua se han obtenido en las categorías de efectos a la salud humana y ecotoxicidad al agua.

Original abstract

A life cycle assessment was conducted for the Zn-electroplating products passivated by different processes in a small and medium enterprise. The goal was to evaluate and to compare the environmental impact associated to the conventional and alternative passivation process from a "cradle to grave" analysis. The assessment was divided into "cradle to gate", "gate to gate", and "gate to grave" steps for natural resources usage and environmental burdens. The innovative process was based on the integration of emulsion pertraction technology to the passivation bath in order to extend its lifetime. Results showed that the transferred hazardous waste from the process to the landfill was the major contributor to the environmental impact of the conventional and innovative passivation. The manufacture of the sodium hydroxide needed in the wastewater treatment process had a main role in the impacts of the "cradle to gate" cycle. This work concluded that the innovative passivation decreased most of the generated waste (92%) during the manufacture cycle of the passivated product as a

consequence of the extension of the lifetime of the passivation bath. A reduction of the total environmental burdens to air and to water and the resource usage during the whole manufacture cycle of the product was stated. The environmental burdens to air and to water were mainly connected to the environmental impacts: human health effects and ecotoxicity to aquatic life, respectively.

4.3 Margallo M, Aldaco R, Irabien A (2014) A case study for environmental impact assessment in the process industry: Municipal solid waste incineration (MSWI). Chem Eng Trans 39, 613-8.

Resumen

El análisis de ciclo de vida ha sido utilizado en la evaluación de productos y procesos químicos con el considerar toda la cadena de suministro, así como los principales impactos o problemas ambientales generados. En relación a la sostenibilidad ambiental, es necesario tener en cuenta dos variables: la sostenibilidad de los recursos naturales (NRS) y de las cargas ambientales (EBS). NRS incluye el uso de energía, agua y materiales, mientras que las cargas ambientales están compuestas por los indicadores de sostenibilidad ambiental desarrollados por la Institución de Ingenieros Químicos (IChemE). Los principales componentes de estas cargas ambientales se clasifican en cinco impactos al aire (acidificación, calentamiento global, efectos a la salud humana, agotamiento de la capa de ozono y formación de ozono fotoquímico), cinco impactos al agua (acidificación al agua, demanda acuática de oxígeno, ecotoxicidad (metales), ecotoxicidad (otros) y eutrofización), y dos impactos al suelo (residuos peligros y no peligrosos). Con el fin de reducir el número de variables y por tanto la complejidad de los resultados, es necesario llevar a cabo una normalización y ponderación. En concreto, este trabajo propone la normalización de las cargas ambientales mediante el uso de los valores umbrales del Registro Europeo de Emisiones y Transferencia Contamines (E-PRTR) y un procedimiento similar para los recursos naturales basado en los valores propuestos en el documento BREF de incineración. Este procedimiento ayudará a la toma de decisiones en el ámbito de la gestión de residuos y más concretamente en la incineración de residuos.

Original abstract

Life cycle assessment (LCA) has been introduced in the evaluation of chemical processes and or products in order to take into account the supply chain and its environmental constraints and burdens. Regarding to the environmental assessment of chemical processes and/or products two main variables need to be taken into account: Natural Resources Sustainability (NRS) and Environmental Burdens Sustainability (EBS). NRS includes the use of energy, water and materials whereas EBS is given by the environmental sustainability metrics developed by the Institution of Chemical Engineers (IChemE). The main components of EBS have been classified in 5 environmental impacts to the atmosphere (acidification, global warming, human health effects, stratospheric ozone depletion and photochemical ozone formation), 5 aquatic media impacts (aquatic acidification, aquatic oxygen demand, ecotoxicity (metals), ecotoxicity (others) and eutrophication) and 2 land impacts (hazardous and non-hazardous waste disposal). To reduce the number of variables and thus, the complexity, the development of a normalisation and weighting procedure is required.

This work proposes the normalization of EB based on the threshold values of the European Pollutant Release and Transfer Register (E-PRTR) and a similar procedure based on the values given by the BREF document on waste incineration for the NRS normalisation. This procedure will help in the decision making process in the waste management field and in the particular, in municipal solid waste incineration (MSWI).

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A Case Study for Environmental Impact Assessment in the Process Industry: Municipal Solid Waste Incineration (MSWI)

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Life Cycle Assessment (LCA) has been introduced in the evaluation of chemical processes and or products in order to take into account the Supply Chain and its environmental constraints and burdens. Regarding to the environmental assessment of chemical processes and/or products two main variables need to be taken into account: Natural Resources Sustainability (NRS) and Environmental Burdens Sustainability (EBS). NRS includes the use of energy, water and materials whereas EBS is given by the environmental sustainability metrics developed by the Institution of Chemical Engineers (IChemE). The main components of EBS have been classified in 5 environmental impacts to the atmosphere (acidification, global warming, human health effects, stratospheric ozone depletion and photochemical ozone formation), 5 aquatic media impacts (aquatic acidification, aquatic oxygen demand, ecotoxicity (metals), ecotoxicity (others) and eutrophication) and 2 land impacts (hazardous and non-hazardous waste disposal). To reduce the number of variables and thus, the complexity, the development of a normalisation and weighting procedure is required. This work proposes the normalization of EB based on the threshold values of the European Pollutant Release and Transfer Register (E-PRTR) and a similar procedure based on the values given by the BREF document on waste incineration for the NRS normalisation. This procedure will help in the decision making process in the waste management field and in the particular, in Municipal Solid Waste Incineration (MSWI).

1. Introduction

Life cycle Assessment (LCA) is a tool to assess the potential environmental impacts and resources used throughout a product´s life-cycle; i.e., from raw material acquisition, via production and use phases, to waste management (Finnvenden et al., 2009). LCA has been introduced in the evaluation of chemical processes and or products in order to take into account the Supply Chain and its environmental constraints and burdens. This methodology should be applied using the ISO 14040 series (ISO, 2006), describing LCA as a four-phase process:

a) Goal and scope definition: The intended application of the study, system boundaries, functional unit and the level of detail to be considered are defined (Cavallet et al., 2012).
b) Life Cycle Inventory (LCI) analysis: It includes the data collection and modelling of the system.
c) Life Cycle Impact Assessment (LCIA): The inputs and outputs data are translated into an impact indicator results related to human health, natural environment, and resource depletion (EC JRC, 2010b). LCIA includes two mandatory steps and two optional stages:
   - Classification: It includes the selection of the impact categories and characterization models (so-called impact assessment methods) (Bare J.C., 2010). Figure 1 shows the classification of the impact categories into midpoints and endpoints (Rack et al., 2013) and Table 1 presents a summary of the main LCIA methods based on the data published by Rack et al. (2013) and IHOBE (2009).

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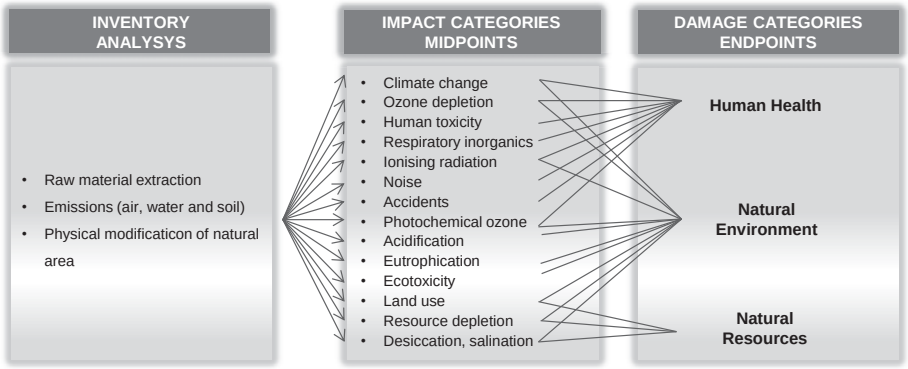

Figure 1: Framework of impact categories at midpoint and endpoint (Adapted from Rack et al., 2013).

Table 1: Review of the LCIA methods

Scope of application		Method	Impact assessment level	Creator	Reference
Asia	Japan	LIME	Combined	-Tokyo City University/ Kogakuin University	Itsubo and Inaba, 2012
	Singapore	Singapore IMPact ASSessment (SIMPASS)	Midpoint	-National University of Singapore/Institute of Chemical and Engineering Sciences of Singapore	Chan et al., 2012
Europe	Switzerland	IMPACT 2002+	Combined	-École polytechnique fédérale de Lausanne (EPFL)	Jollie et al., 2003
		Ecological Scarcity	Midpoint	-Swiss Ministry of Environment (BUWAL)	Frischknecht et al., 2006
	The Netherlands	ReCiPe	Combined	-Pré Consultans	Goedkoop et al., 2012
		Eco-Indicator 99	Endpoint	-Pré Consultans	Goedkoop et al., 2000
		CML 2001	Midpoint	-Centre of Environmental Sciences (CML)	Guinée et al., 2001
		LC-IMPACT	Combined	-Radboud University	LC-IMPACT, 2009
	Denmark	EDIP 2003	Midpoint	-Technical University of Denmark (DTU)	Hauschild and Potting, 2005
America	USA	TRACI	Midpoint	-Environmental Protection Agency (EPA)	Bare et al., 2003
	Canada	LUCAS	Midpoint	-CIRAIG	Toffoletto et al., 2007
Global		IMPACT World+	Combined	-CIRAIG/DTU/Quantis International/University of Michigan /EPFL/Cycleco	IMPACT World+, 2012

- Characterisation: The impact of each emission or resource consumption is modelled quantitatively using a characterisation factor. That factor expresses how much that flow contributes to the impact category indicator (EC JCR, 2010b).
- Normalisation (optional): It related the magnitude of impacts in different impact categories to reference values (Bare, 2011). The characterised impact scores are associated with a common

reference, facilitating comparisons across impact categories (EC JCR, 2010a).

- Weighting (optional): The different environmental impact categories are ranked according to their relative importance. Weighting may be necessary when trade-off situations occur in LCAs which are being used for comparing alternative products (EC JCR, 2010a).

d) Interpretation: The LCI and LCIA results are analysed giving the conclusions and recommendations of the study.

Most LCA studies apply impact methods which comprise several impact categories. The study of different processes by means of group of several impact categories makes difficult the process compassion. To reduce the complexity, this work proposes a LCIA methodology based on the Natural Resources Sustainability (NRS) and the Environmental Burdens Sustainability (EBS) (Irabien et al., 2009). Natural Resources (NR) include the use of primary resources energy, water and materials while Environmental Burdens (EB) is given by the environmental sustainability metrics developed by the Institution of Chemical Engineers (IChemE, 2002). However, as NR and EB are rarely normalized a normalization procedure is proposed. The normalization of EB is based on the threshold values of the European Pollutant Release and Transfer Register E-PRTR (E-PRTR Regulation, 2006) and a similar procedure based on the values given by the BREF document on waste incineration (European Commission, 2006) for the NRS normalization. This procedure will help in the decision making process in the waste management field and in the particular, in the waste incineration process.

2. Application of LCA to the waste management sector

LCA of a waste management system is divided in the same stages (from cradle to grave) that the LCA of a product. The main difference resides in what it is meant by cradle and grave. Whilst they share the same grave, they do not share the same cradle (Fullana and Puig, 1997). LCA methodology has been used to evaluate several types of wastes, such as the management of contaminated dredged sediments (Puccini et al., 2013), sewage sludge (Aranda-Usón et al., 2012), or fly ash from a coal burning power industry (Ondova et al., 2013). However, most of LCAs are focused in the study of Municipal Solid Waste (MSW). Evidence of that are the studies conducted to assess solid waste management systems in China (Zhao et al., 2011), compare different waste treatment options such as incineration and landfill (Hong et al., 2010) and evaluate the environmental feasibility of extending the selective collection of MSW in small villages of Spain (Margallo et al., 2010). Regarding to waste treatment, recently the study of the incineration process has taken off. The aim of these works was to assess the environmental performance of waste incinerators (Scipioni et al., 2009), compare different incineration technologies (Chen and Christensen, 2010), flue gas cleaning processes (Moller et al., 2011), management options of waste from incineration processes (Margallo et al., 2013), and different energy recovery strategies (Guigliano et al., 2008). Although all these studies use the LCA methodology, different impact assessment methods, summarized in Table 2, are applied.

3. Methodology

The LCI methodology includes the 4 steps included in the ISO 14040 (ISO, 2006): classification, characterisation, normalization and weighting.

3.1 Classification and Characterisation

The methodology consider the impact in the environment due to the use (depletion/exhaustion) of natural resources (NR) and the release of pollutants to the environmental compartments, air, water and soil (EB). In this way, NR includes the consumption of resources such as energy, materials and water for the considered process and/or product, so it can be describe by a NR index X_1. On the other side, EB includes

Table 2: Impact methods applied in waste management LCAs

LCIA method	Waste management studies
CML 2001	• Margallo et al., 2013 • Guigliano et al., 2008
EDIP 1997	• Moller et al., 2011 • Zhao et al, 2011 • Chen and Christensen 2010
ReCiPe	• Puccini et al., 2013 • Aranda-Usón et al., 2012
IMPACT 2002+	• Hong et al., 2010
Eco-Indicator 99	• Scipioni et al., 2009

Table 3: Natural Resources variables (X_1)

NR	Variable	Units	Description
Energy	$X_{1,1}$	MJ/t product	Total primary energy involved in the process (imports and exports)
Materials	$X_{1,2}$	kg/t product	The total raw materials involved in the production. Fuel and water are excluded from this variable
Water	$X_{1,3}$	m^2/t product	the modified and occupied land for the process

Table 4: Environmental Burdens variables (X_2)

EB	Variable	Environmental Impact	Units
Air EB ($X_{2,1}$)	$X_{2,1,1}$	Atmospheric Acidification (AA)	kg SO_2 eq.
	$X_{2,1,2}$	Global Warming (GW)	kg CO_2 eq.
	$X_{2,1,3}$	Human Health (HHE)	kg benzene eq.
	$X_{2,1,4}$	Photochemical Ozone Formation (POF)	kg ethylene eq.
	$X_{2,1,5}$	Stratospheric Ozone Depletion (SOF)	kg CFC-11 eq.
Water ($X_{2,2}$)	$X_{2,2,1}$	Aquatic Oxygen Demand (AOD)	kg O_2 eq.
	$X_{2,2,2}$	Aquatic Acidification (Aq. A)	kg H^+ eq.
	$X_{2,2,3,1}$	Ecotoxicity to Aquatic Life (metals) (MEco)	kg Cu eq.
	$X_{2,2,3,2}$	Ecotoxicity to Aquatic Life (others) (NMEco)	kg formaldehyde eq.
	$X_{2,2,4}$	Eutrophication (Eutroph)	kg phosphate eq.
Soil ($X_{2,3}$)	$X_{2,3,1}$	Hazardous waste (HWD)	t/y
	$X_{2,3,2}$	Non-hazardous (NHW)	t/y

the main impacts to the air, water and soil. According to the suggested procedure four variables can describe NRS (Table 3) (Dominguez-Ramos et al., 2013). On the other hand, as displays Table 4 the EB (X_2) considers a total of twelve variables grouped into the release to each environmental compartment. The EBS is based on the based on the sustainability metrics developed by the Institution of Chemical Engineers (ICheme, 2002) that give a balanced view of the environmental impacts of inputs and outputs (Garcia et al., 2013).

3.2 Normalization and Weighting procedure

Table 5 displays the EB normalization procedure that was developed taking into account the threshold values of the European Pollutants Release and Transfer-Register (EPRT-R) (EPRTR Regulation, 2006). The E-PRTR Regulation includes specific information on releases of pollutants to air, water and land and off-site transfers of waste and of pollutants in wastewater. Those data have to be reported by operators of facilities carrying out specific activities. Annex II of the E-PRTR Regulation lists the 91 pollutants that are relevant for reporting and specifies an annual threshold value of each pollutant for releases to each relevant medium (air, water, land). On the other hand, the normalization of variables for NRS ($X_{2,i}$) is carry

Table 5: Normalisation procedure (Irabien et al., 2009)

EB	Environmental Impact	Threshold value (kg/year)	Nº substances
Air EB ($X_{1,1}$)	AA ($X_{2,1,1}$)	150,000	6
	GW ($X_{2,1,2}$)	100,000,000	23
	HHE ($X_{2,1,3}$)	1,000	52
	POF ($X_{2,1,4}$)	1,000	60
	SOD ($X_{2,1,5}$)	1	100
Water EB ($X_{1,2}$)	AOD ($X_{2,2,1}$)	50,000	4
	Aq. A ($X_{2,2,2}$)	100	14
	MEco ($X_{2,2,3}$),1	50	11
	NMEco ($X_{2,2,3,2}$)	50	18
	Eutroph ($X_{2,2,4}$)	5,000	8
Soil EB ($X_{1,3}$)	HW ($X_{2,3,1}$)	2,000	H1…H14
	NHW ($X_{2,3,2}$)	2,000,000	

out using the references available from BREF document on waste incineration (European Commission, 2006). Consequently, the two functions, NR and EB, are converted into variables that can be compared. Finally, to reduce the complexity and to help in the decision making process, the 10 environmental impacts to air and water are reduced by means of weighting factors to two variables: EB to air and EB to water (Dominguez-Ramos et al., 2013).

4. Conclusions

The results of a LCA study are a group of environmental impacts that gives a balanced view of the environmental performance of the process or product under study. However, in some cases, the interpretation of these results is harder, requiring a complexity reduction by means of normalization and weighting methods. This paper aims to help in the decision making process in the waste incineration field proposing a normalization and weighting procedure. In particular, the methodology is based on the use of the threshold values of the E-PRTR and BREF document on waste incineration.

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Resumen

El tratamiento convencional de las cenizas de incineración consiste en un proceso de solidificación utilizando aglutinantes inorgánicos, tales como el cemento. Sin embargo, a pesar el alto contenido en metales pesados, se ha incrementado el uso de las cenizas como un agregado natural. Las cenizas son utilizadas como materia prima en la fabricación de clinker, mortero o vidrio sinterizado, como capa drenante en vertederos o como sub-base en la construcción de carreteras. En este estudio, se ha utilizado un enfoque de ciclo de vida para evaluar y comparar el proceso de solidificación con el reciclaje de las cenizas en la fabricación de cemento Portland, sustituyendo bien parte del clínker o del yeso. Los resultados muestran que la sustitución de cenizas por clínker tiene asociado un menor consumo de recursos y menores cargas ambientales. Cuando se disminuye el porcentaje de clinker, aumenta el consumo de recursos y las cargas ambientales asociadas. En estos procesos de reciclaje, un factor clave es la distancia entre la incineradora y la cementera. En concreto, la solidificación presenta resultados más desfavorables que el reciclaje (con un factor de sustitución del 25 % de clínker substituido), aunque la distancia entre ambas plantas aumentase. Sin embargo, cuando la cantidad de clinker disminuye al 2,5 %, o cuando las cenizas sustituyen al yeso, la distancia juega un papel importante en las cargas ambientales al agua.

Original abstract

Conventional bottom ash (BA) management consists of a solidification process using inorganic binder reagents, such as cement. However, despite the heavy metal content, the use of BA as a natural aggregate has become increasingly more common. In particular, bottom ash is used as a raw material for clinker, cement mortar or frit production, as a drainage layer in landfills or as a sub-base material in road construction. In this study, the life cycle assessment approach was used to evaluate and compare ash solidification with ash recycling in Portland cement production as a clinker and gypsum substitute. The findings showed that the substitution of ash for clinker resulted in the lowest natural resources (NR) consumption and the lowest environmental burdens (EB). The decrease in the clinker substitution percentage generated a higher NR consumption and an increased EB. In ash recycling, the distance between the incinerator and the cement facility is an important parameter in the decision-making process. Specifically, ash solidification presented less favourable results than ash recycling (with a clinker substitution of 25 %), despite the increasing distance between the incinerator and the cement facility. However, when the clinker substitution decreased to 2.5 % or when ash was substituted for gypsum, the distance

played an important role in the water impact.

4.5 Margallo M, Aldaco R, Irabien A, Carrillo V, Fischer M, Bala A, Fullana P (2014) Life cycle assessment modelling of waste-to-energy incineration in Spain and Portugal. Waste Manage Res 32 (6), 492-9.

Resumen

En los últimos años, la evaluación de los sistemas de gestión de residuos se ha realizado mediante el análisis de ciclo de vida (ACV). Una de las principales limitaciones de los primeros estudios era que se analizaba una mezcla de residuos con diferentes características. Por ello, la estimación de las emisiones asociadas a cada una de las fracciones de residuo debe abordarse mediante el uso de asignaciones de carga. En concreto, la incineración de residuos municipales es un ejemplo claro en el que un gran número de materiales son procesados obteniéndose diversas salidas del proceso. Este trabajo investiga un enfoque práctico que permita comprender mejor los procesos de incineración en España y Portugal mediante la aplicación de modelo de asignación de cargas multi-entrada/multi-salida. La aplicación de este modelo permite hacer predicciones de las entradas y salidas del proceso, incluyendo el consumo de materiales auxiliares y combustibles, las emisiones al aire, los residuos sólidos generados y la energía producida en la combustión de cada una de las fracciones de residuos.

Original Abstract

In recent years, waste management systems have been evaluated using a life cycle assessment (LCA) approach. A main shortcoming of prior studies was the focus on a mixture of waste with different characteristics. The estimation of emissions and consumptions associated with each waste fraction in these studies presented allocation problems. Waste-to-energy (WTE) incineration is a clear example in which municipal solid waste (MSW), comprising many types of materials, is processed to produce several outputs.

This paper investigates an approach to better understand incineration processes in Spain and Portugal by applying a multi-input/output allocation model. The application of this model enabled predictions of WTE inputs and outputs, including the consumption of ancillary materials and combustibles, air emissions, solid wastes, and the energy produced during the combustion of each waste fraction.

Original Article

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Life cycle assessment modelling of waste-to-energy incineration in Spain and Portugal

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Abstract

In recent years, waste management systems have been evaluated using a life cycle assessment (LCA) approach. A main shortcoming of prior studies was the focus on a mixture of waste with different characteristics. The estimation of emissions and consumptions associated with each waste fraction in these studies presented allocation problems. Waste-to-energy (WTE) incineration is a clear example in which municipal solid waste (MSW), comprising many types of materials, is processed to produce several outputs. This paper investigates an approach to better understand incineration processes in Spain and Portugal by applying a multi-input/output allocation model. The application of this model enabled predictions of WTE inputs and outputs, including the consumption of ancillary materials and combustibles, air emissions, solid wastes, and the energy produced during the combustion of each waste fraction.

Keywords

Incineration, life cycle assessment, life cycle inventory, municipal solid waste, waste-to-energy

Introduction

Waste generation in Europe has increased regularly in recent years, amounting to more than 2 billion tons of waste in 2010 with an annual per capita generation rate of 502 kg of municipal solid waste (MSW). An identical growth trend is noticed in Spain and Portugal, with 514–535 kg MSW person^{-1}. In these countries, 58–62% of MSW was landfilled, and the remainder was recycled or composted (EUROSTAT, 2011). However, despite landfilling remaining the most common practice, MSW incineration has increased in recent years; currently, Spain and Portugal have 10 and three incinerators, or waste-to-energy (WTE) plants, respectively, with a capacity of more than 3 t h^{-1}. In 2009, nearly 3.5 million tons of MSW were incinerated in these plants, generating more than 2000 GWh of energy. The main advantages of incineration are the reduction in waste mass and the energy recovery (European Commission, 2006). However, incineration has a poor reputation related to environmental impacts because of its emissions of greenhouse gases, acid gases, and dioxins and furans (PCDD/F) (Morselli et al., 2008). To assess the advantages, disadvantages, and environmental impacts of incineration, a life cycle approach is required. Life cycle assessment (LCA) has been used in several studies to assess different waste management systems. A main shortcoming of prior studies was the focus on a mixture of waste with different characteristics (Seyler et al., 2005). In these studies, challenges arose in allocating the emissions and consumptions associated with each waste fraction. Determining how these parameters should be allocated to each waste input fraction is critical (Finnveden et al., 1995). This paper reports on an approach to better understand incineration processes in Spain and Portugal by applying a multi-input/output allocation model. This work is within the framework FENIX-Giving Packaging a New Life, a 3-year European LIFE+ funded project.

Life cycle assessment

Goal and scope

This paper develops a model to better understand incineration processes by investigating the average data from 13 WTE plants

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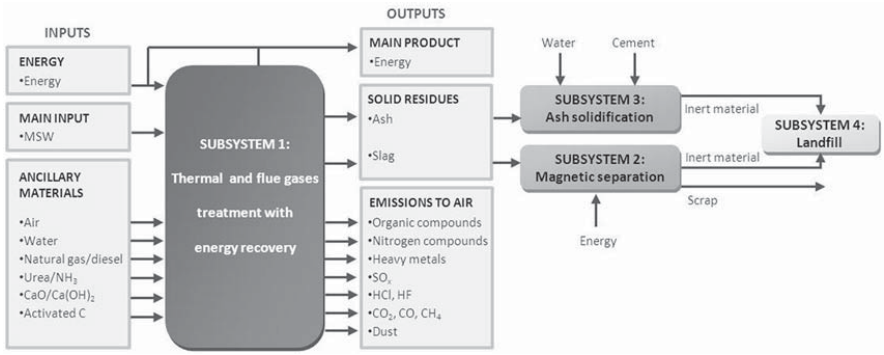

Figure 1. Flow diagram of the incineration process in Spain and Portugal.

in Spain and Portugal. The most common technologies applied in these plants are grate and fluidized beds, the former technology was present in 80% of thermal treatment systems in Spain, whereas 100% of the plants employed the grate-based technologies in Portugal (Margallo et al., 2012).

In particular, the 18 waste fractions that compose the MSW were modelled. Therefore, the functional unit selected was 1 t of each waste fraction incinerated: PET, HDPE packaging (P) and non-packaging (nP), LDPE (P and nP), plastic mix (P and nP), paper and cardboard (PC) (P and nP), beverage carton, steel (P and nP), aluminium (Al) (P and nP), glass (P and nP), organic matter, and remaining materials (wood, construction and demolition wastes, textiles and others). This reference unit was selected taken into account the literature and the fact that the main function of incineration is to treat and reduce the volume and hazard of waste. In relation to cut-offs, all material and energy inputs that have a cumulative total of at least 98% of the total mass and energy inputs have been included. However, those flows that do not meet this criterion but are thought to potentially have a significant environmental impact have also been included. The process comprises thermal treatment with energy recovery, flue gases cleaning, and solid waste treatment. Construction of major capital equipment and, the maintenance and operation of support equipment were excluded from the study. Considering the system boundaries, the incineration process was modelled as a black box including four subsystems (Figure 1).

Subsystem 1: thermal and flue gases treatment. These processes were considered as a single subsystem because the composition and flow of flue gases before treatment is not measured. The inputs of this system are MSW, combustibles, ancillary materials, and reagents; the outputs are energy production, waste, and air emissions. Therefore, because of the lack of significance, the emissions of several pollutants were excluded. Only one plant registered hexachlorobenzene, endrin, heptachlor, naphthalene, pentachlorobenzene, tetrachloroethylene, tetrachloromethane, trichlorobenzene, benzene ethylene oxide, vinyl oxide, and hydrogen cyanide emissions; therefore, these pollutants were not included in the life cycle inventory (LCI). Heavy metals are often significant in toxicity assessment, and the cut-off criteria identified the most representative metal emissions: arsenic (As), cadmiun (Cd), chromium (Cr), cooper (Cu), manganese (Mn), mercury (Hg), nickel (Ni), lead (Pb), and zinc (Zn). Water emissions from waste incineration are only related to plants with an exhaust gas cleaning system (Bjarnadóttir et al., 2002). In these plants, wet scrubbers are not applied; therefore, water emissions were not considered.

Subsystem 2: magnetic separation of slag. The separation entails an energy demand of 0.042 kWhel kg^{-1} iron removed (Doka, 2003). Approximately 10% of slag is recovered as scrap and the remainder as inert slag. Outside of the system boundaries, steel is produced with scrap.

Subsystem 3: ash solidification. Ash is solidified with a mixture of water (30%), cement (20%), and ash (50%) to produce an inert ash that is landfilled.

Subsystem 4: final disposal. Inert slag is sent to a MSW landfill next to the incinerator. Ash is a hazardous material, but once stabilized, it is sent to an inert landfill.

Life cycle inventory

In the LCI, all relevant inputs and outputs for the process in a specified year are collected. The properties of the waste mixture are also determined (Seyler et al., 2005). Data collection was performed from July 2010 to September 2011. The data used resulted from (a) site-specific operating data collected from the Spanish Association of WTE plants (AEVERSU), the Spanish non-profit company resposible for the collection and recovery of packaging waste (Ecoembes), several WTE plants, and the Spanish Pollutant Release and Transfer Register (PRTR) and (b) bibliographic data. The data given in Table 1 consist of annual material and energy inputs and outputs of Spanish and Portuguese plants in 2009. Additionally, the details and quality of this study were analysed by a critical review according to ISO 14040 (ISO, 2006a).

Table 1. Average inputs and outputs at Spanish and Portuguese WTE plants.

INPUTS	***Combustible / ancillary materials (kg t^{-1}MSW)***: *AEVERSU, 2013; WTE plants, 2009*			
	Natural gas	6.88E-01	Urea	3.34
	Diesel	1.99E-01	Ammonia	2.07
	Water	334	CaO	8.29
	Air	3,550	$Ca(OH)_2$	4.04
	Activated Carbon	4.77E-01		
OUTPUTS	***Waste (t t^{-1}MSW)***: *AEVERSU, 2013*		***Products (MJ t^{-1} MSW)***: *AEVERSU, 2013*	
	Slag	1.71E-01	Energy production	3,005
	Ashes	5.05E-02	Self-consumption	325
	Scrap	1.95E-02	Energy sales	2,931
	Air emissions (kg t^{-1}MSW): *PRTR, 2012*			
	Arsenic (As)	3.96E-05	Dioxins and furans (PCDD/F)	2.30E-10
	Cadmium (Cd)	1.16E-05	Carbon dioxide (CO_2)	480
	Chromium (Cr)	7.06E-05	Carbon monoxide (CO)	1.21E-01
	Copper (Cu)	4.53E-05	Methane (CH_4)	5.00E-04
	Lead (Pb)	1.02E-04	Total Organic Compounds (TOC)	1.41E-02
	Manganese (Mn)	7.54E-05	Sulphur oxides (SO_X)	6.90E-02
	Mercury (Hg)	7.35E-06	Nitrogen oxides (NO_X)	9.15E-01
	Nickel (Ni)	3.34E-05	Nitrous oxide (N_2O)	2.51E-02
	Zinc (Zn)	6.32E-04	Ammonia (NH_3)	1.68E-02
	Hydrogen chloride (HCl)	2.56E-02	Total Suspended Particles (TSP)	1.11E-02
	Hydrogen Fluoride (HF)	1.23E-03	Particulate matter (PM_{10})	8.69E-03
	Polycyclic Aromatic Hydrocarbons (PAHs)	1.58E-04	Non Methane Volatile Organic Compounds (NMVOC)	1.20E-02

Model description

Waste incineration is defined as a multifunctional process. In these systems, the environmental burdens associated with a particular process must be partitioned over the various functional flows of that process (Huijungs and Guinée, 2007). There are three basic types of allocations problems: multi-output (co-product systems such as a refinery), multi-input (waste treatment), and open-loop recycling (when a recycled product is transformed into another product) (Finnveden et al., 2009). In addition, some processes could be defined as multi-input/multi-output. Incineration is a clear example, in which several inputs (i.e. waste fractions) and outputs (e.g. energy recovery, waste generation and emissions) coexist. The procedure of establishing a multifunctional model is given in Figure 2 and is described in three steps (Seyler et al., 2005). Table 2 summarizes the allocation rules employed in the model.

1. Compilation of an LCI for the waste mixture.
2. Classification of the input and output parameters according to its product or process dependence. Process-dependent parameters do not rely on the properties of the product but only on the process conditions. Product-dependent parameters rely on the properties of the production, such as the chemical composition or heating value. Likewise, some parameters can depend on both the process and product (Seyler et al., 2005).
3. Calculation of consumptions and emissions from each waste fraction after applying allocations rules. ISO 14044 (ISO, 2006b) proposes as a first solution for allocation problems to expand the system boundaries or divide the process into sub-processes. When this solution is not possible, the allocation should be assigned based on physical causation, which reflects the underlying relation among different flows. The final allocation should be based on other criterion, such as economic-value, mass, or energy. In this work, the allocation was based on the first option with physical causation. Additionally, when that allocation was not possible, a mass or energy allocation was applied.

In this work, this methodology was applied using GaBi 4.4 software (PE International, 2011) by dividing the model into three sections: thermal treatment, flue gases treatment, and solid waste generation (Figure 3).

Thermal treatment

Waste composition. Based on the percentage of each waste fraction, the total amount of MSW and the moisture content, the wet and dry weight of each fraction was calculated according to equations 1 and 2. Additionally, Figure 4 displays the waste composition of the studied WTE plants.

$$\mathrm{In}_{\mathrm{fraction}_i} = \mathrm{In}_{\mathrm{wet\,MSW}} * \frac{\mathrm{Fraction}_i}{100} \quad (1)$$

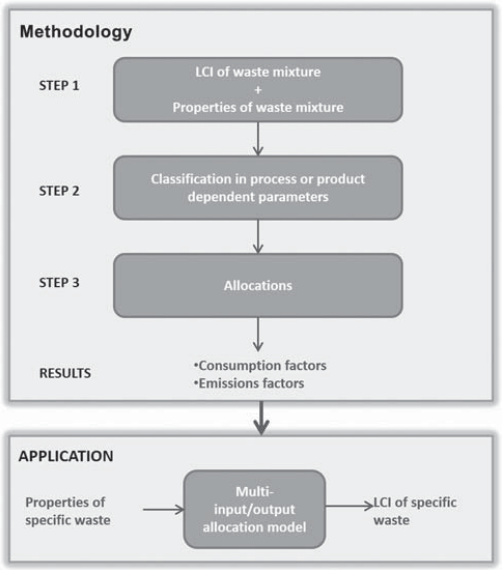

Figure 2. Description of the methodology to develop a multifunctional allocation model.

Table 2. Summary of allocations applied.

Type of allocation	Parameter
Mass	Combustibles, ancillary materials, reagents Ash and slag Emissions of PCDD/F, NO_X, N_2O, NH_3, dust
Heating value	Energy production
Metals content	Metals emission
Fossil C content	CO_2 emission
C content	CO, TOC, CH_4, NMVOC, PAHs emission
Cl content	HCl emission
F content	HF emission
S content	SO_x emission

$$In_{dry\ fraction_i} = In_{fraction_i} * \left(1 - \frac{Moisture_i}{100}\right) \quad (2)$$

where i is the waste fractions, $Fraction_i$ is the percentage of each waste fraction (%), $In_{wet\,MSW}$ is the wet weight of MSW (t), $In_{fraction_i}$ is the wet weight of each waste fraction (t), $In_{dry\ fraction_i}$ is the dry weight of each waste fraction (t), and $Moisture_i$ is the moisture of each waste fraction (%).

Combustibles and ancillary materials consumption. At start up, auxiliary burners that consume diesel or natural gas are used to heat the furnace to a specific temperature. The burners are also switched on if the temperature decreases. Other consumptions include water and air to cool the furnace and to ensure complete combustion. These consumptions are not related to the waste composition; therefore, the mass allocation procedure was used.

$$Consumption_{fraction_i} = \frac{Consumption}{In_{wet\,MSW}} \quad (3)$$

where Consumption is the water, air, diesel and natural gas consumed (kg $year^{-1}$) and $Consumption_{fraction_i}$ is the consumption of water, air, diesel, and natural gas per ton of waste fraction (kg t^{-1} wet waste).

Energy production. The majority of the energy produced in combustion is transferred to flue gases. Cooling of these gases allows energy recovery, which could be used in the production and supply of heat, electricity, or both. In the Iberian Peninsula, approximately 80% of the energy produced is sold to the public grid, with the rest used for self-consumption. The energy produced must be calculated via the energy content of the waste (Riber et al., 2008). In this work, based on the high heating value (HHV) of each waste fraction and the amount of waste incinerated, the theoretical energy produced is calculated according to equation 4. Subsequently, the real energy produced must be allocated (energy allocation) to each waste fraction according to equation 6.

$$E_{theo\ fraction_i} = In_{fraction_i} * HHV_{fraction_i} * 1000 \quad (4)$$

$$E_{theo\,MSW} = \sum_{i}^{n} E_{theo\ fraction_i} \quad (5)$$

$$E_{fraction_i} = \frac{\frac{E_{theo\ fraction_i} * E_{production}}{E_{theo\ MSW}}}{In_{fraction_i}} \quad (6)$$

where $HHV_{fraction_i}$ is the HHV of each waste fraction (MJ kg^{-1}), $E_{production}$ is the energy production (MJ $year^{-1}$), $E_{theo\ fraction_i}$ is the theoretical energy production of each waste fraction based on the HHV (MJ $year^{-1}$), $E_{theo\,MSW}$ is the theoretical energy production of MSW based on the HHV (MJ $year^{-1}$), $E_{fraction_i}$ is the energy production of each waste fraction, that in theory, can be obtained from $E_{theo\,MSW}$ assuming an average efficiency of the incinerator in converting the HHV to energy output (MJ t^{-1} wet waste).

Flue gasesatment

Flue gases are cleaned using a combination of individual process units that together provide overall treatment. The following reagents are consumed: CaO or $Ca(OH)_2$ for acid gases reduction in dry, semi-dry, or wet scrubbers; NH_3 or urea for NO_x in a selective catalytic reduction or selective non-catalytic reduction; and activated carbon for organic compounds (Margallo et al., 2012). The

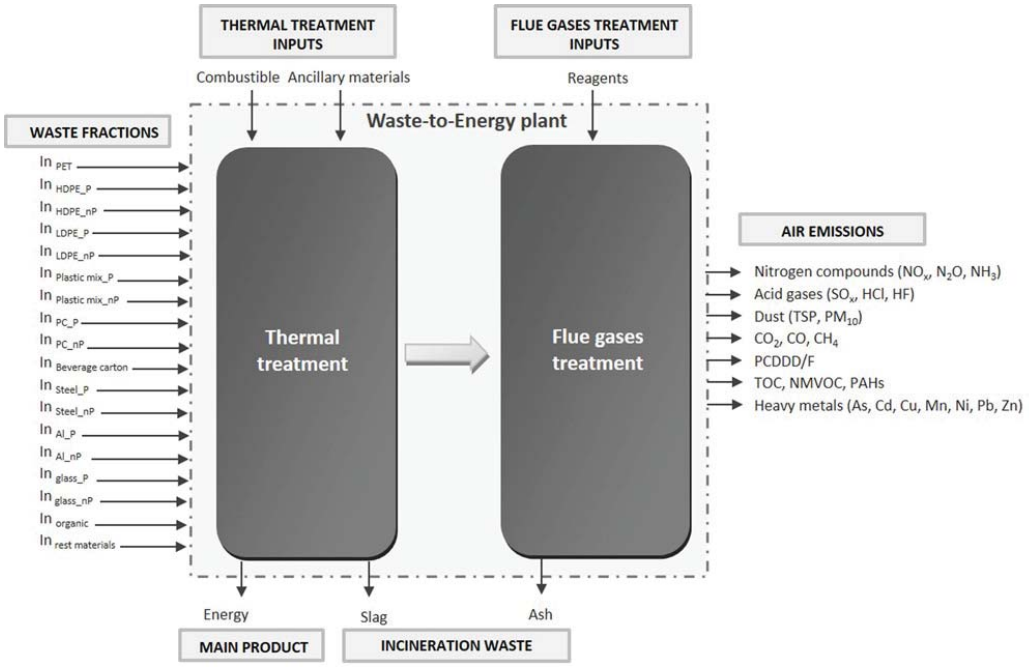

Figure 3. Description of the allocation model for WTE incineration in Spain and Portugal.

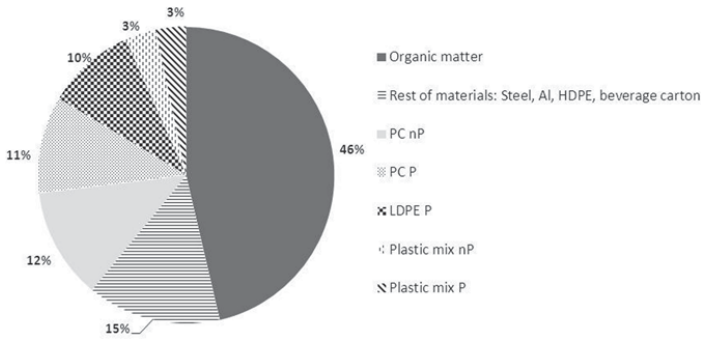

Figure 4. Waste composition of the Spanish and Portuguese WTE plants.

consumptions were allocated by mass to each fraction according to equation 7.

$$\text{Reagents}_{\text{fraction}_i} = \frac{\text{Reagents}}{\text{In}_{\text{wet MSW}}} \quad (7)$$

where Reagents is the consumption of urea, NH_3, CaO, $Ca(OH)_2$ and activated carbon (kg year^{-1}) and $\text{Reagents}_{\text{fraction}_i}$ is the consumption per ton waste fraction (kg t^{-1} waste).

Emissions are generated during the combustion of different waste fractions. Therefore, it is essential to determine the emissions of different pollutants associated with each waste fraction using allocation rules.

Nitrogen compounds (NO_X, NH_3, and N_2O), dioxins and furans (PCDD/F), and dust (TSP and PM_{10}). Emissions of NOx, NH_3, N_2O, and dust depend on applied technology rather than waste composition. A special case is PCDD/F emissions. Often, dioxins are suggested to be allocated to different waste components in relation to the chlorine content of the waste. However, other authors advocate that the emissions of these pollutants are more related to the operating conditions; therefore, PCDD/F should be allocated to the waste component in relation to the mass or energy content of the waste (Finnveden, 1999). In this study, the latter proposal was applied. PCDD/F emissions are thought to depend more on operational conditions and treatment technologies (of

both combustion and flue gases) than on the Cl content of the input waste. Moreover, MSW has a vast surplus of chlorine; the mere fact that the MSW is incinerated under combustion conditions indicates that the process can form PCDD/Fs. Mass allocation was presented as the best option for these pollutants according to equation 8.

$$P_{fraction_i} = \frac{P}{In_{wet\ MSW}} \quad (8)$$

where P is the emission of dust, nitrogen compounds, and PCDD/F (kg year^{-1}) and $P_{fraction_i}$ is the emission per ton waste fraction (kg t^{-1} wet waste).

Metals (Cd, Cr, Cu, Hg, Ni, Pb, Zn, and Mn) and metalloids (As). These compounds are converted primarily into non-volatile oxides. In particular, Hg is emitted usually as metallic Hg or $HgCl_2$ depending on the amount of HCl contained in the flue gas (European Commission, 2006). The remaining metals are usually emitted as oxides or chlorides. Emissions of these compounds depend on the input waste composition; therefore, they were allocated based on the content of the respective elements in the input waste fractions. The metal and metalloid content is based on the data published by Riber et al. (2009).

$$M_{content\ fraction_i} = \frac{M_{fraction_i} * In_{dry\ fraction_i}}{1000} \quad (9)$$

$$M_{emission_i} = \frac{M * \frac{M_{content\ fraction_i}}{\sum_1^i M_{content\ fraction_i}}}{In_{fraction_i}} \quad (10)$$

where M is the emission of As, Cd, Cr, Cu, Hg, Ni, Pb, Zn, and Mn (kg year^{-1}), $M_{fraction_i}$ is the metal and metalloid content in each waste fraction (mg metals kg^{-1} dry waste), $M_{content\ fraction_i}$ is the metal and metalloid content in each waste fraction (kg metals), and $M_{emission_i}$ is the emission of metals per ton waste fraction incinerated (kg metals t^{-1} wet waste).

Acid gases (SO_x, HCl, and HF). These emissions were allocated to the input waste based on the S, Cl, and F content of the waste. Therefore, the calculation shown in equation 11 is identical to those proposed for metals in equations 9 and 10.

$$X_{emission_i} = \frac{X * \frac{X_{content\ fraction_i}}{\sum_1^i X_{content\ fraction_i}}}{In_{fraction_i}} \quad (11)$$

where X is the emission of SOx, HF, and HCl (kg year^{-1}), $X_{content\ fraction_i}$ is the S, F, and Cl content in each waste fraction (mg S, F, and Cl), and $X_{emission_i}$ is the emissions of SOx, HF, and HCl per ton waste fraction incinerated (kg t^{-1} waste).

Carbon compounds (CO_2, CO, CH_4, NMVOC, TOC, and PAHs). The allocation must be performed according to the total or fossil carbon content of the input waste. Carbon dioxide emissions are related to the C content of the waste. However, the climate-relevant CO_2 emissions from waste incineration are determined by the proportion of waste carbon compounds that are of fossil origin. However, the contribution of CH_4, CO, NMVOC, TOC, and PAHs to climate change is only partially dependent on (for CO and CH_4) or completely independent of the fossil C content. Therefore, these compounds were allocated based on the total C content.

$$CO_{2\ fraction_i} = \frac{Fossil\ C_{fraction_i} * In_{dry\ fraction_i} * 1000}{\sum_i^n \left(Fossil\ C_{fraction_i} * In_{dry\ fraction_i} * 1000\right)} * CO_{2\,fossil} \quad (12)$$

$$Ccomp._{fraction_i} = \frac{C_{fraction_i} * In_{dry\ fraction_i} * 1000}{\sum_i^n \left(C_{fraction_i} * In_{dry\ fraction_i} * 1000\right)} * Ccomp. \quad (13)$$

where $C_{fraction_i}$ is the total C content in each waste fraction (g C kg^{-1} dry waste fraction), $Fossil\ C_{fraction_i}$ is the fossil C content in each waste fraction (g fossil C kg^{-1} dry waste fraction), $CO_{2fossil}$ is the fossil CO_2 emission per year (kg CO_2 year^{-1}), $CO_{2\ fraction_i}$ is the fossil CO_2 emission per ton waste fraction incinerated (kg CO_2 t^{-1} wet waste), Ccomp. is the emission of CO, CH_4, NMVOC, TOC, and PAHs per year (kg year^{-1}), and $Ccomp._{fraction_i}$ is the emission per ton waste fraction incinerated (kg t^{-1} wet waste).

Solid waste

Slag and ashes are the main waste generated in combustion. Among all incinerated fractions, only inert materials (steel, Al, glass, and construction and demolition waste) are completely transferred to slag. The remaining materials were allocated by mass according to equation 14.

$$Slag_{fraction\ (non\ inert)} = \frac{Slag_{non\_inert}}{In_{dry\ non\ inert\ MSW}} \quad (14)$$

$$In_{dry\ non\ inert\ MSW} = In_{dry\ MSW} - In_{dry\ inert\ MSW} \quad (15)$$

$$Slag_{non\_inert} = Slag - In_{dry\ inert\ MSW} \quad (16)$$

where $In_{dry\ MSW}$ is the dry weight of MSW (t), $In_{dry\ inert\ MSW}$ is the dry weight of inert waste (t), $In_{dry\ non\ inert\ MSW}$ is the dry weight of non-inert waste (t), Slag is the amount of slag generated (t year^{-1}), $Slag_{non\_inert}$ is the amount of non-inert slag (t year^{-1}), and $Slag_{fraction\ (non\ inert)}$ is the amount of non-inert slag per ton waste (kg t^{-1} waste fraction).

In relation to ash, inert materials are not incinerated; therefore, they are not transferred to ash, whereas, noninert materials are allocated by mass according to equation 17.

$$Ash_{fraction\,(non\ inert)} = \frac{Ash}{In_{dry\ fraction_i}} \quad (17)$$

where Ash is the amount of ashes generated (t year^{-1}) and $Ash_{fraction\,(non\ inert)}$ is the amount of ashes generated in incineration of non-inert material fraction per ton waste (kg t^{-1} waste fraction).

Table 3. Results of the input and output data associated with each waste fraction.

	HDPE	LDPE	Steel	Al	PC	Organic matter
Consumptions (kg t^{-1} waste)						
Air	3105	3105	3105	3105	3105	3105
Water	415	415	415	415	415	415
Diesel	1.74E-01	1.74E-01	1.74E-01	1.74E-01	1.74E-01	1.74E-01
NH_3	2.92	2.92	2.92	2.92	2.92	2.92
Waste (kg t^{-1} waste)						
Slag	153	153	1000	1000	153	153
Ash	78.78	78.78	0	0	78.78	78.78
Products (MJ t^{-1} waste)						
Energy	8844	8844	0	0	1890	850
Air emissions (kg t^{-1} waste)						
NOx	7.64E-01	7.64E-01	7.64E-01	7.64E-01	7.64E-01	7.64E-01
PCDD/F	3.30E–10	3.30E–10	3.30E–10	3.30E–10	3.30E–10	3.30E–10
As	8.38E-06	3.51E-06	4.31E-04	1.57E-04	4.85E-06	9.99E-07
HCl	1.01E-02	7.10E-03	0	0	2.86E-03	1.24E-02
HF	1.33E-03	1.33E-03	0	0	4.99E-03	2.88E-04
SOx	3.37E-02	3.37E-02	0	0	1.20E-01	1.17E-02
CO_2	2237	2236	0	0	0	0
CO	2.88E-01	2.88E-01	0	0	1.37E-01	3.17E-02

Application of the model

The results of the model are the consumption and emissions values and the waste and products of the incineration process associated with each input waste fraction. These factors can be used to calculate the waste fraction LCIs. To display the application of the model, Table 3 details the results obtained for five waste fractions.

According to this method, those input and output parameters, which are process dependent, were allocated by mass. This group included: the consumption of air, water, combustibles, and reagents; the generation of ashes and slag; and the emissions of dust, N compounds, and dioxins. PCDD/F is a special case because the distinction between process and product dependence is not clear. The emissions of this pollutant are suggested to be allocated based on the Cl content; however, the operating conditions have a high influence on the emissions as well. In this case, the choice of allocation methods will have a strong influence on the emissions of each waste fraction. For all of the process-dependent parameters shown in Table 2, identical results were obtained for the different waste fractions, reasserting its independence with the input waste composition (Seyler et al., 2005).

However, the results of the product-dependent parameters, such as energy production, emissions of carbon compounds, heavy metals, and acid gases were different for each fraction. This difference is because process-dependent parameters depend on waste composition and heating value. The energy production assessment shows that, the fractions that generate more energy during combustion are PET, HDPE, and LDPE because of the high energy content (43.47 MJ kg^{-1}), while steel and aluminium fractions do not generate energy as their energy content is null. The following was found in relation to the emissions:

- Steel and Al are the fractions with the highest heavy metals emissions because of the high metal content in the input waste (2 mg As kg^{-1} waste), with lower contributions from the organic matter and plastic mix (0.26 and 0.2 mg As kg^{-1} waste). The combustion of Al and steel do not generate other types of pollutants because Cl, F, C, and S are absent from the input waste.
- C compound emissions are mainly associated with HDPE and LDPE combustion because of the high fossil and biological carbon content. However, some differences are observed in the CO and CO_2 emissions. Al and steel combustion does not generate emissions of CO and CO_2 because this waste lacks fossil and biological C. Nevertheless, PC and organic matter combustion generate only CO emissions because the entire C content is of biological origin. Regarding acid gases, the combustion of PC and the incineration of PET, HDPE, and LDPE contribute to HF emissions. The high Cl content in the plastic mix and organic matter generates the highest emissions of HCl. SOx emissions are primarily associated with paper and plastic combustion.

The results for waste generation displayed, on one hand, that identical amounts of slag is generated for non-inert fractions because a mass allocation was applied. For inert fractions, such as Al and steel, the entire waste is expected to be completely transferred to slag, achieving a value of 1000 kg slag t^{-1} waste. On the other hand, in ash generation, inert materials are not converted to ash, whereas for non-inert materials, a mass allocation was applied, producing identical results for all fractions.

Conclusions

This work presents a multifunctional model to describe the waste incineration process. To establish the model, several

assumptions that influence the outcome of the work were detailed. The most important one is the classification of input and output data based on their dependence on process, product, or both. According to this classification, the allocation relationship was established as another important source of uncertainty. However, occasionally, the distinction between product and process dependence is not clear. In these cases, the selection of allocation will strongly influence the LCI for each waste fraction. From this analysis, the mass, composition, moisture, and heating value of the input waste are critical factors in determining the LCI and the type of allocation to be applied. Therefore, the environmental burdens associated with each waste fraction will depend on these factors. The results show that when a mass allocation was applied, all of the waste fractions obtained identical values in consumptions and emissions. Significant differences were obtained when the allocation was based on the waste composition or heating value.

Finally, the implementation of this model is subject to the interpretation of the practitioners, who may use the paper as a reference guide to conduct future research.

Declaration of conflicting interests

The authors do not have any conflict of interest to declare.

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Resumen

Este estudio propone una metodología basada en el Análisis de Ciclo de Vida (ACV) para la evaluación de la sostenibilidad ambiental (ESA) en diversas plantas incineradoras de España. El procedimiento está basado en el uso de dos variables: la sostenibilidad de los recursos naturales (NRS) y de las cargas ambientales (EBS). NRS incluye el uso de energía, agua y materiales, mientras que las cargas ambientales están compuestas por cinco cargas al aire, cinco al agua y dos al suelo. Para reducir la complejidad de esta metodología, las variables fueron normalizadas y ponderadas utilizando los valores umbrales del Registro Europeo de Emisiones y Transferencia Contaminantes (E-PRTR). Los resultados mostraron que todas las plantas estudiadas tenían un mayor consumo de recursos naturales que España, en concreto entre 1,1 y 2,0 veces mayor que la media española. La comparación de España con el BREF de incineración de residuos, mostró que este país tenía solo tenía un menor consumo que el valor de referencia en la variable de materiales (1,8 veces menor). Respecto a las cargas ambientales, los impactos al aire y al suelo fueros los compartimentos ambientales dominantes. Las plantas estudiadas presentaron cargas al aire y al suelo superiores a la media española, mientras que solo una planta superó la carga al suelo de España. Para concluir, este artículo demuestra la utilidad de la metodología ESA para reducir la complejidad del ACV y ayudar de esta manera en el proceso de decisiones para seleccionar la mejor opción desde un punto de vista ambiental. Este procedimiento puede ser usado para obtener una visión general del comportamiento ambiental de las incineradoras, así como para evaluar las cargas ambientales individuales y por lo tanto determinar los principales problemas ambientales mejorando los puntos críticos del proceso

Original abstract

This study proposes a technical procedure based on a life cycle assessment (LCA) for the implementation of the Environmental Sustainability Assessment (ESA) of several Waste-to-Energy (WtE) plants located in Spain. This methodology uses two main variables: Natural Resources Sustainability (NRS) and Environmental Burdens Sustainability (EBS). NRS includes the consumption of energy, materials, and water, whereas EBS involves five burdens to air, five burdens to water, and two burdens to land. To reduce the complexity of ESA all these variables were normalised and weighted using the threshold values proposed in the European Pollutant Release and Transfer Register regulation. The results showed that all of the studied plants had a greater consumption of natural resources than Spain; it ranges from 1.1 to 2.0 times higher than the Spanish reference. The comparison of Spain with the BREF reference

on waste incineration displayed that only in the variable related to materials, Spain presented a lower consumption (1.80 times lower). In terms of EBS, air and land impacts were the highest contributors to the global burden. The WtE plants presented higher burdens to air and water than Spain, whereas only one plant exceed the average burden to land of Spain. Finally, this paper demonstrated the usefulness of the ESA methodology to reduce the complexity of LCA and assist the decision-making process in choosing the best option from an environmental point of view. This procedure can be used to obtain an overview of the environmental performance of WtE plants, as well as to assess individual burdens and thereby to determine the main environmental hotspots, thereby improving the critical points of the process.

Environmental Sustainability Assessment in the Process Industry: A Case Study of Waste-to-Energy Plants in Spain

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Abstract

This study proposes a technical procedure based on a life cycle approach for implementation of the Environmental Sustainability Assessment (ESA) of several Waste-to-Energy (WtE) plants located in Spain. This methodology uses two main variables: the Natural Resources Sustainability (NRS) and the Environmental Burdens Sustainability (EBS). NRS includes the consumption of energy, materials, and water, whereas EBS considers five burdens to air, five burdens to water, and two burdens to land. To reduce the complexity of ESA, all variables were normalised and weighted using the threshold values proposed in the European Pollutant Release and Transfer Register regulation. The results showed the plants studied had a greater consumption of natural resources than Spain, ranging from 1.1 to 2.0 times higher than the Spanish reference consumption. The comparison of Spain with the BREF reference on waste incineration showed that only in the variable related to materials, did Spain have a lower consumption (1.80 times lower). In terms of EBS, air and land impacts were the highest contributors to global burden. The WtE plants presented higher burdens to air and water than Spain, whereas only one plant exceeded the average burden to land of Spain. Finally, this paper demonstrated the usefulness of the ESA methodology to reduce the complexity of LCA and assist the decision-making process in choosing the best option from an environmental point of view. This procedure can be used to obtain an overview of the environmental performance of WtE plants, as well as to assess individual burdens and thereby determine the main environmental hotspots, thereby improving the critical points of the process.

Keywords: incineration, life cycle assessment, municipal solid waste, waste-to-energy

1. Introduction

The high rate of waste generation in the society today has brought waste management to be a priority in European policies. The European Regulation proposes waste prevention, recycling and reuse, and finally waste incineration and landfilling as fundamental principles to waste management (EC 2008). Despite landfilling remaining the most common practice, waste incineration and recycling have increased in recent years. The primary objective of Municipal Solid Waste Incineration (MSWI) is to treat waste by reducing the solid waste mass and allowing energy recovery. For this reason, the original designation of "incinerator" was dropped, and today it is discussed as "energy from waste" or "waste to energy" (WtE) (Margallo et al. 2014a). The basic linear structure of a WtE plant may include incoming storage and pre-treatment of waste, thermal treatment with energy recovery and conversion, flue gases and wastewater treatment, and the management and treatment of ash and slag (EC-IPPC 2006). Different types of thermal treatments are applied to different types of wastes; however, not all thermal treatments are suited to all wastes. The most common technologies are Grate Incinerators (GI), Rotary Kilns (RK), Fluidised Beds (FB), and pyrolysis and gasification systems. For Municipal Solid Waste (MSW) and Refuse Derived Fuels (RDF), GIs are widely applied; FBs and RKs are also applied, but to a lesser extent (Margallo et al., 2012). Despite the benefits of waste incineration, their high combustion temperatures require very specific materials be used in their construction, increasing installation and maintenance costs. Also, additional combustible material is required when the available waste does not reach the required heating value or when it has high water content (Rodríguez and Irabien, 2013). Moreover, this technology has unfortunately gained a bad reputation because of its environmental impact, specifically due to its emissions of acid gases, dioxins and furans (PCDD/F), and greenhouse gases (Margallo et al., 2012). In this regard, the Environmental Sustainability Assessment (ESA) is a powerful tool to identify the environmental strengths and drawbacks of waste management in WtE plants. Several methodologies, such as Life Cycle Assessment (LCA), Social Cost-Benefit Analysis (CBA), and pricing carbon emissions, are useful to analyse the environmental performance of waste management. All these techniques have advantages and disadvantages, so it is not possible to determine which methodology is more valid to evaluate the ESA. Nevertheless, LCA is one of the most accepted approaches because is a standardised method. In this context, an increasing number of publications related to the LCA of waste management have appeared in recent years (Laurent et al., 2014). Most LCA studies have been conducted in Europe; in particular, several works evaluated MSW management systems in Italy (Arena and Di Gregorio, 2014; De Feo and Malvano, 2009; Buttol et al., 2007), Denmark (Kirkeby et al., 2006), Portugal (Ferrão et al., 2014), and Spain (Aranda-Usón et al., 2013; Bovea et al., 2010; Bovea and Powel, 2006; Muñoz et al., 2004; Rodríguez-Iglesias et al., 2003). Nowadays, an important role is also played by the BRIC countries (Brazil, Russia, India, and China); these are nations that will generate a large amount of MSW in the future. In this context, an important number of works related to the waste management systems in the China (Zhao et al., 2011), India (Mondal et al., 2010), Russia (Tulokhonova and Ulanova, 2014), and Brazil (Leme et al., 2014) have been also reported.

In addition, LCA studies evaluating incineration processes have become common. The aim of the published LCA works on waste incineration was to assess the

advantages, drawbacks, and environmental impacts of the technology. In Italy, the environmental performance of several WtE plants was assessed by Morselli et al. (2007) and (2008), and a prediction of the environmental impacts of a new incineration plant was reported by Scipioni et al. (2009). In France, 110 incinerators have been compared with regard to their environmental impact (Beylot and Villeneuve, 2013), and in China, the environmental impact of waste incineration with auxiliary coal has been evaluated (Zhao et al., 2012). Other studies compared thermal treatment technologies such as GI and FB (Chen and Christensen, 2010), flue gas cleaning systems (Moller et al., 2011; Chevalier et al., 2003), energy recovery strategies (Guigliano et al., 2008; Consoni et al., 2005a and 2005b), management options for pollution control of residues from waste incineration (Fruergaard et al., 2010), and several Bottom Ash (BA) treatments (Margallo et al., 2014a; Huntzinger and Eatmon, 2009; Birgisdottir et al., 2006). Incineration was also compared with other technologies; in particular, the environmental impacts of incineration were compared with those of waste recycling (Merrild et al., 2008) and landfilling in studies conducted in Brazil (Mendes et al., 2004), Thailand (Liamsanguan and Gheewala; 2008), Italy (Cherubini et al., 2009 and 2008), and China (Dong et al., 2014). Other authors extended these comparisons further to include gasification and pyrolysis processes (Zaman, 2010) and the Mechanical Biological Treatment (MBT) (Koci and Trecakova, 2011) in the comparison.

Most existing LCA studies use conventional impact assessment methods such as CML 2001 (Guinée et al., 2001), EDIP 97 (Wenzel et al., 1997), or Eco-indicator 99 (Goedkoop et al., 2000). These methods use a set of metrics, which in some cases could be difficult to understand and thus confuse the process comparisons. A reduction in the complexity of LCA would improve the comprehension of the results and thus assist the decision-making process. In this regard, the goal of the present work is to propose a technical method for conducting an ESA of an organic waste incineration process using two main variables: the Natural Resources Sustainability (NRS) and the Environmental Burdens Sustainability (EBS). NRS includes the consumption of energy, materials, and water; whereas EBS is based on the environmental sustainability metrics proposed by the Institution of Chemical Engineers, IChemE (IChemE, 2002). Currently, NRS and EBS are rarely normalised; thus, they are treated as functions rather than as variables. Considering the previously developed methodology (Irabien et al., 2009) for the normalisation of the Environmental Burdens (EB), which is based on the threshold values proposed in the regulation of the European Pollutant Release and Transfer Register, the so called E-PRTR regulation (EC, 2006), a similar procedure based on the average consumption of natural resources (NR) of Spanish MSWI plants was used for the normalisation of NRS. In this way, NRS and EBS can be normalised, and the comparison between NR and EB can be accomplished. This methodology will help the decision maker choose the best option within ESA, reducing its complexity because the two main functions can be converted into comparable variables that can be used later in a multi-objective optimisation. As a case study, several WtE plants located in Spain were selected to assess and compare the environmental performance of these plants. The analysis was conducted for the Cradle-to-Gate, Gate-to Gate and Gate-to-Grave stages of processing. In particular, the purpose of this paper is firstly to apply a life cycle model of waste incineration (Margallo et al. 2014b) to several WtE plants in Spain. Specifically, the incineration of organic waste fraction was studied to evaluate the environmental impacts of the plants, determining the critical points of the process. The paper also shows a comparison of the environmental performance of the plants by means of the ESA methodology. IChemE metrics were applied successfully to compare the conventional and alternative passivation processes (Garcia et al., 2013), several arsenic removal treatments (Dominguez-Ramos et al. 2014), and BA treatments against ash recycling (Margallo et al., 2014a).This paper also used EB to evaluate land pollution, uses the normalisation of the NR based on the average consumption of resources of the Spanish WtE plants, and employs a weighting procedure to reduce the LCA results to only two variables: NRS and EBS.

2. Methodology

LCA evaluates processes or products from cradle-to-grave (ISO, 2006a). This approach includes three types of analysis (Dominguez-Ramos et al., 2014), described as follows:

- Cradle to Gate (Cr-Ga): This analysis describes the environmental burdens generated by the transformation of natural/primary resources into usable forms of resources, and encompasses all individual transformation processes including raw materials extraction, manufacturing, and transportation.
- Gate to Gate (Ga-Ga): This analysis evaluates the environmental burdens generated by transformation of final resources into a product, process, or service.
- Gate to Grave (Ga-Gr): This analysis considers the burdens from the final emissions to the environment and the burdens from the consumption of the final resources for the selected environmental management practice.

LCA should be applied using the ISO 14040 series (ISO, 2006a), which describes the LCA as a four-stage process involving: a) the definition of the goal and scope of the analysis; b) Life Cycle Inventory (LCI) analysis; c) Life Cycle Impact Assessment (LCIA); and d) interpretation. LCIA is composed of two mandatory (i.e., classification and characterisation) and two optional steps (i.e., normalisation and weighting). This paper developed an ESA methodology that includes the four steps proposed in the LCIA. The advantage of this procedure regarding the conventional methodologies is that the results are divided into NR and EB providing a complete overview of the environmental performance of the process. Moreover, the normalisation factors of conventional LCIA are calculated with the substance emissions and characterised factors per substance. In this work, the normalisation was conducted by means of the thresholds values of the E-PRTR regulation (EC, 2006). This way, the relevance of each EB at a policy and regulatory level was included because the European Commission sets these threshold values for each specific pollutant. This

normalisation and weighting procedure supplies a framework to compare all the European installations included in the industrial sectors of the Integrated Pollutant Prevention and Control Directive (IPPC) using a European policy weighting.

Classification and characterisation. Classification includes the selection of the impact categories, and the characterisation models of the study (Bare 2010). In the characterisation stage the impact of each emission or resource consumption is modelled quantitatively using a characterisation factor or Potency Factor (PF). This factor expresses how much that flow contributes to the impact category indicator (EC JCR 2010). The developed LCIA methodology is based on the use of NR (i.e., depletion/exhaustion) and the release of pollutants to specific environmental compartments (i.e., air, water, and land).

In this way, NRS includes the consumption of final useful resources such as energy (X1,1) [MJ], materials (X1,2) [kg], and water (X1,3) [kg] for the considered process and/or product; thus, it can be described by an NRS dimensionless index X1. Land use and soil carbon stocks were not included as an NR natural resource because the construction and maintenance of the plant was not within the system boundaries of the study because WtE is not considered an intensive-land use process.

On the other hand, EBS includes the primary burdens to the air, water, and land due to the release of pollutants (i.e., gas, liquid, and solid state). EBS is given by the environmental sustainability metrics developed by IChemE. This set of indicators can be used to measure the environmental sustainability performance of an operating unit, providing a balanced view of the environmental impact of inputs (i.e., resource usage), and outputs (i.e., emissions, effluents, and waste) (IChemE 2002). In relation to the outputs, a set of environmental impacts on the atmosphere, aquatic media, and land was chosen. The environmental burden (EB) approach was used to estimate and quantify the potential environmental impacts (Garcia et al. 2013). In particular, the environmental impacts were classified in 12 variables grouped into the release to each environmental compartment (i.e., air, water, and land). According to the suggested procedure shown in Fig. 1, three variables can describe NRS and 12 variables grouped into the release to each environmental compartment can describe EBS.

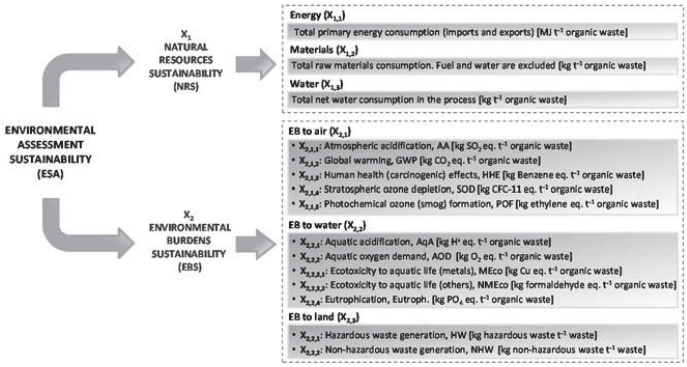

Fig. 1 Life cycle impact assessment methodology based on natural resources sustainability (NRS) and environmental burdens sustainability

Normalisation and Weighting. Normalisation relates the magnitude of impacts in different impact categories to reference values (Bare 2010). The aim of normalisation is two-folded: a) to place the LCIA indicator results into a broader context; and b) to adjust the results to have common dimensions (Bojarski 2010). Specifically, the characterised impact scores associated with a common reference facilitate comparisons across impact categories. Conversely, weighting ranks the different environmental impact categories according to their relative importance (EC JCR 2010a). Consequently, for the variables identified for the assessment of NR and EB, a normalisation and weighting procedure could be performed to assess the contribution of each variable. This procedure allows the decision maker to track the progress towards environmental sustainability and to clarify the optimisation procedure, at least for the environmental pillar.

The consumption of NR (X_1) varies from plant to plant; therefore, to understand whether the consumption of a given plant is acceptable and to compare each plant, a reference should be used. In particular, the average consumption of each i NR in the 10 existing WtE plants located in Spain was selected as the reference value ($X_{1,i}^{ref}$). On the other hand, the variables for EBS were compared using the threshold values taken from the E-PRTR regulation, leading to normalised variables ($X_{2,j,k}^{*}$). The E-PRTR regulation establishes the contaminants for which the European installations must provide notification to the authorities along with the threshold values of those pollutants (Table 1). The threshold values can be used as an important aid in the normalisation process because they provide an overview of the environmental performance of the installation at a European level (Margallo et al. 2014a).

Eqs. 1 and 2 show the basic calculations that were used for the NRS and EBS normalisation:

$$X_{1,i}^{*} = X_{1,i} / X_{1,i}^{ref} \quad (1)$$

$$X_{2,j,k}^{*} = X_{2,j,k} / X_{2,j,k}^{ref} \quad (2)$$

In Eqs. 1 and 2, i represents different NR (energy, materials, and water); j represents different

environmental compartments (air, water, and land); k represents the environmental impacts to air, water, and land described in Fig. 2, respectively; $X_{1,i}$ is the consumption of each i NR; $X^{*}_{1,i}$ is the normalised value of $X_{1,i}$; $X^{ref}_{1,i}$ is the NR taken as reference value; $X_{2,j,k}$ are the environmental burdens to air, water, and land; $X^{*}_{2,j,k}$ is the normalised value of $X_{2,j,k}$; and $X^{ref}_{2,j,k}$ is the reference value used for EBS normalisation.

The three NRS normalised variables ($X^{*}_{1,i}$) that represent energy, materials, and water consumption and the 12 EBS normalised variables ($X^{*}_{2,j,k}$) are subjected to direct summation. Therefore, the NRS index (X_1) can be assessed according to Eq. 3, whereas the calculations of the EBS index to air ($X_{2,1}$), water ($X_{2,2}$) and land ($X_{2,3}$) are based on Eq. 4.

$$X_1 = \gamma\,\alpha_{1,1}X^{*}_{1,1} + \sum_{i=2}^{i=n}\alpha_{1,i}X^{*}_{1,i} \qquad n \in [2,3] \quad (3)$$

$$X_{2,j} = \sum_{j=1}^{j=n}\beta_{2,j,k}X^{*}_{2,j,k} \qquad n \in [1,2] \quad (4)$$

In Eqs. 3 and 4, X_1 is the NRS index that includes energy, materials, and water consumption; $\alpha_{1,i}$ is the weighting factor for the materials and water variables; $X_{2,j}$ are the EBS indexes for air, water, and land; $\alpha_{1,1}$ is the weighting factor for the energy variable; $\beta_{2,j,k}$ is the weighting factor for EBS; and γ is the factor accounting for the energy net importer or exporter character of the plant. This factor has a value of -1 when the plant exports energy and +1 when plant imports energy.

Table 1. Threshold values from the E-PRTR regulation for normalisation and nº of substances included in each impact category. The units are given in kg equivalents (kg eq.)

Environmental Burdens (EB)	Threshold value (kg y^{-1})	Nº substances
EB to air		
Atmospheric acidification (AA) [kg SO_2 eq.]	150,000	6
Global warming (GW) [kg CO_2 eq.]	100,000,000	23
Human health effects (HHE) [kg benzene eq.]	1,000	52
Photochemical ozone formation (POF) [kg ethylene eq.]	1,000	100
Stratospheric ozone depletion (SOF) [kg CFC-11eq.]	1	60
EB to water		
Aquatic oxygen demand (AOD) [kg O_2 eq.]	50,000	14
Aquatic acidification (AqA) [kg H^+ eq.]	100	4
Ecotoxicity to aquatic life (organics) (MEco) [kg Cu eq.]	50	11
Ecotoxicity to aquatic life (metals) (NMEco) [kg formaldehyde eq.]	50	18
Eutrophication (EU) [kg phosphate eq.]	5,000	8
EB to land		
Hazardous waste (HW) (kg hazardous waste)	2,000	-
Non-Hazardous waste (NHW) (kg non-hazardous waste)	2,000,000	-

Consequently, the NRS index depends on the weight assigned to each final resources variable. When the three final resources are equally relevant, $\alpha_{1,i} = 1/3$ for each i. This was assumed because it is the clearest way to obtain a single index that allows a comparison across several plants. The application of a weighting factor of 1/3 to all the components of NR to obtain a single index, yields an overview of the performance of the plant requiring the evaluation of the individual NR to determine the critical points of the process. Other sets are also possible, but different weighting procedures must be discussed.

2.1 Goal and Scope

The goal of this study was to perform an ESA of organic waste incineration in several WtE plants located in Spain using NRS and EBS. The most common thermal treatment technologies for MSWI in Europe are grate incinerators and fluidised beds (FB). In Spain, grate technology is used in 80% of the thermal treatments applied in the country (Margallo et al., 2012). Therefore, 4 Spanish incinerators, namely I1, I2, I3, and I4, all of which are equipped with grate furnaces, were selected as a case study. Specifically, these incinerators were selected because the data available were the most representative and complete of all of the Spanish plants and because the chosen incinerators share similar geographical locations. Nevertheless, they differ in age, pollutant abatement technologies and consumption of reagents and combustibles. Table 2 displays some technical data of the plants (AEVERSU, 2013), whereas Table 3 shows the waste composition of the incinerators as well as the average composition of Spanish MSW (Ecoembes, 2014). According to the goal of the study, one ton of organic waste at the gate of the WtE plants was selected as the functional unit. The system includes the thermal treatment and the cleaning of flue gases, ash solidification and final disposal of ash and slag. Fig. 2 depicts the system under study considering the life cycle stages from cradle-to-grave:

- Cradle to Gate (Cr-Ga): This step includes the final resources of thermal and flue gas treatment: reagents for flue gas treatment, electricity, combustibles and water.
- Gate to Gate (Ga-Ga): In this study, the Ga-Ga step refers to the incineration of organic waste. The EB originated from the emissions to air of greenhouse gases, acid gases, organic compounds, dust, and heavy metals. Water emissions from waste incineration are only related to plants with exhaust gas cleaning systems (Bjarnadóttir et al., 2002). In all of the studied plants, wet scrubbers are not used; therefore, water emissions were not considered. Although a high degree of acid gas removal is reached (near 90%) and the waste products from may be re-usable, wet scrubbers have not been applied in these plants because an extensive equipment is necessary and the wastewater produced would require treatment prior to discharge, translating into higher capital costs. Moreover, the land emissions generated were allocated to the Ga-Gr step related to the final waste treatment.
- Gate to Grave (Ga-Gr): This analysis considers the burdens from the consumption of final resources of ash and slag treatment. Ash treatment includes ash solidification with a mixture of water (30%), cement (20%), and ashes (50%), a process that produces a non-hazardous waste that is later landfilled. Slag from MSWI

requires magnetic separation to recover ferrous materials. In this study, only the organic matter was considered; therefore, slag has a null content of ferrous materials and is sent to a non-hazardous landfill.

Table 1 Technical characteristics of the Spanish WTE plants under study

		I_1	I_2	I_3	I_4
Start-up year		1975	1994	1984	1991
Incineration capacity (t/h)		14.5	10	2.5	9.6
Nº lines		3	2	2	2
Thermal treatment		Roller grate	Travelling grate	Reverse-acting grate	Roller grate
Combustion temperature (ºC)		950	1000-1100	1050	900-1500
Flue gases treat-ment	NO_x	SNCR	SNCR	SNCR	SNCR
	Particles	Electro-filter	Bag filter	Electro-filter/ bag filter	Bag filter
	Acid gases	Semidry scrubber	Semidry/ dry scrubber	Dry scrubber	Semidry scrubber
Rea-gents	Activated carbon	✓	✓	✓	✓
	Urea	✓	✓	✓	
	NH_3				✓
	CaO	✓			✓
	$Ca(OH)_2$		✓	✓	

SNCR: Selective non-catalytic reduction

Table 2 Waste composition of the WTE plants and the Spanish average as percentages

	Type of carbon origin	I_1	I_2	I_3	I_4	Spanish average
PET	Fossil	1.96	1.17	1.89	2.07	2.23
HDPE	Fossil	1.13	0.95	1.34	1.36	1.38
LDPE	Fossil	5.55	4.29	7.15	7.01	7.56
Plastic mix	Fossil	5.31	4.89	4.29	4.91	4.53
Steel	-	3.02	2.72	3.09	4.11	2.81
Aluminium	-	0.46	0.25	0.45	0.30	0.39
Beverage carton	Fossil & biological	1.04	0.57	1.23	0.93	1.40
Glass	-	4.37	5.38	4.94	4.08	3.55
Paper and cardboard	Biological	13.31	9.28	9.74	12.51	14.91
Organic matter	Biological	43.78	53.33	47.89	45.27	40.47
Remaining materials	-	16.37	13.64	14.27	13.57	16.30
Moisture		3.70	3.46	3.76	3.81	4.50

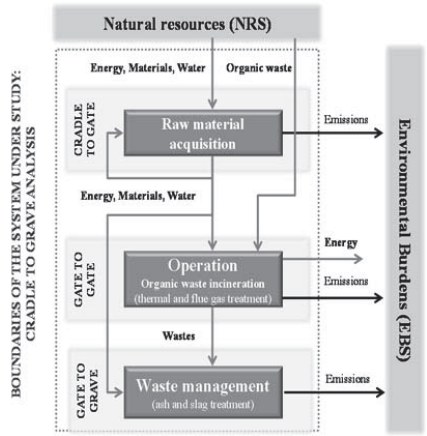

Fig. 2 System description of the Cr-Gr analysis of organic waste incineration

2.2 Allocation procedures

Waste incineration is defined as a multi-functional or as a multi-input/multi-output allocation process in which several inputs (i.e., waste fractions) and outputs (e.g., energy recovery, waste generation, and emissions) coexist. Specifically, in the Spanish incinerators studied, the composition of the MSW includes 18 waste flows: PET, HDPE packaging (P) and non-packaging (nP), LDPE (P and nP), plastic mix (P and nP), paper and cardboard (PC) (P and nP), beverage carton, steel (P and nP), aluminium (Al) (P and nP), glass (P and nP), organic matter, and other remaining materials (wood, construction and demolition wastes, textiles, and others). In this study, only organic matter was assessed; therefore, the total emissions and consumptions associated with MSWI were allocated to the organic fraction. The allocation procedures were based on the waste composition (C, Cl, F, S, and heavy content of the input waste), mass, and heating (Margallo et al., 2014b). In addition, waste incineration involves waste treatment and energy production, providing the system with an additional function. This situation was handled through system expansion by subtracting the function of the alternative system (energy production) from the system under study. In this study, the electric power mix of Spain included in the ELCD-PE GaBi database was selected as the technology replaced in the system expansion (PE International, 2011).

2.3. Life cycle inventory

Table 4 shows the inventory for the 4 Spanish WtE plants. All data are given in reference to 1 ton of organic waste. A zero value for emissions indicates that the plant has not notified that it releases that pollutant; however, the substance could potentially still have been emitted by the plant.

The inventory shown in Table 4 represents the annual material and energy inputs and outputs of the Spanish plants in 2009. Data on the WtE plants were provided by the Spanish non-profit company Ecoembes, which is responsible for the collection and recovery of packaging waste (Ecoembes, 2014), the Spanish and Catalonian associations of MSW valorisation (AEVERSU, 2013; ACEVERSU, 2014), the Spanish Pollutant Release Transfer Register (PRTR, 2012), the IPPC permit of the plants, the WtE plants, and bibliographic data. Finally, the ash solidification data were collected from Doka (2003). Background data for the energy and production of ancillary materials and reagents were taken from the PE database (PE International, 2011). The geographical sources of the study are Spain and Europe, so there is consistency with the goal and scope of this study. As previously indicated, water emissions were not generated. Emissions of carbon dioxide are generated in the MSW incineration; however, in this analysis, CO2 was not considered because the organic matter has a null content of fossil carbon. Finally, the details and quality of this study were analysed in a critical review conducted according to ISO 14040 (ISO, 2006a).

Table 4. Life cycle inventory for the selected WTE plants (values per one ton of organic waste as functional unit)

		I_1	I_2	I_3	I_4	Spain average	Units
THERMAL AND FLUE GAS TREATMENT							
Inputs	**Organic waste**	1.00	1.00	1.00	1.00	1.00	t organic waste
	Energy and combustibles						
	Diesel	2.75	-	1,224	424	205	10^{-3} kg t^{-1} organic waste
	Natural gas	7.46	573	-	-	709	10^{-3} kg t^{-1} organic waste
	Electricity	158	222	241	6.89	31.2	MJ t^{-1} organic waste
	Reagents and auxiliary materials						
	Urea	3.59	11.3	1.68	0.00	3.44	kg t^{-1} organic waste
	Ammonia (NH_3)	-	-	-	0.97	1.69	kg t^{-1} organic waste
	Calcium hydroxide ($Ca(OH)_2$)	0.00	13.1	9.54	0.00	4.16	kg t^{-1} organic waste
	Lime (CaO)	8.38	0.00	0.00	6.52	9.15	kg t^{-1} organic waste
	Activated carbon	0.37	0.80	1.68	0.12	0.48	kg t^{-1} organic waste
	Water	313	449	156	545	344	kg t^{-1} organic waste
	Air	6.70	9.35	6.71	6.71	4.58	10^{3} kg t^{-1} organic waste
Outputs	**Products**						
	Electricity	1,160	1,442	488	574	1,713	MJ t^{-1} organic waste
	Emissions to air						
	Carbon monoxide (CO)	8.30	6.92	8.85	3.88	6.97	10^{-2} kg t^{-1} organic waste
	Carbon dioxide (CO_2)	-	-	-	-	-	kg t^{-1} organic waste
	Methane (CH_4)	0.00	0.00	0.00	0.00	1.26	10^{-3} kg t^{-1} organic waste
	Non-methane volatile organic compounds (NMVOC)	2.48	0.00	0.00	0.00	8.71	10^{-3} kg t^{-1} organic waste
	Polycyclic aromatic hydrocarbons (PAH)	0.00	0.00	10.5	0.00	6.99	10^{-5} kg t^{-1} organic waste
	Total organic compounds (TOC)	2.61	0.00	17.3	3.67	3.12	10^{-3} kg t^{-1} organic waste
	Arsenic (As)	1.34	4.80	19.5	2.08	2.35	10^{-6} kg t^{-1} organic waste
	Cadmium (Cd)	52.4	96.8	9.44	12.5	34.3	10^{-7} kg t^{-1} organic waste
	Chromium (Cr)	6.09	4.19	8.76	3.08	26.9	10^{-7} kg t^{-1} organic waste
	Copper (Cu)	38.2	4.47	3.49	4.52	247	10^{-6} kg t^{-1} organic waste
	Manganese (Mn)	36.3	4.15	1.13	0.00	18.8	10^{-6} kg t^{-1} organic waste
	Mercury (Hg)	4.52	6.09	5.45	3.76	3.92	10^{-6} kg t^{-1} organic waste
	Nickel (Ni)	75.71	5.10	7.06	5.15	18.3	10^{-7} kg t^{-1} organic waste
	Lead (Pb)	9.25	25.7	5.55	3.55	8.20	10^{-6} kg t^{-1} organic waste
	Zinc (Zn)	0.00	0.00	0.00	0.00	2.46	10^{-4} kg t^{-1} organic waste
	Dioxins and furans (PCDD/F)	5.65	1.32	1.61	1.22	3.46	10^{-11} kg t^{-1} organic waste
	Hydrogen chloride (HCl)	11.5	5.52	11.7	8.85	8.52	10^{-3} kg t^{-1} organic waste
	Hydrogen fluoride (HF)	13.9	77.1	213	2.54	58.6	10^{-5} kg t^{-1} organic waste
	Ammonia (NH_3)	1.15	0.00	9.23	6.07	1.26	10^{-2} kg t^{-1} organic waste
	Nitrogen oxides (NOx)	7.08	11.2	17.8	6.57	7.50	10^{-1} kg t^{-1} organic waste
	Nitrous oxide (N_2O)	0.00	0.00	0.00	0.00	6.37	10^{-2} kg t^{-1} organic waste
	Sulphur oxides (SO_X)	1.48	8.52	6.63	4.37	2.38	10^{-2} kg t^{-1} organic waste
	Particles (PM_{10})	4.61	0.00	29.7	6.31	4.82	10^{-3} kg t^{-1} organic waste
	Total suspended particles (TSP)	11.5	189	24	7.32	8.94	10^{-3} kg t^{-1} organic waste
	Waste generation						
	Bottom ashes	9.50	23.6	16.1	18.0	14.04	10^{1} kg t^{-1} organic waste
	Fly ashes	6.59	9.05	4.24	4.59	8.38	10^{1} kg t^{-1} organic waste
ASH SOLIDIFICATION							
Inputs	Cement	26.4	36.2	17.0	18.4	33.5	kg t^{-1} organic waste
	Water	39.5	54.3	25.4	27.5	50.3	kg t^{-1} organic waste
	Ash	65.9	90.5	42.4	45.9	83.8	kg t^{-1} organic waste
Outputs	Inert ash	132	181	84.8	91.8	168	kg t^{-1} organic waste

3. Results and Discussion

The LCIA was conducted following ISO 14040 (ISO, 2006a) and ISO 14044 (ISO, 2006b) requirements using the LCA software GaBi 4 (PE International, 2011). The results were divided into the use of NR and the release of pollutants to different environmental compartments (air, water, and land). The EBs are based on the environmental sustainability metrics developed by the Institution of Chemical Engineers (IChemE) (IChemE, 2002); these metrics provide a balanced view of the environmental burdens of inputs (i.e., resource usage), and outputs (i.e., emissions, effluents, and waste). In particular, the EBs were classified into atmospheric, aquatic, and land burdens. The EBs for air emission air were divided into atmospheric acidification (AA), global warming (GW), human health (carcinogenic) effects (HHE), stratospheric ozone depletion (SOD), and photochemical ozone (smog) formation (POF). The EBs for water emission were defined as aquatic acidification (AqA), aquatic oxygen demand (AOD), ecotoxicity to aquatic life (metals to seawater) (MEco), ecotoxicity to aquatic life (other substances) (NMEco) and eutrophication (EU). The EB to land was given by the amount of generated hazardous and non-hazardous waste (Garcia et al., 2013).

3.1 Natural Resources Sustainability

NRS supports a benchmark comparison in terms of final useful resources including energy, materials, and water. These values were obtained by considering the consumption of energy (X1,1), materials (X1,2), and water (X1,3) in the thermal treatment and flue gas cleaning and during the treatment of slag and ash. Table 5 displays the NRS results normalised to the reference value.

Energy (X1,1) includes the consumption of electricity, diesel, and natural gas and the export of electricity. Negative values are associated with net energy export behaviour because plants are able to export much more energy than the amount obtained from diesel or natural

gas; the net calorific value of the waste is not considered as it was considered an elementary input flow. All of the studied plants showed negative values; nevertheless, greater energy export was observed in I1 and I2. Although no plant presented superior energy performance above the Spanish reference, the X1,1 variable for I2 is very similar to the Spanish value; the plant consumed less combustibles but was able to produce less energy. Conversely, I3 and I4 presented values quite different from the Spanish average, mainly due to the lower energy production of these plants.

The consumption of activated carbon, lime (CaO), calcium hydroxide (Ca(OH)2), urea, and ammonia (NH3) for flue gases cleaning and the consumption of cement for ash solidification were included in the materials variable (X1,2). The consumption of reagents varied in these plants from 0.12 to 1.68 kg t-1 organic waste for activated carbon, from 9.54 to 13.1 kg t-1 organic waste for Ca(OH)2, from 6.52 to 8.40 kg t-1 organic waste for CaO, and from 3.60 to 11.3 kg t-1 organic waste for urea. Conversely, the consumption of cement ranged from 17.0 to 26.4 kg t-1 organic waste depending on the generation of ashes. I3 and I4 had the lowest consumption of materials; in particular, the material consumptions of I1, I3, and I4 were all below the Spanish average. The greatest consumption of materials was observed in I2. Nevertheless, this value should be assessed with respect to the air emissions of the plant so as to determine the efficiency of the flue gas treatment. In this regard, I3 presented a higher rate of emissions of acid gases and NOX per kg of reagent consumed than I2, whereas I1 produced the greatest emission of PCDD/F per kg of reagent consumed. Therefore, plant I3 displays the lowest efficiency of flue gas treatment.

The water variable (X1,2) comprises the consumption of water in the thermal treatment and flue gases cleaning and in ash solidification. For the selected plants, water consumption varied from 155 to 545 kg t-1 organic waste in the incineration process and from 25.44 to 39.5 kg of water t-1 organic waste in the ash solidification process. Plant I4 showed the greatest water consumption linked to the incineration process and the slag cooling. Of all of the incineration plants, only plant I3 presented a lower value than the Spanish average.

Table 5. Comparison of dimensionless NRS variables for the selected WtE plants and the Spanish reference

		Normalised NRS				
Dimensionless NRS variables		***I1***	***I2***	***I3***	***I4***	**Spanish average**
Energy	$X^*_{1,1}$	-0.74	-0.88	-0.14	-0.36	-1.00
Materials	$X^*_{1,2}$	0.83	1.48	0.72	0.58	1.00
Water	$X^*_{1,3}$	0.89	1.27	0.46	1.45	1.00
TOTAL $(X_1 = \frac{\gamma X^*_{1,1}}{3} + \frac{X^*_{1,2}}{3} + \frac{X^*_{1,3}}{3})$	X_1	**0.33**	**0.62**	**0.34**	**0.56**	**0.33**

To obtain a complete overview of incineration consumption in Spain, the Spanish average of NRS was compared with the survey of MSWI facilities included in the document on the best available techniques for waste incineration (BREF document), as shown in Fig. 3 (European Commission, 2006). The references values proposed in the BREF document are -7,760 MJ of energy t-1 waste, 15 kg materials t-1 waste and 250 kg of water t-1 waste.

When the Spanish average was compared with the values included in the BREF document, it was observed that only the variable related to materials (X1,2) presented lower consumption, primarily due to the lower consumption of hydrated lime in the Spanish plants. The plants showed slightly higher water consumption (X1,3); however, these values are close to the European data. The largest difference was observed in the energy variable (X1,1); this was due to the fact that the reference plants included in the BREF document are able to export much more energy than the Spanish plants, and the latter plants do not recover generated heat.

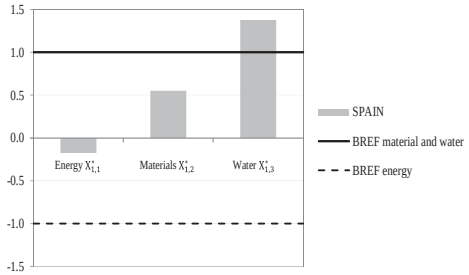

Fig. 3 Comparison of dimensionless NRS variables of Spain and the BREF reference

3.2 Environmental Burdens Sustainability

The EBs to air, water, and land are summarised in Table 6. The results are divided into Cradle to Gate (Cr-Ga), Gate to Gate (Ga-Ga) and Gate to Grave (Ga-Gr). Some values from Table 6 are negative due to the avoided electricity production allocated to the Ga-Ga stage. The characterisation factor from the TRACI method (3.13 108 kg benzene eq./kg PCDD/F) (Bare et al., 2002) was used to incorporate the effect due to the release of PCDD/F in the Human Health Effects (HHE) impact category because it was not originally defined in the reference metrics (IChemE, 2002). In most of the metrics of the air and water compartments, the Cr-Ga step presented the highest burdens; however, the Ga-Gr stage also contributed significantly; in particular, the burdens were of the same order of magnitude in several categories. In the land compartment only, the Ga-Gr stage generated non-hazardous waste, while the Cr-Ga and Ga-Ga steps were associated with a null contribution.

The production and consumption of reagents for flue gas cleaning, fuels and ancillary materials for the thermal treatment are the most significant contributors to the Cr-Ga stage. Overall, the production and consumption of lime and slaked lime for treatment of acid gases generated higher air emissions of CO2 and CO which contribute to global warming (GW), whereas the release of H2SO4 to water contributed to Aquatic Acidification (AqA). The air emissions of acid gases such as SOX, H2SO4, HCl, and HF, CFCs, and other organic compounds during the manufacture of urea and ammonia contributed to Atmospheric Acidification (AA), Stratospheric Ozone

Depletion (SOD) and Photochemical Ozone Formation (POF). In HHE, although urea production caused high emissions of most of the pollutants contributing to HHE, the production of treated water emitted a larger amount of dust to air presenting a greater burden to HHE. In the water categories, the emission of methanol in the production and consumption of ammonia and urea for NOX cleaning contributed to Aquatic Oxygen Demand (AOD). Moreover, the release of pollutants such as ammonia, nitrogen, phosphorus and chemical oxygen demand (COD) to water in ammonia production gave the highest contribution to Eutrophication (EU). With respect to Ecotoxicity to Aquatic Life by Metals (MEco) and Ecotoxicity to Aquatic Life by Non-Metal substances (NMEco), the consumption of diesel, lime or slaked lime, and urea contributed the most due to emissions of Zn and Cu, benzene and xylene to the aquatic media.

Although the Ga-Ga analysis linked with waste combustion process had a high impact in the air categories, the avoided burden that was associated with electricity production compensated for these impacts in most of those categories. From an environmental point of view, without any energy recovery, the Ga-Ga would be the worst stage. In the water categories, waste combustion has a null influence because water emissions were not generated in these plants.

Ga-Gr burdens are associated with the treatment of ash. Among ash solidification processes, cement production and consumption made the largest contribution. Cement manufacturing is an industry that consumes a lot of energy, particularly in the decarbonation and clinkering of raw materials (Margallo et al., 2014a).

For all the WtE plants, the Cr-Ga step displayed the highest burdens in most of the air categories and in all the water categories. Nevertheless, the Ga-Gr stage showed burdens that were 2.7 and 1.8 times higher than those obtained in the Cr-Ga step in the category of HHE for I1 and I4, respectively. Moreover, for the latter plant, in the Cr-Ga step a burden, 2 times higher than in the Cr-Ga stage was observed in SOD. The HHE and SOD burdens in the Ga-Gr analysis were the result of the air emissions of CFCs, HCFCs, heavy metals, dust, and PCDD/F in the consumption of energy and coke in the clinkering process for cement production.

In contrast to the Ga-Gr analysis, the Ga-Ga analysis showed a burden 53 times higher than in the Cr-Ga step in the HHE category for I2, whereas for I3, burdens 1.5 and 13 times higher were obtained in the categories of GW and HHE, respectively. The high impact in the Ga-Ga step in these air categories for I2 and I3 is associated with the emission of greenhouse gases, particularly NOX and CO, that contributed to GW and the emissions of PCDD/F, dust, and heavy metals, specifically As and Cd that contributed to HHE. CO2 emissions were not generated in the organic waste incineration because this waste flow has a null content of fossil carbon. Emissions of NOX depend on the applied technology rather than on the waste composition. Therefore, the high emission of nitrogen oxides by these plants can be associated with poor combustion processes and poor NOX cleaning. NOX is usually formed during combustion in which part of the nitrogen contained in the MSW is oxidised to NOX, but it can also be formed during combustion in which a part of the nitrogen in air is oxidised to NOX (Tillman et al., 1989). NOX can be reduced using furnace control measures that prevent oversupply of air and prevent the use of unnecessarily high furnace temperatures (Pickens, 1996). Moreover, the use of Selective Catalytic reduction (SCR), a more advanced technology than SNCR for reducing NOX to N2, could be useful in reducing the environmental impact. However, in this case, it would be necessary to assess whether the production and the periodic maintenance of the catalyst could be environmentally advantageous compared to the non-catalytic system (Morselli et al., 2007). Similarly, PCDD/F emissions are thought to depend more on operational conditions and treatment technologies related to combustion and flue gases than on the Cl content of the input waste (Margallo et al., 2014b). High levels of PCDD/F formation are associated with poor combustion conditions, feeding of problematic materials or the operation of dust collectors at high temperatures (PNUMA, 2005). To minimise PCDD/F formation, both good combustion and reduction of the time during which flue gases are subjected to temperatures in the range of 400ºC to 200ºC are required. Particularly, WtE plants should be operated at a temperature of 850ºC for two seconds to achieve good burnout of the gases (EC, 2000). Heavy metals and CO emissions depend on the input waste composition (i.e., on the heavy metal and carbon content of the waste). Therefore, the release of large amounts of CO and heavy metals to the atmosphere is related to the presence of high levels of carbon and heavy metal in the input waste. These results are consistent with the NRS values and demonstrate that I3 was the incineration plant with the poorest flue gas cleaning system.

To reduce air emissions, environmentally friendly technologies can be used. To evaluate these techniques the tripod formed by the availability of the technology, economic cost, and environmental issues should be considered; a review of the Best Available Techniques (BAT) compiled in the BREF document is essential. Compared with GI, FB reduces NOX emissions due to the lower temperature and more uniform temperature distribution, which eliminate hotspot and high oxygen zones. Additionally, FB reduces the emissions of SOX because the reagent is added in the bed. However, this technique has a higher energy consumption and higher investment costs. Other thermal technologies such as gasification and pyrolysis generate lower flue gases volume than conventional incineration and produce low-leaching slag. Nevertheless, they are a less proven technology, which are applied to selected waste streams and on smaller scales due to the higher operation and maintenance costs and the requirement of a special pre-treatment for MSW.

In FGT, wet systems generally have the highest absorption capacities and deliver the lowest emission levels for acid gases, but are generally more expensive. Waste and combustion control techniques coupled with SCR generally result in lower emission ranges than SNCR. However, the use of SCR imposes an additional energy demand and associated costs (EC-IPPC 2006).

Table 6. Environmental burdens of the WTE plants I_1, I_2, I_3, and I_4

			Cr-Ga	Ga-Ga		Ga-Gr		Cr-Ga	Ga-Ga		Ga-Gr	
			Manu-facturing of raw materials & supplies	Thermal and flue gas treat-ment	Avoided burden	Ash treat-ment	Slag land-filling	Manu-facturing of raw materials & supplies	Thermal and flue gas treat-ment	Avoided burden	Ash treat-ment	Slag land-filling
			I_1					I_2				
Air EB $X_{2,1}$	AA $X_{2,1,1}$	10^{-3} kg SO_2 eq. t^{-1} org. waste	8.61	36.4	-1,038	4.23	0.00	31.5	85.17	-1,259	5.81	0.00
	GWP $X_{2,1,2}$	10^{1} kg CO_2 eq. t^{-1} org. waste	1.80	2.86	-20.3	0.84	0.00	3.83	4.51	-24.6	1.16	0.00
	HHE $X_{2,1,3}$	10^{-3} kg benzene eq. t^{-1} org. waste	2.50	39.7	-39.1	6.71	0.00	5.06	314	-47.4	9.22	0.00
	POF $X_{2,1,4}$	10^{-4} kg CFC-11 eq. t^{-1} org. waste	19.1	29.5	-746	8.49	0.00	84.4	59.57	-906	11.7	0.00
	SOD $X_{2,1,5}$	10^{-7} kg ethylene eq. t^{-1} org. waste	5.33	0.00	-363	2.78	0.00	18.11	0.00	-440	3.81	0.00
Water EB $X_{2,2}$	AOD $X_{2,2,1}$	10^{-5} kg H^+ eq. t^{-1} org. waste	1,333	0.00	-19.1	0.92	0.00	4,198	0.00	-23	1.27	0.00
	AqA $X_{2,22}$	10^{-8} kg O_2 eq. t^{-1} org. waste	94.9	0.00	-2.93	67.0	0.00	114	0.00	-3.56	92.0	0.00
	MEco $X_{2,2,3,1}$	10^{-8} kg Cu eq. t^{-1} org. waste	28.3	0.00	-1,086	11.5	0.00	113	0.00	-1,318	15.7	0.00
	NMEco $X_{2,2,3,2}$	10^{-8} kg formal. eq. t^{-1} org. waste	18.7	0.00	-394	7.36	0.00	62.2	0.00	-478	10.1	0.00
	Eutroph $X_{2,2,4}$	10^{-5} kg PO4 eq. t^{-1} org. waste	16.4	0.00	-109	2.01	0.00	40.4	0.00	-132	2.76	0.00
Land EB $X_{2,3}$	HW $X_{2,3,1}$	kg hazardous waste t^{-1} org. waste	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	NHW $X_{2,3,2}$	kg non-hazardous waste t^{-1} org. waste	0.00	0.00	0.00	132	95.0	0.00	0.00	0.00	181	236
			I_3					I_4				
Air EB $X_{2,1}$	AA $X_{2,1,1}$	10^{-3} kg SO_2 eq. t^{-1} org. waste	7.41	240	-255	2.72	0.00	3.90	158	-521	2.95	0.00
	GWP $X_{2,1,2}$	10^{1} kg CO_2 eq. t^{-1} org. waste	1.45	7.13	-5.00	0.54	0.00	1.12	2.64	-10.2	0.59	0.00
	HHE $X_{2,1,3}$	10^{-3} kg benzene eq. t^{-1} org. waste	2.92	47.0	-9.62	4.32	0.00	2.57	16.6	-19.6	4.68	0.00
	POF $X_{2,1,4}$	10^{-4} kg CFC-11 eq. t^{-1} org. waste	18.1	55.7	-184	5.46	0.00	7.88	31.47	-375	5.91	0.00
	SOD $X_{2,1,5}$	10^{-8} kg ethylene eq. t^{-1} org. waste	28.4	0.00	-893	17.9	0.00	9.03	0.00	-1,824	19.3	0.00
Water EB $X_{2,2}$	AOD $X_{2,2,1}$	10^{-5} kg H^+ eq. t^{-1} org. waste	624	0.00	-4.71	0.59	0.00	262	0.00	-9.62	0.64	0.00
	AqA $X_{2,2,2}$	10^{-9} kg O_2 eq. t^{-1} org. waste	876	0.00	-7.22	431	0.00	741	0.00	-14.7	467	0.00
	MEco $X_{2,2,3,1}$	10^{-8} kg Cu eq. t^{-1} org. waste	120	0.00	-267	7.37	0.00	23.48	0.00	-546	8.0	0.00
	NMEco $X_{2,2,3,2}$	10^{-8} kg formal. eq. t^{-1} org. waste	73.7	0.00	-96.9	4.74	0.00	13.11	0.00	-198	5.13	0.00
	Eutroph $X_{2,2,4}$	10^{-5} kg PO_4 eq. t^{-1} org. waste	8.75	0.00	-26.8	1.29	0.00	13.54	0.00	-54.7	1.40	0.00
Land EB $X_{2,3}$	HW $X_{2,3,1}$	kg hazardous waste t^{-1} org. waste	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	NHW $X_{2,3,2}$	kg non-hazardous waste t^{-1} org. waste	0.00	0.00	0.00	84.8	161	0.00	0.00	0.00	91.8	180

Table 7. Normalised results of the incineration plants and threshold values from the E-PRTR regulation

		Threshold value (kg y^{-1})	Factor	Normalised results (dimensionless)				
				I_1	I_2	I_3	I_4	Spain average
Air EB	AA $X^*_{2,1,1}$	150,000	10^{-7}	-65.9	-75.8	-0.36	-23.8	-89.0
	GW $X^*_{2,1,2}$	100,000,000	10^{-7}	-14.8	-15.1	4.13	-5.85	-19.5
	HHE $X^*_{2,1,3}$	1,000	10^{-6}	9.86	281	44.7	4.26	-15.0
	POF $X^*_{2,1,4}$	1,000	10^{-5}	-6.89	-7.50	-1.04	-3.30	-9.68
	SOD $X^*_{2,1,5}$	1.00	10^{-6}	35.5	-41.8	-8.47	-18.0	-49.6
	Total EB to air ($X_{2,1}$)		**10^{-6}**	**-103**	**155**	**25.5**	**-49.6**	**-172**
Water EB	AOD $X^*_{2,2,1}$	50,000	10^{-8}	26.3	83.5	12.4	5.07	34.2
	AqA $X^*_{2,2,2}$	100	10^{-8}	1.59	2.02	1.30	1.19	2.21
	MEco $X^*_{2,2,3,1}$	50	10^{-8}	-20.9	-23.8	-2.79	-10.3	-28.7
	NMEco $X^*_{2,2,3,2}$	50	10^{-8}	-7.36	-8.11	-0.37	-3.59	-10.0
	EU $X^*_{2,2,4}$	5,000	10^{-8}	-18.1	-17.8	-3.35	-7.94	-26.1
	Total EB to water ($X_{2,2}$)		**10^{-6}**	**-0.18**	**0.36**	**0.07**	**-0.16**	**-0.28**
Land EB	HW $X^*_{2,3,1}$	2,000						
	NHW $X^*_{2,3,2}$	2,000,000	10^{-6}	113	208	123	136	154
	Total EB to land ($X_{2,3}$)		**10^{-6}**	**113**	**208**	**123**	**136**	**154**
Total EB	**X_2**		**10^{-6}**	**10.6**	**364**	**148**	**86.6**	**-18.2**

To compare the EBs to air, water, and land, the threshold values stated in the E-PRTR were used as weighting factors to obtain dimensionless burdens (Margallo et al., 2014a). The average EB of Spain was calculated with inputs and outputs data collected from the 10 Spanish WtE plants. LCI was obtained from Ecoembes, AEVERSU, the Spanish PRTR, and the IPPC permit of the plants. Table 7 displays the normalised results and the threshold values proposed in the E-PRTR. The factor provided in Table 7 is the multiplicative factor for the normalised results.

In the air compartment, only HHE showed a positive value in all the plants, indicating that this category had the greatest influence due to the emissions of PCDD/F, heavy metals, and dust that occur during waste combustion.
The analysis of this compartment depicted the highest total burden to air, with positive values for I2 and I3, whereas I4 and I1 displayed negative values. This was due to the larger amount of energy produced by I1 and therefore also the higher avoided burden generated, but also to the lower amount of materials consumed by I4. In particular, in all air categories except HHE, in which a greater value was observed in I2, I3 showed the highest impact. This was due to the high emissions of PCDD/F and heavy metals generated by I2. Nevertheless, the total impact to air (X2,1) of I2 was 6.0 times higher than the burden produced by I3; thus, I2 emerged as the plant with the worst air performance.
In the water categories, only AqA and AOD displayed positive values. For I2 and I3, AOD contributed the most to the total water burden. Nevertheless, for I1 and I4, MEco had the highest influence on the total EB to water (X2,2). Specifically, I1 and I4 showed negative values in the total EB to water because a lower burden was obtained in several categories in the Cr-Ga and Ga-Gr steps, and in the Ga-Ga analysis thermal treatment did not generate water emissions, thereby contributing only the avoided burden to this step. The largest total water burden was produced by I2; particularly, this plant presented the highest burdens in AOD and AqA. This was due to the fact that plant I2 showed the greatest consumption of urea and lime; as previously described, these reagents have a strong influence in these categories. In the remaining categories, I3 produced the greatest burden; however, the high influence of AOD in X2,2 made up for these burdens. Specifically, the total water burden from I2 was 9 times higher than that of I3.
Land burden (X2,3) had a positive impact in all of the WtE plants because there were no avoided burdens and because wastes were considered as final flows. The highest burden to land was generated by I2, mainly due to the larger amount of slag and ash generated. In contrast, I1 showed the best land performance with a burden 1.8 times lower than that produced by I2.
The EB and NR are classified in Fig. 4 according to their intensities. Values near the symbol "+" indicate the highest burden, whereas "-" represents the lowest burden. A similar performance in all the environmental compartments was observed in all the studied plants. In particular, I2 presented the highest burden to air, water, and land, whereas I1 displayed the best environmental performance in all compartments. The comparison of EB and NR emphasises the great interdependence of these variables because the plant with the greatest consumption of NR produced the highest EB.

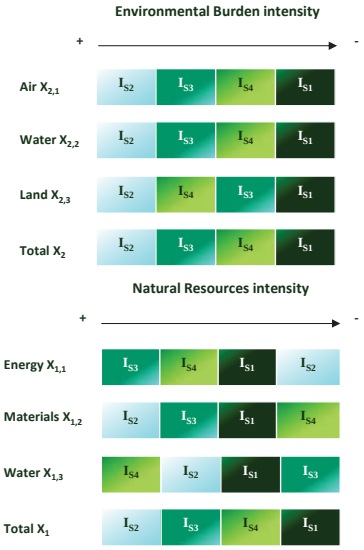

Fig. 4 Intensity of the natural resources and environmental burdens

Finally, to analyse the overall environmental performance of the WtE plants under study, the EB to air (X2,1), EB to water (X2,2), and EB to land (X2,3) are compared to the average EB of Spain in Fig. 5. All the plants present a higher EB to air and water than the Spanish average; this is even more notable for I2 and I3. In the land compartment, only I2 exceeds the burden of Spain, whereas the rest of the plants have values similar to the Spanish average. Generally, the plants under study show worse environmental performance in the air and water compartments and a lower land impact.

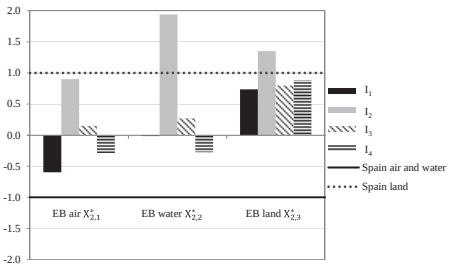

Fig. 5 Air, water, and land EB of the incineration plants, taking the Spanish average EB as a reference

4. Conclusions

LCA has been shown to be a useful tool for conducting an ESA of several WtE plants in Spain. In this regard, the proposed methodology reduces the complexity of LCA and simplifies decision-making through the use of two dimensionless variables, NR X_1^* and the EB X_2^*. These two variables can feed a multi-objective function in which

environmental sustainability is described by two indices. The normalisation and weighting procedure to obtain the EBS variable introduces a policy weighting based on the threshold values of the E-PRTR regulation. Therefore, the WtE plants could compare its environmental performance within the European regulations and the status regarding the Best Available Techniques (BAT) complied in the BREF document.

However, although normalisation and weighting of the EB to obtain a single index renders the decision-making process easier, the use of a global EB could potentially mask certain results. In this sense, a plant with a high burden in one environmental compartment and a low or even negative value in the remaining compartments can make up for the burdens in those compartments and thereby present the best environmental performance. For this reason, the comparison of several plants only by means of a global index can be used to obtain an overview of the environmental performance of the plant and to optimise the process making process. However, individual EBs should also be assessed to determine the main environmental problems and thus provide opportunities for improvement of the critical environmental points of the process.

The outcomes of this study were strongly influenced by the definition of the system boundaries and the data quality. Specifically, in this work, the results would be quite different if energy production had been excluded from the study. In terms of NRS, the plants under study surpassed the X_1^* index of Spain or presented values similar to the Spanish reference data. In particular, this index ranges from 1.1 to 2.0 times the Spanish average. Likewise, when the consumption of NR of Spain was compared with the BREF data, only the variable related to materials ($X_{1,2}^*$) showed a lower value. Therefore, controlling water consumption and improving energy production in Spanish plants will improve their environmental performance and thus the economic aspects of incineration in the country. With respect to the environmental burdens, most of the plants showed land burdens that were similar to the average for Spain. The air compartment ($X_{2,1}^*$) was controlled by HHE due to the emission of PCDD/F, heavy metals, and dust, whereas MEco and AOD presented the greatest contribution to the water compartment due to the high influence of energy production. All the plants applied the same thermal treatment, but they differed in the levels of combustible and in the reagents used for the treatment of flue gases. Therefore, to minimise the EB, it is necessary to optimise the operational conditions and efficiency of flue gases treatment. In this sense, the use of less polluting reagents such as ammonia and slaked lime instead of urea and lime will also contribute to reducing the burden.

Finally, it is remarkable that corporations with an Environmental Management System (EMS) could use this methodology to define its environmental objectives and targets, and to evaluate the degree of compliance of these objectives. The proposed indicators facilitate the dissemination of the performance and progress of industrial plants from an environmental point of view. Moreover, on the basis of these metrics, corporations could develop its environmental policy, a report required in EMS that includes commitments to continually improve the environmental performance, to comply with environmental legislation, and to educate and train employees to enable them to work within the policy.

Acknowledgments

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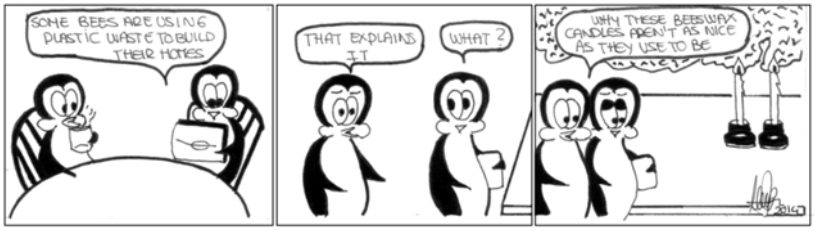
ANNEXES

Life cycle inventory and dissemination of results



Adapted from Alex Hallat

In this Annex are summarised the following aspects of the life cycle inventory of the WtE plants in the Iberian Peninsula:

- Waste composition of the WtE plants in Spain and Portugal.
- Carbon, heavy metals, sulphur, fluorine, and chlorine content of the MSW.
- Input and output data of the Spanish and Portuguese WtE plants.

A1.1 WASTE COMPOSITION

Table A1.1 Waste composition of the Spanish WtE plants.

	Spanish WtE plants									
	I_{S1}	I_{S2}	I_{S3}	I_{S4}	I_{S5}	I_{S6}	I_{S7}	I_{S8}	I_{S9}	I_{S10}
PACKAGING FRACTION										
PET	1.96	1.17	1.89	2.07	2.10	3.89	2.36	2.20	1.51	3.16
HDPE	1.13	0.95	1.34	1.36	0.87	2.69	1.00	1.23	1.38	1.90
LDPE	4.44	2.81	4.91	4.73	5.76	8.89	10.1	7.48	4.55	4.69
Plastic mix	2.10	1.70	2.23	2.19	2.49	3.08	2.06	1.85	1.41	0.96
Steel	2.02	1.28	2.17	2.05	1.93	2.87	1.75	2.03	0.44	2.30
Aluminium	$3.33\ 10^{-1}$	$2.50\ 10^{-1}$	$4.5\ 10^{-1}$	$3.00\ 10^{-1}$	$3.50\ 10^{-1}$	$2.80\ 10^{-1}$	$4.6\ 10^{-1}$	$4.00\ 10^{-1}$	$8.00\ 10^{-2}$	$1.80\ 10^{-1}$
Beverage carton	1.04	$5.70\ 10^{-1}$	1.23	$9.30\ 10^{-1}$	1.07	2.63	1.91	1.37	1.46	1.80
Glass	4.37	5.38	4.94	4.08	4.07	3.98	$6.80\ 10^{-1}$	4.22	$5.10\ 10^{-1}$	3.25
PC	7.58	4.47	4.63	6.15	4.58	6.96	10.8	8.01	6.54	10.5
Wood	$7.8\ 10^{-1}$	$5.7\ 10^{-1}$	$4.6\ 10^{-1}$	$2.5\ 10^{-1}$	$3.5\ 10^{-1}$	$1.23\ 10^{-1}$	$1.33\ 10^{-1}$	$4.10\ 10^{-1}$	$1.00\ 10^{-2}$	6.8010^{-1}
REST OF MATERIALS										
Non packaging										
HDPE nP	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
LDPE nP	1.11	1.48	2.23	2.28	2.56	$8.40\ 10^{-1}$	2.07	1.15	$3.5\ 10^{-1}$	3.17
Pl mix nP	3.21	3.19	2.06	2.72	2.79	1.88	3.36	2.68	1.53	1.80
Steel nP	1.00	1.44	$9.20\ 10^{-1}$	2.06	$6.70\ 10^{-1}$	$6.00\ 10^{-1}$	$5.30\ 10^{-1}$	$7.90\ 10^{-1}$	$1.40\ 10^{-1}$	1.16
Al nP	$8.00\ 10^{-2}$	$7.00\ 10^{-2}$	$3.00\ 10^{-2}$	$7.00\ 10^{-2}$	$6.00\ 10^{-2}$	$8.00\ 10^{-2}$	$2.7\ 10^{-1}$	$7.00\ 10^{-2}$	$4.00\ 10^{-2}$	$3.00\ 10^{-2}$
Glass nP	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
PC non nP	5.73	4.81	5.11	6.37	4.81	4.09	8.29	6.33	28.4	4.96
Wood nP	1.52	1.77	1.24	1.74	1.14	2.15	2.83	1.30	2.20	2.00

Table A1.1 (cont.) Waste composition of the Spanish WtE plants.

	Spanish WtE plants									
	I_{S1}	I_{S2}	I_{S3}	I_{S4}	I_{S5}	I_{S6}	I_{S7}	I_{S8}	I_{S9}	I_{S10}
Other fractions										
Organic matter	36.0	29.0	36.1	32.5	32.5	33.47	9.84	31.3	3.67	37.0
Garden rest	2.54	18.7	3.74	3.18	6.80	1.35	7.83	5.39	1.54	3.11
Cellulose	5.28	5.66	8.05	9.63	8.94	2.68	10.3	6.73	4.65	7.35
Textiles	10.6	5.96	8.06	7.99	6.04	6.92	10.9	6.21	31.6	5.54
Construction and demolition waste	1.94	1.13	1.52	1.00	1.08	$5.50\ 10^{-1}$	1.35	1.03	$1.70\ 10^{-1}$	$6.40\ 10^{-1}$
Other	1.57	4.20	2.98	2.59	3.89	5.21	1.42	2.71	2.60	1.67
INPUT 2009 (tons)	**343,925**	**136,477**	**30,158**	**141,610**	**294,185**	**212,937**	**113,114**	**526,525**	**298,900**	**34,769**

Table A1.2 Waste composition of the Portuguese WtE plants.

	PORTUGUESE INCINERATORS		
	I_{P1}	I_{P2}	I_{P3}
PACKAGING FRACTION			
PET	$7.00\ 10^{-1}$	1.22	$7.90\ 10^{-1}$
HDPE	$5.50\ 10^{-1}$	$7.50\ 10^{-1}$	$4.80\ 10^{-1}$
LDPE	5.13	7.84	7.56
Plastic mix	2.34	3.58	$6.30\ 10^{-1}$
Steel	1.09	1.41	1.23
Aluminium	$3.70\ 10^{-1}$	$3.80\ 10^{-1}$	$2.00\ 10^{-1}$
Beverage carton	1.02	1.46	$7.70\ 10^{-1}$
Glass	5.50	4.47	3.19
PC	5.06	5.87	3.97
Wood	$1.70\ 10^{-1}$	$6.50\ 10^{-1}$	$9.00\ 10^{-2}$
REST OF MATERIALS			
Non packaging waste			
HDPE non packaging			
LDPE non packaging			
Plastic mix non packaging	$7.60\ 10^{-1}$	$8.30\ 10^{-1}$	$7.90\ 10^{-1}$
Steel non packaging	$3.00\ 10^{-1}$	$2.60\ 10^{-1}$	$1.40\ 10^{-1}$
Aluminium non packaging	$1.10\ 10^{-1}$	$2.30\ 10^{-1}$	$5.80\ 10^{-1}$
Glass non packaging	$3.80\ 10^{-1}$	$8.30\ 10^{-1}$	$5.00\ 10^{-2}$
Paper and Cardboard non packaging	12.85	2.92	3.45
Wood non packaging	$6.20\ 10^{-1}$	$2.00\ 10^{-1}$	$5.70\ 10^{-1}$
Other fractions			
Organic matter	32.54	33.05	28.97
Garden rest	2.13	4.45	19.56
Cellulose			
Textiles	4.06	2.76	8.44
Construction and demolition waste			
Other			
INPUTS (tons)	**662,000**	**126,000**	**400,000**

Table A1.3 Average waste composition of the Spanish and Portuguese WtE plants.

	SPANISH AVERAGE	PORTUGUESE AVERAGE	IBERIAN PENINSULA AVERAGE[1]
PACKAGING FRACTION			
PET	2.23	$9.00\ 10^{-1}$	1.92
HDPE	1.38	$5.90\ 10^{-1}$	1.20
LDPE	5.84	6.84	6.07
Plastic mix	2.01	2.18	2.05
Steel	1.88	1.24	1.74
Aluminium	$3.10\ 10^{-1}$	$3.20\ 10^{-1}$	$3.10\ 10^{-1}$
Beverage carton	1.40	1.08	1.33
Glass	3.55	4.39	3.74
PC	7.02	4.97	6.55
Wood	$6.10\ 10^{-1}$	$3.00\ 10^{-1}$	$5.40\ 10^{-1}$
REST OF MATERIALS			
Non packaging waste			
HDPE non packaging	0.00		0.00
LDPE non packaging	1.72		1.72
Plastic mix non packaging	2.52	$7.90\ 10^{-1}$	2.12
Steel non packaging	$9.30\ 10^{-1}$	$2.30\ 10^{-1}$	$7.70\ 10^{-1}$
Aluminium non packaging	$8.00\ 10^{-2}$	$3.10\ 10^{-1}$	$1.30\ 10^{-1}$
Glass non packaging	0.00	$4.20\ 10^{-1}$	$4.20\ 10^{-1}$
Paper and Cardboard non packaging	7.89	6.41	7.55
Wood non packaging	1.79	$4.60\ 10^{-1}$	1.48
Other fractions			
Organic matter	28.1	31.52	28.91
Garden rest	5.42	8.71	6.18
Cellulose	6.93		6.93
Textiles	9.98	5.09	8.85
Construction and demolition waste	1.04		1.04
Other	2.88		2.88
INPUTS (tons)	**213,260**	**396,000**	**255,431**

[1] *The average value of the Iberian Peninsula was calculated as the average value of the Spanish and Portuguese incinerators (I) as:* $(I_{S1}+I_{S2}+I_{S3}+I_{S4}+I_{S5}+I_{S6}+I_{S7}+I_{S8}+I_{S9}+I_{S10}+I_{P1}+I_{P2}+I_{P3})/13$

A1.2 CARBON CONTENT OF WASTE

Table A1.4 Carbon content of MSW (Bjarnadóttir et al. 2002).

	Proportion of C (total) (g/kg dry matter)	Proportion of fossil C (g/kg dry matter)	Proportion of biological C (g/kg dry matter)	Proportion of fossil C (%)
PACKAGING FRACTION				
PET	640	640	0.00	100
HDPE	856	856	0.00	100
LDPE	855.5	855.5	0.00	100
Plastic mix	590	590	0.00	100
Steel	0.00	0.00	0.00	0.00
Aluminium	0.00	0.00	0.00	0.00
Beverage carton	500	125	375	25.0
Glass	0.00	0.00	0.00	0.00
Paper and cardboard	433	0.00	433	0.00
Wood	495	0.00	495	0.00
REST OF MATERIALS				
Non packaging fractions				
HDPE non packaging	856	856	0.00	100
LDPE non packaging	855.5	855.5	0.00	100
Plastic mix non packaging	590	590	0.00	100
Steel non packaging	0.00	0.00	0.00	0.00
Aluminium non packaging	0.00	0.00	0.00	0.00
Glass non packaging	0.00	0.00	0.00	0.00
Paper and cardboard non packaging	422	0.00	422	0.00
Wood non packaging	495	0.00	495	0.00
Other fractions				
Organic matter	434	0.00	434	0.00
Garden and pruning rest	500	0.00	500	0.00
Cellulose	479	0.00	479	0.00
Textiles	556	278	278	50.0
Construction and demolition waste	0.00	0.00	0.00	0.00
Others	0.00	0.00	0.00	
Total	**9.557**	**5.646**	**3.911**	

A1.3 HEAVY METALS CONTENT OF WASTE

Table A1.5 Heavy metals content of MSW (Riber et al. 2009).

	Heavy metals content (mg/kg dry fraction)								
	As	**Cd**	**Cr**	**Cu**	**Hg**	**Ni**	**Pb**	**Zn**	**Mn**
PACKAGING FRACTION									
PET	$2.00\ 10^{-1}$	$3.40\ 10^{-2}$	24.3	34.3	$2.00\ 10^{-2}$	4.53	4.35	87.3	195
HDPE	$4.78\ 10^{-1}$	$3.51\ 10^{-1}$	73.2	224	$2.00\ 10^{-2}$	4.66	319	331	13.6
LDPE	$2.00\ 10^{-1}$	$3.40\ 10^{-2}$	8.14	39.2	$4.34\ 10^{-2}$	1.06	26.2	95.3	7.65
Plastic mix	$2.00\ 10^{-1}$	$3.40\ 10^{-2}$	2.87	94.8	$1.98\ 10^{-2}$	4.44	1.28	74	10.1
Steel	25.0	1.49	172	420	$2.00\ 10^{-1}$	152	24.1	275	3540
Aluminium	9.13	$4.71\ 10^{-1}$	109	892	$2.00\ 10^{-1}$	61.2	18.4	101	5140
Beverage carton	$2.00\ 10^{-1}$	$2.70\ 10^{-2}$	2.89	17.6	$2.00\ 10^{-2}$	2.76	1.76	11.3	15.7
Glass	4.35	$1.25\ 10^{-1}$	412	9.08	$1.00\ 10^{-1}$	153.725	77	38.4	118
Paper and cardboard	$2.94\ 10^{-1}$	$9.80\ 10^{-2}$	32.7	135	$6.61\ 10^{-2}$	28.2	11	83.4	36.8
Wood	$3.05\ 10^{-1}$	$3.41\ 10^{-1}$	34.1	34.3	$2.13\ 10^{-1}$	4.17	18.1	436	248
REST OF MATERIALS									
Non packaging waste									
HDPE non packaging	$4.78\ 10^{-1}$	$3.51\ 10^{-1}$	73.2	224	$2.00\ 10^{-2}$	4.66	319	331	13.6
LDPE non packaging	$2.00\ 10^{-1}$	$3.40\ 10^{-2}$	8.14	39.2	$4.34\ 10^{-2}$	1.06	26.2	95.3	7.65
Plastic mix non packaging	$2.00\ 10^{-1}$	$3.40\ 10^{-2}$	2.87	94.8	$1.98\ 10^{-2}$	4.44	1.28	74	10.1
Steel non packaging	2.00	$1.00\ 10^{-1}$	1.01	65.6	$2.00\ 10^{-1}$	8.00	1.33	15.5	6.29
Aluminium non packaging	2.00	$5.48\ 10^{-1}$	57.9	674	$2.00\ 10^{-1}$	201	67.0	292	4,270
Glass non packaging	4.35	$1.25\ 10^{-1}$	412	9.08	$1.00\ 10^{-1}$	154	77.0	38.4	129
PC non packaging	$4.83\ 10^{-1}$	$7.40\ 10^{-2}$	15	41.9	$3.31\ 10^{-2}$	6.58	2.68	83.0	61.1
Wood non packaging	$3.05\ 10^{-1}$	$3.41\ 10^{-1}$	34.1	34.3	$2.13\ 10^{-1}$	4.17	18.1	436	248

Table A1.5 (cont.) Heavy metals content of MSW (Riber et al. 2009).

	Heavy metals content (mg/kg dry fraction)								
	As	**Cd**	**Cr**	**Cu**	**Hg**	**Ni**	**Pb**	**Zn**	**Mn**
Other fractions									
Organic matter	$2.62\ 10^{-1}$	$9.50\ 10^{-2}$	5.24	12.5	$2.00\ 10^{-2}$	2.57	1.04	25	86.1
Garden and pruning rest	$9.41\ 10^{-1}$	$3.57\ 10^{-1}$	4.51	20.2	$2.62\ 10^{-1}$	3.24	24.4	208	115
Cellulose	$2.00\ 10^{-1}$	$2.0\ 10^{-2}$	5.82	52.9	$2.00\ 10^{-2}$	2.64	4.32	42.8	17.2
Textiles	$2.00\ 10^{-1}$	$4.87\ 10^{-1}$	475	21.4	$8.47\ 10^{-2}$	1.47	149	211	15.1
Construction and demolition waste	2.85	$1.20\ 10^{-1}$	13.1	13.2	0.00	8.12	12.59	36.44	456

A1.4 SULPHUR, FLUORINE, AND CHLORINE CONTENT OF WASTE

Table A1.6 S, F, and Cl content of MSW (Riber et al. 2009).

	F, Cl, and S content		
	F (mg/kg dry fraction)	**Cl (mg/kg dry fraction)**	**S (%)**
PACKAGING FRACTION			
PET	100	1,700	$5.00\ 10^{-2}$
HDPE	100	1,000	$5.00\ 10^{-2}$
LDPE	100	700	$5.00\ 10^{-2}$
Plastic mix	100	46.1	0.00
Steel	0.00	0.00	0.00
Aluminium	0.00	0.00	0.00
Beverage carton	100	1,100	1.0010^{-1}
Glass	0.00	0.00	$4.00\ 10^{-2}$
Paper and cardboard	400	300	$1.90\ 10^{-1}$
Wood	100	1,400	$4.00\ 10^{-2}$
REST OF MATERIALS			
Non packaging waste			
HDPE non packaging	100	1000	$5.00\ 10^{-2}$
LDPE non packaging	100	700	$5.00\ 10^{-2}$
Plastic mix non packaging	100	46.1	0.00
Steel non packaging	0.00	0.00	0.00
Aluminium non packaging	100	2,400	0.00
Glass non packaging	0.00	0.00	$4.00\ 10^{-2}$
PC non packaging	100	300	$1.80\ 10^{-1}$
Wood non packaging	100	1,400	$4.00\ 10^{-2}$
Other fractions			
Organic matter	100	5,600	$8.00\ 10^{-2}$
Garden and pruning rest	100	2,800	$1.30\ 10^{-1}$
Cellulose	100	2,600	$5.00\ 10^{-2}$
Textiles	100	3,500	$1.80\ 10^{-1}$
Construction and demolition waste	0.00	10.7	

A1.5 MEDIUM INPUTS AND OUTPUTS DATA OF THE WtE PLANTS

Table A1.7 Medium input and output data of the Spanish WtE plants.

	SPANISH INCINERATORS									
	I_{S1}	I_{S2}	I_{S3}	I_{S4}	I_{S5}	I_{S6}	I_{S7}	I_{S8}	I_{S9}	I_{S10}
Consumption of auxiliary materials and combustibles (kg/t MSW)										
Diesel	$2.67\ 10^{-3}$		1.19	$4.12\ 10^{-1}$	$2.01\ 10^{-1}$			$1.74\ 10^{-1}$		1.66
Total air		9,089			2,310	7,282	6,196			6,903
Natural gas	$7.25\ 10^{-3}$	$5.57\ 10^{-1}$					3.20	72.84[2]		
Water	304	436	151	530	478		895	2,593		391
Consumption of FGT reagents (kg/t MSW)										
Urea	3.49	10.9	1.62		2.55		$8.77\ 10^{-1}$			
NH_3				$9.39\ 10^{-1}$	10.54	$7.84\ 10^{-1}$				
CaO	8.14		9.25	6.33	$6.80\ 10^{-2}$	11.27				
$Ca(OH)_2$		12.76			$7.72\ 10^{-1}$		7.27			
Activated carbon	3.6310^{-1}	7.7710^{-1}	1.63	$1.13\ 10^{-1}$			$7.88\ 10^{-1}$	$1,88\ 10^{-1}$		
Electricity production										
Energy production (MJ/t MSW)	1,753	2,271	907	1,133	1,865	11,178	2,635	2,291	2,828	859
% Sold	86.0	85.0	51.0		79.0	96.0	98.0	99.0	72.0	71.0
% Self-consumption	14.0	15.0	49.0		21.0	4.0	2.42	0.69	28.0	29.0

[2]*Values of consumption of natural gas and water collected from the IPPC permit of I_{S8}. They could be referred to the consumption of all the installations that compound the plant. Therefore, they were not included in the average value.*

Table A1.7 (cont.) Input and output data of the Spanish WtE plants.

	SPANISH INCINERATORS									
	I_{S1}	I_{S2}	I_{S3}	I_{S4}	I_{S5}	I_{S6}	I_{S7}	I_{S8}	I_{S9}	I_{S10}
Waste generation										
Slag (t/t MSW)	$1.62\ 10^{-1}$	$2.47\ 10^{-1}$	$2.10\ 10^{-1}$	$2.18\ 10^{-1}$	$2.35\ 10^{-1}$	$1.90\ 10^{-1}$	$1.32\ 10^{-1}$	$1.31\ 10^{-1}$		$2.40\ 10^{-1}$
Ashes (t/t MSW)	$3.50\ 10^{-2}$	$4.25\ 10^{-2}$	$2.16\ 10^{-2}$	$2.48\ 10^{-2}$	$9.60\ 10^{-2}$	$3.74\ 10^{-2}$	$4.01\ 10^{-2}$	$6.31\ 10^{-2}$		$2.66\ 10^{-2}$
Scrap (t/t MSW)	$2.04\ 10^{-2}$	$6.26\ 10^{-3}$	$3.56\ 10^{-2}$	$2.79\ 10^{-2}$	$1.42\ 10^{-2}$	$2.00\ 10^{-2}$			$2.35\ 10^{-2}$	$1.72\ 10^{-2}$
% Slag	16.2	24.7	21	21.8	23.5	19.0	13.2	13.1		24.0
% Ashes	3.50	4.25	2.16	2.48	9.60	3.74	4.01	6.31		2.66
% Scrap in MSW	2.04	0.63	3.56	2.79	1.42	2.00			2.35	1.72
% Scrap in slag	12.6	2.54	16.9	12.8	6.05	10.5				7.17
Emissions to air (kg/t MSW)										
CO_2	338	416	325	326	324	1,277	335	162	338	964
As	$1.40\ 10^{-5}$	$2.16\ 10^{-5}$	$1.89\ 10^{-4}$	$1.84\ 10^{-5}$		$3.42\ 10^{-6}$	$1.64\ 10^{-5}$	$1.47\ 10^{-5}$		
Cd	$2.39\ 10^{-5}$	$1.72\ 10^{-5}$	$3.65\ 10^{-6}$	$4.94\ 10^{-6}$	$3.50\ 10^{-6}$	$2.04\ 10^{-6}$	$3.54\ 10^{-5}$	$1.90\ 10^{-6}$		
HCl	$3.19\ 10^{-2}$	$1.44\ 10^{-2}$	$2.70\ 10^{-2}$	$2.10\ 10^{-2}$	$1.02\ 10^{-2}$		$3.61\ 10^{-2}$	9.1010^{-3}	$3.71\ 10^{-2}$	$4.31\ 10^{-2}$
Cu	$1.93\ 10^{-4}$	$1.60\ 10^{-5}$	$1.33\ 10^{-5}$	$1.55\ 10^{-5}$	$5.93\ 10^{-6}$	$7.74\ 10^{-6}$	$1.02\ 10^{-4}$	$1.37\ 10^{-5}$	$4.12\ 10^{-5}$	
Cr	$2.68\ 10^{-5}$	$1.29\ 10^{-5}$	$3.02\ 10^{-5}$	$9.18\ 10^{-6}$	$4.78\ 10^{-5}$	$8.39\ 10^{-6}$	$2.65\ 10^{-5}$	$1.87\ 10^{-5}$	$4.55\ 10^{-4}$	
HF	$3.43\ 10^{-4}$	$1.33\ 10^{-3}$	$3.98\ 10^{-3}$	$4.94\ 10^{-5}$	$6.23\ 10^{-4}$		$1.46\ 10^{-3}$	$2.28\ 10^{-3}$	$8.43\ 10^{-4}$	$2.04\ 10^{-4}$
PAHs						$4.37\ 10^{-6}$	$4.95\ 10^{-5}$		$2.93\ 10^{-4}$	$2.84\ 10^{-4}$
Hg	$1.52\ 10^{-5}$	$7.47\ 10^{-6}$	1.4310^{-5}	$1.06\ 10^{-5}$	$2.43\ 10^{-6}$	$4.21\ 10^{-6}$	$6.19\ 10^{-6}$	$1.05\ 10^{-5}$	$1.78\ 10^{-6}$	$8.63\ 10^{-7}$
CO	$1.69\ 10^{-1}$	$1.05\ 10^{-1}$	$1.56\ 10^{-1}$	$6.68\ 10^{-2}$	$1.48\ 10^{-2}$	$3.77\ 10^{-1}$	$9.37\ 10^{-2}$	$7.41\ 10^{-2}$	$1.26\ 10^{-1}$	$2.38\ 10^{-2}$
Ni	$1.86\ 10^{-4}$	$1.20\ 10^{-5}$	$1.76\ 10^{-5}$	$1.20\ 10^{-5}$	$8.80\ 10^{-6}$	$6.51\ 10^{-6}$	$3.89\ 10^{-5}$	$1.31\ 10^{-5}$	$5.39\ 10^{-6}$	
SO_x	$3.88\ 10^{-2}$	$1.35\ 10^{-1}$	$1.39\ 10^{-1}$	$9.58\ 10^{-2}$	$4.52\ 10^{-2}$	$1.18\ 10^{-1}$	$1.77\ 10^{-2}$	$2.34\ 10^{-2}$	$1.90\ 10^{-3}$	$7.51\ 10^{-2}$

Table A1.7 (cont.) Medium input and output data of the Spanish WtE plants.

	SPANISH INCINERATORS									
	I_{S1}	I_{S2}	I_{S3}	I_{S4}	I_{S5}	I_{S6}	I_{S7}	I_{S8}	I_{S9}	I_{S10}
Emissions to air										
NO_X	$6.89\ 10^{-1}$	1.09	1.72	$6.38\ 10^{-1}$	$3.31\ 10^{-1}$	1.07	$9.11\ 10^{-1}$	$4.39\ 10^{-1}$	1.09	1.17
PCDD/F	$1.57\ 10^{-10}$	$3.44\ 10^{-11}$	$3.71\ 10^{-11}$	$2.90\ 10^{-11}$	$2.66\ 10^{-11}$	$9.39\ 10^{-12}$	$7.96\ 10^{-11}$	$3.04\ 10^{-11}$	$8.10\ 10^{-11}$	$1.81\ 10^{-9}$
Pb	$1.95\ 10^{-4}$	$1.33\ 10^{-4}$	$8.26\ 10^{-5}$	$4.87\ 10^{-5}$	$9.38\ 10^{-6}$	$3.10\ 10^{-5}$	$3.62\ 10^{-4}$	$1.96\ 10^{-5}$	$4.05\ 10^{-5}$	
Zn							$2.48\ 10^{-4}$		$1.48\ 10^{-3}$	$1.69\ 10^{-4}$
Mn	$2.38\ 10^{-4}$	$1.50\ 10^{-5}$	$7.63\ 10^{-6}$				$4.07\ 10^{-5}$			
TSP	$1.12\ 10^{-2}$	$1.84\ 10^{-2}$		$7.10\ 10^{-3}$			$6.19\ 10^{-3}$		$8.40\ 10^{-3}$	$1.50\ 10^{-2}$
NH_3	$1.12\ 10^{-2}$			$5.89\ 10^{-2}$	$1.29\ 10^{-3}$	$9.62\ 10^{-4}$	$2.39\ 10^{-2}$			$4.43\ 10^{-3}$
TOC	$5.31\ 10^{-3}$		$3.05\ 10^{-2}$	$6.31\ 10^{-3}$			$1.50\ 10^{-2}$			$1.34\ 10^{-2}$
CH_4						$7.84\ 10^{-4}$		$1.00\ 10^{-4}$	$6.16\ 10^{-4}$	
N_2O								$2.51\ 10^{-2}$		
NMVOC	$5.04\ 10^{-3}$				$1.02\ 10^{-2}$	$1.89\ 10^{-2}$		$1.28\ 10^{-2}$	$1.17\ 10^{-2}$	$1.37\ 10^{-2}$

Table A1.8 Input and output data of the Portuguese WtE plants.

	PORTUGUESE INCINERATORS		
	I_{P1}	I_{P2}	I_{P3}
Consumption of auxiliary materials and combustibles (kg/t MSW)			
Diesel (kg/t MSW)			
Total air (kg/t MSW)			
Natural gas (kg/t MSW)			
Water (kg/t MSW)	319		500
Electricity production			
Energy production (MJ/t MSW)	1,876		1,800
% Sold	86.0		90
% Self-consumption	14.0		10.0
Waste generation			
Slag (t/t MSW)	$2.00\ 10^{-1}$	$1.60\ 10^{-1}$	
Ashes (t/t MSW)	$3.00\ 10^{-2}$	$5.90\ 10^{-2}$	
Scrap (t/t MSW)			
% Slag	20.0	16.0	
% Ashes	3.00	5.90	
Emissions to air (kg/t MSW)			
CO_2	681		
HCl	$2.90\ 10^{-2}$		
Hg	$3.32\ 10^{-5}$		
NO_X	$6.95\ 10^{-1}$		$5.55\ 10^{-1}$
PCDD/F		$5.56\ 10^{-9}$	
CH_4		3.79	
N_2O	$7.87\ 10^{-2}$		

In this annex are listed other publications, as well as the contributions to international congress.

Scientific publications

1. **Margallo M**, Aldaco R, Irabien A **(2013)** Life cycle assessment of bottom ash management from a municipal solid waste incinerator (MSWI). *Chem Eng Trans* 35, 871-6.
2. **Margallo M**, Aldaco R, Bala A, Fullana P, Irabien A **(2010)** Implementation of the selective collection in small villages of less than 50 Inhabitants in Cantabria region (Spain): Preliminary viability study. *Chem Eng Trans* 21, 733-8.

Guidebooks

1. Aldaco R, **Margallo M,** Ibañez R, Miñambres C **(2008)** Guía de implantación del registro europeo de emisiones y transferencia de contaminantes (PRTR) en la comunidad autónoma de Cantabria. Consejería de Medio Ambiente, Gobierno de Cantabria.

Contributions to international congress

1. Aldaco R, Diban N, **Margallo M**, Barceló A, Ortiz I, Irabien A. Environmental sustainability assessment of an innovative process for partial dealcoholization of wines. *9th International Conference LCA of food (LCA Food 2014).* San Francisco (USA), 8-10 October, **2014**.
2. Aldaco R, **Margallo M**, Gazulla C, Fullana P, Irabien A. Developing a model of Sustainable Production and Consumption of Cantabrian Anchovies: a case study of Life Cycle Management in the Fish Canning

Industry. *LCA Food 2014*. San Francisco (USA), 8-10 October, **2014**.

3. Hernández A, **Margallo M**, Aldaco R, Irabien A. European regulatory framework of MSWI fly and bottom ash valorization. *13th Mediterranean Congress of Chemical Engineering, 13 MCCE.* Barcelona (Spain), 30 September-3 October, **2014**.

4. **Margallo M**, Hernández A, Aldaco R, Irabien A. Sustainability evaluation of municipal solid waste incineration using a life cycle assessment approach. *13 MCCE.* Barcelona (Spain), 30 September-3 October, **2014**.

5. **Margallo M**, Aldaco R, Irabien A. A case study for environmental impact assessment in the process industry: Municipal solid waste incineration (MSWI). *17th Conference Process Integration, Modelling and Optimisation for Energy Saving and Pollution Reduction, PRES 2014.* Prague (Czech Republic), 25-29 August, **2014**.

6. Aldaco R, **Margallo** M, Fernandez JA, Gazulla C, Fullana P, Irabien A. Carbon Footprint of Cantabria Anchovy: Global Environmental Considerations for a Local Sector Development. 17th World Congress of Food Science & Technology. Montreal (Canada), 17-21 August, **2014**.

7. **Margallo M**, Hernández A, Aldaco R, Irabien A. Environmental assessment of ash treatment from municipal solid waste incineration. *II International Conference on Chemical Engineering, ICCE 2014.* Madrid (Spain), 1-4 July, **2014**.

8. Hernández A, **Margallo M**, Aldaco R, Irabien A. Towards zero landfilling of municipal wastes with energy recovery. *ICCE 2014*. Madrid (Spain), 1-4 July, **2014**.

9. Dominguez-Ramos A, **Margallo M**, Aldaco R, Irabien A. Sustainability assessment of chemical processes and/or products using life cycle assessment. *I Simposio de la Red Española de ACV: "ACV & bioenergía.* Madrid (Spain), 15 October, **2013**.

10. **Margallo M**, Aldaco R, Irabien A. Environmental assessment of bottom ash management from a municipal solid waste incinerator (MSWI). *PRES 2013.* Rhodes (Greece), 29 September-2 October, **2013**.

11. **Margallo M**, Aldaco R, Bala A, Fullana P, Irabien A. Application of life cycle assessment (LCA) to waste management: Municipal solid waste

(MSW) incineration in Spain and Portugal. *XXXIV Reunión Bienal de la Real Sociedad Española de Química.* Santander (Spain), 15-18 September, **2013**.

12. **Margallo M**, Aldaco R, Bala A, Fullana P, Irabien A. Modeling of municipal solid waste (MSW) incineration in Spain and Portugal. *6th International Conference on Life Cycle Management, LCM 2013*. Gothenburg (Sweden), 25-28 August, **2013**.

13. Irabien A, **Margallo M**, Aldaco R. Sustainability evaluation of processes using life cycle methodologies (LCM). *9th World Congress of Chemical Engineering (WCCE9)*. Seoul (Korea), 18-23 August, **2013**.

14. Dominguez-Ramos A, **Margallo M**, Aldaco R, Irabien A. Sustainability assessment of chemical processes and/or products using life cycle assessment. *V International Conference on Life Cycle Assessment, CILCA 2013*. Mendoza (Argentina), 24-27 March, **2013**.

15. **Margallo M**, Aldaco R, Bala A, Fullana P, Irabien A. Municipal solid waste (MSW) incineration in Spain and Portugal: A multi-input/output allocation model. *CILCA 2013*. Mendoza (Argentina), 24-27 March, **2013**.

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18. **Margallo M**, Aldaco R, Bala A, Fullana P, Irabien A. Best available techniques in municipal solid waste incineration: State of the art in Spain and Portugal. *PRES 2012.* Prague (Czech Republic), 25-29 August, **2012**.

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34. Aldaco R, **Margallo M**, Irabien A. Environmental sustainability assessment in the process industry. *III CILCA.* Pucón (Chile), 27-29 April, **2009**.

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engineering. *LCM 2007*. Zurich (Switzerland), 27-29 August, **2007**.

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It's sad to think that nature speaks and

mankind does not listen

(Victor Hugo)