Universitat Autònoma de Barcelona

PhD Programme in Environmental Science and Technology
Institute of Environmental Science and Technology

DOCTORAL DISSERTATION: METHODOLOGICAL ADVANCEMENTS IN LCA OF WASTE MANAGEMENT SYSTEMS

Alba Bala Gala
UNESCO Chair in Life Cycle and Climate Change

Supervisors:
Dr. Marco Raugei
Faculty of Technology, Design and Environment
Oxford Brookes University, UK.

Dr. Pere Fullana i Palmer
Càtedra UNESCO de Cicle de Vida i Canvi Climàtic
Escola Superior de Comerç Internacional, UPF.
Barcelona, Spain.

Tutor at UAB:
Dr. Pere Ysern Comas
Oficina de Medi Ambient
Bellaterra (Barcelona), Spain.

Barcelona, February 2015

Universitat Autònoma de Barcelona

DR. MARCO RAUGEI, Senior Research Fellow a la Oxford Brookes University,

DR. PERE FULLANA- I- PALMER, Director de la Càtedra UNESCO de Cicle de Vida i Canvi Climàtic de l'Escola Superior de Comerç Internacional (ESCI-UPF)

i

DR. PERE YSERN COMAS, Cap de l'Oficina de Medi Ambient de la UAB.

CERTIFIQUEM:
Que la present memòria titulada **"Methodological advancements in LCA of Waste Management Systems"**, ha estat realitzada sota la nostra direcció i supervisió per la llicenciada Alba Mª Bala Gala, i constitueix la seva Tesi per optar al Grau de Doctor en Ciència i Tecnologia Ambientals.

I perquè així consti, signem el present certificat a Bellaterra, 2 de febrer de 2015.

Dr. Marco Raugei

Dr. Pere Fullana-i-Palmer

Dr. Pere Ysern Comas

Dedicada, amb tot el meu amor,
a la meva filla Marina,
nascuda durant aquests estudis de doctorat,
i als meus avis Pepe i Hortènsia,
que van morir durant el mateix període de temps.
Gràcies per la vostra força,
la vostra bondat i el vostre amor.

Dedicated, with all my love,
to my daugther Marina,
born during this PhD study,
and to my grandparents Pepe and Hortènsia,
who died during the same time period.
Thanks you for all your strength,
your kindness and your love.

ACKNOWLEDGMENTS/AGRADECIMIENTOS/AGRAÏMENTS

En primer lloc voldria donar les gràcies als meus directors de tesi, Marco Raugei i Pere Fullana. Marco, gracias por haberme apoyado y animado siempre, por las discusiones metodológicas, por tus valiosos consejos y por estar ahí siempre, a pesar de la distancia. Pere, gràcies per haver confiat en mi, per accollir-me dins el grup de recerca d'ESCI i la Càtedra i per dedicar-me el teu preciat temps. També voldria donar les gràcies al meu tutor a la UAB, en Pere Ysern, per haver-me donat suport i consells en tot aquest temps.

Voldria expressar el meu sincer agraïment a totes les persones que durant aquest llarg trajecte han aportat el seu granet de sorra, m'han animat, m'han tret el somriure en moltes ocasions i m'ham ajudat a créixer com a persona i com a professional. D'entre totes elles voldria destacar a en Joan Rieradevall i a en Xavier Domènech, per haver-me posat en contacte amb el món de l'ACV fa gairebé 15 anys, donar-me l'oportunitat de treballar amb ells, i també per animar-me a iniciar els estudis de doctorat l'any 2000 que, després de molts esforços - i de formar una familia pel mig - , he aconseguit per fi concloure. *Gràcies de tot cor!* I molt especialmente a la Cristina Gazulla (*amiga, companya de fatiques des del principi fins al final a qui respecto molt com a profesional i aprecio encara més com a persona*), a en Llorenç Milà, a l'Ivan Muñoz i a la Marta Arrufí, tots quatre companys en l'inici de la meva aventura en el món de l'ACV.

Durant l'època que vaig pasar a l'ICTA vaig tenir també l'oportunitat de compartir coneixements i experiencies amb molts investigadors –molts del quals he tingut el plaer d'anar retrobant després en la meva vida professional –. D'entre tots ells, voldria destacar la Montse Núñez (*por nuestros cafés y conversaciones profundas que recuerdo con mucho cariño*), en Carles Martínez, en Jordi Olivé, en Raúl García, en Ramón Farreny i la Marta Albet, sense oblidar a la resta de companys amb qui vaig compartir aquells anys. També a altres persones com en Paco Muñoz de l'Oficina de Medi Ambient (*gràcies per donar-me confiança…ja saps que t'aprecio molt!*).

Durant els darrers 7 anys, i ja més centrada de nou en l'ACV i en particular el l'ACV de residus, voldria mencionar a aquelles persones amb les que he compartit el dia a dia i amb les que he tingut l'oportunitat d'aprendre (i també el plaer de conèixer com a persones). D'entre totes elles, i sense menysprear a la resta de companys passats i actuals de la Càtedra, voldria destacar (de nou) a la Cristina Gazulla, a la Jenna Watson, a la Marta Anglada (*la meva castanyera particular…ja saps per on vaig!*), a la Marina Isasa, a la Susana Leao (*compañera de aventuras de FENIX de inicio o fin*), a la Gabriela Bienveniste i a en Miguel Pérez.

También querría expresar un profundo agradecimiento a mis compañeros del proyecto FENIX, y en especial a Silvia Ayerbe, a Alberto Fernández y a Noelia Martin de Ecoembes por compartir sus ámplios conocimientos en materia de recogida y gestión de residuos de envase, y también a Francisco Pan-Montojo, por su persistencia, su dedicación y sus aportaciones y conocimientos para mejorar el modelo de recogida incluido en esta tesis doctoral. A Susana Ángelo de Sociedade Punto verde por su apoyo y aportaciones, y a Carlos Afonso Teixeira de UTAD por sus valiosos datos para la validación del modelo de recogida así como por sus ánimos persistentes (*muito obrigada!*). También a Viviana Carrillo de PE International y a los compañeros de la Universidad de Cantabria, Rubén Aldaco y María Margallo (*aprecio mucho tu apoyo moral en estos últimos meses que hemos compartido finalizando nuestras respectivas tesis doctorales*).

Special thanks also to the whole staff of DTU Environment for their warm welcome and for teaching me a lot about LCA of waste management systems, and especially to Thomas H. Christensen, Michael Hauschild, Emmanuel Gentil (*thanks for your valuable advice and for sharing your experience in the development of software packages for LCA of waste management*) and Anders Damgaard (*also for your advice and for your hospitality…I will never forget the night we spent in your home watching the match between Italy and Spain the year Spain won the World Championship!*). I would also like to express special thanks to the International Expert Group on Waste Management and LCA for giving me the oportunity to attend and participate actively in the last 4 annual meetings of the Group, and more especially to Terry Colemman, Roland Clift, Tomas Ekwall and Susana Xarà.

Finalment, no hauria pogut completar aquesta tesi sense l'ajuda de la meva familia -dels que encara hi són com dels que ens han deixat- així com dels amics, que m'han ajudat a agafar forces i a oxigenar-me en determinats moments amb unes paraules d'ànim, un cafè, un sopar o algún assaig musical, i també d'altres persones desinteressades que m'han ajudat al llarg d'aquests anys com ara l'Al Fisher (*the best English teacher I have ever had!)*.

Molt especialmente vull donar les gràcies als meus pares, sense els quals ni tans sols seria aquí, per l'educació que m'han donat per convertir-me en la persona que sóc, per ensenyar-me a no rendir-me mai i també per fer de cangurs de la Marina en tots els moments en que ho he necessitat per poder enllestir aquesta tesi. *Us estimo molt!*

I encara més especialmente al Manel, per compartir amb mi tots aquests anys, donar-me ànims constantment i per donar-me el més important que tinc a la vida, la meva filla Marina i l'altre que està en camí, que també m'han donat forces per no tirar la tovallola en cap moment. I a la Marina, amb tot el meu amor, per ser un cascavell que m'anima cada dia.

Moltes gràcies a tots!

LIST OF CONTENTS

List of Tables

List of Figures

Appendixes

PREFACE

The work reported in this PhD thesis, entitled "Methodological advancements in LCA of Waste Management Systems", was mainly undertaken at the UNESCO Chair in Life Cycle and Climate Change (ESCI-UPF) under the framework of the LIFE+ FENIX Project (2010-2013; LIFE08 ENV/E/000135) financed by the European Commission and coordinated by the author of this thesis. The main objective of this project was to develop a flexible and easy to use tool to help municipalities in Spain and Portugal to perform environmental analyses of their integrated waste management systems (WMS) under a Life Cycle Thinking (LCT) approach. Further details about this project are available at www.life-fenix.eu/.

All the research work was supervised by Dr. Marco Raugei (Senior Research Fellow at Oxford Brookes University) and Dr. Pere Fullana-i-Palmer (Director of the UNESCO Chair in Life Cycle and Climate Change, ESCI-UPF), and also by Dr. Pere Ysern Comas (Director of the Environmental Office, UAB), acting as a tutor.

This PhD thesis mainly consists of a compilation of three papers submitted to international peer-reviewed scientific journals. In the text, these papers are referred to by the names of the authors and their corresponding number in Chapter 2, according to the following list, written with Roman numerals and in bold.

I. **Bala A.**, Raugei M., Fullana-i-Palmer P., 2015. Introducing new method for calculating environmental credits of end-of-life material recovery in attributional LCA. Submitted to The International Journal of Life Cycle Assessment (accepted, DOI: 10.1007/s11367-015-0861-3).

II. **Bala A.**, Raugei M., Afonso C., Fullana-i-Palmer P., 2015. Assessing the environmental performance of municipal solid waste collection: a new predictive model. Submitted to Waste Management (currently undergoing a second round of review).

III. **Bala A.,** Raugei M., Ripa M., Ulgiati, S., 2015. Dealing with waste flows in Life Cycle Assessment and Emergy Accounting: methodological overview and synergies. Submitted to Ecological Modeling (currently undergoing a second round of review).

In addition to the three papers which form the main body of this PhD thesis, the following publications dealing with LCA and waste management were also developed during the PhD study:

- **Bala A.** and Raugei M., 2009. Waste Management. Chapter 5 of the Book "Life Cycle Assessment applications: results from COST Action 530. A joint effort by COST Action 530: Life-Cycle Inventories for Environmentally-Conscious Manufacturing Processes". Ed. Fulana, P., Betz, M., Hischer. R., Puig, R. AENOR, Madrid. ISBN: 978-84-8143-611-2

- 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. Chemical Engineering Transactions, 21, 733-738.

- Lehmann A., Russi D., **Bala A.**, Finkbeiner M., Fullana-i-Palmer, P., 2011. Integration of Social Aspects in Decision Suport, Based on Life Cycle Thinking. Sustainability, 3 (4), 562-577.

- Fullana-i-Palmer P., Puig R., **Bala A.**, Baquero G., Riba J., Raugei M., 2011. From Life Cycle Assessment to Life Cycle Management. A case study on Industrial Waste Management Policy Making. Journal of Industrial Ecology, 15 (3), 458-475.

- Margallo M., Aldaco R., **Bala A.**, Fullana P., Irabien A., 2012. Best Available Techniques in municipal solid waste incineration: state of the art. Chemical Engineering Transactions, 29, 1345-1350.

- Puig R., Fullana-i-Palmer P., Baquero G., Riba J.R., **Bala A.,** 2013. A Cumulative Energy Demand Indicator (CED), life cycle based, for industrial waste management decision making. Waste Management, 33 (12), 2789-2797.

- Camba A., González-García S., **Bala A.**, Fullana-i-Palmer P., Moreira M.T., Feijoo G., 2014. Modeling the leachate flow and aggregated emissions from municipal waste landfills under life cycle thinking in the Oceanic region of the Iberian Peninsula. Journal of Cleaner Production, 67, 98-106.

- 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 Management & Research, 32(6), 492-499.

- Margallo M., Dominguez-Ramos A., Aldaco R., **Bala A.**, Fullana P., Irabien A., 2014. Environmental sustainability assessment in process industry: A case study of waste-to-energy plants in Spain. Resources, Conservation and Recycling, 93, 144-155.

February, 2015

Alba Bala

LIST OF ACRONYMS

AFNOR	French standardization system
AP	Acidification Potential
BSI	British Standards Institution
BUWAL	Federal Office for the Environment, Forests and Landscape
CED	Cumulative Energy Demand
CML	Institute of Environmental Sciences, Leiden University, Netherlands
EC	European Comission
ELCD	European reference Life Cycle Database
ELR	Environmental Loading Ratio
EMA	Emergy Accounting
EPLCA	European Platform on LCA
EU	European Union
EYR	Emergy Yield Ratio
FU	Functional Unit
GWP	Global Warming Potential
IEEP	Institute for European Environmental Policy
IES	Institute for Environment and Sustainability
ILCD	International Reference Life Cycle Data System
ISO	International Organization for Standardization
IWM	Integrated Waste Management
JRC	Joint Research Centre
LCA	Life Cycle Assessment
LCDN	Life Cycle Data Network
LCI	Life Cycle Inventory Analysis
LCIA	Life Cycle Impact Assessment
LCSA	Life Cycle Sustainability Analysis
LCT	Life Cycle Thinking
MRI	Midwest Research Institute
MSW	Municipal Solid Waste
NIR	National Inventory Reports
NMVOC	Non methane volatile organic compounds
OECD	Organization for Economic Cooperation and Development
PEF	Product Environmental Footpring
PROGRIC	Industrial Waste Management Programme of Catalonia
REPA	Resources and Environmental Profile Analysis
RPM	Revolutions per Minute
SETAC	Society of Environmental Toxicology and Chemistry

SI	International System of units
SPOLD	Society for the Promotion of LCA Development
TNO	Netherlands Organization for Applied Scientific Research
U.S. EPA	Environmental Protection Agency of the United States
UEV	Unit Emergy Value
UK	United Kingdom
UNEP	United Nations Environment Programme
UNFCCC	United Nations Framework Convention on Climate Change
USA	United States of America
WMS	Waste Management Systems

SUMMARY

Waste management has been identified by the European Commission, the United States Environmental Protection Agency and the Organization for Economic Cooperation and Development, among other institutions, as a key issue for the achievement of a resource-efficient society and for achieving a sustainable economy. Waste management practices need to be improved in order to reinforce material recycling, closing essential material loops and also recovering energy from waste, while at the same time ensuring that toxic substances are not released to the environment. This essentially means moving from a linear extraction-use-throw away economy model to a more circular one.

Life Cycle Assessment (LCA) has been gaining acceptance in recent years as the best tool to assist decision making for waste management policy and planning in Europe. It can provide a global and expanded view of the system, taking into account all the processes involved and its interactions with the economy (by means of reintroducing recycled materials or energy recovered from the waste management system), as well as accounting for a complete set of environmental effects (i.e. climate change, ozone layer depletion, acidification or eutrophication).

During the last decades, some authors have identified methodological issues when performing LCA of Waste Management Systems (WMS) that may have a great influence on the final results of the analysis. These issues have been addressed in the last methodological guidelines about LCA and waste management developed by the Joint Research Centre of the European Union (published in 2011). However, according to the author, there are still some important issues when applying LCA to evaluate the environmental efficiency of WMS that merit special attention. In particular, the way in which collecting systems and credits due to material recovery have been historically modelled needs to be reviewed, especially if the focus of the analysis is the collection system itself and if we are performing an analysis of the system from an attributional LCA point of view. On the other hand, how resource depletion is assessed in LCA has been recently questioned in the literature, and needs also to be analysed.

The work done in this thesis aims to identity these important issues and shed some light on how to address them. The outcomes of the thesis are (1) an alternative method to account for the credits of material recovery in LCA, more in line with the fundamental aim of the attributional approach in LCA; (2) a new predictive model for evaluating the environmental perfomance of waste collection, which produces more accurate results when compared with real collection routes than other existing models; and (3) a contribution to the ongoing

discussion for the future development of a more robust method for evaluating Resource Scarcity impact category in LCA, by looking at synergies with Emergy Accounting methodology.

RESUMEN

La gestión de residuos ha sido identificada por la Comisión Europea, la Agencia de Protección Ambiental de los Estados Unidos y la Organización para la Cooperación Económica y el Desarrollo, entre otras instituciones, como un tema clave para el logro de una sociedad eficiente en el uso de recursos y para lograr una economía sostenible. Las prácticas de gestión de residuos deben ser mejoradas con el fin de reforzar el reciclaje de materiales, cerrando ciclos de materiales esenciales y también recuperando energía a partir de los residuos, al tiempo que se garantiza que no se liberan sustancias tóxicas al medio ambiente. En esencia, esto significa pasar de un modelo lineal de economía basado en extraer, usar y tirar a un modelo más circular.

El Análisis de Ciclo de Vida (ACV) ha ido ganando aceptación en los últimos años en Europa como la mejor herramienta para ayudar en la toma de decisiones sobre políticas de gestión de residuos y planificación. Éste proporciona una visión global y ampliada del sistema, tiene en cuenta todos los procesos que intervienen y sus interacciones con la economía (mediante la reintroducción de materiales reciclados o de la energía recuperada del sistema de gestión de residuos), así como la quantificación del impacto de un conjunto completo de los efectos ambientales (p. ej. el cambio climático, el agotamiento de la capa de ozono, la acidificación o la eutrofización).

Durante las últimas décadas, algunos autores han identificado cuestiones metodológicas al realizar ACV de Sistemas de Gestión de Residuos (SGR) que pueden tener una gran influencia en los resultados finales del análisis. Estas cuestiones se han abordado en las últimas guías metodológicas sobre ACV y gestión de residuos elaboradas por el *Joint Research Centre* de la Unión Europea (publicadas en 2011). Sin embargo, según la autora, todavía hay algunas cuestiones importantes en la aplicación del ACV para evaluar la eficiencia ambiental de SGR que merecen especial atención. En particular, la forma en que los sistemas de recogida y los créditos asociados a la recuperación de materiales han sido modelizados históricamente necesitan ser revisados, especialmente si el foco del análisis es el sistema de recogida en sí mismo y si estamos llevando a cabo un análisis del sistema desde el punto de vista del ACV atribucional. Por otro lado, la forma en que se está evaluando el agotamiento de recursos en ACV ha sido cuestionada recientemente en la literatura, y necesita también ser analizada.

El trabajo realizado en esta tesis pretende resaltar estas importantes cuestiones y arrojar algo de luz sobre cómo abordarlas. Los resultados de la tesis son: (1) un método alternativo para contabilizar los créditos debidos a la recuperación de materiales en ACV, más en línea con el objetivo fundamental o el enfoque del ACV atribucional; (2) un nuevo modelo predictivo para

evaluar el comportamiento ambiental de la etapa de recogida de residuos, que produce resultados más precisos que otros modelos existentes al compararlos con rutas reales de recogida; y (3) una contribución al debate abierto sobre el futuro desarrollo de un método más robusto para la evaluación de la categoría de impacto de escasez de recursos en ACV, mediante la búsqueda de sinergias con la metodología de *Emergy Accounting*.

RESUM

La gestió de residus ha estat identificada per la Comissió Europea, l'Agència de Protecció Ambiental dels Estats Units i l'Organització per a la Cooperació Econòmica i el Desenvolupament, entre altres institucions, com un tema clau per a l'assoliment d'una societat eficient en l'ús de recursos i per aconseguir una economia sostenible. Les pràctiques de gestió de residus han de ser millorades per tal de reforçar el reciclatge de materials, tancant cicles de materials essencials i també recuperant energia a partir dels residus, alhora que es garanteix que no s'alliberen substàncies tòxiques al medi ambient. En essència, això significa passar d'un model lineal d'economia basat en extreure, usar i llençar a un model més circular.

L'Anàlisi de Cicle de Vida (ACV) ha anat guanyant acceptació en els últims anys a Europa com la millor eina per ajudar en la presa de decisions sobre polítiques de gestió de residus i planificació. Aquesta proporciona una visió global i ampliada del sistema, té en compte tots els processos que intervenen i les seves interaccions amb l'economia (mitjançant la reintroducció de materials reciclats o de l'energia recuperada del sistema de gestió de residus), així com la quantificació de l'impacte d'un conjunt complet dels efectes ambientals (p. ex. el canvi climàtic, l'esgotament de la capa d'ozó, l'acidificació o l'eutrofització).

Durant les últimes dècades, alguns autors han identificat qüestions metodològiques en realitzar ACV de Sistemes de Gestió de Residus (SGR) que poden tenir una gran influència en els resultats finals de l'anàlisi. Aquestes qüestions s'han abordat en les últimes guies metodològiques sobre ACV i gestió de residus elaborades pel *Joint Research Centre* de la Unió Europea (publicades el 2011). No obstant això, segons l'autora, encara hi ha algunes qüestions importants en l'aplicació d'ACV per avaluar l'eficiència ambiental de SGR que mereixen especial atenció. En particular, la forma en què els sistemes de recollida i els crèdits associats a la recuperació de materials han estat modelitzats històricament necessiten ser revisats, especialment si el focus de l'anàlisi és el sistema de recollida en si mateix i si estem duent a terme una anàlisi del sistema des del punt de vista de l'ACV atribucional. D'altra banda, la forma en què s'està avaluant l'esgotament de recursos en ACV ha estat qüestionada recentment en la literatura, i necessita també ser analitzada.

El treball realitzat en aquesta tesi pretén ressaltar aquestes importants qüestions i donar una mica de llum sobre com abordar-les. Els resultats de la tesi són: (1) un mètode alternatiu per comptabilitzar els crèdits deguts a la recuperació de materials en ACV, més en línia amb l'objectiu fonamental o l'enfocament de l'ACV atribucional; (2) un nou model predictiu per avaluar el comportament ambiental de l'etapa recollida de residus, que produeix resultats més

precisos que altres models existents en comparar-los amb rutes reals de recollida; i (3) una contribució al debat obert sobre el futur desenvolupament d'un mètode més robust per a l'avaluació de la categoria d'impacte d'escassetat de recursos en ACV, mitjançant la recerca de sinèrgies amb la metodologia de l'*Emergy Accounting*.

CHAPTER 1. Introduction

«The important thing in science is not so much to obtain new facts as to discover new ways of thinking about them.»

[William Lawrence Bragg, 1890-1971, *Nobel Prize of Physics in 1915*]

1.1 Structure of the thesis

This thesis is structured in four main chapters:

Chapter 1 provides the necessary background to understand the drivers and objectives of the research. A theoretical framework explaining the origin of the problem, how it has been tackled by policy at European level and how Life Cycle Assessment (LCA) can help in this context is stated (section 1.2). In this chapter the reasons for carrying out the study are also expounded, and the hypotheses and the research objectives for each one of the scientific papers included are formulated (section 1.3).

Chapter 2 consists on the compilation of the papers listed in the preface section that are the core part of this thesis. The contents of the papers, figures and tables, have been extracted literally from the versions of the papers submitted to the international peer-reviewed scientific journals, and adapted to the format of this thesis.

Chapter 3 summarizes and synthesizes the discussion and results sections of the three papers, giving a global overview of the connections among them under a Life Cycle Thinking (LCT) approach.

Chapter 4 includes the conclusions of the thesis, providing overall recommendations and discussion of its limitations, and future research needs.

Additionally, the thesis includes an **APPENDIX** section, where additional information about LCA history and methodology, and its application to waste management systems is provided. Also a general description of the Emergy Accounting methodology (EMA) is included.

1.2 Background

1.2.1 The origin of the waste problem

Waste generation is nothing new, as it is inherent to human activities. So much so, that the wastes generated by our ancestors have become a valuable source of information for archaeologists to study the culture and lifestyle of ancient civilizations [Stewart & Gifford - Gonzalez, 1994; McCorriston & Weisberg, 2002]. Throughout history, and as inhabited areas have become bigger and more concentrated, the problems associated to waste have been growing and worsening as well. In the Middle Ages, the accumulation of food residues in cities caused serious health problems due to the attraction and rapid growing of rat populations that hosted fleas that transmitted the plague to humans [McGovern & Friedlander, 1997; Prentice & Rahalison, 2007]. Such problems forced big cities to develop methods to improve sanitation. The remains of food and faeces were transported out of the cities to prevent infections, but also to return nutrients to agricultural soils [Klang, 2005; Güereca, 2006]. Such measures could be considered the beginnings of what we now call "waste management practices".

The advent of industrialization caused an added problem to waste. Until then, waste consisted mainly of natural and renewable materials (wood, wool, fur, hemp...) and products made out of these materials were reused until they could no longer be repaired. For that reason the volume of waste generated was not enough to cause major problems. However, with the arrival of industrialization and mass production at the end of the eighteenth century, the negative consequences of the increased use of non-renewable resources became increasingly evident. The burning of fossil fuels for energy led to the decline of coal reserves and the massive pollution of the cities, causing many health problems [Te Brake, 1975; Nye, 1994]. The twentieth century coincided with the development of artificial materials [Barceló et al., 2000], mainly derived from petroleum, and the growing of the throw-away culture. The deposit of waste in a landfill became the preferred choice for solving the problem of waste [Eldredge & Hickman, 1999], and sometimes combined with the incineration of waste to reduce the volume of waste sent to the landfill. Thus, the continuous growth in the use of single-use products, in the use of non-renewable artificial materials - or renewable materials at a higher rate than their natural replenishment - and the treatment of waste by its burial in landfills or incineration, caused a new environmental effect added to waste generation: the progressive depletion of natural resources.

1.2.2 Brief historical outline of the European political approach to the waste problem

The waste problem was recognized early in the history of the European Union as one of the key factors to be addressed. The first Waste Directive was adopted in 1975[1]. However, as declared by the Institute for European Environmental Policy, «as with much other environmental legislation, the early waste measures were generally "end-of-pipe", i.e., attempting to address problems once waste has already been generated»[2] [IEEP, 2008], and did not provide a solution to the second problem associated with waste generation, the progressive depletion of resources.

---

[1] Council Directive 75/442/ECC of 15 July 1975 on waste. Official Journal L 194, 1975-07-25 pp.: 0039-0041.

[2] Citation from the website of the Institute of European Environmental Policy (IEEP) (http://213.198.115.240/activities/policyresearch/wastemanagement.php) (last consultation: 2/03/2012).

Fortunately, European waste policy orientation has moved towards a more Life Cycle Thinking approach (LCT hereafter), focusing on waste prevention and recycling instead of waste treatment and end-of-pipe solutions [Bala and Raugei, 2009]. This change started with the adoption of the landfill Directive in 1999[3] and the Directive on packaging and packaging waste 94/62/EC[4], in which it is recommended to prioritize prevention in the production of packaging waste, followed by reuse, recycling, incineration and landfilling, and where the establishment of material recovery systems is mandated. In 2001 the EU adopted the Sixth Environmental Action Programme of the European Union [EC, 2001], and included sustainability in resource use and waste management as one of the four priority lines of action. As a result of this program, in December 2005 the Thematic Strategy on Waste Prevention and Recycling was adopted. In this document it is stated that waste must no longer be considered strictly as a problem to be controlled and disposed of, but as a source of «valuable resource for industry» [EC, 2005]. Directive 2008/98/EC[5] on waste further enhances the measures to be taken regarding the prevention of waste, specifying that one must «introduce an approach that takes into account the whole life-cycle of products and materials and not only the waste phase, and to focus on reducing the environmental impacts of waste generation and waste management (…). Furthermore, the recovery of waste and the use of recovered materials should be encouraged in order to conserve natural resources[6]». A step beyond this change from seeing waste as a problem to be solved to seeing it as a potential source of resources has been made by "Europe 2020", the strategy proposed by the European Commission on March 2010 for the next decade [EC, 2010]. This strategy includes the Societal Challenge "Climate action, environment, resource efficiency and raw materials" with the aim of achieving a resource efficient economy and society, the protection and sustainable management of natural resources and ecosystems, and a sustainable supply and use of raw materials by 2020 [EC, 2014]. The idea behind this challenge is to favour the development of a green economy – a circular economy in sync with the natural environment –, trying to decouple economic growth from the progressive resources depletion, by giving an added value to recycled waste flows and reintroducing them into the economy.

[3] Council Directive 1999/31/EC of 26 April 1999 on the landfill of waste. OJ L 182 de 16.7.1999. The deadline for its transposition to all member states was 16.07.2001. The reduction targets to meet were: for 2006 the reduction to 75% of the amount of biodegradable waste landfilled in 1999; for 2009 50% of 1999 levels; and by 2016 35% of 1999 levels.

[4] European Parliament and Council Directive 94/62/EC of 20 December 1994 on packaging and packaging waste (OJ L 365, 31.12.1994, p. 10), amended by Directives 2004/12/EC and 2005/20/EC. In this Directive quantitative targets for energy recovery or incineration and recycling were set by weight. Since its entry into force, each member state has been forced to create return and/or collect packaging waste in order to treat them using the best treatment options. In Spain, this has led to the creation of the Integrated Management Systems: ECOEMBES (for light packaging), ECOVIDRIO (for glass bottles) and SIGRE (for pharmaceutical packaging).

[5] European Parliament and Council Directive 2008/98/EC of 19 November 2008 on waste, repealing Directive 2006/12/EC.

[6] Citation extracted from page 312/4 of the Directive 2008/98/EC (8th preliminary consideration).

This change of orientation in policies related to waste management has not taken place in Europe alone. The Environmental Protection Agency of the United States and the Organization for Economic Cooperation and Development (OECD), for instance, adopted, respectively, the Resource Conservation Challenge in 2002 and the Environmentally Sound Management of Waste in 2004 to promote waste prevention, promote recycling and preserving natural resources [U.S. EPA, 2004, OECD, 2004]. The former emphasizes the need to shift the focus from an end-of-pipe approach of waste management issues to a resources or materials management approach [Thorneloe et al., 2005, 2007]. The latter seeks to promote among its members «a scheme for ensuring that wastes and used and scrap materials are managed in a manner that will save natural resources, and protect human health and the environment against adverse effects that may result from such wastes and materials[7]» [OECD, 2007].

1.2.3 Life Cycle Assessment applied to waste management systems

Life Cycle Assessment (LCA hereafter) is the most commonly used methodology for integrating LCT on the analysis of products and systems (a brief description of the methodology is provided in Appendix B). Its formal development began in the 1980s and it is currently regulated by ISO 14040[8] and ISO 14044[9] standards. Although LCA was originally developed for analysing the environmental performance of products (see Appendix A where its history is detailed), since the end the 90s, this methodology has also been used for the analysis of waste management [Hauschild & Barlaz, 2011].

Right from the start, LCA was used to compare different alternatives for the treatment of one specific waste flow [just to mention some of them, i.e. Finnveden & Ekvall, 1998, 1999; Wollny et al., 2001; Güereca et al., 2006; Villanueva & Wenzel, 2007; Merrid et al., 2008], but also to compare more complex systems such as integrated waste management systems, including all waste fractions [i.e. Denison, 1996; Rodrigo & Castells, 2000; Eriksson, 2003; Beigl & Salhofer, 2004; Muñoz et al., 2004; Emery et al., 2007; Thorneloe et al., 2007; Banar et al., 2009; Iriarte et al., 2009; Koci & Trecakova, 2011]. The benefit of using LCA in this context is that it helps to expand the perspective of the analysis and to have a complete view of the entire system, gathering all processes and environmental impacts associated. This approach can avoid the unintentional shifting of environmental loads between different steps of the waste management system, geographic areas, environmental compartments (air, land and water) or impact categories (e.g. global warming, acidification, etc.) [**Bala et al., 2015; II**].

[7] Citation extracted from page 8 of the "Guidance Manual for the Implementation of the OECD Recommendation C(2004)100 on Environmentally Sound Management (ESM) of Waste" OECD, 2007.

[8] ISO 14.040: 2006. Environmental Management – Life Cycle Assessment –Principles and framework.

[9] ISO 14.044: 2006. Environmental Management – Life Cycle Assessment – Requeriments and Guidelines.

In addition to the practical application of LCA to waste management systems mentioned above, some research groups have been working actively on its methodological development (for further details see Bala & Raugei, 2009). As a result of their activities, since the end of the 1990's, a wide range of guidelines and methodological recommendations for LCA practitioners have been written [Finnveden, 1999; Clift et al., 2000; Bjarnadóttir et al, 2002; JRC, 2007a; JRC, 2007b; JRC, 2011a; JRC, 2011b; JRC, 2011c], in an attempt to standardize the methodology and provide clear methodological guidelines on how it should be implemented for waste management systems, gathering the latest scientific consensus at each point in time. In parallel, some of the groups have also been promoting its proper use and providing quality-assured life-cycle data inventories for LCA practitioners. Among them, the *International Expert Group on LCA for Integrated Waste Management*[10], the *European Commission's Institute for Environment and Sustainability – ISPRA Joint Research Centre (JRC-IES)*[11] and the *European Platform on LCA (*EPLCA hereafter*) (JRC and DG Environment)*[12] merit special attention.

LCA has been gaining acceptance in recent years as a tool to assist decision making for waste management policy and planning in Europe [EC, 2005a; Rigamonti et al. 2009]. Examples of this can be found on:

- the repealed Directive 75/439/EC on waste oils, in which the regeneration of waste oils was always recommended over other methods of treatment or disposal. When developing the new Waste Directive[13] it was proposed to change the waste

[10] The International Expert Group on LCA for Integrated Waste Management consists on a discussion forum which meets regularly with the intention of helping the implementation of more sustainable waste management practices by supporting the development and proper use of LCA tools applied to integrated management systems [Coleman et al, 2003]. It is comprised of about 30 members from 14 different countries (Australia, Canada, France, Germany, Ireland, Italy, Netherlands, Sweden, Switzerland, Austria, Spain, Portugal, USA and UK). The access to this group is by invitation.

[11] The Institute for Environment and Sustainability (IES) is one of the seven scientific institutes of the European Commission's Joint Research Centre (JRC). Its mission is to provide scientific and technical support to EU policies for the protection of the European and global environment. It combines this in-house expertise with its role as a scientific catalyst in order to provide the knowledge base necessary to assess the social, environmental and economic aspects of policy options (information extracted from its website http://ies.jrc.ec.europa.eu/, 23/03/2014).

[12] The European Platform on Life Cycle Assessment is a project of the European Commission, carried out by the Commission's Joint Research Centre (JRC), Institute for Environment and Sustainability (IES) in collaboration with DG Environment Directorate Green Economy (information extracted from its website: http://eplca.jrc.ec.europa.eu/, 23/03/2014). Through the EPLCA it has been facilitated the development of the European Reference Life Cycle Database (ELCD), which comprises Life Cycle Inventory (LCI) data from front-running business associations and other sources for key materials, energy carriers, transport, and waste management; the International Reference Life Cycle Data System (ILCD) Handbook [IES, 2010]; and the Life Cycle Data Network (LCDN) launched the 6th of February 2014, aimed at providing globally usable infrastructure for consistent and quality assured LCA data from different organizations.

[13] Directive 2008/98/EC of the European Parliament and of the Council of 19 November 2008 on waste and repealing certain Directives. OJ, L 312, 22.11.2008.

management hierarchy based on a review of old LCA studies [Monier & Labouze, 2001]. This proposal was rejected based on a new LCA study developed by Fullana-i-Palmer et al. (2005);

- in the last two revisions of the Industrial Waste Management Programme of Catalonia (PROGRIC), in which the use of a LCT approach to waste policy was mandated [Fullana-i-Palmer et al., 2011];

- in the extensive use of LCA as a tool to support decision making on the best waste treatment methods in many countries, including Italy, Spain, Sweden, Germany, UK, Turley, USA, Kuwait, Singapore and China, among others [Bovea et al., 2010; Al-Salem & Lettieri, 2009].

The methodological development of LCA for waste management and its acceptance as a tool for assisting decision making have gone hand in hand with the development of models and software tools to facilitate its implementation also by non-LCA experts. These models have been developed almost independently, and mainly in Europe and North America, from the mid 90´s to the present time by a wide range of universities, consultancies or environmental protection agencies [**Bala et al., 2015, II**] (for further details see Table 2.2 in Chapter 2, **II**.).

1.3 Rationale and research objectives

1.3.1 Rationale

Waste management has been identified by the European Commission, the United States Environmental Protection Agency and the Organization for Economic Cooperation and Development, among other institutions, as a key issue for the achievement of a resource-efficient society and for achieving a sustainable economy. Waste management practices need to be improved in order to reinforce material recycling, closing essential material loops and also recovering energy from waste, while at the same time ensuring that toxic substances are not released to the environment. This essentially means moving from a linear 'extraction-use-disposal' economic model to a more circular one (see Figure 1.1), in which wastes from different sources become raw materials for other activities, fostering what is often referred to as an *industrial ecology*, a *smart economy*, a *green economy* or (more recently) a *circular economy.*

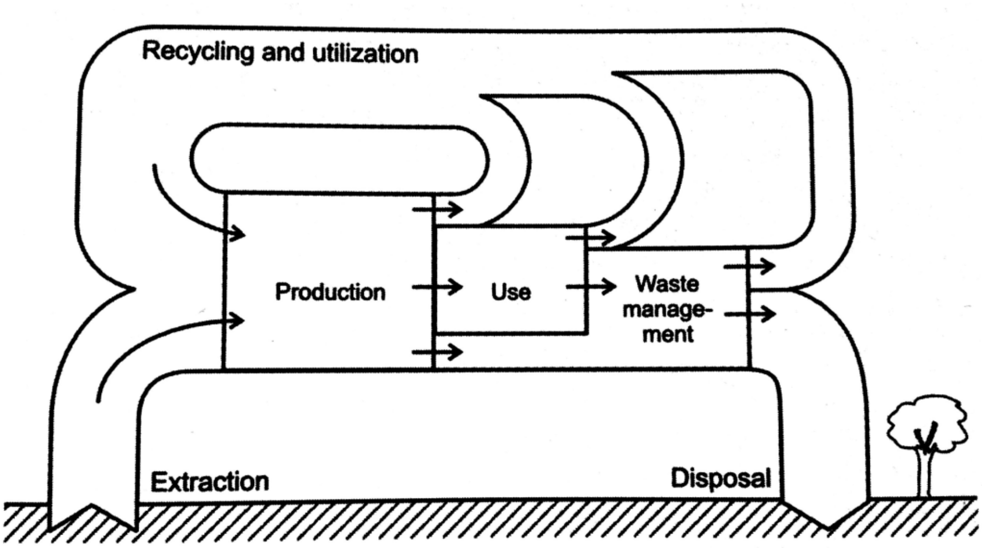

Figure 1.1: Schematic material flow in society showing extraction of resources, production, use, waste management and disposal into the environment [Extracted from Christensen (2011); previously adapted from Vesilind et al. (2002)]

Nevertheless, in order to obtain raw materials or energy from waste flows and close cycles, it is necessary to collect, transport, classify and finally process them, by means of recycling or energy recovery processes, which imply the consumption of water, energy and additional materials as well. From an environmental point of view, raw materials, energy consumption and emissions during waste collection and treatment, are to be compared to the levels consumed and emitted during the recycling or energy recovery processes. Thus, a waste management strategy should only be considered *efficient* if the amount of resources employed for the waste management itself is less than the amount of energy and materials that can eventually be recovered from the waste. Under this premise, efficient planning for integrated solid waste management systems (IWM) requires accounting for the complete set of environmental effects associated with the entire life cycle of solid wastes [Wassermann et al., 2005], considering a holistic approach.

As stated in the previous section, LCA is widely accepted as the best tool to provide such global and expanded view of the system, taking into account all the processes involved and its interactions with the economy (by means of reintroducing recycled materials or energy recovered from the waste management system), as well as accounting for a complete set of environmental effects (i.e. climate change, ozone layer depletion, acidification or eutrophication).

Methodological choices affecting the final results

Since the beginning of the use of LCA for the analysis of WMS, a number of methodological issues have been raised, and improvements suggested, by the scientific community, allowing the development of the methodology in order to better describe and be able to account for the real environmental loads of WMS. This fact is exemplified by the relevant differences observed in some studies conducting a comparison of the same waste management systems, or the comparison of different waste treatment technologies, using some of the existing models to perform LCA of WMS (further details are included in **Bala et al., (2015) [II]**).

The first step to perform an LCA of a WMS is to know the amount and composition of waste[14]. In fact, this is crucial for performing LCA and proper WMS [Coleman et al, 2003; Ekvall et al. 2007] and policy-making [Del Borghi et al, 2009]. However, given a) the large number and complexity of the products which eventually end up becoming waste, b) the high speed of generation of waste, and c) changes in its composition because of seasonality reasons or differences between consumer’s behaviors, it is practically impossible to know the real amount and composition of waste that is being treated by a WMS. This fact leads to two major technical difficulties: uncertainties about the behavior of waste in waste treatment facilities, and particularly in landfills, and uncertainties for the calculation of the toxicity effect of waste on human and natural ecosystems.

In addition to the technical difficulties related to the impossibility of knowing the real amount and composition of waste, there are also methodological issues when applying LCA methodology to the analysis of WMS that may have a great influence on the final results of the analysis and have already been identified (see Table 1.1). The most important methodological issues to be considered in LCA of WMS are described in more detail in Appendix C.

Most of these issues are discussed and modeling solutions are suggested in the specific guidelines about Waste Management and LCA developed by the Joint Research Centre of the European Union [JRC, 2011a, 2011b and 2011c], which gather the most recent scientific consensus, and which are built on the International Organization for Standardization (ISO) 14040 and 14044 standards for LCA and the International Reference Life Cycle Data System (ILCD) Handbook [IES, 2010].

[14] By “composition” I refer either at the primary waste composition level (paper, plastics, metals, etc.), the secondary composition level (types of plastics, types of papers, etc.) as well as the tertiary composition level (waste specific composition in terms of elementary elements: nitrogen (N), carbon (C), phosphorus (P), etc.).

Table 1.1: Identified methodological issues in LCA of WMS with a strong effect on the results

Methodological issue	Comments	Authors who have mentioned it
Allocation of environmental burdens to different life cycles	Important for LCA of products when recycling is considered (more than for LCA of WMS itself).	[Ekvall and Tillman, 1997; Winkler, 2007; Björklund & Bjuggren, 1998; Björklund, 1998; Eriksson, 2003]
Assumptions made in the study	Hypothesis, allocations, etc.	[Ekvall et al., 2007]
Biotic Carbon and Carbon Sinks	The common practice is to count the GWP of biotic CO_2 as zero.	[Björklund et al., 2011]
Choice of time perspective	This choice has more to do with characterization factors and landfill models.	[Finnveden et al., 1995; Obersteiner et al., 2007; Hyks et al. (2009); Gentil, 2011; Björklund et al., 2011]
Energy systems	Energy systems considered to be displaced by the energy recovered from the WMS (marginal/average).	[Björklund & Bjuggren, 1998; Björklund, 1998; Eriksson, 2003; Gentil et al., 2010]
Functional unit definition	To which inventory and results are related to.	[Björklund & Bjuggren, 1998; Björklund, 1998; Eriksson, 2003; Gentil et al., 2010]
Modelling of environmental impacts	Related to the characterization factors applied, and especially human toxicity [(a)].	[Ekvall et al., 2007]
System boundaries	Including boundaries of the system and the environment, time horizon, energy system boundaries and cut-off criteria.	[Wenzel and Villanueva, 2006; Guinée, 2002, Gentil et al., 2010; Björklund et al., 2011]
Waste composition	Primary, secondary and tertiary levels [(b)].	[Björklund & Bjuggren, 1998; Björklund, 1998; Eriksson, 2003, Del Borghi et al., 2009; Coleman et al, 2003; Ekvall et al. 2007; Gentil et al., 2010]
Waste management processes	Different inventories, technologies, modeling principles and calculation methods.	[Björklund & Bjuggren, 1998; Björklund, 1998; Eriksson, 2003; Gentil et al., 2010]
	And especially mechanical biological treatment, landfill and incineration.	[Gentil, 2011]

(a) Toxicity characterization factors were identified by the Apeldoorn Declaration[15] as critical, due to the impossibility of knowing the behavior of all chemical substances included in products and wastes. For that reason, it was suggested to perform sensitivity analyses of human and ecological toxicities using different characterization methods. Knowing this problem, the UNEP/SETAC Life Cycle Initiative endorsed the USETox Project. USEtox is a scientific consensus model for characterizing human and ecotoxicological impacts of chemicals in life cycle impact assessment. Main output is a database of recommended and interim characterization factors including fate, exposure, and effect parameters for human toxicity and ecotoxicity (more information can be obtained from www.usetox.org and Hauschild et al., 2008).

(b) Efforts for gathering data about waste generation have been done by the European Union. The European Union of Waste Statistics Regulations requires data to all Member States every 2 years on the generation and treatment of waste. Eurostat, the Statistical Office of the European Communities, compiles the data to provide statistics at European level. About composition, public information is scarcer. If the composition is not known, the National Inventory Reports (NIR) to the UNFCCC can be used. However, it is important to mention that those inventories assumes that the composition of waste has not changed in the period 1950-2004 and that it will not change in the period 2005-2020.

More possible areas of improvement identified

Even though important methodological issues have been identified and solved, and some of them are still under discussion, I consider that there are still three main issues that merit special attention.

The first one is related to the calculation of the environmental performance of a recycling system. Usually, its environmental performance is calculated by means of subtracting the environmental load associated to collect and transport waste from the environmental benefits due to the recycling of those materials, considering that the secondary materials equal virgin materials in terms of quantity and quality [see for instance Prognos et al., 2008; Smith et al., 2001; US EPA, 2006]. Or, in other words, that one tonne of a recycled material displaces a tonne of the virgin material from which it primarily origins. Even though some authors have corrected this amount by introducing a correction factor considering technical properties of the recycled materials versus the virgin ones, this is essentially the way how the benefits from recycling have been calculated in the majority of LCA of WMS.

In LCA of products there is currently an open debate about how to allocate the environmental burdens of the end-of-life stage and the production stage when recycled materials are used or recyclable materials are produced in the same product system. Different approaches are under discussion [AFNOR, 2011; BSI, 2011; EC, 2012; Wolf and Chomkhamsri, 2014] as a result of the *Single Market for Green Products Initiative* which establishes a method to measure environmental performance of products throughout the lifecycle, the so-called *Product Environmental Footprint* (PEF), an attempt by the DG Environment, the European

[15] The April 15th, 2004 a group of specialists in the areas of LCA, LCIA and Risk Assessment went toghether at TNO (Netherlands Organization for Applied Scientific Research) in Apeldoorn to discuss current practices and complication of LCIA methodologies for non-ferro metals. The complete text can be found at http://media.leidenuniv.nl/legacy/declaration_of_apeldoorn.pdf.

Commission's Joint Research Centre (JRC IES) and other European Commission services to harmonize methodology for the calculation of the environmental footprint of products (including carbon footprint), which is currently under development[16]. Conversely, the effects of moving towards a more circular economy and how credits due to material recovery should be calculated are not being called into question in LCA of WMS to the same extent[17].

The second issue is strictly connected to the first one stated, but focuses the attention on the collection system itself. Whereas this part of the WMS accounts for the majority of the economic costs of the system [Sonesson, 2000], a large amount of LCAs have shown that its overall environmental effect remains comparatively small, assuming that the collection and the transport system are reasonably efficient. However, some authors questioned this premise of efficiency and pointed out that this stage of the WMS can have a major influence depending on the conditions in which it is implemented. For instance, Klang (2005) states that in rural and sparsely populated areas of Sweden it is «more difficult to transform their waste systems in a more sustainable direction», due to small waste volumes, long collection routes and distant treatment facilities. Along the same lines, Tanskanen & Kaila (2001) pointed out that the increasing complexity of the collection systems in terms of the amount of different fractions collected separately and, as a consequence, the needed transport and consumption of diesel may increase the importance or this stage in WMS. Additionally, Salhofer and colleagues (2007) demonstrated how transport distances may influence the environmental benefit of recycling different waste flows (refrigerators, waste paper, polyethylene films and expanded polystyrene). All this calls for a better focus on the collection stage to assess how it is addressed in LCA of WMS and how to calculate its environmental loads.

The third issue to be addressed is the capability of the current life cycle impact assessment methods to answer the challenge of resource sustainability. This matter is currently debated in the literature [Klinglmair et al., 2014]. In LCA, flows outside the market dynamics (such as

[16] The Product Environmental Footprint (PEF) is one of the actions included in the *Single Market for Green Products* initiative of the European Comission [EC, 2013]. The aim of this initiative is to harmonize different existing methodologies (ISO 14025:2006, BP X30-323-0:2011, GHP Protocol, PAS 2050, ISO/TS 1067:2013, ISO 14020:2000, ISO 14021:1999, ISO 14040:2006, ISO 14044:2006, ISO 14050:2006 and ISO 17024:2003) in order to obtain environmental footprints of products and organizations applicable to the whole European Market [EC, 2012]. Two waves of pilot tests are currently under development. The fist one started on the 1st of November 2013 and the second one on the 2nd of June 2014. Final Product Category Rules are foreseen to be available by December 2016 [Imola Bedo, 2014]. More information can be found in the website of the European Comission:
http://ec.europa.eu/environment/eussd/smgp/index.htm

[17] One exception can be found in the PhD of Sevigné, E. (2014), who analyses the effects in savings of green house gases emissions (GHG) associated to material recycling. She takes into account the market and the international trade for waste paper, aluminium old scrap and plastic waste, and considers synergies between LCA and material flow analysis (MFA) methodologies. However, the study is restricted to a consequential LCA approach.

environmental services and renewable resources that do not flow through human controlled devices) are not generally included. LCA usually operates under a natural environment supply-side perspective, starting to account for environmental impacts at the point where materials are extracted from the earth (cradle). As a consequence, the supply-side quality and degree of renewability of resources, in terms of biosphere activity leading to resource generation processes, are not generally taken into account in LCA [Ulgiati et al., 2006]. Even where renewable flows are included (e.g. in the calculation of cumulative energy demand (CED)), their inclusion only refers to the renewable fraction captured under human control (e.g. the amount of sunlight actually captured by photovoltaic modules). Under a circular economy perspective, and in order to convert our society to a more sustainable one, it would arguably be important to be able to estimate the "real" environmental load of producing materials, in terms of the efforts employed by natural ecosystems to produce such raw materials. In other words, the larger the effort made by nature to provide the resources used, the more responsibility we should take on the use of those materials [Raugei et al., 2014]. Because of the current impossibility of LCA characterization methods to account for this effort of natural ecosystems, synergies with other methodologies were sought in order to complete the analysis (**Bala et al., 2015 [III]**).

1.3.2 Research objectives

The research objectives of this thesis are listed below, as well as the particular paper in which they are tackled (in brackets) and the main hypothesis behind them:

- Analysing the existing methods for calculating the avoided impacts of WMS associated to material recovery, and proposing a new method (Paper I, Chapter 2). This first objective aims at correcting the potential overestimation of the benefits of material recycling when the commonly used rule of considering that recycled materials displace virgin material production with a 1:1 substitution ratio is applied.

- Assessing the environmental performance of the waste collection phase and developing a new predictive model for evaluating its environmental load (Paper II, Chapter 2). This second objective aims at improving on the simplistic ways in which waste collection is modelled in most LCA studies and software packages for conducting LCA of WMS.

- Performing a methodological overview and analysing synergies between LCA and Emergy Accounting in order to help to improve the Resources Depletion impact category in LCA (Paper III, Chapter 2).

1.4 References

- AFNOR, 2011. BPX 30-323.0: Affichage environnemental des produits de grande consummation – Principles généraux et cadre méthodologique. Ed. AFNOR, La Plaine Saint-Denis, France.
- Al-Salem S.M. and Lettieri P., 2009. Life Cycle Assessment (LCA) of Municipal Solid Waste Management in the State of Kuwait. European Journal of Scientific Research 34 (3), 395-405. ISSN: 1450-216X.
- Bala A. and Raugei M., 2009. Waste Management. Chapter 5 of the Book "Life Cycle Assessment applications: results from COST Action 530. A joint effort by COST Action 530: Life-Cycle Inventories for Environmentally-Conscious Manufacturing Processes". Ed. Fullana P., Betz M., Hischier R., Puig R. AENOR, Madrid. ISBN: 978-84-8143-611-2.
- Banar M., Cokaygil Z., Ozkan A., 2009. Life cycle assessment of solid waste management options for Eskisehir, Turkey. Waste Management 29, 54-62.
- Barceló M., Barranco M., Huguet Ll., Navarro J., Nieto A., Pascual P., Pla J., Poves A., Puigdomènech P., Rifà J., Ripoll A., Roqué X., Salvador E., Serra F., Solanes J.M., Valverde Ll., 2000. 10 Impactes de la Ciència del Segle XX. Joaquim Pla (Ed.) Eumo Editorial. Vic. ISBN: 9788476025697.
- Beigl P. and Salhofer S., 2004. Comparison of ecological effects and cost communal waste management systems. Resources, Conservation and Recycling 41, 83-102.
- Bjarnadóttir H.J., Fridriksson G.B., Johnsen T., Sletsen H., 2002. Guideline for the use of LCA in the waste management sector, Nordtest, Espoo, Finland (Nordtest Report 517).
- Björklund A. and Bjuggren C., 1998. Waste modelling using substance flow analysis and life cycle assessment. In: Proceedings of the Air and Waste Management Associations Annual Meeting, June 14–18, San Diego, CA, USA (Paper 98-A431).
- Björklund A., 1998. Environmental Systems Analysis Waste Management with Emphasis on Substance Flows and Environmental Impact. Licentiate Thesis. Royal Institute of Technology, Industrial Ecology, Stockholm, Sweden.
- Björklund A., Finnveden G., Roth L., 2011. 3.2. Application of LCA in Waste Management. Christensen, T.H. (Ed.) Solid Waste Technology and Management. Blackwell Publishing.
- Bovea M.D., Ibáñez-Forés V., Gallardo A., Colomer-Mendoza F.J., 2010. Environmental assessment of alternative municipal solid waste management strategies. A Spanish case study. Waste Management. DOI: 10.1016/j.wasman.2010.03.001.
- British Standards Institution (BSI), 2011. PAS 2050: Specification for the assessment of the life cycle greenhouse gas emissions of goods and services.
- Christensen T.H., 2011. Introduction to Waste Management. In Christensen, T.H. (Ed.), Solid Waste Technology and Management, Chapter 1.1. John Wiley & Sons, Ltd, Chichester. ISBN: 978-1-405-17517-3.

- Clift R., Doig A., Finnveden G., 2000. The application of life cycle assessment to integrated solid waste management. Part 1 - Methodology. Process Safety and Environmental Protection 78 (4), 279-287.
- Coleman T., Masoni P., Drier A., McDougall F., 2003. International Expert Group on Life Cycle Assessment for Integrated Waste Management. The International Journal of Life Cycle Assessment 8 (3) 175-178.
- Del Borghi A., Gallo M., Del Borghi M., 2009. A survey of life cycle approaches in waste management. The International Journal of Life Cycle Assessment 14, 597-610. DOI: 10.1007/s11367-009-0111-7.
- Denison R., 1996. Environmental Life Cycle comparisons of recycling, landfill and incineration: a review of recent studies. Annual Review of Energy and the Environment 21, 191-237.
- Ekvall T. and Tillman A.M., 1997. Open-loop recycling: criteria for allocation procedures. The International Journal of Life Cycle Assessment 2 (3), 155–162.
- Ekvall T., Assefa G., Björklund A., Eriksson O., Finnveden G., 2007. What life-cycle assessment does and does not do in assessments of waste management. Waste Management 27, 989-996.
- Eldredge R.W. and Hichman Jr. H.L., 1999. A Brief History of Solid Waste Management in the US during the last 50 years. MSW Management: The Journal for Municipal Solid Waste Professionals (September) 9 (5).
- Emery A., Davies A., Griffiths A., Williams K., 2007. Environmental and economic modelling: A case study of municipal solid waste management scenarios in Wales. Resources Conservation and Recycling 49, 244-263.
- Eriksson O., 2003. Environmental and Economic Assessment of Swedish Municipal Solid Waste Management. PhD Thesis. Industrial Ecology, Royal Institute of Technology, Stockholm, Sweden.
- European Comission, 2005. Communication from the Commission to the Council, the European Parliament, the European Economic and Social Committee and The Committee of the Regions - Taking sustainable use of resources forward - A Thematic Strategy on the prevention and recycling of waste. COM (2005) 666 final.
- European Comission, 2010. Comunication from the Comission. Europe 2020. A strategy for smart, sustainable and inclusive growth. Brussels, 3.3. 2010. COM (2010) 2020 final.
- European Comission, 2013. Communication from the Commission to the European Parliament and the Council. Building the Single Market for Green Products. Facilitating better information on the environmental performance of products and organizations. Brussels, 9.4.2013. COM (2013) 196 final.
- European Commission, 2001. Environment 2010: our future, our choice. 6th EU Environmental Action Program. Luxembourg: Office for Official Publications of the European Communities. ISBN: 92-894-0261-X.
- European Commission, 2014. Horizon 2020. Work Programme 2014-2015. 12. Climate action, environment, resource efficiency and raw materials. Available online in:

http://ec.europa.eu/research/participants/data/ref/h2020/wp/2014_2015/main/h2020-wp1415-climate_en.pdf (Consulted in: 07/06/2014).

- European Commission, 2012. Product Environmenatl Footprint (PEF) Guide. Deliverable 2 and 4A of the Administrative Arrangement between DG Environment and the Joint Research Centre Nº N 070307/2009/552517, including Amendment Nº 1 from December 2010. Ares (2012)873782-17/07/2012.
- Finnveden G. and Ekvall T., 1998. Life Cycle Assessment as a decision-support tool – The case of recycling versus incineration of paper. Resources, Conservation and Recycling 24 (3-4), 235-256.
- Finnveden G. and Ekvall T., 1999. Environmental aspects of energy and material recovery of paper – today and in a more sustainable future. Appendix VII in Ekvall, T.: System Expansion and Allocation in Life Cycle Assessment, with implications for Waste Paper Management. Dissertation, Chalmers University of Technology.
- Finnveden G., 1999. Methodological Aspects of Life Cycle Assessment of Integrated Solid Waste Management Systems. Resources, Conservation and Recycling, 26, 173-187.
- Finnveden G., Albertsson A.C., Berendson J., Eriksson E., Höglund L.O., Karlsson S., Sundqvist J.O., 1995. Solid waste treatment within the framework of life-cycle assessment. Journal of Cleaner Production 3, 189–199.
- Fullana-i-Palmer P., Puig R., Bala A., Baquero G., Riba J., Raugei M., 2011. From Life Cycle Assessment to Life Cycle Management. A Case Study on Industrial Waste. Journal of Industrial Ecology 15 (3), 458-475.
- Fullana-i-Palmer P, Weidema B., Frankle P, Wolf M.A., 2005. Paper on the position of the Waste Agency of Catalonia, GOVERNMENT OF CATALONIA, in response to the Consultation issued by the EC DG Environment on the Directive 75/439/EEC on waste oils. Waste Agency of Catalonia. Ministry of Environment. Government of Catalonia. March, 2005.
- Gentil E., 2011. Life-cycle modelling of waste management in Europe: tools, climate change and waste prevention. PhD Thesis. Technical University of Denmark: Kgs. Lyngby, Denmark.
- Gentil E.C., Damgaard A., Hauschild M., Finnveden G., Eriksoon O., Thorneloe S., Kaplan P.O., Barlaz M., Muller O., Matsui Y., Li R., Christensen T.H., 2010. Models for waste life cycle assessment: Review of technical assumptions. Waste Management 30 (12), 2636-2648. DOI: 10.1016/j.wasman.2010.06.004.
- Güereca L.P., 2006. Desarrollo de una metodología para la valoración en el análisis de ciclo de vida aplicada a la gestión integral de residuos municipales. Tesis Doctoral. UPC. Barcelona.
- Güereca L.P., Gassó S., Baldasano J.M., Jimenez-Guerrero P., 2006. Life Cycle Assessment of two biowaste management systems for Barcelona, Spain. Resources, Conservation and Recycling 49 (1), 32-48.
- Guinée J.B., 2002. Handbook on Life Cycle Assessment: Operational Guide to the ISO Standards. Dordrecht, The Netherlands, p. 692.

- Hauschild M. and Barlaz M.A., 2011. LCA in waste management: Introduction to principle and method. In Christensen, T.H. (Eds.), Solid Waste Technology and Management, Chapter 3.1. John Wiley & Sons, Ltd, Chichester. ISBN: 978-1-405-17517-3.
- Hauschild M. Z., Olsen S.I., Hansen E., Schmidt A., 2008. Gone…but not away – addressing the problem of long-term impacts from landfills in LCA. The International Journal of Life Cycle Assessment 13 (7), 547-554.
- Hyks J., Astrup T., Christensen T.H., 2009. Long-term leaching from MSWI air-pollution-control residues: Leaching characterization and modeling. Journal of Hazardous Materials 162 (1), 80-91. DOI: 10.1016/j.jhazmat.2008.05.011.
- Imola Bedo, 2014. Production Coordinator at European Commission DG Environment.
- Institute for Environment and Sustainability (IES), Joint Research Centre, European Commission, 2010. International Reference Life Cycle Data System (ILCD) handbook. General guide for Life Cycle Assessment – Detailed guidance. First edition, 2010.
- Institute for European Environmental Policy (IEEP), 2008. www.ieep.eu (http://213.198.115.240/activities/policyresearch/wastemanagement.php) (last consultation: 2/03/2012)
- Iriarte A., Gabarrell X., Rieradevall J., 2009. LCA of selective waste collection systems in dense urban areas. Waste Management 29 (2), 903-914
- Joint Research Centre, 2007a. Scientific and Technical Reports. Environmental Assessment of Municipal Waste Management Scenarios: Part I – Data collection and preliminary assessment for life cycle thinking pilot studies. DOI: 10.2788/4915.
- Joint Research Centre, 2007b. Scientific and Technical Reports. Environmental Assessment of Municipal Waste Management Scenarios: Part II Detailed Life Cycle Assessments. DOI: 10.2788/49570.
- Joint Research Centre, 2011a. Supporting Environmentally Sound Decisions for Waste Management. A technical guide to Life Cycle Thinking (LCT) and Life Cycle Assessment (LCA) for waste experts and LCA practitioners. JRC scientific and Technical Reports. EUR 24917 EN- 2011. Manfredi, S. and Plant, R. Editors. Institute for Environmentant and Sustainability. Joint Research Centre. European Comission. DOI: 10.2788/52632.
- Joint Research Centre, 2011b. Supporting Environmentally Sound Decisions for Bio-Waste Management. A practical guide to Life Cycle Thinking (LCT) and Life Cycle Assessment (LCA). JRC scientific and Technical Reports. EUR 24916 EN- 2011. Manfredi, S. and Plant, R. Editors. Institute for Environmentant and Sustainability. Joint Research Centre. European Comission. DOI: 10.2788/53942.
- Joint Research Centre, 2011c. Supporting Environmentally Sound Decisions for Construction and Demolition (C&D) Waste Management. A practical guide to Life Cycle Thinking (LCT) and Life Cycle Assessment (LCA). JRC scientific and Technical Reports. EUR 24918 EN- 2011. Manfredi, S. and Plant, R. Editors. Institute for Environmentant and Sustainability. Joint Research Centre. European Comission. DOI: 10.2788/54618.

- Klang A. G., 2005. Sustainable Waste Management. Methods and framework for analysis. PhD in Industrial Ecology. Dissertation. Norwegian University of Science and Technology. Department of Hydraulic and Environmental Engineering. NTNU-trykk, Trondheim. ISBN: 82-471-6894-4.
- Klinglmair M., Sala S., Brandao M., 2014. Assessing resource depletion in LCA: a review of methods and methodological issues. The International Journal of Life Cycle Assessment 19, 580-592. DOI: 10.1007/s11367-013-0650-9.
- Koci V. and Trecakova T., 2011. Mixed municipal waste management in the Czech Republic from the point of view of the LCA method. The International Journal of Life Cycle Assessment 16, 113-124.
- McCorriston J. and Weisberg S., 2002. Spatial and Temporal Variation in Mesopotamian Agricultural Practices in the Khabur Basin, Syrian Jazira. Journal of Archeological Science 29 (5), 485-498.
- McGovern T.W and Friedlander A.M., 1997. Plague. In Textbook of Military Medicine: Medical aspects of Chemical and Biological Warfare. Ed. F. R., Sidell, E.T. Takafuji and D.R. Franz. Office of the Surgeon General, U.S. Department of the Army, 479-502.
- Merrid H., Damgaard A., Christensen T.H., 2008. Life cycle assessment of waste paper management: the importance of technology data and system boundaries in assessing recycling and incineration. Resources, Conservation and Recycling 52 (12), 1391-1398.
- Monier V. and Labouze E., 2001. Critical Review of Existing Studies and Life Cycle Analysis on the Regeneration and Incineration of Waste. European Commission. DG Environment. A2-Sustainable Resources Consumption and Waste. TAYLOR NELSON SOFRES CONSULTING and BIO INTELLIGENCE SERVICE. December.
- Muñoz I., Rieradevall J., Domènech X., Milà L., 2004. LCA application to integrated waste management planning in Gipuzkoa (Spain). The International Journal of Life Cycle Assessment 9 (4), 272-280.
- Nye D.E., 1994. American Technological sublime. Third printing (1999). The MIT Press. Massachusetts Institute of Technology. ISBN: 0-262-14056-X (HB), 0-262-64034-1 (PB).
- Obersteiner G., Binner E., Mostbauer P., Salhofer S., 2007. Landfill modelling in LCA – A contribution based on empirical data. Waste Management 27 (7), 939–955.
- Organization for Economic Co-operation and Development (OECD), 2004. Recommendation of the Council on the Environmentally Sound Management of Waste. 9 June 2004- C(2004)100.
- Organization for Economic Co-operation and Development (OECD), 2007. Guidance Manual for the Implementation of the OECD Recommendation C(2004)100 on Environmentally Sound Management (ESM) of Waste. The Organisation for Economic Co-operation and Development.
- Prentice M.B and Rahalison L., 2007. Plague. Lancet. Apr 7; 369(9568):1196-207.
- Prognos AG, Ifeu and INFU, 2008. Resource savings and CO2 reduction potential in waste management in Europe and the possible contribution to the CO2 reduction target in 2020.

- Raugei M., Rugani B., Benetto E., Ingwersen W.W., 2014. Integrating emergy into LCA: Potential added value and lingering obstacles. Ecological Modelling 271, 4-9.
- Rigamonti L., Grosso M., Sunseri M.C., 2009. Influence of assumptions about selection and recycling efficiencies on the LCA of integrated waste management systems. Int J Life Cycle Assessment 14: 411-419.
- Rodrigo J. and Castells F., 2000. Environmental evaluation of different strategies for the management of municipal waste in the region of Catalonia. Project founded by Junta de Residus, the regional Waste Agency of Catalonia.
- Salhofer S., Schneider F., Obersteiner G., 2007. The ecological relevance of transport in waste disposal systems in Western Europe. Waste Management 27, 47-S57.
- Sevigné E., 2014. Potential savings of resources and greenhouse gas emissions from waste management: a case study of Spain in a global Economy. PhD on Environmental Science and Technology. Institute of Environmental Science and Technology (ICTA) at Universitat Autònoma de Barcelona (UAB).
- Smith A., Brown K., Ogilvie S., Rushton K., Bates J., 2001. Waste Management Options and Climate Change. Final Report to the European Commission. AEA Technology Environment. Luxemburg: Office for Official Publications of the European Communities.
- Sonesson U., 2000. Modelling of waste collection – a general approach to calculate fuel consumption and time. Waste Management and Research 18 (2), 115-123.
- Steward K.M and Gifford-Gonzalez D., 1994. An Ethnoarcheological Contribution to Identifying Hominid Fish Processing Sites. Journal of Archeological Science 21(2), 237-248.
- Tanskanen J. H. and Kaila J., 2001. Comparison of methods used in the collection of source-separated household waste. Waste Management and Research 19 (6), 486-497.
- Te Brake W.H., 1975. Air Pollution and Fuel Crises in Preindustrial London, 1250-1650. Technology and Culture 16 (3), 337-359.
- Thorneloe S.A., Weitz K., Jambeck J., 2005. Moving from Solid Waste Disposal to Materials Management in the United States. Presentation at the tenth International Waste Management and Landfill Symposium in Cagliari, Italy on October 3-7.
- Thorneloe S.A., Weitz K., Jambeck J., 2007. Application of the US decision support tool for materials and waste management. Waste Management 27 (8), 1006-1020.
- U.S. EPA, 2004. Resource Conservatioon Challenge – A Year of Progress, EPA 530-R-04-001, February.
- Ulgiati S., Raugei M., Bargigli S., 2006. Overcoming the inadequacy of single-criterion approaches to Life Cycle Assessment. Ecological Modelling 190 (3-4), 432–442.
- US EPA, 2006. Solid Waste Management and Greenhouse Gases. A Life-Cycle Assessment of Emissions and Sinks. Executive Summary.
- Vesilind P.A., Worrell W., Reinhart D., 2002. Solid Waste Engineering. Brooks/Cole, Pacific Grove, USA.

- Villanueva A. and Wenzel H., 2007. Paper waste - recycling, incineration or landfilling? A review of existing life cycle assessments. Waste Management 27 (8), S29-S46.
- Wassermann F., Binner E., Salhofer S., 2005. Influencing Waste Management Policy by using LCA in a SEA process. LCM 2005 – Innovation by Life Cycle Management International Conference. Vol I. Barcelona, September 5-7.
- Wenzel H. and Villanueva A., 2006. The significance of boundary conditions and assumptions in the environmental life cycle assessment of waste management strategies. In: NorLCA Proceedings, Lund, Sweden.
- Winkler J., 2007. Comparative evaluation of life cycle assessment models for solid waste management. Waste Management 27 (7), 921–931.
- Wolf M.A. and Chomkhamsri K., 2014. The “integrated formula” for modeling recycling, energy recovery and reuse in LCA. White paper. Berlin, 12 October.
- Wollny V., Dehoust G., Fritsche U.R., Weinem P., 2001. Comparison of Plastic Packaging Waste Management Options: Feedstock Recycling versus Energy Recovery in Germany. Journal of Industrial Ecology 5 (3), 49–63.

CHAPTER 2. Papers

«Testing* occurs as part of the competition
between rival paradigms for the allegiance of the scientific community.»

** referred to a model or a theory*

[Thomas Kuhn, 1922-19961,
American physicist, historian and philosopher of science]

PAPER I: INTRODUCING A NEW METHOD FOR CALCULATING ENVIRONMENTAL CREDITS OF END-OF-LIFE MATERIAL RECOVERY IN ATTRIBUTIONAL LCA.

Submitted to: The International Journal of Life Cycle Assessment.

Current state: accepted.

Authors: Alba Bala Gala, Marco Raugei and Pere Fullana-i-Palmer

ABSTRACT

Purpose This paper aims to provide an alternative method for calculating the environmental credits associated with material recycling in life cycle assessment (LCA) of waste management systems. The method proposed here is more consistent with the general attributional approach in LCA than the hitherto common practice of simply assuming a 1:1 substitution of primary material production.

Methods The formula proposed for estimating the environmental credit is applicable for the recovered materials that are reintroduced into the market (outputs of the recycling facilities), after all process losses in the various stages of the waste management system have been accounted for. It considers the displacement of materials by using the mix of virgin and recycled materials for each individual material that is used in the market for the production of goods. Moreover, it also considers the changes in the inherent properties of the materials undergoing a recycling process ('down-cycling'), by introducing a quality (Q) factor, affecting the proportion of virgin material that is accounted for.

Results and discussion Example applications of the proposed formula to a number of different materials (aluminium, steel, paper and cardboard and plastics) illustrate the range of possible results obtained. The environmental credit calculated using the proposed formula can be interpreted as an indication of the remaining margin for improvement, since it depends on the existing mix of virgin and recycled materials already on the market, and on the potential of the recycled material to actually replace the primary one on a functional basis. We also discuss the possible use of a material's Q factor to estimate the maximum allowable % of recycled material in a product consistent with the quality demands of selected applications.

Conclusions and recommendations We have introduced here a consistent and unified formula for the evaluation of the credits associated with material recovery of all waste materials in waste management systems (paper, glass, plastics, metals, etc.). Such a formula requires the knowledge of the current average market consumption mixes of primary and secondary materials (or the application-specific average mixes when the final application of the recovered materials is known), and of suitable Q factors for the material(s) that are recycled. As the latter are often not readily available, more research is called for to arrive at a ready-to-use Q factors database.

KEYWORDS

Attributional LCA, avoided impact, environmental credit, LCA, material recycling, system expansion, waste management.

1. INTRODUCTION

Integrated waste management systems can be seen as multi-functional systems, from a life cycle assessment (LCA) perspective, in which the treatment of waste is the main function of the system and the recovered energy and materials are additional functions. To be able to compare different waste management alternatives and maintain the same functional unit, it is necessary to take into account both the credits of material and energy recovery as well as the environmental impacts due to the collection and treatment of all waste fractions. In this context, system expansion (also referred to using synonymous terms such as 'substitution', 'crediting', and 'system enlargement') is the common method to avoid resorting to the allocation of environmental impacts in all sub-steps of the waste management system and to maintain the same functional unit for the comparison (see for instance, Bjarnadóttir et al. 2002; JRC 2010; JRC 2011).

When carrying out system expansion, uncertainty in identifying the alternative system to produce the same product is introduced, regardless of the application of a consequential or attributional LCA perspective (see section 2.1). As mentioned by several authors (i.e. Finnveden & Ekvall 1998; Shonfield 2008; Michaud et al. 2010), deciding which systems are displaced may have a strong influence on the results of an LCA. Different ways of modelling recycling have been extensively discussed over the past two decades (JRC 2010). Whereas the effects of using different assumptions or approaches in relation to the energy that is substituted or displaced in waste management systems have been widely analysed (see for instance Finnveden et al., 2005; Smith et al. 2001; Eriksson et al. 2005; Bernstad & la Cour Jansen 2011; Laurent et al. 2014), the effects of material substitution have not been studied to the same extent. The vast majority of the LCA studies analysing the effects of recycling so far have assumed a 1:1 substitution ratio of recycled to virgin materials (Laurent et al., 2014). Such a substitution ratio applies at the point where the recycled materials are reintroduced into the market, after all losses due to impurities and process inefficiencies have been considered. A

1:1 substitution ratio implicitly means that recycled materials are supposed to replace the same amount of virgin materials with the same quality. Examples of this common practice can be found in: Björklund et al. 1999; Bovea et al. 2010; Dodbiba et al. 2008; Finnveden et al. 2005; Grant, et al. 2001; Merrild et al. 2008; Michaud et al. 2010; Muñoz et al. 2004; Perugini et al. 2005; Shen et al. 2010; Shonfield 2008; Smith et al. 2001; and US EPA 2006. A few studies also account for a decrease in the quality of the recycled materials and use varied assumptions and reduced substitution ratios (see for instance Bernstad, A. et al 2011 for paper recycling), sometimes applying different criteria depending on the type of material (plastics, paper, metals or glass) recovered (e.g., Finnveden et al. 2000; Prognos et al. 2008; Smith et al. 2001; and US EPA 2006).

However, only a few authors have addressed the influence of the substitution ratio through a sensitivity analysis, especially for those materials for which a 'down-cycling' occurs when they are recycled (i.e. for which a direct substitution on a like-for-like basis is not possible), such as paper and plastics. Among them, Gentil et al. (2009) undertook a sensitivity analysis for a range of substitution ratios ranging from 1:2 (i.e. 50% replacement of virgin material) to 1:1 (100% replacement). Although no substantial effects on the results were identified, compared with the effects associated with changes in other technology parameters, it was concluded that a country relying strongly on material recovery with a poor substitution ratio would have a higher GWP, compared to systems with better substitution ratios. Rigamonti et al. (2009), analysed the effects of a substitution ratio <1 for paper and plastics and observed a worsening of around 15-20% in several impact category indicators and up to 45% for GWP. Using the same substitution factors, the sensitivity analysis performed by Bovea et al. (2010) concluded that this choice has a significant influence on the results, up to 20-42% in some impact categories. Thus, it seems that the employed substitution ratio is a significant factor to take into account.

When analysing all these facts, two potential methodological issues arise: (1) whether the recycled materials effectively displace virgin materials in all cases (or a mix of virgin and secondary materials, see section 2.2), and (2) whether the technical quality of the recycled materials remains the same as that of the original virgin materials.

The aim of this paper is to propose a novel method for calculating the environmental credits due to material recycling in LCA of waste management systems, applicable to all waste materials, and more in line with the attributional approach in LCA than the extended practice of simply assuming the substitution of primary material production. The proposed formula takes into account the average mix of virgin and recycled materials used in the market for the production of new goods as the displaced material to be considered. Moreover, it also considers the changes in the inherent properties of the materials undergoing a recycling process ('down-cycling'), by introducing a quality (Q) factor, affecting the proportion of virgin material that is accounted for.

2. METHODOLOGICAL KEY POINTS

2.1. Attributional vs. Consequential approach

As mentioned above, two main modelling approaches to LCA are possible, namely attributional and consequential. The choice between using the former or the latter should be based on the fundamental context and purpose of the study. The ILCD Handbook (JRC, 2010) defines four major types of contexts: Situation A (micro-level decision support), Situation B (Meso/macro-level decision support) and Situations C1 and C2 (accounting with no decision support). As depicted in Figure 2.1, all approaches can be used for assessing the environmental performance of a system or to compare different waste treatment alternatives. In addition to the micro-, meso-, or macro-level decision context, we agree with Brandão and colleagues (2014) in claiming that attributional LCA is not an appropriate basis for policy development, but may be applicable in the context of policy implementation. Thus, a clear analysis of the purpose of the study and whether it is intended for policy development or implementation must also be taken into account for deciding the most appropriate modelling approach.

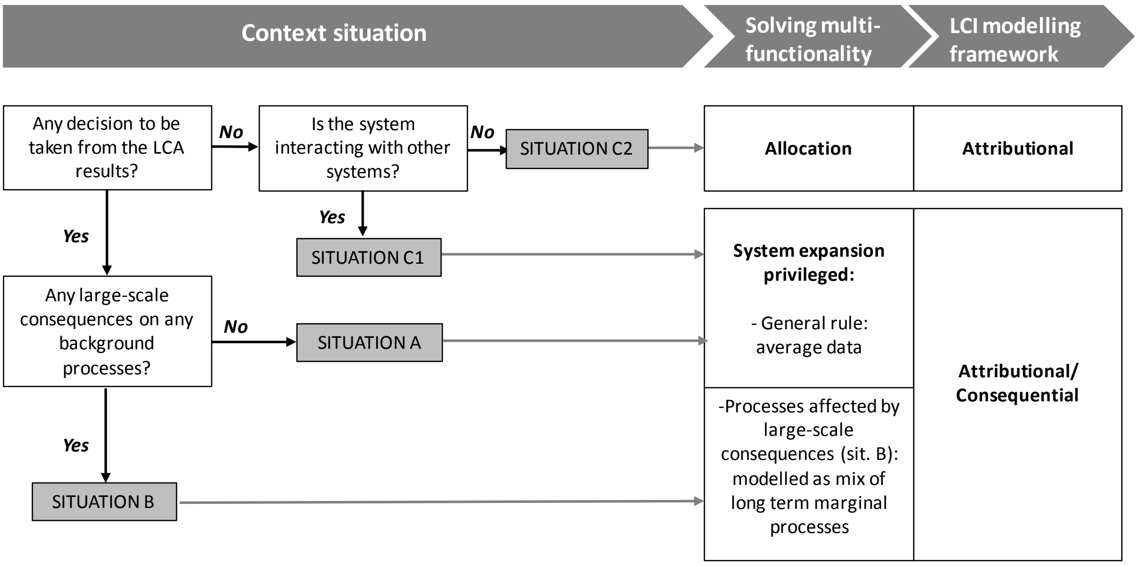

Figure 2.1: Identification of context situations from the ILCD Handbook (Source: Laurent et al., 2014)

A consequential approach assumes that the changes in the system under study have large-scale effects on the background system. Accordingly, the "avoided impacts" are estimated on the basis of the displaced marginal technologies – those that are directly affected by changes in demand (Weidema et al., 1999). This is arguably the most appropriate approach to be used for strategic decisions (situation B), including decisions on new investment policies (Finnveden et al. 2005), and to answer questions of the type: "what would be the consequences of developing a policy that would achieve an overall increased recycling rate for a given waste product/material?" In this case, the "what if" scenario would clearly entail a change in the background system, a change in the composition of the virgin/recycled mix for the particular material consumed, and the additional recycled material recovered would clearly displace its virgin counterpart.

At the same time, an attributional LCA approach is more appropriate to answer questions like: "what would happen if an existing source-sorting waste collection policy were implemented in additional 'X' sites/villages/etc.?" This is because the attributional LCA assumes that the analyzed system does not modify its environment or, in other words, does not affect in a significant way the environmental performance of the background systems that supply the materials and energy inputs required (situations C1 and A). In this case, the system should be modelled as it is (or was, or is forecast to be) using historical data. In this context, one may claim that each additional unit of waste material collected and recycled would displace an equivalent quantity of the current mix of virgin+recycled material being used as raw material by the market, without significantly affecting the composition of the overall mix; accordingly, the "environmental credits" of the recycled material should be calculated on the basis of the same mix of virgin+recycled (and not as the "avoided" 100% virgin) material.

Admittedly, if, in the same example above, the number of additional collection sites were large enough, the composition of the mix might end up being affected anyway, so the boundary of application of the two approaches is not clear-cut, but rather blurred. Moreover, as stated by Zamagni et al. (2012), "One should be careful, however, to note that the attributional/consequential dichotomy is constructed for the sake of argument. In practice, many LCAs are prospective based on scenarios for identified variables or explore the effect of identified causal changes while modelling the remainder of the system in an attributional manner".

This paper is however strictly meant to be confined to attributional LCA, and applicable to situations C1, A and B (if no large scale consequences in the background processes are produced). Also, it is recommended that the formula be applied in the context of waste policy implementation.

2.2. Virgin (marginal) vs. market mix (attributional) substitution

As discussed above, it has so far been common practice in LCA to assume a 1:1 substitution ratio of recycled to virgin materials (albeit sometimes accounting for a loss of technical properties in the recycled materials leading to a reduced ratio). Additionally, and as stated by Laurent et al. (2014), this practice has often been accompanied by a lack of transparency about whether average or case-specific primary production data were used to perform the system expansion.

From a strictly theoretical point of view, using primary production as the displaced process entails a linear vision of the economy, since it assumes that every single unit of secondary material that is introduced into the market always avoids the production of the primary material. This can be interpreted in some way as the potential or marginal gain that is sought by implementing a recycling system.

However, it is arguably more fitting for a strictly attributional analysis, and from the point of view of a more 'circular' economy (Stahel and Reday-Mulvey 1981; Ayres 1998), to assume that each time a material is reintroduced into the market (i.e. at each cycle), it does not displace the primary production of the virgin material, but the average mix of technologies that provide an average unit of the material itself. According to this latter view, the environmental credits of one unit of recycled material should be calculated as the weighted average of the impacts of producing the primary (i.e. virgin) and secondary (i.e. recycled) materials being used by the market as input materials for the production of new goods. This is methodologically similar to the calculation of the credits associated with energy recovery in attributional LCA, where, if the technology mix that is effectively being displaced is not known, the average mix of technologies (e.g. grid mix) should be employed (Ripa et al. 2014).

Let us illustrate the difference between using the 100% primary vs. the market mix substitution approach by comparing the environmental credits of recycling aluminium and steel in a simplified example, using a single impact metric (Cumulative Energy Demand). The cumulative energy demand of virgin aluminium production is 194 MJ/kg and that of recycled aluminium is 23.8 MJ/kg; for steel those values are respectively 30 MJ/kg and 8.9 MJ/kg (Classen et al. 2009). If we apply a 'marginal' gain approach (primary substitution), the net impact in each case is simply the energy used for recycling the material minus that for primary production (23.8-194 = -170.2 MJ/kg for aluminium and 8.9-30 = -21.1 MJ/kg for steel, where a resulting negative sign indicates an environmental gain); hence, collecting 1 kg of aluminium for recycling is always 8 times more beneficial than collecting 1 kg of steel (170.2/21.1 = 8). In contrast, if we apply the 'attributional' approach based on (variable) market mixes, the relative benefit of collecting 1 kg of aluminium or steel for recycling changes depending on how much of those metals are already being recycled in their respective market mixes (Figure 2.2). For instance, only 50% of the steel and as little as 25% of the aluminium used in the packaging sector in Europe is of primary (virgin) origin (Table 2.1). Using these percentages, Figure 2.2 shows that the net gain of recycling aluminium is only 42 MJ/kg, while that of recycling steel is 12 MJ/kg; the relative benefit ratio in such sector-specific real-life conditions is thus still in favour of aluminium, but only 42/12 = 3.5. It could then be argued that if, hypothetically, the two market mixes became sufficiently different from one another, collecting 1 kg of steel for recycling might become more beneficial than collecting 1 kg of aluminium. Specifically, this would happen if the amount of virgin aluminium in the aluminium mix were to fall below 10% and, at the same time, the amount of virgin steel in the steel mix remained higher than 80% (see Figure 2.2).

From a policy point of view, it can easily be argued that a marginal approach encourages material recycling, which was the original aim of starting up an integrated waste management system, whereas using a market mix approach can lead to the seeming paradox that the more we recycle the less credit we get. However, we argue here that moving from the 'marginal' to the 'market mix / attributional' approach can lead to a better evaluation of what happens in reality due to waste policy implementation, especially if we are at a stage where a more circular economy is in place (almost fully closed recycling loops).

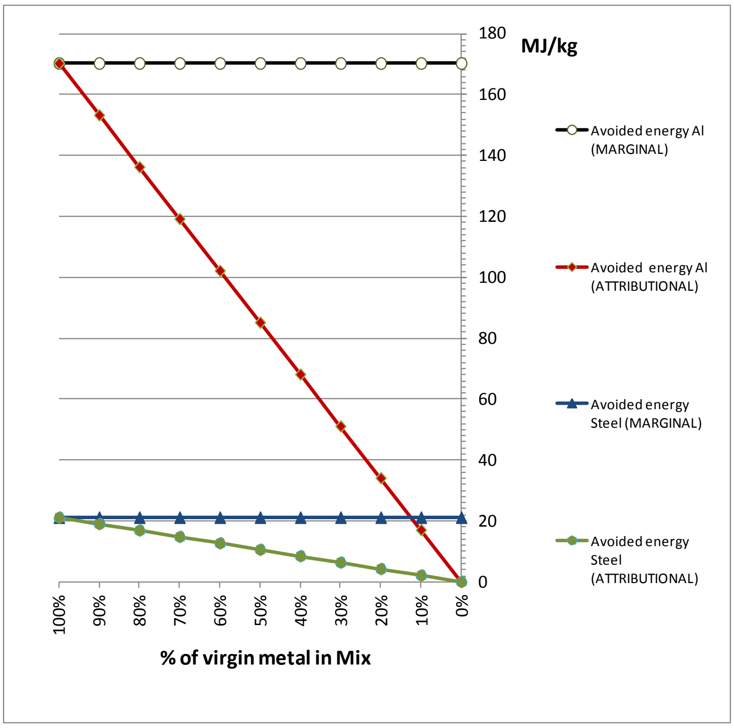

Figure 2.2: Net benefits of recycling aluminium and steel under a marginal approach (1:1 replacement of virgin material) vs. an 'attributional' approach (replacement of virgin and recycled market mix).

2.3. Accounting for quality

In line with the recommendations of the JRC (2010), in order to correct the possible overestimation of the environmental credits associated with material recycling (which often produces lower-quality secondary materials), some authors calculate the amount of primary production displaced by applying correction factors based on technical properties of the secondary material, or on its price (for further details see Rigamonti et al. 2009), leading to the use of substitution ratios < 1. However, using market prices for calculating the substitution ratios is based on the assumption that price elasticity or, in other words, the way a change in price affects the demand, is equal for recycled and virgin materials, which has been demonstrated by some authors (Ekvall 1999; Weidema 2001; Frees 2008) to be wrong. Bearing this in mind, using a physical basis seems to be more appropriate for accounting for the substitution ratios and credits of material recovery.

3. METHODS

The formula proposed here (Eq.1) estimates the credits associated with the recovery of materials by means of using the actual mix of virgin and recycled materials that is used as a source of raw materials in the market (cf. section 2.2). Moreover, it also considers the deterioration of the inherent properties of the materials undergoing the recycling process ('down-cycling'), by introducing a quality factor (cf. section 2.3). This factor is used as 'proxy' to indirectly take into account that, because of its lower technical quality, the recycled material cannot replace an equal quantity of virgin material being part of the mix, but only a smaller quantity thereof (quality factor ≤ 1).

The formula to calculate the environmental credit associated to 1 tonne of recycled material is:

$$\textbf{Environmental credit} = \mathbf{x * REC + (1\text{-}x) * Q * VIR} \qquad \text{[Eq.1]}$$

Where:

- x = proportion of recycled material in the average market mix
- (1-x) = proportion of virgin material in the average market mix
- Q = quality factor of recycled material *vs*. virgin material (Q ≤ 1)
- REC = environmental load of the recycling process (1 tonne of recycled material in output)
- VIR = environmental load of the production process of the virgin material (1 tonne in output)

This same approach is to be applied consistently to all recovered materials, and for all life-cycle impact categories/metrics.

4. PUTTING THE FORMULA INTO PRACTICE

4.1. Representative mixes

The first step to apply the proposed formula is to identify the average mix of virgin and recycled materials that is displaced. If the appropriate mix for a particular application or sector is known, and this is where the recovered material effectively ends up, then such a mix should be used. If not, average market-mix data such as those in Table 2.1 may be used instead. Import and export effects are considered in the model by adopting suitable material consumption (as opposed to production) mixes.

Table 2.1: Average European market mixes for different materials

Material	% virgin	% recycled	Source
Aluminium*	63	37	Calculated from EAA, 2011
Steel	50	50	EUROFER, 2014
Glass	55	45	Roldán & Pino, 2012
Cardboard	16	84	Calculated from CEPI, 2010
Paper	71	29	Calculated from CEPI, 2010
Beverage cardboard	57	43	Calculated from CEPI, 2010
Plastics**	**	**	-

* *For the packaging sector these percentages move to 25% of virgin and 75% of recycled.*
** *Tne percentage of recycled plastic is difficult to quantify.*

4.2. Quality factors

The second step for applying the formula is to determine the quality factors for those materials for which a down-cycling occurs. These should reflect the loss of quality of recycled vs. virgin materials. Obviously, this is not an easy task. In fact, we have identified a lack of studies in which the properties of recycled vs. primary materials are compared, especially in the case of plastics.

These quality factors can be likened to the technical correction factors used by some authors in the ‘marginal’ approach. In the case of paper products, for instance, the European Topic Centre on Waste Materials Flows (2004) suggests using a ratio not higher than 1:1.25 (i.e. Q = 0.8) for paper and cardboard, very close to the 1:1.23 (i.e. Q = 0.81) ratio calculated by

Rigammonti et al. (2009). Instead, other authors such as Gentil et al. (2008) suggest using a ratio of 1:1.11 (i.e. Q = 0.9) for paper and also for plastics. However, we propose that the Q factors should always be strictly calculated on the basis of the actual physical properties of the materials and their contamination levels (to be determined by appropriate laboratory tests).

4.2.1. An example for calculating a quality factor based on mechanical properties of the materials

The process whereby recycled wood fibres behave differently from virgin ones is in itself complex and, contrary to common belief, cannot be reduced to a simple matter of 'fibre shortening'. Other properties related to the quality of the product such as water retention, tensile strength or tear index can also be significant, depending on the final application of the recycled pulp (Wistara & Young 1999). What is undeniable is that recycling paper products always results in down-cycling, and additional cycles (beyond the first one) result in progressively worse properties. Since it is impossible to distinguish between fibres that have undergone one, two or three recycling processes, it is common practice to counteract this loss of quality by adding a certain amount of virgin paper to the recycled products. Villanueva & Wenzel (2007) for instance, quantified this amount as about 20%, which means a Q = 0.8. Based on the study by Wistara and Young (1999), and taking into account the tensile strength indicator, we have arrived at a similar number (Q ≈ 0.83), which implies a loss of quality of about 17% compared to the virgin paper (see Figure 2.3).

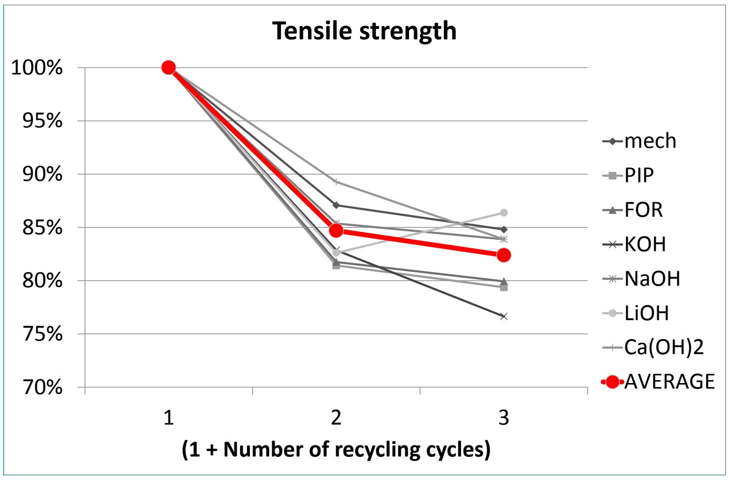

Figure 2.3: Change in Tensile Strength as a proxy of quality factor for paper and cardboard produced from (1) virgin pulp, (2) first-cycle secondary pulp and (3) second-cycle secondary pulp [after Wistara and Young, 1999]. mech = purely mechanical recycling; PIP = recycling with piperidine treatment; FOR = recycling with formide treatment; KOH = recycling with potassium hydroxide treatment; NaOH = recycling with sodium hydroxide treatment; LiOH = recycling with lithium hydroxide treatment; Ca(OH)2 = recycling with calcium hydroxide treatment; AVERAGE = average of all of the above.

4.3. Examples of application

In this section, our proposed formula is applied to a set of materials, which serve as typical examples of different situations that may occur in the market: aluminium, paper and high density polyethylene (HDPE), considering an average market consumption mix substitution.

Figure 2.4 illustrates the varying trends of, respectively:

- a) the impact of the average market mix (red dotted line), calculated as per Eq.2, and
- b) the credit corresponding to one unit of recycled material (green dashed line), calculated according to Eq.1.

Production impact of mix = x * REC + (1-x) * VIR [Eq.2]

The horizontal axis shows the percentage of secondary material present in the market mix, whereas the vertical axis shows the % of Global Warming Potential (GWP), normalized to the GWP of the virgin production (expressed as 100%).

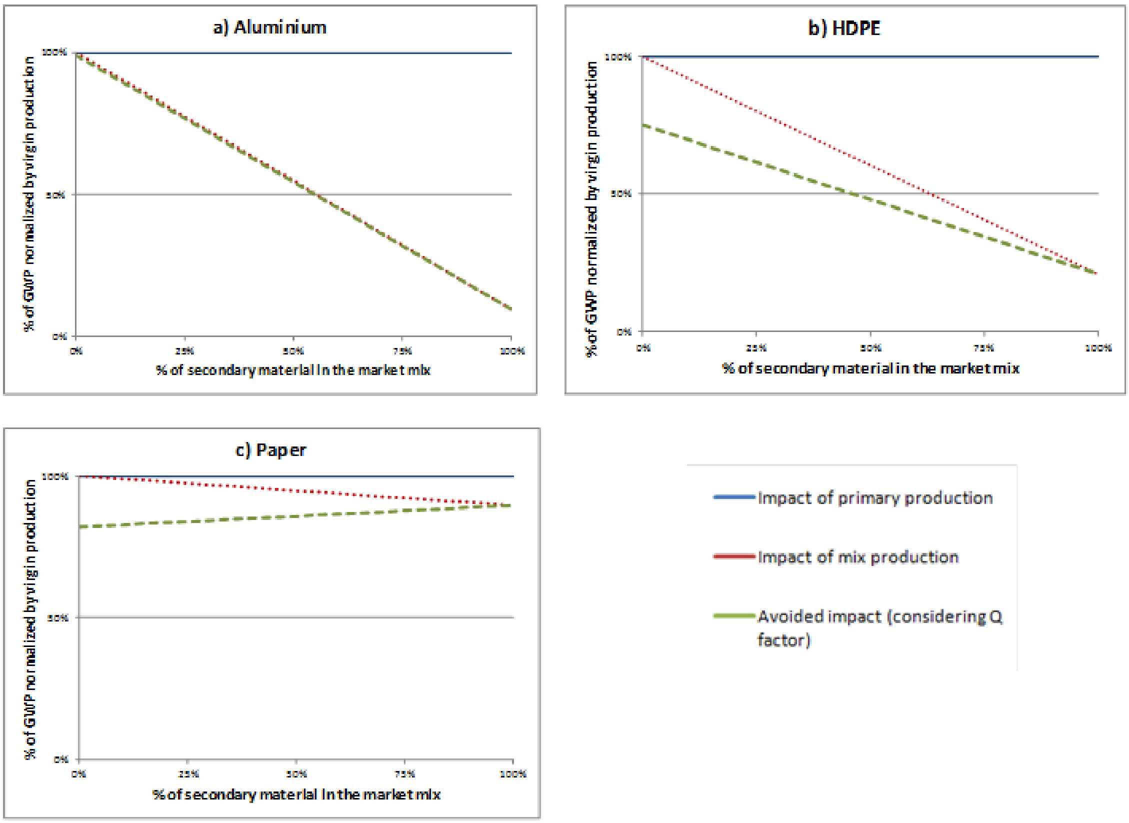

Figure 2.4: Comparison of (1) GWP impact of primary (virgin material) production; (2) GWP impact of the representative market mix of primary (virgin material) and secondary (recycled material) production, calculated according to Eq. 2; and (3) avoided GWP impact relative to the representative mix, calculated according to Eq. 1 (Q factor aluminium=0.99; Q factor HDPE=0.75; Q factor paper=0.83).

Three classes of situations may occur when applying the proposed formula.

- **Situation a)** (illustrated by the case of aluminium), in which the calculation of the credit mainly depends on the market mix. In this case, the impact of virgin production (VIR) is about 10 times higher than that of the recycling process (REC/VIR ≈ 0.1). At the same time, the quality factor is virtually equivalent to 1 (the same also applies to many other metals and glass). Thus, from a pragmatic point of view, in these cases, using the market mix alone is considered a reasonably good proxy, and the credit closely matches that of the simple weighted average of the mix itself.

- **Situation b)** (illustrated by the case of high density polyethylene). In this case, the impact of the recycling process is still lower than that of virgin production, but the difference is not so large (REC/VIR ≈ 0.2). Additionally, the credit is strongly influenced by the application of the quality factor (Q ≈ 0.75, as obtained through laboratory tests, to be published shortly). Thus, the lower the quality of the recovered material, the less credit one has. The result is that the credit line lies lower than that indicating the production impact of one unit of material according to the market mix. This is typically the case for most other plastics too.

- **Situation c)** (illustrated by the case of paper). This situation merits special attention due to the fact that the line indicating the credit ends up having a positive slope, instead of the normal negative one seen in all other cases. This counterintuitive result is due to the fact that the Q factor is actually lower than the ratio of the impact of recycling to that of virgin production (Q ≈ 0.83, and REC/VIR ≈ 0.9). As a consequence, the credit actually increases as the recycling replaces more and more secondary material (since the quality reduction only affects the replacement of the virgin material). This indicates that, because of the inevitable quality loss inherent in the recycling process, recycling waste paper is actually more beneficial (in terms of credits) if the output can be used to contribute to a well-established mix of already mainly secondary paper products (e.g. in the packaging sector) than if it were employed to provide its inevitably low-quality fibre to a production mix still dominated by virgin paper (e.g. in the publishing sector).

4.4. Minimum acceptable quality for selected applications

A further issue that may be analysed by properly taking into account the relative difference in quality between the recycled and virgin forms of a material is the minimum acceptable technical quality of the mix of the two for specific applications (this discussion does not take into account other possibly important but unrelated ‘quality’ considerations, including aesthetics, colour homogeneity, etc., which may lead to a lower amount of recycled material being acceptable in a specific final product). Knowing this minimum acceptable relative quality (i.e. assuming that the quality of the primary material is 1) for a specific application, and expressing the average quality of the mix of virgin + secondary material ($\bar{Q}$) as dependent on the fraction (x) of recycled material in the mix itself:

$$\bar{Q} = Q * x * REC + 1 * (1-x) * VIR \qquad [Eq.3]$$

using a figure similar to Figure 2.4, the cross-over point between the line indicating such average quality of the mix ($\bar{Q}$) and the horizontal line indicating the minimum acceptable quality for the particular application at hand will point to the percentage of secondary material (x) that may be accepted in input (along the horizontal axis). Such estimates can be used for specific analyses where the final application and the minimum acceptable quality of the material mix in input are known. Figure 2.5 shows three simple examples for aluminium, paper and HDPE, assuming for instance minimum acceptable relative qualities Q = 0.9, 0.83 and 0.8, respectively.

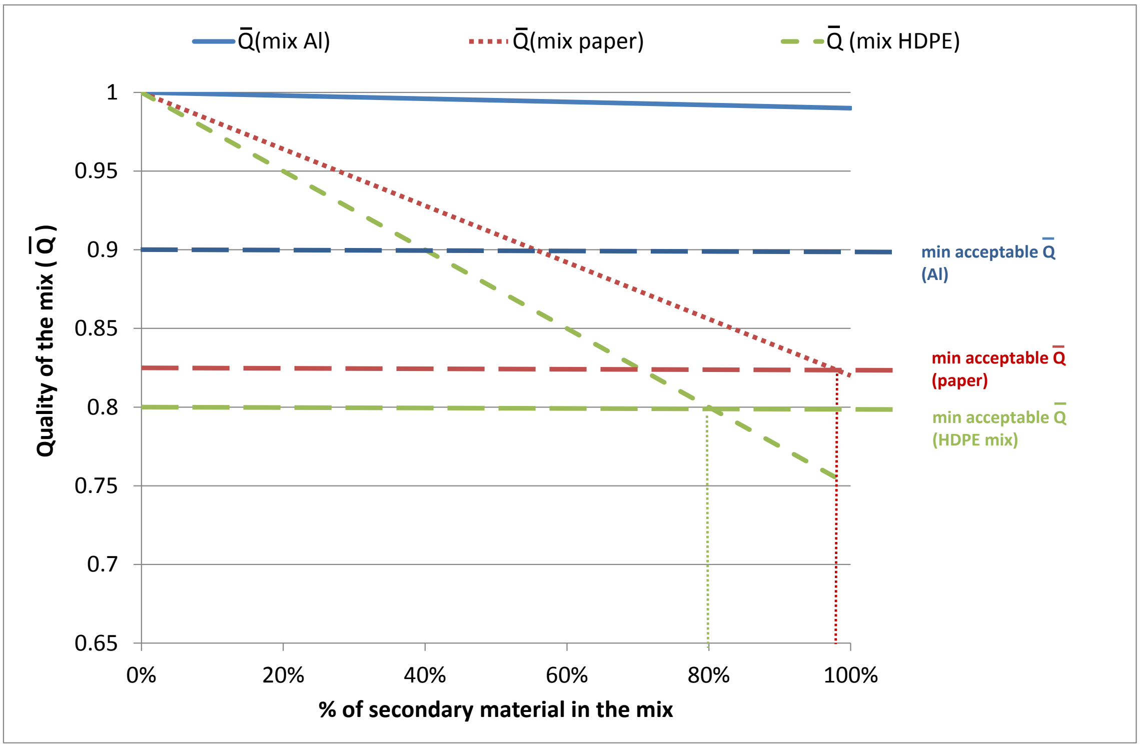

Figure 2.5: **Example of estimation of the maximum acceptable % of secondary material in the mix in order to comply with a pre-set minimum average quality demand. Examples for Aluminium (blue solid line); Paper (red dotted line); and HDPE (green dashed line). The minimum values employed in the figure are only for illustrative purpose and do not correspond to any real case.**

5. DISCUSSION

We stated that there is a common practice of using a substitution factor of 1:1 in LCAs of waste management, considered at the point where the recycled materials are ready to be reintroduced into the market, after having considered all process losses because of impurities in the input waste materials or technology efficiencies. We argue that this practice originates from a time when the market for recycled goods and materials was very limited, and the economy was perceived and described as a linear chain of processes. However, the waste management systems for recovering and recycling goods and the effective reintroduction of secondary materials in the market have improved and become more widespread in many countries, thereby moving towards the goal of a 'circular economy'. As a result, continuing with the use of this simple substitution factor can lead to a misrepresentation of reality, and in particular to an overestimation of the environmental credits associated with recycling practices. Let us illustrate this fact by focussing for instance on the case of platinum. This valuable metal is used by the automotive industry in the production of catalytic converters, and is recovered and reused by the industry in an almost perfectly closed loop. Thus, when analysing a car recycling facility, it no longer makes sense to assume that by recovering platinum we are displacing the extraction and production of virgin platinum every time we recover it - because this is not what is happening in reality. Considering primary production as the displaced impact would thus lead to an inaccurate estimation of the immediate environmental consequences of the recovering facility, when we are under the framework of an attributional analysis.

Applying the formula proposed here to all LCAs of waste management systems to calculate the credits for all recycled materials may lead to the seeming paradox that the more one substitutes, the less credit one gets. This, according to some authors (IFEU & Öko-Institut 2012), may be problematic when comparing LCAs performed in different countries, because in those countries where the percentages of recycled materials in the market mixes are still small, the credit will end up being larger than in those countries where the recycling practices are more established and the amounts of recycled materials in the mixes are already larger. However, in our opinion this should not be considered a 'problem', but instead a necessary consequence of methodological consistency in strictly adhering to the attributional approach in LCA. The credit calculated by using the formula proposed here (Eq. 1) can essentially be interpreted as an indication of the remaining margin for improvement, since it depends on the existing mixes of virgin and recycled materials on the market, and on the potential of the recycled material to actually replace the primary ones on a functional basis.

As briefly mentioned in section 2.2, another reason for adopting this approach is the fact that it is strictly consistent with common practice in attributional LCAs when dealing with electricity production from waste management, where the national grid mix is used to calculate the environmental credits when the real substitution is not known (JRC 2011). Let us imagine a case in which one wishes to compare the gains of recycling to the gains of incineration with energy recovery. Applying a 'marginal' approach to material recycling (1:1 substitution ratio) while adopting the common attributional praxis of assuming grid mix replacement for electricity production, would result in a methodological bias against energy recovery. While favouring material recycling may in fact be a good decision in many cases, especially when the recycling market is still in its infancy, applying the same, strictly attributional, approach to both waste management alternatives is unquestionably more even-handed and allows the analysis of the situation from a more neutral starting point.

6. CONCLUSIONS AND RECOMMENDATIONS

We have introduced here a unified formula for the evaluation of the environmental credits associated with material recovery in waste management, which represents a viable methodological alternative to the common marginal replacement approach (1:1 substitution factor) for many practical case studies. This formula is in line with the fundamental aim of the attributional approach in LCA, and may be applied to all waste materials, thereby ensuring methodological consistency among them. Such formula relies on the knowledge of the application-specific or market-average mix of primary and secondary material currently in use, which is assumed to be displaced by the recycled material. It also requires the evaluation of a quality factor (Q) to account for the reduced relative technical quality of the recycled material (vs. that of the virgin one). While information on the composition of the average market-consumption material mixes for many common materials is easily obtained, there is a dearth of specific studies addressing the quality of secondary vs. primary materials, and more research is called for to arrive at a ready-to-use database of suitable Q factors for many materials and applications.

Finally, the same approach recommended here for waste management systems is, in principle, equally valid for LCAs of product systems. However, while the system boundary for the former is almost invariably the same (namely, a cut-off rule is invoked whereby all input waste materials carry no environmental burdens), many alternatives exist when dealing with product systems. In fact, products may be parts of complex chains or even webs of other upstream and downstream processes and systems, and may already have secondary, as well as primary, material inputs. Utmost care is therefore needed in order to avoid any implicit or even explicit double counting, where the same product system is credited twice for the same amount of recovered material used as raw material and being recycled at the end of the product’s life. A more in-depth discussion of all the possible intricacies arising from the application of system expansion in the LCA of products is however beyond the scope of the present paper, which is confined to LCA of waste management systems.

Acknowledgments

The authors wish to extend their acknowledgment to all people involved in the FENIX Project as well as the European Life + Financing Programme.

References

- Ayres R.U., 1998. Viewpoint: Toward a Zero-Emissions Economy. Environmenatl Science and Technology 32(15), 366-367.
- Bernstad A., la Cour Jansen J., 2011. A life cycle approach to the management of household food waste – A Swedish full-scale case study. Waste Management 31, 1879-1896.
- Bernstad A., la Cour Jansen J., Aspegren H., 2011. Life cycle assessment of a household solid waste source separation programme: a Swedish case study. Waste Management and Research 29(10), 1027-1042.
- Bjarnadóttir H.J., Fridriksoon G.B., Johnsen T., Sletsen H., 2002. Guidelines for the use of LCA in the waste management sector. Nordtest Report. TR 517.
- Björklund A., Dalemo M., Sonesson U., 1999. Evaluating a municipal waste management plan using ORWARE. Journal of Cleaner Production 7, 271-280.
- Bovea M.D., Ibáñez-Forés V., Gallardo A., Colomer-Mendoza F.J., 2010. Environmental assessment of alternative municial solid waste management strategies. A Spanish case study. Waste Management 30, 2383-2395.
- Brandão M., Clift R., Cowie A., Greenhalgh S., 2014. Letter to the editor. “The Use of Life Cycle Assessment in the Support of Robust (Climate) Policy Making: Comment on “Using Attributional Life Cycle Assessment to Estimate Climate-Change Mitigation... ”. Journal of Industrial Ecology 18(3), 461-463.
- CEPI- European Pulp and Paper Industry (2010) Annual Statitics 2010 (http://www.cepi.org). Accessed, July 2012.
- Classen M., Althaus H.J., Blaser S., Tuchschmid M., Jungbluth N., Doka G., Faist Emmenegger M., Scharnhorst W., 2009. Life Cycle Inventories of Metals. Final Report Ecoinvent data v2.1, Nº 10. EMPA Dübendorf, Swiss Centre for Life Cycle Inventories, Dübendorf, CH, Online-Version under: www.ecoinvent.ch
- Dodbiba G., Takahashi K., Sadaki J., Fujita T., 2008. The recycling of plastic wastes from discarded TV sets: comparing energy recovery with mechanical recycling in the context of life cycle assessment. Journal of Cleaner Production 16, 458-470.
- EAA- European Aluminium Association, 2011. Activity Report, 2011. Brussels.
- Ekvall T., 1999. System expansion and allocation in life cycle assessment. Göteborg: Chalmers University of Technology, ARF Report 245.
- Eriksson O., Carlsson Reich M., Frostell B., Björklund A., Assefa G., Sundqvist J.O., Granath J., Baky A., Thyselius L., 2005. Municipal solid waste management from a system perspective. Journal of Cleaner Production 13, 241–252
- Joint Research Centre - Institute for Environment and Sustainability, 2010. International Reference Life Cycle Data System (ILCD) Handbook - General guide for Life Cycle Assessment - Detailed guidance. First edition March 2010. EUR 24708 EN. Luxembourg. Publications Office of the European Union.

- Joint Research Centre, 2011. Supporting environmentally sound decisions for waste management – A technical guide to Life Cycle Thinking (LCT) and Life Cycle Assessment (LCA) for waste experts and LCA practitioners. Luxembourg: Publications Office of the European Union.
- EUROFER- European Steel Association, 2014. http://www.eurofer.org/Sustainable%20Steel/Steel%20Recycling.fhtml. Accessed 10 January 2014.
- European Topic Centre on Waste and Material Flows, 2004. Review of existing LCA studies on the recycling and disposal of paper and cardboard. http://waste.eionet.europa.eu/. Accessed 10 January 2014
- Finnveden G., Johansson J., Lind P., Moberg Å., 2000. Life Cycle Assessment of Energy from Solid Waste. Forskningsgruppen för Miljöstrategiska Studier (FMS), report nr. 137, Stockholm
- Finnveden G. and Ekvall T., 1998. “Life Cycle Assessment as a decision-support tool – The case of recycling versus incineration of paper”. Resources, Conservation and Recycling 24 (3-4), 235-256.
- Finnveden G., Johansson J., Lind P., Moberg A., 2005. Life cycle assessment of energy from solid waste – part 1: general methodology and results. Journal of Cleaner Production 13, 213-229.
- Frees N., 2008. Crediting aluminium recycling in LCA by demand or by disposal. The International Journal of Life Cycle Assessment 13(3), 212-218.
- Gentil E., Potter A., Boldrin A., 2008. Carbon footprinting of export of second-life materials using life-cycle thinking. Proceedings of Waste 2008 Conference, Stratford-upon-Avon (England), 16–17, September 2008.
- Gentil E., Clavreul J., Christensen T.H., 2009. Global warming factor of municipal solid waste management in Europe. Waste Management and Research 27, 850-860.
- Grant T., James K.L., Lundie S., Sonneveld K., 2001. Stage 2 Report for Life Cycle Assessment for Paper and Packaging Waste Management Scenarios in Victoria. Report to EcoRecycle Victoria, Melbourne. Centre for Design at RMIT University, Centre for Packaging Transportation and Storage at Victoria University and the Centre for Water and Waste Technology at University of New South Wales, Melbourne and Sydney.
- IFEU & Öko-Institut e.V., 2012. Minutes of the Workshop “Methods to calculate GHG mitigation potentials in Solid Waste Management” on behalf of a project commissioned by Unweltbundesamt (Federal Environment Agency) Germany. 18 June 2012, Berlin, Germany.
- Laurent A., Clavreul J., Bernstad A., Bakas I., Niero M., Gentil E., Christensen T.H., Hauschild M.Z., 2014. Review of LCA studies of solid waste management systems – Part II: Methodological guidance for a better practice. Waste Management 34, 589-606.
- Merrild H., Damgaard A., Christensen T.H., 2008. Life cycle assessment of waste paper management: The importance of technology data and system boundaries in assessing recycling and incineration. Resources, Conservation and Recycling 52, 1391-1398.

- Michaud J.C., Farrant L., Jan O., Kjaer B., Bakas I., 2010. Waste and Resources Action Program – WRAP (2006): Environmental benefits of recycling- 2010 update.
- Muñoz I., Rieradevall J., Domenech X., Milà L., 2004. LCA application to integrated waste management planning in Gipuzkoa (Spain). The International Journal of Life Cycle Assessment 9, 272-280.
- Perugini F., Mastellone M.L., Arena U., 2005. A life cycle assessment of mechanical and feedstock recycling options for management of plastic packaging wastes. Environmental Progress 24(2), 137-154.
- Prognos A.G-, Ifeu and INFU, 2008. Resource savings and CO2 reduction potential in waste management in Europe and the possible contribution to the CO2 reduction target in 2020. Brussels.
- Rigamonti L., Grosso M., Sunseri M.C., 2009. Influence of assumptions about selection and recycling efficiencies on the LCA of integrated waste management systems.). The International Journal of Life Cycle Assessment 14, 411-419.
- Ripa M., Raugei M., Bala A., Ulgiati S., 2014. Dealing with waste management in Emergy Accounting and LCA: methodological overview and synergies. Presented at: 8th Biennial Emergy Research Conference. January 16-18, 2014, University of Florida, USA.
- Roldán E. and Pino M.R., 2012. FENIX-Giving Packaging a New Life Project. Life Cycle Inventory of pretreatment and glass recycling. Universidad San Jorge, unpublished.
- Shen L., Worrell E., Patel M.K., 2010. Open-loop recycling: a LCA case study of PET bottle-to-fibre recycling. Resources, Conservation and Recycling 55, 34-52.
- Shonfield P., 2008. LCA of Management Options for Mixed Waste Plastics-Final Report. WRAP, 2008. ISBN: 1-84405-397-0.
- Smith A., Brown K., Ogilvie S., Rushton K., Bates J., 2001. Waste Management Options and Climate Change. Final report to the European Commission, DG Environment. http://europa.eu.int
- Stahel W. and Reday-Mulvey G., 1981. Jobs for tomorrow: The Potential for Substituting Manpower for Energy.Vantage Press. New York. Publication in book form of: The potential for substituting manpower for energy (1977), Report to DG V for Social Affairs, Commission of the EC, Brussels.
- United States Environmental Protection Agency, 2006. Solid Waste Management and Greenhouse Gases: A life Cycle Assessment of Emissions and Sinks. 3rd Edicion. September, 2006.
- Villanueva A. and Wenzel H., 2007. Paper waste-Recycling, incineration or landfill? A review of existing life cycle assessments. Waste Management 27 (8), S29-S46.
- Weidema B., Frees N., Nielsen A.M., 1999. Marginal Production Technologies for Life Cycle Inventories. The International Journal of Life Cycle Assessment 4 (1), 48-56.
- Weidema B., 2001. Avoiding co-product allocation in Life-Cycle Assessment. Journal of Industrial Ecology 4 (3), 11-33.

- Wistara N. and Young R.A., 1999. Properties and treatments of pulps from recycled paper. Part I. Physical and chemical properties of pulps. Cellulose 6, 291-324.
- Zamagni A., Guinée J., Heijungs R., Masoni P., Raggi A., 2012. Lights and shadows in consequential LCA. The International Journal of Life Cycle Assessment 17 (7), 904–918.

PAPER II: ASSESSING THE ENVIRONMENTAL PERFORMANCE OF MUNICIPAL SOLID WASTE COLLECTION: A NEW PREDICTIVE LCA MODEL

Submitted to: Waste Management.

Current state: undergoing a second round of review.

Authors: Alba Bala Gala, Marco Raugei, Carlos Afonso Texeira, Alberto Fernández, Francisco Pan-Montojo and Pere Fullana-i-Palmer

ABSTRACT

Most existing life cycle assessment models of waste management have so far underplayed the importance of the waste collection phase, addressing it only in a simplified fashion, either by requesting the total amount of fuel used as a direct user input or by calculating it based on a set of input parameters that the user has to enter, and the use of fixed diesel consumption factors (with units of l/(t.km), l/km or kg/h). However, in situations where a large percentage of the municipalities are very small and scattered on the territory and especially in the case of source-separated waste fractions, a more detailed analysis of the collection phase is required, lest oversimplified and potentially misleading conclusions are drawn. Hence the development of the new collection LCA model being presented here, which relies on a large number of parameters (number and type of waste containers, collection frequency, individual distances for the various legs of transport, etc.) and allows the detailed predictive analysis of alternative collection scenarios before they are implemented. The results of applying such newly-developed model to a number of experimental case studies in Portugal are analysed, discussed and compared to those produced by a selection of pre-existing, more simplified models. The new model is confirmed as being the most accurate and, importantly, as the only one capable of predicting the consequences of a range of possible changes in the collection parameters.

KEYWORDS

Fuel consumption, LCA, model, predictive, waste collection.

1. INTRODUCTION

1.1. Background

LCA was originally developed to analyse the environmental performance of product systems; however, since the end of the 1990s this methodology has also been used for the analysis of waste management systems (Hauschild & Barlaz, 2011). LCA has been used to compare different alternatives for the treatment of a specific waste flow (e.g. Finnveden & Ekvall, 1998, 1999; Güereca et al., 2006; Merrid et al., 2008; Villanueva & Wenzel, 2007; Wollny et al., 2001) and also to compare more complex systems such as integrated waste management systems, including all waste fractions (i.e Banar et al., 2009; Beigl & Salhofer, 2004; Denison, 1996; Emery et al., 2007; Eriksson, 2003; Iriarte et al., 2009; Koci & Trecakova, 2011; Muñoz et al., 2004; Rodrigo & Castells, 2000; Thorneloe et al., 2007). The benefit of using LCA in this context is that it helps to expand the perspective of the analysis and to obtain a complete view of the entire system, including all processes and associated environmental impacts. This approach can avoid the unintentional shifting of environmental loads between different steps of the waste management system, geographic areas, environmental compartments (air, land and water) or impact categories (e.g. global warming, acidification, etc.)

LCA has gained importance in recent years as a tool to assist decision making for waste management policy and planning in Europe (EU, 2005; Rigamonti et al. 2009). Methodological development of LCA for waste management has gone hand in hand with the development of models and software tools to facilitate its implementation by non-LCA experts. These models have been developed almost independently, and mainly in Europe and North America, from the mid-90s onwards by a wide range of universities, consultancy firms and environmental protection agencies (Table 2.2).

Table 2.2: The most widely known, used and complete waste LCA models

Software	Country	Launch time	Still in active development [e]	Reference
ORWARE [a]	SW	1997	Yes	[Dalermo et al, 1997; Erikson et al., 2002]
EPIC/CSR	CA	1999	No	[Haight, 1999, 2004; EPIC & CSR, 2000]
MSW-DTS	USA	1999	Yes	[Weitz et al., 1999; Thorneloe et al., 2007]
WIZARD	UK, FR, NZ	1999	No	[Ecobilan, 1997]
IWM-2[b]	UK	2001	No	[McDougall et al. 2007]
LCA IWM [c]	EU	2005	No	Den Boer et al., 2005; 2007
WRATE [d]	UK	2007	Yes	[Gentil et al, 2005; Coleman, 2006.]
EASEWASTE	DK	2008-2009	No	[Kirkeby et al, 2006]
EASETECH	DK	2013	Yes	[Clavreul et al., 2014]
SWOLF	USA	2014	Yes	[Levis et al., 2013]

Notes: (a) Initially focused on organic waste but extended afterwards to other waste fractions. (b) IWM first version was launched in 1995 [White et al., 1995]. (c) This software was developed under a research project financed by the European Commission between 2002 and 2005. It also includes a prognostic tool for estimating future generation of waste in European Cities. (d) This software is an evolution of the older WIZARD. (e) Information extracted from Gentil et al, 2010 and updated.

All these software packages have in common the inclusion of specific datasets for a wide range of unit processes (waste collection, sorting, recycling, incineration, landfilling, composting or anaerobic digestion), and the possibility for users to build their own waste management systems by combining these unit processes and specifying waste generation, waste composition and/or recovery rates to arrive at specific results for their system(s) of interest.
An in-depth review of the existing models was carried out by Björklund et al. (2011) and by Gentil et al., (2010). Both studies featured comparisons based on methodological issues, input parameters and modelling assumptions, and concluded that there are substantial differences in the models, often linked to the date of development and the current level of knowledge at that time. Along the same lines, other authors evaluated the same management systems or waste management processes using different models to check if the results were comparable. Examples of such comparative work may be found in Hansen & Christensen (2006 - comparison of organic waste treatment), Winkler and Bilitewski (2007 - comparison of the entire management system of the city of Dresden, Germany), and Rimaityté et al. (2007 - comparison of the outcomes of waste incineration using different models and also vs. experimental data).

However, we found no comparative meta-analyses of the results of applying different LCA models to the waste collection phase. This is probably due to the fact that, while this latter phase of the management system accounts for a major part of the total costs of modern waste management systems (Sonesson, 2000), several LCAs (Ekvall & Finnveden, 2000; Eriksson et al., 2005; EU, 2011; Larsen, 2009) have shown that its overall effect in terms of energy demand and emissions of CO_2, SO_2 and NO_x remains comparatively small, provided that the collection and transport systems are reasonably efficient.

1.2. Aims and methodology

The drive for implementing a source-separated collection system for different waste fractions rests on the assumption that the amount and quality of the waste collected in such way are sufficient to overcompensate for the additional environmental burdens entailed by the more complex collection system (thanks to the environmental credits arising from the recovered materials and/or energy). Or in other words, that the collection and transport systems are reasonably efficient. The national implementation of the new EC waste directive (EU, 2008) has led many countries to introduce a legal obligation of establishing a source-separated collection for plastics, glass, paper and cardboard, and metals (e.g. in Spain by the end of 2015), and also for organic waste (e.g. in Spain by the end of 2020). However, in countries like Spain or Portugal, where a large percentage of the municipalities are very small (under 5,000 inhabitants) and scattered on the territory, whether source-separated collection is really environmentally sound and effective in all cases remains to be carefully assessed.

Some authors have questioned the premise of efficiency of the collection and pointed out that this stage of the waste management system can have a major influence on the overall outcome, depending on how it is implemented. For instance, Klang (2005) states that in rural and sparsely populated areas of Sweden it is "more difficult to transform their waste systems in a more sustainable direction", due to small waste volumes, long collection routes and distant treatment facilities. Along the same lines, Tanskanen & Kaila (2001) pointed out that the increasing complexity of the collection systems in terms of the amount of different waste fractions collected separately, and the associated transport and fuel consumption, may increase the relevance of this stage. Additionally, Salhofer and colleagues (2007) demonstrated how transport distances may affect the environmental benefit of recycling a range of waste flows (refrigerators, waste paper, polyethylene films and expanded polystyrene). All this calls for a better focus on the collection phase to assess whether (or the extent to which) the additional environmental burdens associated to the source-separated collection of municipal solid waste are in fact offset by the attainable higher recovery rates of materials and energy from waste.

The aim of this paper is thus two-fold. Firstly, it provides a careful review of the existing models for the LCA of waste management systems, and looks at whether or not they do a satisfactory job of estimating the fuel consumption and the associated impacts in the waste collection phase. Secondly, it introduces a new, more complete and detailed model to predict the environmental performance of the waste collection phase, in which changes in the operational parameters of the system (such as e.g. the number or volume of waste containers, changes in the distances between containers or between the collection area and unloading site, etc.) have a direct effect on the modelled fuel consumption and emission rates.

The steps followed in order to achieve the aims of this research are summarized in Figure 2.6.

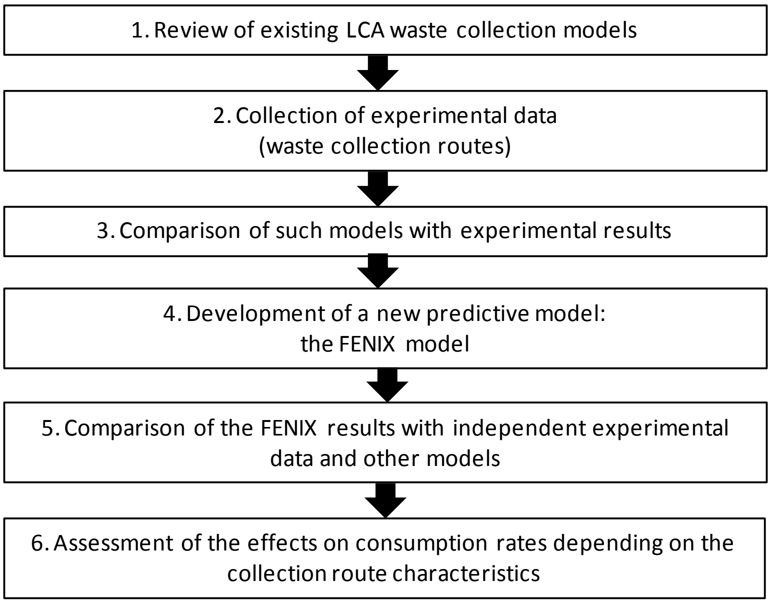

Figure 2.6: Research steps

2. LCA MODELS FOR WASTE COLLECTION

2.1. Review of existing models

When dealing with the environmental performance of the waste collection phase, three different levels of complexity can be found among the LCA software packages listed in Table 2.2. The fist level applies to those models that require that the user inputs the total amount of fuel (e.g. diesel) consumed directly (such is the case of IWM-2), and then applies fixed emission factors associated to the combustion of this fuel. Second level models require the input of fuel consumption (or efficiency) rates (e.g. in terms of km travelled per litre of diesel, litres of diesel consumed per tonne of collected waste or litres of diesel consumed per hour of service) and the input of the amount of waste and/or km travelled and/or the overall collection time in order to calculate the total fuel consumption. Then, corresponding emission factors are also applied. In this second type of models, we can distinguish models in which the user is required to specify their own consumption factors (for instance IWM-Canada and EASEWASTE/EASETECH), and others in which default values are provided, which may be changed by the user either manually or by selecting different truck types from a database (WRATE and LCA-IWM). Third level models are those in which the distances or time spent in the collection stage are not directly entered by the user but instead calculated by means of a set of operational parameters like travel speed, distances between individual collection points along the route, time spent in different operations, etc. Such km or hours spent are then multiplied by fixed consumption factors (l/km, l/h or l/(t.km)), either introduced by the user or provided by default (as in MSW-DST and ORWARE). Like in the previous cases, emission factors are then applied to convert the diesel consumption into airborne emissions in order to evaluate the environmental impact of the collection stage.

For the purpose of this paper, the third level of complexity was selected for the comparison, alongside the built-in model for waste collection trucks included in the Ecoinvent database, since the latter is arguably the most widely-used life cycle inventory database used by LCA practitioners. The selected models are described below:

- The **ORWARE** model is based on the calculation of fuel consumption and emissions for waste collection trucks, considering two different situations: while collecting waste, and while travelling from the collection area to the unloading site. Data on average load, average speed, etc. are used as input parameters. Data on energy consumption [MJ/(t.km)] were obtained from average data provided by the Uppsala Public Service Work in 1994, and emissions from a simulation of an average bus tour in an urban area with many stop-and-go cycles and a rather low average speed. The author explicitly mentions that this model is only valid for simulating the collection of waste in urban areas. Data for energy consumption (converted to the international system of units (SI)) are shown in Table 2.3. A complete description can be found in Sonesson (1996).

- The **MSW-DST** model includes a set of equations to calculate the time required for the individual activities of the collection vehicles in a typical working day (driving to the collection area, driving in stop-and-go cycles, and idling at the stops). These times are then used to calculate the associated fuel consumption, based on corresponding fixed consumption factors (gallons per mile and gallons per hour) - see Table 2.3 for the values converted to SI. A complete description of the model and equations can be found in Curtis & Dumas (2000).

- The **Ecoinvent** dataset for collection trucks ('CH: transport, municipal waste collection, lorry 21t') is based on 5 case studies for German and Swiss municipalities, whence an average consumption rate of 4 l/t was obtained. The (fixed) transportation distance was estimated from the standard transport distance to municipal solid waste incineration plants in Ecoinvent, i.e. 10 km. From these values, an average fuel consumption factor expressed as [l/(t.km)] was derived - also included in Table 2.3. For further details, the reader is referred to Doka (2007).

It is worth noting that, in the end, existing simplified models (both 'levels 2' and 'level 3' types) are still similarly limited in assuming a straightforward direct correlation between the distance travelled, waste collected or time spent collecting (in the case of MSW-DST), and the fuel consumed. These models may be regarded as arguably sufficient only for a quick estimate of the environmental performance of the collection phase within the framework of a broader analysis of a complete waste management system, especially when the average 'collection performance' (in terms of l(fuel)/t(waste), l(fuel)/km or total diesel consumption) is already known. However, they fail to provide a sufficient level of detail if the focus of the analysis is on the optimization of the waste collection system itself. Questions like the following cannot be answered using such simplified models if no real data are available: "what if rear-loading trucks were replaced by side-loading trucks, using fewer larger containers?" or "what if the number of containers were reduced and the collection frequency were doubled?" In order to answer these types of questions, more sophisticated models are needed, which must be able to predict the ensuing changes in the performance of the waste collection system. The original model presented here in section 3 is one such models.

Table 2.3: Default diesel consumption rates used in the analysed models

	Units of energy consumption	Waste truck, in collecting route	Waste truck, from the collection area to the unloading point	Waste truck, idling
ORWARE[(a)]	l/(t.km)	0.24	0.13	
MSW-DST[(a)]	l/km	1.18	0.47	
	l/h			0.26
Ecoinvent	l/(t.km)	0.40		

Notes: (a) original consumption rates were transformed to international system (SI) units.

2.2. Comparison of experimental data to results of existing models

Weekly experimental data from 14 different kerbside collection routes were gathered during all 2012 in Portugal (Lisbon and surrounding areas). These routes refer to the source-separated collection of different waste fractions, namely: mixed municipal solid waste (MSW), light packaging waste (LPW), paper and cardboard (P) and glass (G). For all routes, specific data were gathered, including total amount of waste collected, number of containers, distances between different parts of the collection route, and total fuel (diesel) consumption. Average numbers are presented in Table 2.4. The experimental data on total fuel consumption were then compared with the results obtained by multiplying the default consumption rates assumed by ORWARE, MSW-DST and Ecoinvent, by the experimental data gathered in terms of total amount of waste (t), transport and effective collection distances (km), number of containers (-) and idling containers time (h). Results of this comparison are shown in Figure 2.7. Vertical axes are expressed in a base-2 logarithmic scale.

Table 2.4: Average experimental data of different kerbside collection routes in Portugal

Waste fraction collected	Route Code	Amount collected (t/year)	Annual Distance (km/year)	Annual Diesel consumption (l/year)	Performance Indicators	
					l/100 km	l/t
Glass	G1	189	5,996	1,716	28.6	9.1
	G2	124	2,637	755	28.6	6.1
MSW	MSW1	2039	22,919	24,935	108.8	12.2
	MSW2	3015	47,265	30,216	63.9	10.0
	MSW3	1990	35,078	23,475	66.9	11.8
Light Packaging Waste	LP1	76	3,877	1,956	50.4	25.6
	LP2	139	2,990	1,765	59.1	12.7
	LP3	165	5,347	4,390	82.1	26.6
	LP4	92	2,147	2,335	108.8	25.3
Paper & Cardboard	P1	144	2,773	1,606	57.9	11.2
	P2	248	2,584	1,525	59.1	6.2
	P3	293	6,747	5,540	82.1	18.9
	P4	201	2,651	2,884	108.8	14.4
	P5	332	5,788	3,873	66.9	11.7

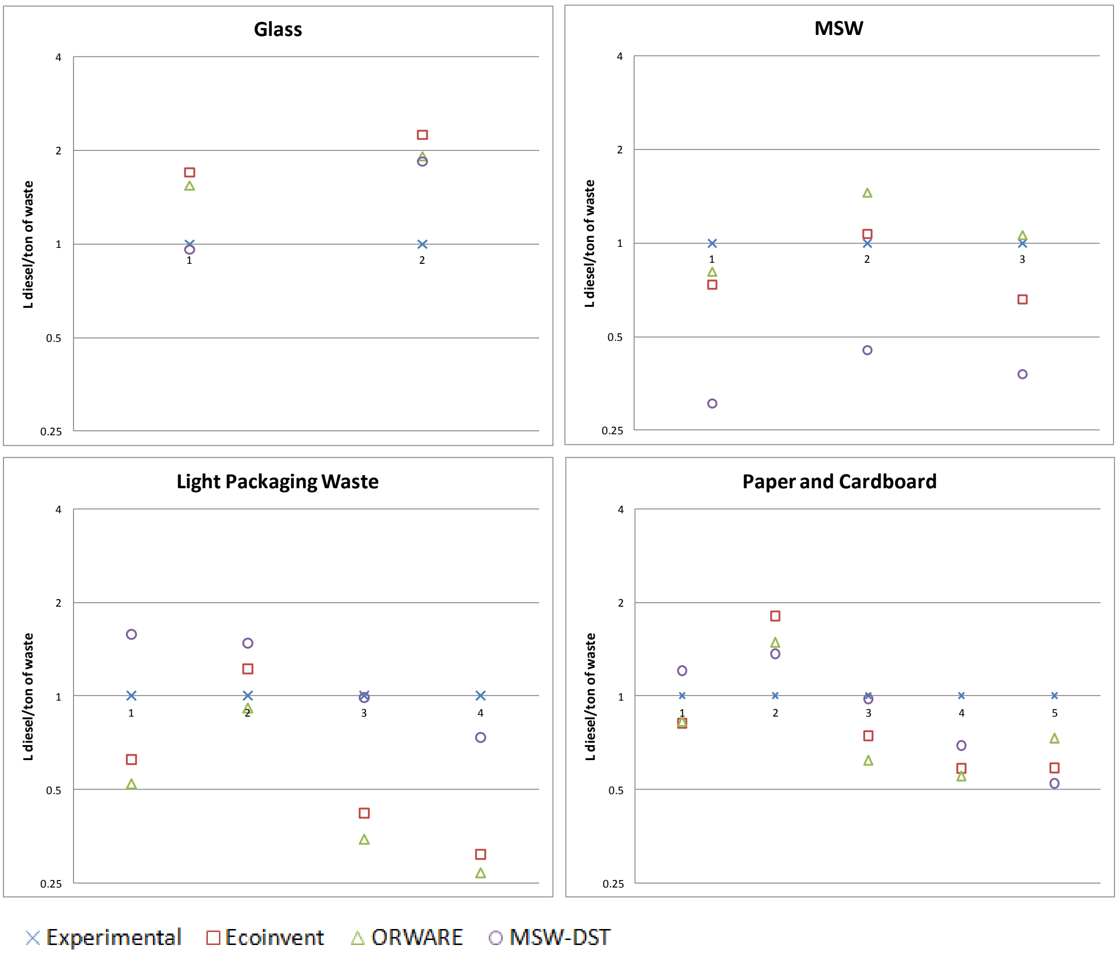

Figure 2.7: Comparison of the results of experimental data with the results using models Ecoinvent, ORWARE and MSW-DST.

An analysis of the results reveals that, overall, none of the total fuel consumption figures produced by these models match the experimental data. Only in those cases in which the characteristics of the experimental routes happened to coincide somehow with those of the calibration routes on which the average fuel consumption rates used in the models are based, were the results any better. In other cases, for instance for light packaging waste, the model estimates were found to be low by a factor of 4, e.g by using Ecoinvent and ORWARE. In general, it was confirmed that if the objective of an LCA is to predict the potential impacts of a specific waste collection option, or to compare and choose among different alternatives, the use of models based on fixed 'average' fuel consumption factors may lead to rather inaccurate results.

3. DEVELOPMENT OF A NEW MODEL: THE FENIX MODEL

Within the EU Life+ project 'FENIX', we developed a new predictive LCA model for the assessment of the environmental performance of the waste collection stage (hereinafter, the FENIX model). In this model, the collection stage includes both the effective collection leg of the route (within urban areas) as well as other distances travelled by the collection trucks, from the moment they leave the parking up to the moment in which they return to it (the latter distances collectively referred to as 'transportation' in the model) (see Figure 2.8).

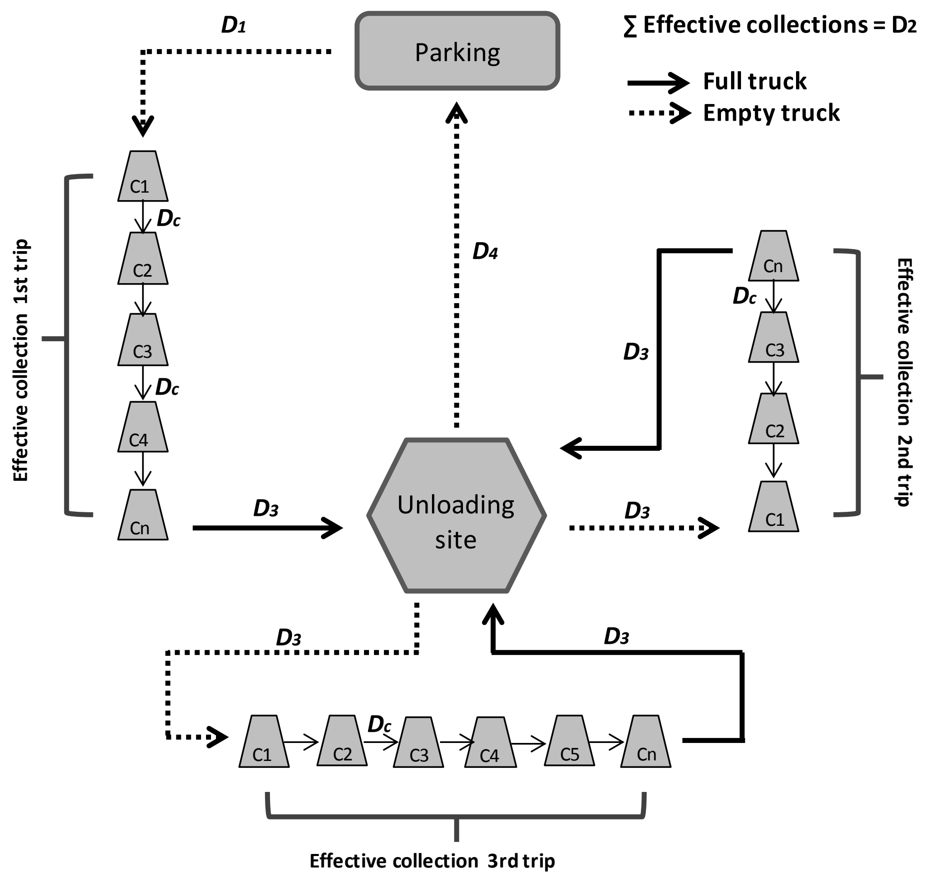

Figure 2.8: Simplified diagram of the collection model to calculate distances, share of km in each type of service and utilization ratio

The development started by modifying a pre-existing model for a conventional transport truck, to factor in the additional fuel consumption due to the specific stop-and-go driving cycles and other intrinsic characteristics of the waste collection truck, as well as to lifting the waste containers and compacting the waste. Finally, a detailed model for the calculation of the needed input parameters to run the modified collection truck model was developed, based on a set of operational parameters and limiting factors.

3.1. Starting point: a conventional commercial truck

The FENIX model is based on the parameterized truck models developed by PE International and available in the built-in database of the GaBi LCA software package, as well as in the European Reference Life Cycle Database (ELCD). Essentially, those models calculate variable fuel (diesel) consumption and emission factors (CO_2, CO, N_2O, NH_3, NMVOC, CH_4, NOx, SO_2, Toluene, Xylene, and PM) in terms of mass units per (kg.km) of transport. The fuel consumption factors are computed according to Eq.4:

$$\boldsymbol{Diesel_{cf}} = \sum_{j=1}^{3}\left[\boldsymbol{\alpha_j} * \left(\boldsymbol{A_j} + \left(\boldsymbol{B_j} - \boldsymbol{A_j}\right) * \boldsymbol{U_r}\right)/\left(\boldsymbol{P_l} * \boldsymbol{U_r}\right)\right] \qquad \text{[Eq. 4]}$$

Where:

- Dieselcf is the diesel consumption factor [kgdiesel/(kgload.km)];
- j is the type of road (1=urban; 2=extra-urban; 3=motorway;);
- αj is the share of km travelled in each type of road [-];
- Aj is the diesel consumption of the empty truck, depending on the type of road (speed and driving conditions) [kgdiesel/km];
- Bj is the diesel consumption of the full truck, depending on the type of road (speed and driving conditions) [kgdiesel/km];
- Pl is the maximum payload capacity of the truck [kgload]; and
- Ur is the utilization (fill) ratio of the truck by mass [-].

Then, the total amount of diesel consumed to transport goods is calculated as detailed in Eq. 5:

$$\boldsymbol{Diesel_T = Diesel_{cf} * Load * D_T} \qquad \text{[Eq. 5]}$$

Where:

- DieselT is the total diesel consumption [kgdiesel];
- Load refers to the total transported mass [kg]; and
- DT refers to the total distance travelled to transport the load [km].

In such base models, Eq. 1 and 2 are formulated in the same way for calculating the emissions of CO_2, CO, NMVOC, CH_4, NOx, Toluene, Xylene, and PM. The only difference is that factors Aj and Bj are referred to the correspondent mass of substance emitted per km. The remaining emission factors, namely those for N_2O, NH_3 and SO_2 are calculated according to Eq. 6, 7 and 8 respectively.

$$N_2O_{ef} = \sum_{j=1}^{3} [\alpha_j * E_j/(P_l * U_r)]$$ [Eq. 6]

$$NH_{3ef} = E/(P_l * U_r)$$ [Eq. 7]

$$SO_{2ef} = PPM * 2 * Diesel_{cf}$$ [Eq. 8]

Where:

- Ej is the emission factor, depending on the type of road (speed and driving conditions) [mgsubstance/km];
- E is the average emission factor [mgsubstance/km];
- PPM is the proportion of sulphur in diesel [ppm] and
- 2 is the ratio of the molecular mass of SO_2 to that of S [kg (SO2)/kg(S)].

3.2. Adaptation of the conventional truck to waste collection vehicles

Waste collection vehicles differ from conventional trucks in their performance because they have different intrinsic characteristics. First of all, waste collection vehicles continuously carry the additional load of the heavy box and equipment to collect and compact the waste. Moreover, their operation entails more stop-and-go cycles in comparison to conventional trucks, since they have to stop and start again every time they collect the waste at each collection point. Additionally, while stopped, the engines of the waste collection trucks still operate at high revolutions per minute (RPMs) in order to lift the waste containers from the kerb and compress their content. Another important difference is related to the utilization ratio (Ur) of the truck. Whereas the Ur of a conventional truck would remain constant, from the loading site all the way to the unloading site, the same parameter for a waste collection truck varies along the collection route (increasing as more and more waste is collected). For all these reasons, it was necessary to modify the basic truck model described above in order to incorporate the additional diesel consumption and related emissions.

The additional consumption due to the heavy box and equipment, the additional stop-and-go cycles and the variable utilization ratio was considered by including a correction parameter (βj), which is the ratio of consumption/emission factor of a waste collection truck to that of a conventional one (Eq. 9).

$$\beta_j = Diesel_{cf}'/Diesel_{cf}$$ [Eq. 9]

Where

- Dieselcf ' is the diesel consumption factor [kgdiesel/(kgwaste.km)] of the collection truck

$$Diesel_{cf}' = \sum_{j=1}^{2} [\alpha_j * (A_j' + (B_j' - A_j') * U_r')/(P_l' * U_r')] \qquad \text{[Eq. 10]}$$

Where:

- α_j is the share of km travelled in each leg of the collection route (respectively, j = 1 for transport and j = 2 for effective collection),
- A_j' is the diesel consumption of the empty collection truck (including the box) in each leg [kgdiesel/km];
- B_j' is the diesel consumption of the full collection truck (including the box) in each leg, also depending on the number of additional stops per trip [kgdiesel/km];
- U_r' is the utilization (fill) ratio of the collection truck by mass [-]; and
- $P_l'=(P_l - W_{box})$ is the maximum effective payload capacity of the truck (discounting the weight of the box) [kgwaste]; and
- W_{box} is the weight of the box used to store and compact the waste [kg].

The share of km travelled during the effective collection and the transportation legs of the route (αj) and the total distance per collection trip (DT) are calculated as detailed in Eq. 11 to 14.

$$\alpha_2 = 1 - \alpha_1 \qquad \text{[Eq. 11]}$$

$$\alpha_1 = (D_T - D_2)/ D_T \qquad \text{[Eq. 12]}$$

$$D_T = [D_1 + D_2 + (N-1) * 2D_3 + D_3 + D_4]/ N \qquad \text{[Eq. 13]}$$

$$D_2 = (C - 1) * D_c \qquad \text{[Eq. 14]}$$

Where, again:

- α_2 is the share of km travelled during the effective collection [-];
- α_1 is the share of km travelled the rest of the collection route [-];
- C is the number of containers (or collection stops) [-];
- D_1 is the distance between the parking lot and the collection route [km];
- D_2 is the total distance while collecting waste (effective collection) [km];
- D_3 is the distance from the collection area to the unloading site [km];

- D_4 is the distance between the unloading site and the parking lot [km]; and
- D_c is the average distance in between individual containers or collection stops [km].

All calculations are included in an Excel worksheet. The number of trips per truck (N) is calculated taking into account the following limiting factors in an iterative mode:

(1) the maximum number of containers in the collection route,
(2) the maximum volume or weight capacity of the truck, and
(3) the maximum duration of the working day.

These and other default parameters and intermediate calculations needed to obtain the output data are described in Table 2.5. It is first assumed that, after collecting all the waste from the kerbside waste containers, if neither the volume, mass or time limits have been reached, then the truck will travel to the unloading site, unload its content there and then go back to the parking lot. If, however, during the collection route the truck reaches its maximum capacity either by weight or volume, then it will also go to unload its content, and then the algorithm in the model evaluates whether there is still enough time (the third limiting factor) to go back and continue the collection. If yes, the same truck is then assumed to return to the collection area and continue collecting - this iterative process is allowed to occur up to 3 times.

When adopting the same basic model as for a conventional truck, the utilization ratio may be calculated as:

$$\boldsymbol{U_r = W_T/(N * P_l)} \qquad \text{[Eq. 15]}$$

Where:

- W_T is the total waste mass collected per year and N is the number of trips per year.

A variable utilization ratio of the collection truck could instead be calculated as described in Eq. 16.

$$\boldsymbol{U_r' = \left(\frac{WT}{N(P_l - W_{box})} * \frac{(D_1 + D_4 + (N-1)D_3)}{D_T} * r_1\right) + \left(\frac{WT}{N(P_l - W_{box})} * \frac{ND_3}{D_T} * r_2\right) + \left(\frac{WT}{N(P_l - W_{box})} * \frac{D_2}{D_T} * r_3\right)} \qquad \text{[Eq. 16]}$$

Where:

- D_T is the distance of one complete trip [km];
- N is the number of trips [-];

- r_1 is the effective load while the truck travels empty [-];
- r_2 is the effective load while the truck travels full[-]; and
- r_3 is the effective load while the truck is collecting waste [-].

Taking into account the values for the effective load of the truck in each individual leg of the route (r_1=0, r_2=1 and r_3=0.5), Eq. 16 may be simplified as described in Eq. 17.

$$U_r' = \left(\frac{WT}{N(P_l - W_{box})}\right) * \left(\frac{ND_3}{D_T} + \frac{D_2}{2D_T}\right) \quad \text{[Eq. 17]}$$

However, based on the experimental fuel consumption values obtained for a number of collection routes in the north of Spain (Galicia), we found that in virtually all cases the resulting β_j factor for a truck of 20-26t of maximum authorized weight and 17.3 of payload capacity (in relation to the corresponding collection truck) was invariably around 2. We therefore decided to refrain from implementing this additional level of complexity in the model, and instead settled for using the simpler Eq. 15 for the calculation of Ur, and then applying a fixed parameter β_j = 2 throughout.

The additional fuel consumption to lift the containers and compress the waste (Add_diesel) was modelled assuming that the truck uses the same amount of fuel per hour as when travelling on urban roads, since it was impossible to obtain the additional fuel consumption due to these operations from experimental sources:

$$Add\_diesel = C * T_{comp} * F_{comp} \quad \text{[Eq. 18]}$$

Where:

- C is the number of containers (or collection stops) [-];
- T_{comp} is the average time spent in empting one container and compacting the waste contained therein (or the correspondent amount spent at a collection stop) [h], which depends on the type of container used; and
- F_{comp} is the diesel consumption factor while the truck is lifting containers and compacting waste [kgdiesel/h], which was set by adopting the average fuel consumption of conventional trucks when travelling on urban roads at a speed used of 27 km/h.

Finally, the total fuel consumption of a waste collection vehicle ($Diesel_{TCT}$) [kgdiesel] results from Eq. 19:

$$Diesel_{TCT} = \left(\sum_{j=1}^{2} \beta_j * Diesel_{cf}\right) * W_T * D_T + Add\_diesel \qquad \text{[Eq. 19]}$$

Where:

- W_T is the total amount of waste collected [kg] and
- D_T refers to one complete trip of the waste collection truck [km].

In the same way, the formulas to calculate the emissions of substances associated to diesel consumption were corrected using adapted β_j factors and additional emission amounts.

Table 2.5: Main operational data included in the model to calculate model key parameters

Input Data	**Default parameters and intermediate calculations**
W_T: total amount of waste collected [kg] *dens*: waste density [kg/m^3] *Freq*: collection frequency [year-1] *C:* number of containers [-] V_c: average volume per container [m^3/C] V_t: volume capacity of the truck [m^3/truck] D_1: Distance between the parking lot and the collection route [(a)] [km] D_3: Distance from the collection area [(a)] to the unloading site [km] D_c: Distance in between individual containers [km/C] D_4: Distance between the unloading site and the parking lot [km] T_T: Duration of the working day [h] **Output Data** α_j : share of km travelled in each type of road [-] U_r : utilization (fill) ratio of the truck by mass [-] D_T :distance of one full trip of the waste collection truck [km]	βj : consumption truck correction factor [-] *Pl:* maximum payload capacity of the truck [kg_{load}] A_j: diesel consumption of the empty truck, depending on the type of road [kg_{diesel}/km] B_j: diesel consumption of the full truck, depending on the type of road [kg_{diesel}/km] F_{comp}:diesel consumption factor while the truck is lifting containers and compacting waste [kg_{diesel}/h] W_{box}: weight of the box truck [kg] $Fill_c$: average container fill ratio [%] cr_t: compaction ratio of the truck [-] ef: collection efficiency in number of containers collected per hour [C/ h] S_{col}: average speed while collecting [km/h] S_{transp}: average speed while transporting [km/h] T_{comp}: time spent loading and compacting waste [h] T_{unload}: time spent unloading waste [h] T_{luch}: time for lunch brake [h] T_{transp}: total time spent while transporting [h] T_{col}: time spent collecting [h] D_2: Total distance spent while collecting waste (effective collection distance) [km] *N:* number of trips per truck [-]

[(a)] This distance is modelled as the weighted mean distance to the different collection points within the collection area.

4. RESULTS

The behaviour of the FENIX model in actually predicting the performance of the collection trucks in terms of fuel (diesel) consumption is discussed in this section. The evaluation was conducted on two levels. Firstly, the model was run and its results were compared to the experimental data from the known collection routes in Portugal (see Section 2.2), and also to the results obtained using the fixed consumption factors of the previously selected models. Secondly, the model's ability to predict the environmental performance of the collection phase when selected changes are made to the operational parameters was analyzed as well.

4.1. Comparison of FENIX model results to those produced by previous models and to experimental data

The results produced by the FENIX model were checked against the experimental data from the 14 kerbside collection routes in Portugal, for which all required input data included in Table 2.5 had already been gathered. The results of this comparison are shown in Table 2.6. What this reveals is that, on average, the FENIX model produced much more balanced and accurate results than all other models, and for all waste routes. This is quantitatively proven by the sum of the squared deviations of the results of each individual model vs. the experimental data.

Table 2.6: Comparison of FENIX results with experimental and existing models

Waste fraction collected	Route Code	Experimental (l/t)	Relative deviation from experimental data			
			FENIX	ORWARE	MSW-DST	Ecoinvent
Glass	G1	9.1	**-0.34**	0.54	-0.04	0.70
	G2	6.1	**0.20**	0.92	0.85	1.25
MSW	MSW1	12.2	**0.44**	-0.19	-0.69	-0.26
	MSW2	10.0	**-0.11**	0.46	-0.55	0.07
	MSW3	11.8	**0.63**	0.06	-0.62	-0.34
Light Packaging Waste	LP1	25.6	**-0.11**	-0.48	0.57	-0.38
	LP2	12.7	**-0.13**	-0.09	0.48	0.22
	LP3	26.6	**-0.25**	-0.66	-0.01	-0.58
	LP4	25.3	**0.26**	-0.73	-0.27	-0.69
Paper & Cardboard	P1	11.2	**-0.01**	-0.17	0.20	-0.19
	P2	6.2	**0.12**	0.48	0.36	0.80
	P3	18.9	**-0.11**	-0.38	-0.02	-0.26
	P4	14.4	**0.19**	-0.45	-0.31	-0.42
	P5	11.7	**0.65**	-0.27	-0.48	-0.41
Sum of the squared deviations			**1.41**	3.27	3.01	4.35

4.2. Sensitivity to model parameters

The main reason for developing the FENIX model was the identified need for a model which would be able to predict changes in the environmental performance of the waste collecting phase before changes in operational parameters are applied in real life, in order to help decision makers to make environmentally and economically sound choices.

Table 2.7 presents some illustrative examples in which the potential effects of the variation of some of these operational parameters are assessed. Specifically, we analyzed the influence of the number of containers (scenarios 1-2), the collection frequency (3-4), the length of the working day (5-6) and, finally, the collection in more rural areas (7-8), which is parameterized by increasing the distances between the parking lot and the collection area, between the collection area and the unloading point, and from the unloading point back to the parking lot, as well as increasing the distance between the individual collection sites within the collection area (a distance along which no effective waste collection is carried out).

Table 2.7: Effects of selected operational parameters of the route on the waste collection performance

INPUT DATA											CALCULATED DATA					
	Waste [t]	**Working day[h]**	**Yearly Freq.**	**N. cont**	**container volume [m^3]**	**truck volume [m^3]**	**Dist parking-fist cont**	**Dist end coll-unloading**	**Dist unloading point -parking**	**Dist between collection areas**	**Performance [n.cont/h]**	**[kg diesel/ (t.km)]**	**Annual km travelled**	**km/trip**	**l/year**	**l/t**
Baseline	92.2	8	55	449	0.25	22	1.7	10.2	10.7	0	41.2	0.86	3462	31	2928.48	31.76
1	92.2	8	55	**269**	0.25	22	1.7	10.2	10.7	0	41.2	0.49	2184	40	2171.13	23.55
2	92.2	8	55	**494**	0.25	22	1.7	10.2	10.7	0	41.2	1.00	3462	31	3419.55	37.09
3	92.2	8	**70**	449	0.25	22	1.7	10.2	10.7	0	41.2	1.20	4406	31	4123.59	44.72
4	92.2	8	**110**	449	0.25	22	1.7	10.2	10.7	0	41.2	1.88	6923	31	6435.03	69.79
5	92.2	**6**	55	449	0.25	22	1.7	10.2	10.7	0	41.2	1.20	4739	29	3853.33	41.79
6	92.2	**4**	55	449	0.25	22	1.7	10.2	10.7	0	41.2	1.44	6016	27	4280.64	46.43
7	92.2	8	55	449	0.25	22	**3.4**	**20.4**	**21.4**	**25**	39.3	0.67	7257	66	4915.81	53.32
8	92.2	8	55	449	0.25	22	**3.4**	**20.4**	**21.4**	**50**	37.5	0.64	8602	78	5469.50	59.32

Note: Changes with respect to the baseline scenario are marked in **bold**.

As it can be seen, by reducing the number of containers in scenario 1 from 449 to 269 (the latter being the minimum number of containers calculated by the model as necessary for this particular route), or by increasing the number of containers by 10% (scenario 2), the performance of the collection phase [l/t] is respectively improved by ~25% or worsened by ~17%. Increasing the collection frequency by 25% (scenario 3), or doubling it (scenario 4), produces a worsening in the performance by ~41% and ~120%, respectively. Changes in the duration of the working day (the new limiting factor included in this model) also have a large effect on the results. Reducing the available time by 2 hours (scenario 5) or by one half (scenario 6), leads to a worsening in the performance by ~32% and ~46%, respectively. Finally, collecting the same amount of waste with the same operational parameters but in more rural areas, exemplified by doubling the distances in the collection route and also considering 25 (scenario 7) or 50 (scenario 8) additional km travelled to reach the collection site, produces a worsening of the performance by ~68% and ~87% respectively.

5. DISCUSSION

When performing LCAs of waste management systems, the environmental benefits of recycling must often be balanced against the additional environmental impacts arising from increased transportation (Baumann & Tillman, 2004). As it has been demonstrated by experimental results, using fixed consumption factors may lead to some serious shortcomings in calculating the environmental impact of the collection phase, especially when the characteristics of the real collection routes do not coincide with those that were used to derive the default consumption factors. In particular, the effects of changes in operational parameters of the waste collection route on the diesel consumption rates have been already discussed in the previous section and shown in Table 2.7. However, the effect of time (in terms of the overall duration of the waste collection trip) merits special attention and additional discussion.

In the guidelines for conducing LCA of waste management systems developed in the last decades [i.e. Bjarnadóttir et al, 2002; EU, 2011], the time dimension is not given any consideration, and only the overall volume and weight of the waste are mentioned as limiting factors to be considered for the calculation of the environmental impacts of the collection and transport of waste. In addition, recent studies evaluating the significance of the collection and transport phases in LCAs of integrated waste management systems indicate that ”time is not relevant for assessing the environmental loads of waste collection“ (Larsen et al., 2009; Christensen et al., 2010). However, this assertion appears to be questionable based on the results illustrated in Table 2.7, where changes in the duration of the waste collection journey were found to have an important effect on the results. In fact, once the maximum allotted time for the collection of waste is reached, it no longer matters whether there is any capacity available left in the truck, since the truck will still return to the parking lot and another truck will resume the collection on the following day from where it was interrupted. In so doing, the total cumulative distance travelled to collect the same overall amount of waste will increase, thereby negatively affecting the average collection performance in terms of l(diesel)/t(waste collected). The time factor may not have a significant influence in urban areas, but it does have a larger influence in more rural areas, where long distances are driven waste collection is less efficient to begin with.

This is particularly relevant for countries like Spain and Portugal, where the performance of the waste collection phase is often far from optimal. In these countries a large percentage of municipalities under the obligation of implementing source-separated collection systems have less than 5,000 inhabitants. Moreover, collection frequencies are much higher than in other countries. Whereas in Denmark, for instance, collection for paper and glass wastes are carried out once or two times per month (Larsen et al., 2009), in Spain the collection frequency is typically once per week and in Portugal twice per month. In the case of MSW the situation is even more extreme. In Denmark MSW is collected 2 or around 4 times per month, whereas in Spain and Portugal MSW is typically collected every 1 to 3 days. This situation leads to higher fuel consumption rates, in terms of litres of diesel per tonne of collected waste, than in other countries. The higher intensity in collection frequency may be due to specific climate conditions with result in a more rapid decomposition of organic waste (especially in summer), which causes undesirable odours and inconvenience to citizens, but may also be partly due to perceived citizen demand.

6. CONCLUSIONS

As amply discussed in the previous sections, it is safe to conclude that the fixed consumption rates included in some LCA tools or databases for waste management to calculate the environmental performance of the waste collection phase are to be used warily. If the aim of the study at hand is to evaluate the environmental performance of the currently implemented waste collection system, we would suggest using real fuel consumption data instead, trying to adjust as much as possible the characteristics of the route on which the average parameters are based to the characteristics of the system under analysis. Conversely, for some applications, especially if the focus of the study is the optimization of the collection phase itself, and where predictive results are needed before implementing changes in the system, it is arguably necessary to use a more complex model (such as the FENIX model presented here), duly taking into account many more characteristics of the collection scheme and the associated operational parameters in the model itself, in which consumption and emission factors depend on the characteristics of the route and are not fixed. Specifically, one parameter which has been identified as particularly relevant in the development of the FENIX model, but which had hitherto been neglected in most previously available models, is the duration of the working day. We thus recommend paying special attention to this parameter in all future LCA modelling of waste collection.

Acknowledgements

The authors of this paper want to extend their acknowledgements to the FENIX Project (LIFE08/ENV/E/000135), and especially to the Green Dot Holders of Spain and Portugal, Ecoembalajes España and Sociedade Ponto Verde, for their continuous help throughout the project and the critical review of the model, using their broad and practical experience.

References

- Banar M., Cokaygil Z., Ozkan A., 2009. Life cycle assessment of solid waste management options for Eskisehir, Turkey. Waste Management 29, 54-62.
- Baumann H. and Tillman A.M, 2004. The Hitch Hiker's Guide to LCA. An orientation in life cycle assessment methodology and application. Studenlitteratur AB. Lund, Sweden.
- Beigl P. and Salhofer S. 2004. Comparison of ecological effects and cost communal waste management systems. Resources, Conservation and Recycling 41, 83-102.
- Bjarnadóttir H. J., Fridriksoon G. B., Johnsen, T, Sletsen, H., 2002. Guidelines for the use of LCA in the waste management sector. Nordtest Report. TR 517.
- Björklund A., Finnveden G., Roth L., 2010. Application of LCA in Waste Management, in: Christensen, T.H. (ed), Solid Waste Technology & Management, Volume 1&2, John Wiley & Sons, Ltd, Chichester, UK, pp.137-160. DOI: 10.1002/9780470666883.
- Clavreul J., Baumeister H., Christensen T. H., Damgaard, A., 2014. An environmental assessment system for environmental technologies. Environmental Modelling and Software 60, 18-30.
- Christensen T.H. (Ed), 2010. Solid Waste Technology & Management, Volume 1&2, John Wiley & Sons, Ltd, Chichester, UK. DOI: 10.1002/9780470666883.
- Coleman T., 2006. Life Cycle Assessment for Municipal Waste: Supporting Decisions. Resources Recovery Forum. Annual General Meeting, July 19, London, UK, 2006.
- Curtis E.M. and Dumas R.D., 2000. A spreadsheet process model for analysis of costs and life-cycle inventory parameters associated with collection of municipal solid waste. North Carolina State University for the Department of Civil Engineering. [Available at: https://mswdst.rti.org/resources.htm]
- Dalermo M., Sonesson U., Björklund A., Mingarini K., Frostell B., Jönsson H., Nybrant T., Sundqvist J.O., Thyselius L., 1997. ORWARE – a simulation model for organic waste handling systems, part 1: model description. Resources, Conservation and Recycling 21, 17-37.
- Den Boer E., Den Boer J., Jager J., 2007. LCA-IWM: a decision support tool for sustainability assessment of waste management Systems. Waste Management 27, 1032-1045.
- Denison R., 1996. Environmental Life Cycle comparisons of recycling, landfill and incineration: a review of recent studies. Annual Review of Energy and the Environment 21, 191-237.

- Doka G., 2007. Life Cycle Inventories of Waste Treatment Services. Ecoinvent report nº 13. Swiss Centre for Life Cycle Inventories, Dübendorf.
- Ecobilan, 1997. Life Cycle Research Programme for Waste Management: Inventory Development for Waste Management Operations: Landfill, Final Report. Environment Agency, Bristol, UK.
- Ekvall T. and Finnveden G., 2000. The application of Life Cycle Assessment to Integrated Solid Waste Management: Part 2- Perspective on Energy and Material Recovery from Paper. Process Safety and Environmental Protection 78 (4), 288-294.
- Emery A., Davies A., Griffiths A., Williams K., 2007. Environmental and economic modelling: A case study of municipal solid waste management scenarios in Wales. Resources, Conservation and Recycling 49, 244-263.
- EPIC and CSR, 2000. Integrated solid waste management's tool: Measuring the environmental performance of waste management systems. Environment and Plastics Industry Council & Corporations Supporting Recycling, Mississauda, ON, Canada. http://www.iwm-model.uwaterloo.ca/iswm_booklet.pdf.
- Eriksson O., 2003. Environmental and Economic Assessment of Swedish Municipal Solid Waste Management. PhD Thesis. Industrial Ecology, Royal Institute of Technology, Stockholm, Sweden.
- Eriksson O., Carlsson Reich M., Frostell B., Björklund A., Assefa G., Sundqvist J-O., Granath J., Baky A., Thyselius, L., 2005. Municipal solid waste management from a systems perspective. Journal of Cleaner Production 13, 241–252.
- Eriksson O., Frostell B., Björklund A., Assefa G., Sundqvist J.O.,Granath J., Carlsson M., Baky A., Thyselius L., 2002. ORWARE – a simulation model for waste management. Resources, Conservation and Recycling 36, 287-307.
- EU, 2005. Communication from the Commission to the Council, the European Parliament, the European Economic and Social Committee and The Committee of the Regions - Taking sustainable use of resources forward - A Thematic Strategy on the prevention and recycling of waste. COM (2005) 666 final.
- EU, 2008. Directive 2008/98/EC of the European Parliament and of the Council of 19 November 2008 on waste and repealing certain Directives. (OJ L 312/3, 22.11.2008).
- EU, 2011. Supporting environmentally sound decisions for waste management – A technical guide to Life Cycle Thinking (LCT) and Life Cycle Assessment (LCA) for waste experts and LCA practitioners. Luxembourg: Publications Office of the European Union, 2011.
- Finnveden G. and Ekvall T., 1998. Life Cycle Assessment as a decision-support tool – The case of recycling versus incineration of paper. Resources, Conservation and Recycling 24 (3-4), 235-256.
- Finnveden G. and Ekvall T., 1999. Environmental aspects of energy and material recovery of paper – today and in a more sustainable future. Appendix VII in Ekvall, T.: System Expansion

and Allocation in Life Cycle Assessment, With Implications for Waste Paper Management. Dissertation, Chalmers University of Technology, 1999.

- Gentil E., Hall D., Thomas B., Shiels S., Collins M., 2005. LCA tool in waste management: new features and functionalities, Sardinia. In: Tenth International Waste Management and Landfill Symposium, Sardinia, Italy.
- Gentil E.C., Damgaard, A., Hauschild M., Finnveden G., Eriksoon O., Thorneloe S., Kaplan P.O., Barlaz M., Muller O., Matsui Y., Li R., Christensen T.H., 2010. Models for waste life cycle assessment: Review of technical assumptions. Waste Management 30, 2636-2648.
- Güereca L.P., Gassó S., Baldasano J.M., Jimenez-Guerrero P., 2006. Life Cycle Assessment of two biowaste management systems for Barcelona, Spain. Resources, Conservation and Recycling 49, 32-48.
- Haight M., 1999. EPIC/CSR Integrated Solid Waste Management Model. Final Report. Waterloo, Canada, pp. 23.
- Hansen T.L. and Christensen T.H., 2006. Environmental modelling of use of treated organic waste on agricultural land: a comparison of existing models for life cycle assessment of waste systems. Waste Management and Research 24, 141-152.
- Hauschild M. and Barlaz M.A., 2010. 3.1. LCA in waste management: introduction to principle and method. Christensen, T.H. (Ed). Solid Waste Technology and Management. Blackwell Publishing.
- Iriarte A., Gabarrell X., Rieradevall J., 2009. LCA of selective waste collection systems in dense urban areas. Waste Management 29 (2), 903-914.
- INE, 2012 (Spanish National Institute of Statistics). www.ine.es
- Kirkeby J.T., Birgisdóttir H., Hansen T.L., Christensen T. H., Bhander G.S., Hauschild M.Z., 2006. Environmental assessment of solid waste systems and technologies: EASEWASTE. Waste Management and Research 24, 3-15.
- Klang A. G., 2005. Sustainable Waste Management. Methods and framework for analysis. PhD in Industrial Ecology. Dissertation. Norwegian University of Science and Technology. Department of Hydraulic and Environmental Engineering. NTNU-trykk, Trondheim. ISBN: 82-471-6894-4.
- Koci V. and Trecakova T., 2011. Mixed municipal waste management in the Czech Republic from the point of view of the LCA method. The Internatinal Journal of Life Cycle Assessment 16, 113-124.
- Larsen A.W., 2009. Environmental Assessment of waste collection seen in a system perspective. PhD Thesis. Technical University of Denmark: Kgs. Lyngby, Denmark.
- Larsen A.W., Vrgoc M., Lieberknecht P., Christensen T.H., 2009. Diesel consumption in waste collection and transport and its environmental significance. Waste Management and Research 27, 652-659.

- Levis J.W., Barlaz M.A., DeCarolis J.F., Ranjithan S.R., 2013. A generalized multistage optimization modelling framework for life cycle assessment-based integrated solid waste management. Environmental Modelling and Software 50, 51-65.
- McDougall F., White P., Franke M., Hindle P., 2007. Integrated Solid Waste Management: a Life Cycle Inventory. 2nd Edition. Blakwell Publishing. United Kingdom.
- Merrid H., Damgaard A., Christensen T.H., 2008. Life cycle assessment of waste paper management: the importance of technology data and system boundaries in assessing recycling and incineration. Resources, Conservation and Recycling 52, 1391-1398.
- Muñoz I., Rieradevall J., Domènech X., Milà L., 2004. LCA application to integrated waste management planning in Gipuzkoa (Spain). The International Journal of Life Cycle Assessment 9, 272-280.
- Rigamonti L., Grosso M., Sunseri M.C., 2009. Influence of assumptions about selection and recycling efficiencies on the LCA of integrated waste management systems. The International Journal of Life Cycle Assessment 14, 411-419.
- Rimaityté I., Denafas G., Jager J., 2007. Report: environmental assessment of Darmstadt (Germany) municipal waste incineration plant. Waste Management and Research 25, 177-182.
- Rodrigo J. and Castells F., 2000. Environmental evaluation of different strategies for the management of municipal waste in the region of Catalonia. Project founded by Junta de Residus, the regional Waste Agency of Catalonia).
- Salhofer S., Schneider F., Obersteiner G., 2007. The ecological relevance of transport in waste disposal systems in Western Europe. Waste Management 27, S47-S57.
- Sonesson U., 1996. Modelling of the compost and transport process in the ORWARE simulation model. Rapport 214. Institutionen för lantbruksteknik. Swedish University of Agricultural Sciences. Uppsala.
- Sonesson U., 2000. Modelling of waste collection – a general approach to calculate fuel consumption and time. Waste Management and Research 18, 115-123.
- Tanskanen J. H. and Kaila J., 2001. Comparison of methods used in the collection of source-separated household waste. Waste Management and Research 19, 486-497.
- Thorneloe S.A., Weitz K., Jambeck J., 2007. Application of the US decision support tool for materials and waste management. Waste Management 27, 1006-1020.
- Villanueva A. and Wenzel H., 2007. Paper waste- recycling, incineration or landfilling? A review of existing life cycle assessments. Waste Management 27, S29-S46.
- Weitz K.A., Barlaz M.A., Ranjithan S., Brill E.D., Thorneloe S.A., Ham R., 1999. Life Cycle Management of municipal solid waste. The International Journal of Life Cycle Assessment 4 (4), 195-201.
- White P. R., Franke M. and Hindle P., 1995. Integrated Solid Waste Management: a Lifecycle Inventory. London, UK, Blackie Academic and Professional.
- Winkler J. and Bilitewski B., 2007. Comparative evaluation of life cycle assessment models for solid waste management. Waste Management 27, 1021-1031.

- Wollny V., Dehoust G., Fritsche U.R., Weinem P., 2001.Comparison of Plastic Packaging Waste Management Options: Feedstock Recycling versus Energy Recovery in Germany, Journal of Industrial Ecology 5 (3), 49–63.

PAPER III: DEALING WITH WASTE FLOWS IN LIFE CYCLE ASSESSMENT AND EMERGY ACCOUNTING: METHODOLOGICAL OVERVIEW AND SYNERGIES

Submitted to: Ecological Modeling.

Current state: undergoing a second round of review.

Authors: Alba Bala Gala, Marco Raugei, Maddalena Ripa and Sergio Ulgiati

ABSTRACT

This paper considers the different approaches taken in dealing with waste products and flows in Life Cycle Assessment (LCA) and Emergy Accounting (EMA), from a methodological point of view, and aims to develop more standardized and synergistic procedures. LCA deals with the waste issue from the point of view of the impact of their disposal, as well as the potential benefit ('environmental credit') afforded by the avoided extraction and processing of additional primary resources when waste is recycled or its energy content recovered. The 'environmental burden' associated to the entire production and consumption chain leading to the waste item is generally not included in LCAs of waste management systems, due to the boundary being placed – consistently with the intended goal – around the actual disposal processes (including recycling alternatives and associated environmental credits). Instead, Emergy Accounting, a donor-side approach with its implicit boundary set at the biosphere level, in principle keeps track of the entire supply-chain at all times, considering even waste flows as products (or co-products), and calculating their intensity factors and assessing their role within the ecosystem's web and hierarchy.

However, when the focus is limited to evaluating processes under human control, within the narrower space and time boundary of human-dominated production and consumption processes, waste products can arguably be regarded as something to be recycled or disposed of to minimize the environmental burden. When this is the case, and particularly in comparative analyses, the emergy perspective thus becomes closer to the LCA perspective and interesting methodological synergies may emerge. A clearly defined set of emergy algebra rules for waste products and flows, and specifically for recycling, was found to be still lacking in the available emergy literature. We propose here that a better and more consistent methodological solution may be arrived at by leveraging the work done in LCA.

1. INTRODUCTION

In natural ecosystems, all material flows are circular and the very concept of waste does not apply: 'waste' products and flows from a process always become inputs to other processes. Instead, human-dominated systems are typically incapable of continuously re-using all waste flows, which puts increased pressure on the environment in terms of pollution as well as ever-increasing depletion of natural resources. Waste management strategies are aimed at minimizing such problems, but they entail additional resource use too, and so must be carefully assessed and optimized.

As already advocated and explained elsewhere (Ulgiati et al., 2006; 2011), there is much to be gained from the comparison, parallel application and, where appropriate, integration (Raugei et al., 2006; Ingwersen, 2011; Rugani and Benetto, 2012; Marvuglia et al., 2013; Arbault et al., 2014; Raugei et al., 2014) of life cycle assessment (LCA) and emergy accounting (EMA), when the intended object of analysis is human-dominated systems. Waste management systems are often especially complex, and therefore require extra care when making all the necessary methodological choices and assumptions, in order to ensure both strict internal adherence to the dictates of the underlying theories, and, no less importantly, external consistency and comparability to pre-existing and possible follow-up studies.

While a number of waste management case studies have already been investigated by emergy analysts (Brown and Buranakarn, 2003; Marchettini et al., 2007; Lei et al., 2008; Amponsah et al., 2011; Yuan et al, 2011; Zhang et al., 2011; Mu et al., 2011; Agostinho et al., 2013; Giannetti et al., 2013; Liu et al., 2013; Song et al., 2013), it seems reasonably safe to conclude that coherent and agreed-upon methodological guidelines as to how to approach this particular field of application are still lacking. On the other hand, a large body of scientific and technical literature exists in which LCA has been used as the method of choice when tackling waste management systems from the point of view of their energy and environmental performance from a user-side perspective (e.g. Finnveden & Ekvall, 1998; Eriksson, 2003; Coleman, 2006; Thorneloe et al, 2007; Gentil, 2011; Koci & Trecakova, 2011). Additionally, in recent years a considerable effort has been made to standardize LCA and provide clear methodological guidelines on how it should be implemented for waste management systems (Bjarnadóttir et al., 2002; JRC, 2010; 2011a; 2011b; 2011c), and on the trade-offs that are inherent in the adoption of alternative assumptions in those cases where no single clear-cut distinction can be made between absolutely 'right' or 'wrong' approaches.

We herein provide a brief overview of the main critical points that are specific to waste products and flows (with selected illustrative examples) and of how they have so far been addressed in LCA. We then discuss the extent to which the work done in the LCA community may be leveraged to improve the clarity and consistency of EMA when applied to waste management. At the same time, we also highlight and discuss those instances where underlying perspective of LCA conflicts with that of EMA, thereby rendering some of the assumptions and solutions proposed by the former essentially inapplicable within the framework of the latter.

2. METHODS

2.1. Life Cycle Assessment

Life Cycle Assessment (LCA) is a relatively recent methodology that has rapidly grown to become a standard tool to investigate the environmental performance of a wide range of human-dominated processes (ISO, 2006a,b; JRC, 2010). LCA is based on the basic principle that in order to accurately assess the environmental impact of the analysed system or product, all its life stages must be addressed, also including in the analysis, where appropriate, the end-of-life recovery and/or recycling of the system's components (for subsequent re-use in other product systems).

Methodologically, a LCA is structured in four consecutive phases, namely:

(i) **goal and scope definition** (including a clear definition of the functional unit, system boundaries and associated assumptions);

(ii) **life cycle inventory** (the compilation of all the inputs and outputs respectively from and to nature associated to all processes that form part of the system's life cycle);

(iii) **life cycle impact assessment** (in which the full inventory of inputs and outputs is translated into a number of aggregated metrics of environmental impact); and

(iv) **Interpretation** (in which results are discussed and compared to suitable benchmarks).

As simple as it may sound when taken at face value, most of the key methodological dilemmas in the application of LCA to waste management arise in that first all-important step of a clear and unambiguous definition of the intended goal and scope of the study.
In fact, all that LCA requires is that whatever the stated goal and scope of the analysis is, the analysis be then carried through in strict adherence to those same goal and scope at all times. In other words, it is perfectly permissible to carry out two independent LCAs of the very same system starting with different 'questions' in mind and, consequently, arriving at quite different 'answers' in the end. Indeed, this is the principal reason why not all methodological assumptions and alternatives that have legitimately been adopted in LCA may be equally

applicable to EMA (whether specifically dealing with end-of-life and waste management processes or otherwise).

In all cases, LCA only accounts for matter and energy flows occurring under human control, whereas flows outside of market dynamics (such as environmental services and renewable resources that do not flows through human controlled devices) as well as flows which are not associated to significant matter and energy carriers (such as labour, culture, information) are not generally included. Moreover, the supply-side 'quality' and degree of renewability of resources, in terms of biosphere activity leading to resource generation processes, are not explicitly taken into account in LCA evaluations (Ulgiati et al., 2006). Where renewable flows are included, such as e.g. in the calculation of the CED metric (VDI, 1997), their inclusion only refers to the renewable fraction captured under human control (e.g. the amount of sunlight actually captured by photovoltaic modules).

2.2. Emergy Accounting

Emergy is defined as the available energy (exergy) of one kind (usually solar) previously required, directly and indirectly, to make a service or product (Odum, 1996). The boundary of the analysis is always set at the biosphere level, thereby keeping track of the entire supply chain (from resource generation to processing and disposal), and accounting for the environmental support needed to generate all the storages and flows of (renewable and non-renewable) raw natural resources which flow through the web of natural processes supporting the analysed process either directly or indirectly (e.g. in the form of ecosystem services). The unit of emergy is the solar emergy Joule (seJ), and the emergy to generate one unit of available energy or mass along a particular pathway is named tranformity (units: seJ/J) or, more generally, Unit Emergy Value (UEV, units: seJ/unit). Incidentally, it is worth noting that in a natural ecosystem, which is not only subject to, but the product of natural selection, the transformity also indicates the position of each type of energy flow in the ecosystem‘s energy hierarchy (Brown et al., 2006), while this only applies loosely and at a very coarse level to human-dominated systems, many of which co-exist without having yet been vetted by long-term natural selection. The total emergy driving a system, calculated as the sum of all emergy inflows, is assigned to the product or service delivered (for further details see Odum, 1996; Brown and Ulgiati, 2004, 2010). After all the flows of interest have been quantified, a set of additional indicators: Environmental Loading Ratio (ELR), Emergy Yield Ratio (EYR), etc., can be developed for better understanding of a system‘s dynamics as well as for environmental policy making (sustainable resource use), by assessing the environmental performance of the process itself (Brown and Ulgiati, 2004).

One fundamental difference between LCA and EMA is arguably that in the latter, unlike in the former, the analyst is required to always abide by the same underlying 'donor side perspective' that is at the very core of emergy theory. Also, the concept of waste (something useless and devoid of any ability to drive further transformation processes) has little meaning from an emergy point of view, because every flow or residue from a process inevitably has a 'history' of its own (hence the concept of 'energy memory' introduced by Brown and Herendeen, 1996), becomes an input to and has an impact on some other (human-dominated or natural) process (Genoni et al., 2003).

As a consequence, EMA should always consider all waste flows as products or coproducts, and calculate their intensity factors accordingly (but paying careful attention not to double-count the emergy inputs when dealing with multiple functional units). On the contrary, LCA distinguishes between 'waste flows', 'waste products' and co-products based on market value (Guinée et al., 2004), and applies different allocation rules accordingly. This is better detailed in section 3 below.

Lastly, when the focus is on evaluating processes under human control, within the narrower space and time boundaries of the economy (production and consumption processes), and when the goal is no longer to assign a value to a given material flow based on its biosphere-scale generation pattern (e.g.: production cost, hierarchical position, ability to exert a feedback control, etc.), but to assess the needed or avoided investment of high-value resources for its disposal, reuse, and/or recycling (in comparative analyses only), then the emergy perspective becomes even closer to that of LCA, and additional interesting methodological synergies may emerge.

3. KEY METHODOLOGICAL ASPECTS

3.1. Treatment of elementary flows vs. products and waste products

LCA makes a fundamental distinction between what it calls 'elementary flows', i.e. flows which are directly sourced and/or emitted to the environment as is (including 'waste flows'), and 'products' (including 'waste products'), which on the other hand are the product of, and are output to, a range of human-dominated systems (the latter collectively referred to as the 'technosphere'). While it is the elementary flows which directly contribute to environmental impact (in terms of resource depletion, and of a number of emission-related impact categories such as global warming potential, acidification potential, etc.), a life-cycle impact potential is computed and assigned to products and waste materials, depending on the inputs and outputs of elementary flows that they have been 'responsible for' along their life cycle. The rules for the allocation of such 'responsibility' amongst (co)-products and waste materials in LCA are detailed in the following sub-section.

EMA, on the other hand, by virtue of its intrinsic 'historical' perspective on the exergy cumulatively spent to provide any given flow at any given moment, has no use for such distinction, and treats flows from/to the environment and those from/to the technosphere in the same way, from a methodological point of view.

3.2. Different approaches to multi-functionality

Based on their market value, LCA then also clearly differentiates between: (i) useful (co)-products, which jointly carry the environmental burden of a production system, and (ii) waste products, which (like waste elementary flows) are considered devoid of any useful value, and whose environmental impact is therefore re-distributed amongst the (useful) (co)-products.

The general recommended way to tackle co-products in LCA (both those of the same physico-chemical nature – which are usually named 'splits' in EMA - and those of different physico-chemical nature) is by system expansion (ISO 2006b; JRC, 2010). When adopting the system expansion approach in LCA, the analyst is free to select those output products which are considered to be of primary interest, and the impact associated to the remaining co-products is removed by (i) expanding the analysis to also assess alternative product systems which generate those same (and only those) outputs whose impact needs to be removed, and then (ii) subtracting the impact associated to the latter systems from that of the original system under study (on a per functional unit basis).

If such system expansion is impossible or impractical, then allocation may alternatively be employed (similarly to what is done by default in EMA in the case of product splits – see below); however, in LCA the analyst has a choice to opt for either energy-, mass-, or economic-based allocation. In fact, depending on the specific system under study and on the goal of the analysis, any of these options may be preferable in order to better reflect the user-side perspective (i.e. "to which degree is each co-product responsible for the operation of the entire system?").

Contrary to what happens in LCA, in EMA all system outputs (including waste products) are, at least in principle, always considered to be either co-products or 'splits'. Additionally, according to the basic emergy definition, computation procedures in EMA follow a special 'algebra' that keeps track of all steps from resource generation up to the product at stake, and differentiates between 'co-products' (two or more products or flows characterized by different physico-chemical nature and generated simultaneously: one cannot be generated without also generating the other one) and 'product splits' (two more products or flows sharing the same physico-chemical nature: in principle it is possible to generate only one of them without also generating the others). When only one product is obtained from a process, all source-emergy is assigned to it. Instead, when two split products are generated, the source-emergy is assigned (allocated) to them according to their available energy (or mass). Finally, when two or more coproducts are generated, the total source-emergy is assigned to all of them (no allocation). Consequently, when two co-product pathways re-unite in a downstream process, the emergy carried by those converging flows must not be added together, lest their common original driving source be double-counted. In such cases, the traditional approach has been to only account for the largest flow when computing the total emergy of the final product.

This peculiarity of the 'emergy algebra' represents a potential stumbling block for the seamless integration of EMA into an existing LCA workflow. Marvuglia et al. (2013) proposed an interesting way to address and solve this issue with their SCALE software. However, the fly in the ointment of their solution is that in so doing, they had to resort to treating all those flows which appear to be co-products in an LCA database as if they were actual co-products of the same real process, while in reality, more often than not, the same database process is used as a proxy for independent processes taking place at different locations and at different moments in time, thereby removing the requirement for any special emergy algebra rule in the first place. So, while worthy of praise from a theoretical point of view, the solution proposed in SCALE tends to 'compensate' and hence may lead to more uncertainty and loss of accuracy than if the emergy algebra rule on co-products were simply ignored.

3.3. End-of-life processes, avoided impact and environmental credit

When specifically dealing with those end-of-life processes that result in the production of secondary materials (recycling) or recovered energy (incineration and sometimes landfilling), the recommended way to address them in LCA, in order to maintain the functional unit in comparative scenarios, is again similar to system expansion. The analysis is extended to also include a representative mix of conventional technologies that is assumed to be displaced, and the impact associated to the latter is then subtracted from that of the original system under study. In so doing, one issue that remains open is that of the choice of the most appropriate process (es) to be displaced.

One line of reasoning, which is often referred to as 'marginal replacement', leads to the identification of the production of virgin material(s), and of energy produced by those technologies whose use it is the industry‘s or government‘s intention to curb, as the best candidates for the calculation of such 'avoided impact'. This corresponds to arguing that, after all, it is essentially in order to reduce the demand for primary materials (and in order to replace polluting energy technologies) that, respectively, recycling and energy recovery are implemented.

Alternatively, and arguably more fittingly for a strictly attributional analysis (i.e. one whose goal is not to investigate the potential long-term consequences of large-scale policy choices, but to actually assess the real impact associated to the life cycle of a system as it is now), the analyst may choose to assume the displacement of the average mix of technologies that at the time of the analysis provide an average unit of, respectively, the material and/or energy that is being replaced. Figure 2.9 illustrates this logic in the case of aluminium. From to this latter viewpoint, the 'environmental credit' associated to one unit of recycled material is calculated as the weighted average of the impacts of producing the primary (i.e. virgin) and secondary (i.e. recycled) material used in the market. Likewise, for energy, the appropriate average mix of technologies (e.g. the grid mix) should be employed.

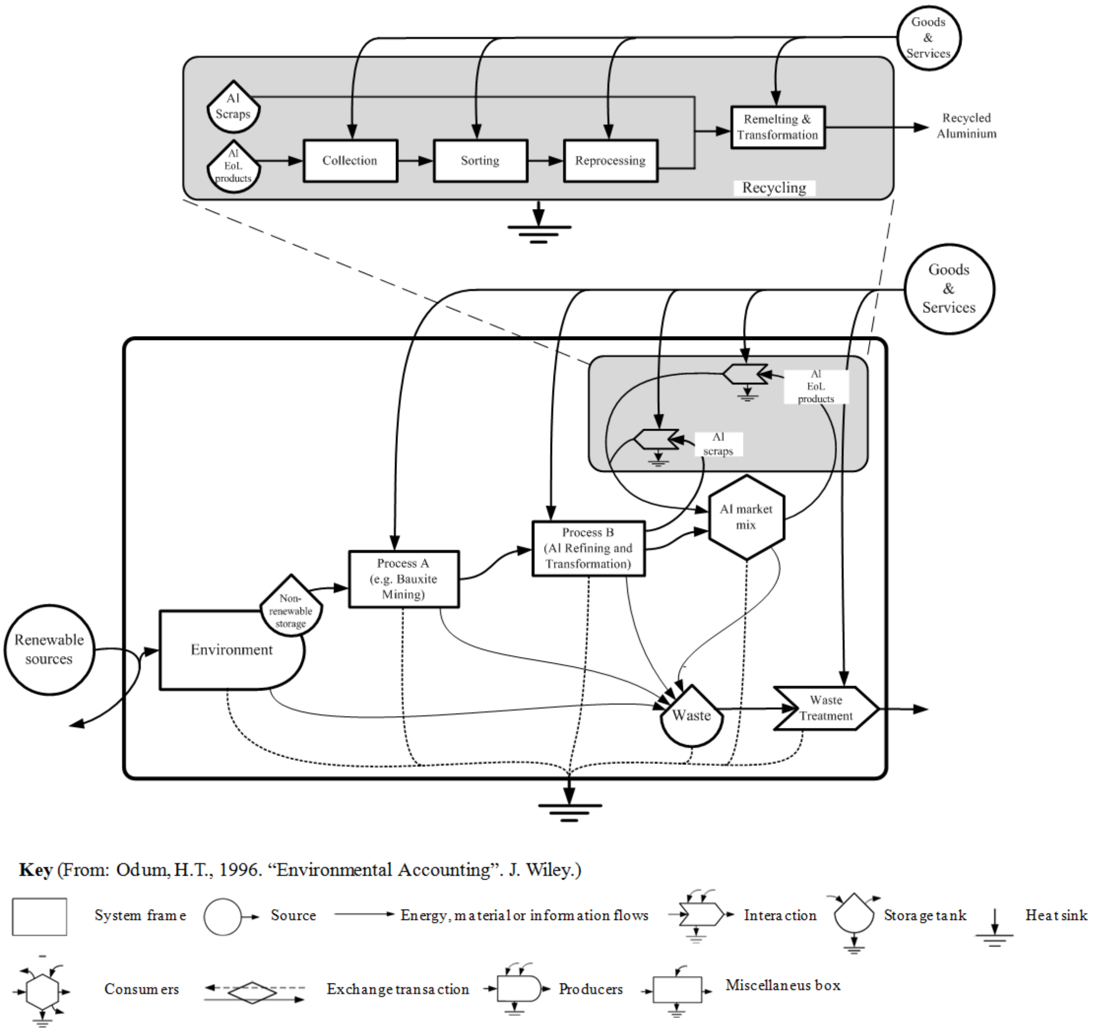

Figure 2.9: Energy system diagram for primary and secondary aluminium production, both contributing to an average mix of Al on the market (hexagon-shaped symbol on the right hand side of the main diagram).

At least in principle, an 'environmental credit' logic similar to that of LCA illustrated above could also be applied in EMA. For instance, when waste materials are produced which could be recycled or put to new use elsewhere (via open-loop recycling), be they categorized as co-products (e.g. corn straw which could be used as soil fertilizers in another system) or split flows (e.g. saw dust of wood processing, which could be used as a source of energy), a virtual decrease of input emergy to the analysed system could be considered. In the two examples above, such 'credited emergy' would be respectively that for the production of chemical fertilizers, and that for the production of conventional thermal energy.

3.4. System boundary and closed-loop vs. open-loop recycling

In LCA, when materials are used in more than one product cycle, it is crucial to always set inter-system boundaries in such a way as to clearly separate the life cycles of the different product systems that make successive use of the same materials (Figure 2.10). A number of options are available as to where to locate such 'cut-off' points (Ekvall and Tillman, 1997).

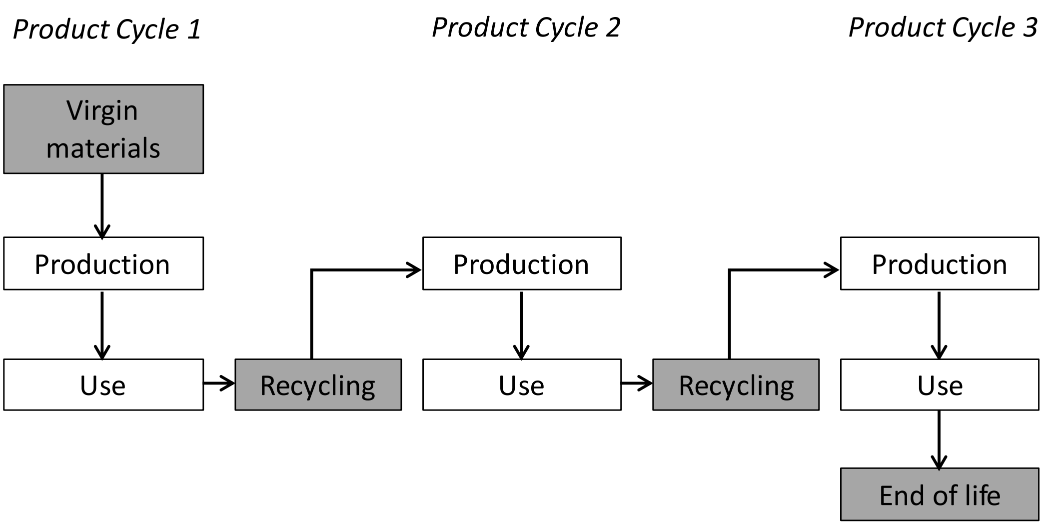

Figure 2.10: Simplified example of successive product cycles.
Processes in grey are those susceptible to be assigned to different product cycles of shared among them.

One approach that is sometimes adopted when analysing one particular product system which happens to be located along any such chains of multiple material uses is to assign the impact associated to the first stages of its waste management (i.e. its collection, disassembly and transport to landfill, incinerator and/or sorting facilities) to the first product system, and then the additional impact due to the pre-treatment and recycling of those materials that are re-used in subsequent product systems to the latter systems. This corresponds to adopting the 'rule' that secondary scrap used as input material carries 'zero embodied impact'. In so doing, though, the analyst foregoes the possibility to claim back any 'environmental credit' for the first product system (cf. previous sub-section) due to the recovery of materials at its end-of-life.

Alternatively, in many cases the system boundaries are often set so as to include all of the waste management in the life cycle of the first product system (including the recycling processes), and then an 'environmental credit' is claimed back for the same product system, based on what the recycled materials are assumed to replace. It is interesting to note, however, that whenever this second approach is adopted, a potential external inconsistency issue arises when results from independent analyses are combined. This, of course, is because the impacts of the recycling process and the associated 'credits' can only be assigned at any given time to either product system 1 or product system 2, along the chain.

In EMA, the following two basic scenarios are distinguished:

a) **Recycling within the same process** (i.e. 'closed-loop recycling'), analysed assuming a steady state. When a recycled flow (waste or co-product) is fed back to a process' earlier step, its emergy should not be double counted and only the additional emergy investment for collection, feedback and pre-treatment should be added. This essentially coincides with the LCA logic.

b) **Waste flows from other processes** (i.e. 'open loop recycling'). The rule to prevent double-counting does not automatically apply to this situation, and at first it might seem that if the recycled/reused material were allowed to carry its entire 'emergy memory', each reuse cycle would increase the emergy of the recycled fraction, in principle increasing its UEV without a limit - and in fact, a similar argument has sometimes been made in the literature (Amponsah et al., 2011). However, more careful scrutiny reveals that such interpretation stems from a fundamental misconception of the fundamentals of emergy theory (Ulgiati et al., 2004). In general terms, the emergy of a 'virgin' resource in input to a production process may be decomposed into: (Ef + Ep), where Ef is the emergy of natural resource 'formation', and Ep is the emergy of the subsequent processes taking place in the technosphere (i.e. extraction, refining/pre-treatment and delivery). It should be noted that Ef is in fact the contribution of nature's own work to slowly 'recycle' the resource once on the geological scale (e.g. through sedimentary deposition, or through remelting in the mantle, etc.), and does not take into account more than one successive 'loop' of such natural recycling process. According to the same logic, the emergy of a 'secondary' (i.e. recycled) resource in input to a process at any given moment should only be Er = the emergy of (technological) recycling. A secondary input should not be assigned any additional emergy besides Er, because:

   (1) The material is already in the technosphere, and therefore its use does not entail any additional resource depletion; in other words, it does not require nature to perform another 'loop' of its slow 'recycling' work on the geological scale. Hence, in this case Ef = 0; to include this contribution again would be double counting.
   (2) The material does not need to be extracted, refined and delivered again from its natural source in the geobiosphere (e.g. from the ore in the ground). Hence, in this case Ep = 0; to include this contribution again would be double counting.

It should be noted that the same fundamental logic applies throughout emergy theory, and specifically to all natural ecosystem processes, where multiple recycling loops are ubiquitous. For instance, the emergy of the inorganic nutrients uptaken by a plant at any given moment do not carry the emergy that went into growing the previous generations of plants that grew and then decayed in the past, thereby releasing (i.e. recycling) the nutrients back into the soil. Nor does a blade of grass being fertilized by the decaying carcass of a lion see its emergy propelled to any higher level by virtue of the emergy accrued during the former 'life cycle' of its 'donor system' (i.e. the lion).

Additionally, it should also be considered that with each consecutive cycle, a new use is made (i.e. a new 'functional unit' is created) for the same amount of (recycled) material (assuming for the moment for the sake of simplicity that the recycling itself is 100% efficient). Thus, on average, the emergy of a unit of material after N cycles (ErN) would amount to its original emergy of the 'virgin' material (Ef+Ep), plus N times the additional emergy required to recycle it once (Er), divided by (N+1) total functional units (Eq. 20):

$$Er_N = \frac{(Ef + Ep) + N * Er}{(N + 1)} \qquad \text{[Eq. 20]}$$

For N >> 1, the expression above reduces to ErN ≈ Er. In other words, for those materials that may routinely be recycled multiple times (like e.g. glass and virtually all metals), the average emergy of one unit of recycled material is demonstrated to be approximated by the sole additional emergy required for the recycling process itself.

Operationally, this essentially coincides with adopting a simple 'cut-off' rule like is done in LCA, but, importantly, without calling for any special 'ad hoc rules' or exceptions to the general emergy theory. For those materials for which the recycling process entails some degree of structural degradation, thereby limiting the maximum number of cycles (N) before terminal disposal becomes inevitable, Eq. 20 also provides a theoretically sound way to compute the average emergy of a unit of recycled material. Since, typically, Er << (Ef+Ep) (otherwise recycling would not make sense in the first place), we will have in these more general cases: Er < ErN << (Ef+Ep).

4. EXAMPLES OF APPLICATION

The streamlined example below is provided as a simple illustration of some of the theoretical points discussed in the previous section. For the sake of simplicity, we shall restrict ourselves to considering only the Cumulative Energy Demand (CED) indicator (MJ of primary energy per FU) in LCA, and the Unit Emergy Value (UEV) (seJ per FU) in EMA. The former indicator allows a comparison of alternative systems and scenarios on the basis of their different demand for existing commercial energy sources. The latter, instead, provides an overall assessment of the energy 'cost' of the analysed systems over the full evolutionary time scale of the biosphere (i.e. including resource generation in addition to resource processing), and may be used as a different measure of sustainability.

It is however important to note that the overall assessment of a system‘s environmental performance typically calls for more indicators in both LCA (e.g. Global Warming Potential (GWP), Acidification Potential (AP), etc.) and EMA (e.g. Emergy Loading Ratio (ELR), Emergy Yield Ratio (EYR), etc.). In this simple, idealized example, we shall consider a factory that manufactures products made entirely of aluminium, and define our functional unit (FU) as 1 kg of product (for instance, we may refer to a 1 kg section of aluminium pipe). Virgin aluminium ingots are melted, cast, extruded and cut into the final products, which are then anodised. An amount of 0.5 kg of scraps and trimmings from the above processes per FU are reintroduced into the furnace, leading to what may be referred to as closed-loop recycling. The first time the aluminium product is produced (cycle N=0), an input of 1.5 kg of virgin aluminium is needed. Already in the first cycle (N=1), though, 0.5 kg of scraps from the first production run are reused, and the demand for virgin Al is down to 1 kg (Figure 2.11a). From then on, the average steady state amount of virgin Al that is required will tend to be reduced as the number of cycles increases (as $1+[1/(N+1)]*0.5$, where N is the number of cycles), up to a point in which a stable situation is reached (e.g. $N > 10$) where the average amount of virgin Al needed is ~ 1 kg (Figure 3b). In order to further simplify the example, we shall then analyse a case in which such a stable situation has already been reached (Figure 2.12).

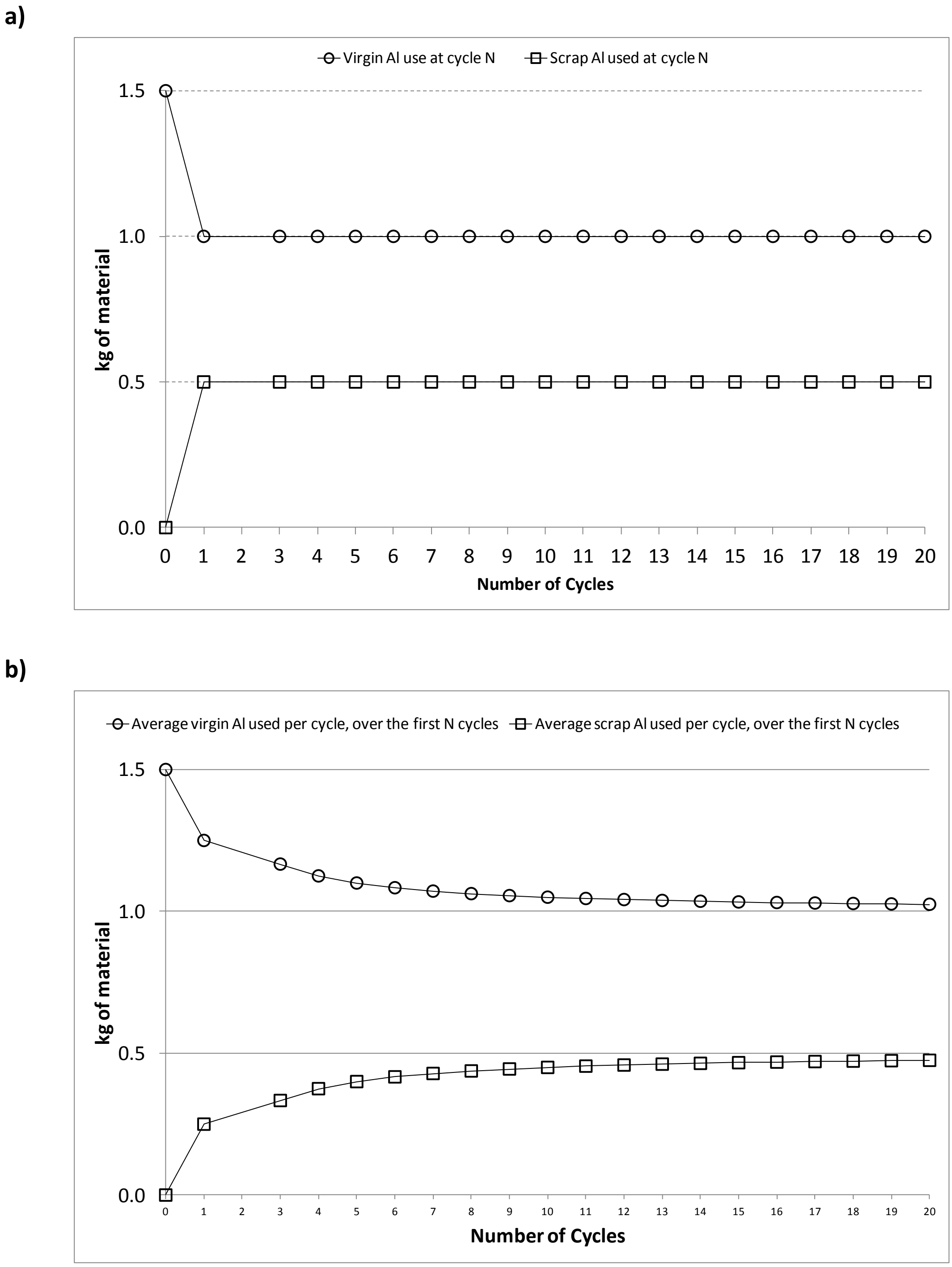

Figure 2.11: Input of virgin and secondary materials in a closed-loop industrial recycling waste process. a) in each cycle; b) average over the first N cycles.

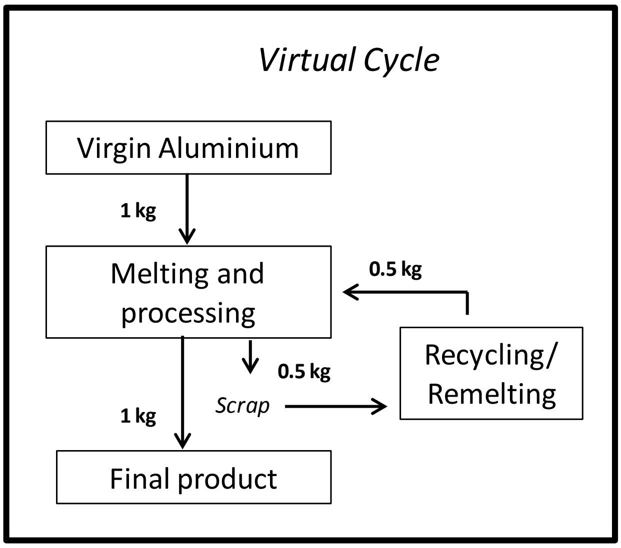

Figure 2.12: Closed-loop recycling of industrial waste (aluminium) when a steady state is reached (N>>1).

Table 2.8: Calculations for no recycling scenario.

	Amount	CED (MJ_{PE}/FU)[(a)]	EMERGY (seJ/FU)
INPUTS			
Virgin Al (kg/FU)	1.5	235.5	$2.43{\cdot}10^{13}$ (b)
Product manufacturing (electricity, kWh/FU)	1.2	14	$5.33{\cdot}10^{11}$ (c)
Total Impact		**249.5**	$\mathbf{2.49{\cdot}10^{13}}$

(a) Cumulative Energy Demand from CED impact assessment procedure in GaBi 6, based on PE International Database included in the GaBi 6 LCA software package (update: 1/12/2013)
(b) Unit Emergy Values of resource extraction, transport and processing to ingot, including biosphere work for ore concentration (Bargigli, 2003)
(c) Based on current ENTSO-E European mix; Unit Emergy Values of electricity production after Brown and Ulgiati (2002)

Table 2.9: Calculations for closed-loop recycling of industrial waste.

	Amount	CED (MJ_{PE}/FU)[(a)]	EMERGY (seJ/FU)
INPUTS			
Virgin Al (kg/FU)	1	157	$1.62{\cdot}10^{13}$ (b)
Product manufacturing (electricity, kWh/FU)	1.2	14	$5.33{\cdot}10^{11}$ (c)
Al scrap recycling process (kg/FU)	0.5	3	$1.23{\cdot}10^{11}$ (d)
Total Impact		**174**	$\mathbf{1.69{\cdot}10^{13}}$

(a) Cumulative Energy Demand from CED impact assessment procedure in GaBi 6, based on PE International Database included in the GaBi 6 LCA software package (update: 1/12/2013)

(b) Unit Emergy Values of resource extraction, transport and processing to ingot, including biosphere work for ore concentration (Bargigli, 2003)
(c) Based on current ENTSO-E European mix; Unit Emergy Values of electricity production after Brown and Ulgiati (2002)
(d) Calculated assuming 0.27 kWh/FU electricity use (Ecoinvent 2010); Unit Emergy Values of electricity production (European mix) after Brown and Ulgiati (2002)

5. CONCLUSIONS

As previously discussed a number of times elsewhere, life cycle assessment and emergy accounting are independently developed methods that have a lot in common, but which also differ in some fundamental ways, making neither expendable and instead both potentially complementary to one another in many applications.

When dealing with end-of-life and waste management processes and systems, we have found that a comparative methodological review of LCA and EMA, as presented here, points to a significant convergence of the two methods, which represents a valuable opportunity for their integration. Specifically, LCA's clear and non-contradictory treatment of system and inter-system boundaries (as applies to chains of processes that are linked in ways that make the output and waste products of one the direct or indirect inputs of the next) may lead to a better understanding and to a less potentially ambiguous statement of emergy algebra rules as they apply to waste and recycled products. Additionally, the availability of a large body of LCA literature specifically focused on waste products and systems provides a valuable opportunity for EMA researchers and practitioners to reflect on a number of complex and sometimes subtle issues, thereby potentially improving the methodology further and facilitating its applicability to policy.

However, in spite of the many steps already made towards the fruitful comparison and integration of LCA and EMA, well-framed and carried out waste management case studies are still few and far between in the existing EMA literature, and there are still a number of unresolved issues that call for further research. On one hand, there is the need for further standardization, in order to arrive at fully consistent and comparison friendly boundary and accounting procedures in LCA and EMA. On the other hand, though, there is also a need for a better and more widespread understanding and awareness of the different inherent perspectives offered by the two methods. In fact, in our opinion there is no need for a forced integration in those cases when the intended goal of the study does not require it. Also, it makes little sense to always adopt the largest possible system boundaries in those cases when the goal and scope of the analysis is intentionally restricted (e.g. when dealing with two alternative options for steel recycling).

Our systematic discussion of the main key methodological aspects of the analysis of waste products and systems in both LCA and EMA has helped identify a number of clear and non-contradictory practical guidelines that apply to both methods. We suggest that in the future such guidelines be vetted and, if confirmed to be sound, followed in all analyses of human-dominated systems that either focus on waste products and flows, or in which, in any case, the latter play a prominent role.

Acknowledgements

The authors Ripa and Ulgiati gratefully acknowledge the financial support received from the EU Project LIFE11 ENV/DE/343, MARSS —Material Advanced Recovery Sustainable Systems, LIFE+ Environment Policy and Governance.

References

- Agostinho F., Almeida M.V.B C., Bonilla S. H., Sacomano J. B., Giannetti B. F., 2013. Urban solid waste plant treatment in Brazil: Is there a net emergy yield on the recovered materials? Resources, Conservation and Recycling 73,143-155
- Amponsah N.Y., Le Corre O. and Lacarriere B., 2011. Recycling flows in emergy evaluation: A mathematical paradox? Ecological Modelling 222 (17), 3071-3081
- Arbault D., Rugani B., Marvuglia A., Benetto E., Tiruta-Barna L., 2014. Emergy evaluation using the calculation software SCALE: case study, added value and potential improvements. Science of the Total Environment 472, 608-619.
- Bargigli S., 2003. Analisi del ciclo di vita e valutazione di impatto ambientale della produzione ed uso diidrogeno combustibile. PhD thesis (Italian language).
- Bjarnadóttir H. J., Fridriksoon G. B., Johnsen T., Sletsen H., 2002. Guidelines for the use of LCA in thewaste management sector. Nordtest Report. TR 517.
- Brown M.T. and Buranakarn V., 2003. Emergy indices and ratios for sustainable material cycles and recycle options. Resources, Conservation and Recycling 38 (1), 1-22
- Brown M.T. and Ulgiati S., 2002. Emergy evaluations and environmental loading of electricity production systems. Journal of Cleaner Production 10, 321–334
- Brown M.T. and Ulgiati S., 2004. Energy quality, emergy, and transformity: H.T. Odum‘s contributions to quantifying and understanding systems. Ecological Modelling 178 (1-2), 201-213.
- Brown M.T. and Ulgiati S., 2010. Updated evaluation of exergy and emergy driving the geobiosphere: A review and refinement of the emergy baseline Original Research Article. Ecological Modelling 221, 2501-2508.

- Brown M.T. and Herendeen R.A., 1996. Embodied energy analysis and EMERGY analysis: a comparative view. Ecological Economics 19, 219-235.
- Brown M.T., Cohen M., Bardi E. and Ingwersen W., 2006. Species diversity 501 in the Florida Everglades, USA: A systems approach to calculating biodiversity. Aquatic Sciences 68, 254-277.
- Coleman T., 2006. Life Cycle Assessment for Municipal Waste: Supporting Decisions. Resources Recovery Forum. Annual General Meeting, July 19, London, UK, 2006.
- Ecoinvent, 2010. LCI database version 2.2; Swiss Centre for Life Cycle inventories, Duebendorf, CH. http://www.ecoinvent.org/database/.
- Ekvall T. and Tillman A.M., 1997. Open-Loop Recycling: Criteria for Allocation Procedures. The International Journal of Life Cycle Assessment 2 (3), 155-162.
- Eriksson O., 2003. Environmental and Economic Assessment of Swedish Municipal Solid Waste Management. PhD Thesis. Industrial Ecology, Royal Institute of Technology, Stockholm, Sweden.
- Finnveden G. and Ekvall T., 1998. Life Cycle Assessment as a decision-support tool – The case of recycling versus incineration of paper. Resources, Conservation and Recycling 4 (3-4), 235-256.
- Genoni G.P., E.I. Meyer, A. Ulrich., 2003. Energy flow and elemental concentrations in the Steina River ecosystem (Black Forest, Germany). Aquatic Sciences 6, 143-157.
- Gentil E., 2011. Life-cycle modelling of waste management in Europe: tools, climate change and waste prevention. PhD Thesis. Technical University of Denmark: Kgs. Lyngby, Denmark.
- Giannetti B.F., Bonilla S.H. and Almeida C.M.V.B., 2013. An emergy-based evaluation of a reverse logistics network for steel recycling. Journal of Cleaner Production 46, 48-57.
- Guinée J., Heijungs R., Huppes G., 2004. Economic allocation: examples and derived decision tree. The International Journal of Life Cycle Assessment 9 (1), 23-33.
- Ingwersen W.W., 2011. Emergy as a Life Cycle Impact Assessment Indicator. A Gold Mining Case Study. Journal of Industrial Ecology 15 (4), 550-567.
- ISO 2006a. 14040—Environmental Management. Life Cycle Assessment. Principles and Framework. International Organization for Standardization.
- ISO 2006b. 14044—Environmental Management. Life Cycle Assessment. Requirements and Guidelines. International Organization for Standardization.
- JRC 2010. ILCD Handbook: General guide for Life Cycle Assessment: detailed guidance. Joint ResearchCenter-Institute of Environment and Sustainability, European Commission, Ispra, Italy. 414pp. Downloaded from: http://lct.jrc.ec.europa.eu/pdf-directory/ILCD-Handbook-General-guide-for-LCA-DETAIL-online-12March2010.pdf.
- JRC 2011a. Supporting Environmentally Sound Decisions for Waste Management – A technical guide to Life Cycle Thinking (LCT) and Life Cycle Assessment (LCA) for waste experts and LCA practitioners.Joint Research Center-Institute of Environment and Sustainability, European Commission. Luxembourg: Publications Office of the European Union, 2011.

- JRC 2011b. Supporting Environmentally Sound Decisions for Bio-Waste Management – A practical guide to Life Cycle Thinking (LCT) and Life Cycle Assessment (LCA). Joint Research Center-Institute of Environment and Sustainability, European Commission, Luxembourg: Publications Office of the European Union, 2011.
- JRC 2011c. Supporting Environmentally Sound Decisions for Construction and Demolition (C&D) Waste Management – A practical guide to Life Cycle Thinking (LCT) and Life Cycle Assessment (LCA). Joint Research Center-Institute of Environment and Sustainability, European Commission, Luxembourg: Publications Office of the European Union, 2011.
- Koci V. and Trecakova T., 2011. Mixed municipal waste management in the Czech Republic from the point of view of the LCA method. The International Journal of Life Cycle Assessment 16, 113-124.
- Lei K. and Wang Z., 2008. Municipal wastes and their solar transformities: An emergy synthesis for Macao. Waste Management 28 (12), 2522-2531.
- Liu G., Yang Z., Chen B., Zhang Y., Su M., Zhang L., 2013. Emergy Evaluation of the Urban Solid Waste Handling in Liaoning Province, China. Energies 6, 5486-5506.
- Lotka A., 1922a. Contribution to the Energetics of Evolution. Proceedings of the National Academy of Sciences 8, 147-150.
- Marchettini N., Ridolfi R., Rustici M., 2007. An environmental analysis for comparing waste management options and strategies, Waste Management 27(4), 562-571.
- Marvuglia A., Benetto E., Rios G., Rugani B., 2013. SCALE: Software for CALculating Emergy based on life cycle inventories. Ecological Modelling 248, 80-91.
- Mu H., Feng X., Chu K. H., 2011. Improved emergy indices for the evaluation of industrial systems incorporating waste management. Ecological Engineering 37 (2), 335-342.
- Odum H.T., 1996. Environmental Accounting. Emergy and Environmental Decision Making. John Wiley and Sons, N.Y.
- Raugei M., Bargigli S., Ulgiati S., 2006. ―Nested emergy analyses : moving ahead from the spreadsheet platform. Presented at 4th Biennial Emergy Analysis and Research Conference, University of Florida, Gainesville, FL.
- Raugei M., Rugani B., Benetto E., Ingwersen W.W., 2014. Integrating Emergy into LCA: potential added value and lingering obstacles. Ecological Modelling 271, 4-9.
- Rugani B., Benetto E., 2012. Improvements to Emergy evaluations by using Life Cycle Assessment. Environmental Science and Technology 46, 4701-4712.
- Song Q.B., Wang Z.S., Li J.H., 2013. Sustainability evaluation of e-waste treatment based on emergy analysis and the LCA method: A case study of a trial project in Macau. Ecological Indicators 30, 138–147
- Thorneloe S.A., Weitz K.A., Jambeck J., 2007. Application of the US decision support tool for materials and waste management. Waste Management 27, 1006-1020.

- Ulgiati S. and Brown M.T., 2002. Quantifying the environmental support for dilution and abatement of process emissions: The case of electricity production. Journal of Cleaner Production 10, 335–348.
- Ulgiati S., Ascione M., Bargigli S., Cherubini F., Franzese P.P., Raugei M., Viglia S., Zucaro A., 2011. Material, energy and environmental performance of technological and social systems under a Life Cycle Assessment perspective. Ecological Modelling 222(1), 176-189.
- Ulgiati S., Raugei M., Bargigli S., 2006. Overcoming the inadequacy of single-criterion approaches to Life Cycle Assessment. Ecological Modelling 190 (3-4), 432–442.
- Ulgiati S., Bargigli S., Raugei M., 2004. Dotting the I‘s and Crossing the T‘s of Emergy Synthesis: Material Flows, Information and Memory Aspects, and Performance Indicators. In: Brown, M.T. (Ed.), Proceedings from the Third Biennial Emergy Evaluation Research Conference, Gainesville, Florida.
- VDI 1997. Cumulative Energy Demand - Terms, Definitions, Methods of Calculation. 581 In: VDI-Richtlinien 4600. Verein Deutscher Ingenieure, Düsseldorf.
- Yuan F., Shen L.Y., Li M.Q., 2011. Emergy analysis of the recycling options for construction and demolition waste. Waste Management 31 (12), 2503–2511.
- Zhang X.H., Deng S.H., Zhang Y.Z., Yang G., Li L., Qi H., Xiao H., Wu J., Wang Y.L., Shen F., 2011. Emergy evaluation of the impact of waste exchanges on the sustainability of industrial systems. Ecological Engineering 37, 206–216.

CHAPTER 3. DISCUSSION

«When a distinguished but elderly scientist states that something is possible, he is almost certainly right. When he states that something is impossible, he is very probably wrong.»

[Arthur C. Clarke, 1917-2008,
British Scientist and Writer]

«Science is a way of thinking much more than it is a body of knowledge.»

[Carl Sagan, 1934-1996,
American astronomer, astrophysicist, cosmologist]

3.1 General introduction

Preventing waste generation and promoting recycling and recovery of waste will increase the resource efficiency of the European economy and reduce the negative environmental impact associated with the use of natural resources. Promoting an efficient use of resources has become one of the main priorities of the Europe 2020 Strategy, which aims the EU to become a smart, sustainable and inclusive economy by 2020 (EC, 2010).

> *«In a world with growing pressures on resources and the environment, the* ***EU has no choice but to go for the transition to a resource-efficient*** *and ultimately regenerative* ***circular economy****. This will contribute to maintaining the resource base, essential for sustained economic growth».*
>
> *Manifesto for a resource-efficient Europe (EC, 2012)*

In this context, and to be able to quantify the environmental gains associated with these practices, it is essential to apply a LCT approach. LCA has become the most supported and valuated tool for this purpose. The risk of sub-optimization by problem-shifting, which occurs when the solution of one problem associated to a process in a system inadvertently creates other problems somewhere else, is minimized because LCA requires the consideration of the entire system, as well a full range of relevant impacts [Hauschild & Barlaz, 2011]. LCA also helps to move beyond the view of WMS as isolated systems, because sinergies with the productive system by means of energy recovery and material recycling from waste flows are considered and included in the accounting of the environmental balance of waste management practices.

However, there are still some methodological issues when applying LCA to evaluate the environmental efficiency of the WMS that merit special attention. In particular, the way in which both waste collection systems and the credits associated to material recovery have been historically modelled needs to be reviewed. Additionally, how resource depletion is assessed in LCA has been recently questioned by Klinglmair and colleagues (2014), and needs also to be improved. The three papers included in this thesis aim suggesting possible alternatives to address these important issues. In the following sections a specific discussion, summarizing and synthesizing the information contained in the three papers, is provided.

3.2 Environmental credits of material recovery

When evaluating the performance of entire WMS, it is important to find an optimum for minimizing the environmental loads caused by the operations of the system itself (collection, sorting and treatment) and, at the same time, maximizing the environmental credits obtained through the recovery of materials and energy from waste. The common practice to deal with the credits associated with recovered materials in the LCA of waste systems is to subtract the environmental impact of producing the same amount of virgin materials. That means to apply a 1:1 substitution ratio, referred at the point where the recycled materials are ready to be reintroduced into the market, after having considered all process losses because of impurities in the input waste materials or because of technology efficiencies. Although some authors have evaluated the effects of using different substitution ratios and discovered that those ratios may have a great influence in the final results (Rigamonti et al., 2009, Bovea et al., 2010), the 1:1 substitution ratio continues being the preferred choice.

This common practice originated from a time when the market for recycled goods and materials was very limited, and the economy was perceived and described as a linear chain of processes. However, the waste management systems for recovering and recycling goods and the effective reintroduction of secondary materials in the market have improved and become more widespread in many countries, thereby moving towards the goal of a more circular economy. As a result, continuing with the use of this simple substitution factor can lead to a misrepresentation of reality, and in particular to an overestimation of the environmental credits associated with recycling practices. This fact can be illustrated, for instance, with the case of platinum. This valuable metal is used by the automotive industry in the production of catalytic converters, and is recovered and reused by the industry in an almost perfectly closed loop. Thus, when analysing a car recycling facility under an attributional LCA framework, it no longer makes sense to assume that by recovering platinum we are displacing the extraction and production of virgin platinum every time we recover it - because this is not what is happening in reality.

Applying the formula proposed in this thesis to all LCAs of waste management systems to calculate the credits for all recycled materials may lead to the seeming paradox that the more one substitutes, the less credit one gets. As stated by the IFEU & Öko-Institut (2012), this may be problematic when comparing LCAs performed in different countries, because in those countries where the percentages of recycled materials in the market mixes are still small, the credit will end up being larger than in those countries where the recycling practices are more established and the amounts of recycled materials in the mixes are already large. However, this should not be considered as a 'problem', but instead as a necessary consequence of methodological consistency in strictly adhering to the attributional approach in LCA. The credit calculated by using the proposed formula can essentially be interpreted as an indication of the remaining margin for improvement, since it depends on the existing consumption mixes of virgin and recycled materials used by the market, and on the potential of the recycled material to actually replace the primary ones on a functional basis (Q factor).

Another reason for adopting this approach is the fact that it is strictly consistent with common practice in attributional LCAs when dealing with electricity production from waste management, where the national grid mix is used to calculate the environmental credits when the real substitution is not known (JRC, 2011). Let us imagine a case in which one wishes to compare the gains of recycling to the gains of incineration with energy recovery. Applying a 'marginal' approach to material recycling (1:1 substitution ratio) while adopting the common attributional praxis of assuming the grid mix replacement for electricity production, would result in a methodological bias against energy recovery. While favouring material recycling may in fact be a good decision in many cases, especially when the recycling market is still in its infancy, applying the same, strictly attributional, approach to both waste management alternatives is unquestionably more even-handed and allows the analysis of the situation from a more neutral starting point.

3.3 New predictive waste collection model

Although the majority of LCAs of waste management found in the literature reveal that the collection phase is not overly relevant when studying a full waste management system, one may suppose that this stage can gain in importance as more and more complex source-separated collection systems are established. Some studies and references in which, under some conditions, the collection stage seems to have a more relevant impact can be found in the literature as well.

A source-separated collection system for different waste fractions is implemented on the premise that the additional environmental credits gained through the recovery of materials because of the quantity and quality of waste collected this way is sufficient to compensate the additional environmental loads of the more complex collection system. However, this premise may be called into question in a country like Spain, for example, where a large percentage of municipalities are very small (under 5,000 inhabitants) and scattered on the territory, and where there is a legal obligation of establishing source-separate collection systems for plastics, glass, paper and cardboard, and organic waste (the latter by the end of 2020). Within this context, focusing the analysis on the collection phase and assessing whether or not the environmental burdens associated to the source-separated collection of municipal solid waste are in fact offset by the effective recovery of materials and energy from waste seems to be necessary.

In this thesis it has been demonstrated by experimental results that using the fixed consumption factors included in the collection stage of some LCA software packages (namely ORWARE and MSW-DST) as well as the Ecoinvent database may lead to some serious shortcomings in calculating the environmental impact of the collection phase, because usually the characteristics of the routes from which the default consumption factors where obtained do not coincide with the characteristics of the system under study.

The new predictive model for the collection stage developed within the framework of this thesis, the FENIX model, has been demonstrated to be more stable in predicting the environmental performance of collection systems in comparison to the above mentioned models and databases for all fractions.

Contrary to what some authors have recently published [Larsen et al., 2009; Crhistensen et al., 2010] and to the recommendations of guidelines for conducting LCA of waste management systems [Bjarnadóttir et al, 2002; JRC, 2011], within which the time dimension is not given any consideration, and only the overall volume and weight of the waste are mentioned as limiting factors to be considered for the calculation of the environmental impacts of the collection and transport of waste, the results of this thesis reveal that changes in the duration of the waste collection journey may have an important effect on the results. In fact, once the maximum allotted time for the collection of waste is reached, it no longer matters whether there is any capacity available left in the truck, since the waste collection truck will still return to the parking lot and another truck will resume the collection on the following day from where it was interrupted. In so doing, the total cumulative distance travelled to collect the same overall amount of waste will increase, thereby negatively affecting the average collection performance in terms of litres of diesel consumed per ton of waste collected. Thus, the time factor, just like volume or weight, can be a limiting factor for waste collection systems and has more relevance than expected from the environmental point of view in the overall life cycle of WMS, especially when we are analysing WMS in rural areas, where the transport for collecting waste is less effective.

Finally, considering other relevant issues such as the number of containers, the effective payload of the trucks or the amount of km travelled in urban areas or outside them, makes the model being able to predict changes in diesel consumption and emissions if the operational characteristics of the collection route change. This is particularly interesting when the focus of the study is put on optimizing the collection stage itself.

3.4 LCA vs. Emergy Accounting[18]

In spite of the many steps already made towards the fruitful comparison and integration of LCA and EMA, well-framed and carried out waste management case studies are still few and far between in the existing EMA literature, and there are still a number of unresolved issues that call for further research. Moreover, there is the need for further standardization, in order to arrive at fully consistent and comparison-friendly boundary and accounting procedures in LCA and EMA. Lastly, there is also a need for a better and more widespread understanding and awareness of the different inherent perspectives offered by the two methods.

In this section, the main key methodological aspects of the analysis of waste systems in both LCA and EMA are discussed.

TREATMENT OF ELEMENTARY FLOWS VS. PRODUCTS AND WASTE PRODUCTS

LCA and EMA differ in a fundamental approach related to waste. Whereas LCA makes a distinction between what it calls 'elementary flows' – flows directly sourced or emitted to the environment, including 'waste flows' – and 'products' (including waste products) derived from human activities, EMA, by definition, considers flows from/to the environment and those from/to the technosphere in the same way from a methodological point of view. Elementary flows in LCA directly contribute to environmental impact (in terms of resources depletion, and of a number of emission-related impact categories such as global warming potential or acidification potential). On the contrary, depending on the consideration of waste products as co-products (waste products with a secondary life) or waste (waste products without any possibility of further exploitation) the environmental loads of waste products differ. As co-products, waste products carry part of the elementary flows associated to the system under study. As waste, they do not have any environmental burden per se, and their environmental impact is therefore re-distributed amongst the products (or-coproducts) leaving the system.

DIFFERENT APPROACHES TO MULTI-FUNCTINALITY

When a system generates more than one product a problem of multi-functionality appears. If the object of the study is only one of such products, then, a method for allocating the environmental burdens of the system amongst the different co-products must be applied. In LCA this is solved by system expansion. When applying it, the analyst has to select which products are of primary interest and, consequently, remove the environmental impact of the rest of products by means of substracting the impact associated to the production of such

[18] This paper focuses on methodological differences between LCA and EMA, and does not contain an experimental section. Consequently, its discussion here essentially reprises (and in part duplicates) the considerations made in the paper itself. In order to maintain a consistent structure of the thesis, I have chosen to take this route rather than avoid such repetition by having a truncated discussion chapter.

products from alternative sources. If system expansion is impracticable, then allocation based on physical or economic properties may alternatively be applied.

Conversely, in EMA waste products leaving the system are considered to be either co-products or 'splits' and, depending on this, different allocation rules are employed. By co-products EMA considers two or more products (or flows) characterized by different physico-chemical nature that are gererated simultaneously. That means that one can not be produced without producing the others at the same time. In this case, according to EMA algebra rules, the total source-emergy is assigned to all of them (no allocation is applied). By 'splits' EMA considers two or more products sharing the same physico-chemical nature which are able to be produced independently. When this occurs, the total source-emergy of the system is allocated to them according to their available energy (or mass). In order to avoid double counting, EMA rules state that when two co-products reunite in a system, only the emergy of the largest flow is computed in the emergy calculation of the global system.

The way how emergy differentiates between co-products and splits may represent an important barrier for the integration of EMA and LCA inventories. Marvuglia et al. (2013) proposed an interesting way to solve this issue with their SCALE software, but this solution is methodologically questionable because this software may lead to more uncertainty and loss of accuracy than if emergy algebra rule on co-products is simply ignored, since in real life most co-products are produced in different facilities and in different periods of time.

END-OF-LIFE PROCESSES, AVOIDED IMPACT AND ENVIRONMENTAL CREDIT

In case of waste systems, the recovery of secondary materials and energy from waste is methodologically similar to multi-functionality, and the way to address it in LCA is again system expansion. The analyst must include a representative mix of conventional technologies that is assumed to be displaced by the recovery of materials and energy and, therefore, substract it from the environmental impact of the system under study.

One open debate in LCA is the selection of the most appropriate process (es) to be displaced. Two main lines of reasoning are discussed. The first one, referred as 'marginal replacement', considers the production of virgin material(s), and energy produced by those technologies whose use can be slowed by industries or governments as the best candidates to be considered for the calculation of the avoided impact. The second one is more related to situations in which the goal is not to investigate the long-term consequences of a policy, for instance, but to assess the environmental impact associated to the system as it is now. In those cases, assuming the displacement of the mix of technologies that at the time of the analysis provide an average unit of, respectively, material and/or energy seems to be more reasonable.

In principle, an 'environmental credit logic' similar to that of LCA could also be applied in EMA.

SYSTEM BOUNDARY AND CLOSED-LOOP VS. OPEN-LOOP RECYCLING

In LCA, when materials are used in more than one product cycle, it is cruzial to always set the inter-system boundaries in such a way as to clearly separate the life cycles of the different product systems that make successive use of the same materials. However, there are some procecess (the extraction of virgin materials, the recycling stages and the end of life treatments) that can alternatively be assigned to one or other of the systems, or allocated among them. One of the most used solutions for this problem is to assign the impact associated to the first stages of its waste management (collection, disassembly and transport to landfill, incineration or sorting facilities) to the first product system, and the additional impacts due to the pre-treatment and recycling of those materials to the subsequent products systems that use them. This corresponds to what in LCA is named a 'zero embodied impact'.

In EMA two basic scenarios are distinguished:

a) **Recycling with the same process** (i.e. 'closed-loop recycling') analysed assuming a steady state. When a recycled flow (waste or co-product) is fed back to a previous stage of the system, its emergy should not be double-counted and only the additional emergy invested for collection, feedback and pre-treatment should be added. This essentially coincides with the LCA logic.
b) **Waste flows from other processes** (i.e. 'open-loop recycling'). The rule to prevent double-counting does not automatically apply to this situation. At first it might seem that if the recycled/reused material was allowed to carry its entire 'emergy memory', each reuse cycle would increase the emergy of the recycled fraction without a limit. However, this interpretation is derived from a misconception of the fundamentals of emergy theory (Ulgiati et al., 2004). In general terms, the emergy of a virgin resource may be descomposed into: the emergy of natural resource formation on the geological scale (Ef) + the emergy of the subsequent processes taking place in the technosphere (Ep). Acording to the same logic, the emergy of a secondary resource should only be Er = the emergy of (technological) recycling. Any additional emergy should be considered in this case because the material is already in the technosphere and it is not needed an additional natural cycle to perform it (Ef=0) and the material does not need to be extracted, refined and delivered again from its natural source (Ep=0).

On average, the emergy of a unit of material after N cycles (Er_N) can be calculated as follows:

$$Er_N = \frac{(Ef + Ep) + \ N * Er}{(N + 1)}$$

According with this expression, for those materials that may be recycled multiple times (like glass and virtually all metals) (N>>1), the average emergy of une unit of recycled material is demonstrated to be approximated by the sole additional emergy required for the recycling process itself ($Er_N \approx Er$). Operationally, this essentially coincides whith adopting a simple 'cut-off' rule like is done in LCA, without calling any special 'ad hoc rule' or exeption in EMA theory.

3.5 References

- Bjarnadóttir H. J., Fridriksoon G. B., Johnsen, T., Sletsen, H., 2002. Guidelines for the use of LCA in the waste management sector. Nordtest Report. TR 517.
- Bovea M.D., Ibáñez-Forés V., Gallardo A., Colomer-Mendoza F.J., 2010. Environmental assessment of alternative municial solid waste management strategies. A Spanish case study. Waste Management 30, 2383-2395.
- Christensen T.H. (Ed.), 2010. Solid Waste Technology & Management Volume 1&2, John Wiley & Sons, Ltd, Chichester, UK. DOI: 10.1002/9780470666883.
- Joint Research Centre, 2011. Supporting environmentally sound decisions for waste management – A technical guide to Life Cycle Thinking (LCT) and Life Cycle Assessment (LCA) for waste experts and LCA practitioners. Luxembourg: Publications Office of the European Union.
- European Commission, 2010. Communication from the Commission. EUROPE 2020. A strategy for smart, sustainable and inclusive growth. COM (2010) 2020 final.
- European Commission, 2012. Manifesto & Policy Recommendations. European Resource Efficiency Platform (EREP).
- Hauschild M. and Barlaz MA., 2011. LCA in waste management: Introduction to principle and method. In Christensen, T.H. (Eds.), Solid Waste Technology and Management, Chapter 3.1. John Wiley & Sons, Ltd, Chichester. ISBN: 978-1-405-17517-3.
- IFEU & Öko-Institut e.V, 2012. Minutes of the Workshop "Methods to calculate GHG mitigation potentials in Solid Waste Management" on behalf of a project commissioned by Unweltbundesamt (Federal Environment Agency) Germany. 18 June 2012, Berlin, Germany.
- Klinglmair M., Sala S., Brandao M., 2014. Assessing resource depletion in LCA: a review of methods and methodological issues. The International Journal of Life Cycle Assessment 19, 580-592. DOI 10.1007/s11367-013-0650-9.
- Larsen A.W., Vrgoc M., Lieberknecht P., Christensen T.H., 2009. Diesel consumption in waste collection and transport and its environmental significance. Waste Management and Research 27, 652-659.
- Marvuglia A., Benetto E., Rios G., Rugani B., 2013. SCALE: Software for CALculating Emergy based on life cycle inventories. Ecological Modelling 248, 80-91.

- Rigamonti L., Grosso M., Sunseri M.C., 2009. Influence of assumptions about selection and recycling efficiencies on the LCA of integrated waste management systems. The International Journal of Life Cycle Assessment 14, 411-419.
- Ulgiati S., Bargigli S., Raugei M., 2004. Dotting the I‘s and Crossing the T‘s of Emergy Synthesis: Material Flows, Information and Memory Aspects, and Performance Indicators. In: Brown, M.T. (Ed.), Proceedings from the Third Biennial Emergy Evaluation Research Conference, Gainesville, Florida.

CHAPTER 4. CONCLUSIONS

«We often think that when we have completed our study of one
we know all about two, because "two" is "one and one”.
We forget that we still have to make a study of "*and*".»

[Arthur Eddington, 1882-1944,
British astronomer, physicist, and mathematician]

4.1 Introduction

This PhD thesis largely ensues from an in-depth reflection on the main unresolved methodological issues when applying LCA to WMS that emerged during the development of the LIFE+ FENIX project (LIFE08 ENV/E/000135; 2010-2013). These methodological issues were directly transposed into the main objectives of this thesis, which are outlined in section 1.3.2 and are refreshed below:

- Correcting the potential overestimation of the benefits of material recycling when the commonly used rule of considering that recycled materials displace virgin material production with a 1:1 substitution ratio is applied.
- Improving on the simplistic ways in which waste collection is modelled in most LCA studies and software packages for conducting LCA of WMS.
- Contributing to improving the Resources Depletion impact category in LCA.

Each one of these objectives was specifically dealt with in a corresponding scientific paper submitted to a peer-reviewed international journal. The main conclusions of this thesis are outlined in the following sections, as is a description of its limitations and future research lines.

4.2 Overall Conclusions

This thesis highlights some methodological aspects in LCA of WMS that may lead to a poor estimation of the environmental impacts associated to those WMS. The combined effect of the (often likely) underestimation of the impacts of the waste collection stage (when oversimplified models based on a direct correlation between km travelled and fixed fuel consumption factors are used) and of the overestimation of the credits due to the recovery of materials (by means of using a simply 1:1 substitution ratio) whenever an attributional or a consequential approach is used, may lead to inaccurate LCA conclusions and therefore potentially misguided policy decisions. This is particularly risky in countries like Spain and Portugal, where the performance of the waste collection stage (liters of diesel per ton of waste collected) is often far from optimal, and in which its impacts may end up being of the same order of magnitude as the credits associated to the recovered materials.

To try to solve this problem, in this thesis, a viable methodological alternative to account for the credits associated to material recovery in LCA of WMS and a predictive model to evaluate the environmental burdens of the collection stage, which produces much more accurate results that other models when compared with real routes, have been developed.

The alternative method for material credit accounting is intended to open a scientific debate about possible methodological inconsistencies when LCA of WMS are carried out, specially related to the sometimes practical difficulty to identify if the study should be modelled under an attributional or a consequential approach or, what is the same, if the background system will be affected in a significant way or not by our decisions. What is defended in this thesis is that the 'best' practice should be chosen on a case-by-case base, so as to be the most representative possible for the reality at hand, and for the specific research question being asked.

Another important outcome of this thesis is the identification of the Emergy Accounting methodology (EMA) as a possible complement to LCA in order to better address the challenge of resource scarcity. Taking advantage of the donor-side perspective always considered in EMA, the efforts employed by natural ecosystems to produce raw materials could be in the near future integrated in Resources Depletion impact categories in LCA. This could help us to be more conscious of the responsibility we should take on the use of those materials if we are to move towards a more efficient society in the use of resources.

4.3 Specific conclusions

In this section, the specific conclusions derived from each one of the three papers included in Chapter 2 are summarized.

AVOIDED IMPACTS DUE TO MATERIAL RECOVERY

- A unified formula for the evaluation of the environmental credits associated with material recovery in waste management, which represents a viable methodological alternative to the common marginal replacement approach (1:1 substitution factor) has been presented.

- This formula is in line with the fundamental aim of the attributional approach in LCA, as well as with a more circular economy concept, and may be applied to all waste materials, thereby ensuring methodological consistency among them.

- The formula is advocated to be used under an attributional approach, mainly for policy implementation, and for decision situations which rely on situations C1 (accounting) and A (decisions with no large-scale consequences on the background system) according the context situations described in the ILCD Handbook. It is not directly applicable to LCA adopting a LCA consequential approach (situation B).

- The formula relies on the knowledge of the application-specific or market-average consumption mixes of primary and secondary materials currently in use, which is assumed to be displaced by the recycled material. It also considers the changes in the inherent properties of the materials undergoing a recycling process ('down-cycling'), by introducing a quality (Q) factor, affecting the proportion of virgin material that is displaced.

- The same approach recommended for waste management systems is, in principle, equally valid for LCAs of product systems. However, utmost care is needed in this case in order to avoid any implicit or even explicit double counting.

NEW PREDICTIVE TRANPORT MODEL

- A new predictive model to calculate the environmental burdens related to the collection stage of WMS has been developed. This model produces more accurate results when compared with real collection routes than other existing models (ORWARE, MSW-DST) and datasets (Ecoinvent).

- The fixed diesel consumption rates included in those other models and datasets to calculate the environmental performance of the waste collection phase are to be used with caution.

- If the purpose of the study is to evaluate the environmental performance of the current implemented waste collection system, using real fuel consumption data instead of the above mentioned fixed data is suggested, trying to adjust as much as possible the characteristics of the route on which the average parameters are based to the characteristics of the system under study.

- However, if the purpose of the analysis is the environmental optimization of the collection phase itself - and the analysis of predictive results are needed before implementing changes in the system -, it is necessary to use more complex models (such the one developed in this thesis) in which the consumption and emission factors are dependent on the characteristics of the route and are not fixed.

- The duration of the working day may have a significant incluence in the environmental performance of the collection stage in LCA of WMS in rural areas. It is recommended to pay special attention to this parameter in all future LCA modelling the waste collection stage.

LCA vs. EMERGY ACCOUNTING

- LCA and EMA are independently developed methods that have a lot in common, but which also differ in some fundamental ways, making them not completely extendable but potentially complementary to one another in many applications.

- When dealing with end-of-life and waste management processes and systems, a significant convergence of the two methods, which represents a valuable opportunity for their integration appears.

- LCA's clear and non-contradictory treatment of system and inter-system boundaries (as applies to chains of processes that are linked in ways that make the output and waste products of one the direct or indirect inputs of the next) may lead to a better understanding and to a less potentially ambiguous statement of emergy algebra rules as they apply to waste and recycled products.

- For the integration of LCA and EMA, there are still a number of unresolved issues that call for further research. However, this is a great opportunity for LCA practitioners to improve the resource depletion impact by means of taking advantage of the donor-side perspective of the EMA and including the nature efforts for providing goods to human dominated systems.

4.4 Limitations of the study and future research

This PhD specifically focuses on the analisis of WMS and does not address the complex issues of defining system boundaries between products and waste management systems. In fact, such boundaries are not clear-cut and, consequently, the applicability of the system expansion suggested in this thesis needs to be carefully considered in studies where the focus is on product systems or potentially ambiguous. Additionally, a similar limitation applies in terms of a clear definition of the intended goal and scope of the study (leading to an attributional or consequential LCA approach). For instance, as discussed in Paper I (Chapter 2), the choice of the alternative system to be considered for the calculation of the environmental credits due to material recovery is instrinsically dependent on the question to be addressed or the intended application of the results. In this thesis only the attributional approach is tackled. It is noteworthy, though, that this issue of goal definition applies at least to some extent to a large proportion of real world situations, where neither a 100% attributional nor a 100% consequential approach are in fact completely applicable. This is not only the case for material recycling, but also for energy recovery, as well as in fact any situation where a product displaces a similar competitive product from the market.

This PhD opens the door to further investigate two important issues that may have a strong influence on the results of a complete LCA of a WMS. In particular:

- To investigate in the calculation of Q factors for different materials and for different applications in order to have a database that could be used by LCA practitioners to calculate the environmental credits of the recovery of materials and, particularly, for plastics.
- To work on the combination of LCA and Emergy Accounting methods to develop new characterization factors which goes beyond the current practice of only including the renewable fraction of energy captured under human control in the resource scarcity or resource depletion indicators. That means also to take into account the energy effort of nature to provide some natural resources.

Additionaly, and in order to refine the developed collection model and extend its applicability, another identified task for future research is:

- To test and validate the collection model with a larger number of routes in different Spanish and Portuguese locations, and find specific calibration factors for different collection trucks.

APPENDIXES

A. Brief history of the development of the LCA methodology and current state

LCA origins can be sought in the energy crisis of the late sixties and early seventies, which forced industries to look for more energy efficient solutions for their products [Milà i Canals, 2003]. However, the name of LCA as such did not begin to be used until 1990 [SETAC, 1990].

In 1969 the Midwest Research Institute (MRI) conducted a study for Coca-Cola® to compare different bottles for beverage packaging. This study was called *Resources and Environmental Profile Analysis* (REPA) and is considered by many authors as the "father" of current LCA [Hunt and Franklin, 1996; Fullana and Puig, 1997]. Other authors, however, consider the study by MRI for the Environmental Protection Agency of the United States in 1974 as the true precursor of LCA [Assies, 1992].

Between 1972 and 1976, the REPA methodology was described and a large number of databases were developed [Franklin & Hunt, 1972; Hunt & Franklin, 1973; Hunt & Welch, 1974; Cross et al. 1974; Hunt & Franklin, 1996]. During this initial period, the studies were simply restricted to the calculation of energy and resources consumption and the calculation of solid waste [Muñoz, 2006]. From 1976, the civil society begins to lose interest in the REPA, perhaps due to some economic recovery [Fullana and Puig, 1997], although several studies for some private companies in Sweden, Switzerland and USA were carried out [Bider et al, 1980; Huppes 1996; Udo de Haes, 1993].

From the mid eighties, though, the interest in resources and energy consumption increased again. Different studies, many of them comparing alternative packaging systems for household distribution of milk, were carried out [Bundesamt für Umweltschutz, 1984 unfold; Franke, 1984; Lundholm & Sundström, 1985; Mekel & Huppes, 1990; Pommer et al., 1991]. All these studies tried to answer the same question, i.e comparing returnable glass bottles or polycarbonate bottles with milk cartons. A thorough analysis of these studies reveals that, although the majority of them used the same bottling technology, they came up with different conclusions about which of the analyzed packaging systems had the lowest environmental impact. It became clearly apparent, therefore, that if LCA should retain its credibility and be able to fulfill society and industry's needs for life-cycle-oriented analyses, the development of consensus on central parts of the methodology was indispensable [Hauschild and Barlaz, 2011]. In those years, a study which firstly introduced a method to add (weight) different environmental impacts, the "method of critical volumes", was developed [Druijff, 1984]. At the same time, different institutions, BUWAL[19] and EMPA[20] in Switzerland; or CML[21] in The Netherlands, began to develop methods for adding substances in different impact categories [Milà i Canals, 2003].

It was in the nineties that the true methodological development of LCA began [Fullana and Puig, 1997]. In the working group organized by SETAC in 1990, the name of Life Cycle Assessment was used for the first time [SETAC, 1990]. From that year, many international groups under the supervision of SETAC and its LCA Steering Committee (a good description of those early years can be found in Jensen and Postlethwaite, 2008) furthered the methodology and the construction of a common frame of reference [Consoli et al, 1993; Vigon et al, 1993; Elkington, 1993; Fava et al, 1994; Udo de Haes, 1996; Udo de Haes et al. 2002]. In 1992, SPOLD (Society for the Promotion of LCA Development) was created. This merged 20 major European companies, with the aim of promoting the development and application of LCA. In parallel, ISO developed the standards for LCA (ISO 14.04x series, see Table A. 0.1) setting the minimum requirements for conducting an LCA.

Table A. 0.1: LCA ISO Standards list

- ISO 14.040: 1997. Environmental Management – Life Cycle Assessment – Principles and framework ([a]).
- ISO 14.041: 1998. Environmental Management – Life Cycle Assessment – Goal and Scope Definition and Inventory Analysis ([2]).
- ISO 14.042: 2000. Environmental Management – Life Cycle Assessment – Life Cycle Impact Assessment ([b]).
- ISO 14.043: 2000. Environmental Management – Life Cycle Assessment – Life Cycle Interpretation ([b]).
- ISO/TR 14.049:2000. Environmental Management – Life Cycle Assessment – Examples of application of ISO 14.041 to goal and scope definition and inventory analysis ([c]).
- ISO/TR 14.048: 2002. Environmental Management – Life Cycle Assessment – Data Documentation Format ([d]).
- ISO/TR 14.047: 2003. Environmental Management – Life Cycle Assessment – Examples of application of ISO 14.042 ([e]).
- ISO 14040:2006. Environmental Management – Life Cycle Assessment – Principles and framework ([f]).
- ISO 14044: 2006. Environmental Management – Life Cycle Assessment – Requirements and Guidelines ([g]).
- ISO/TR 14047: 2012. Environmental Management – Life Cycle Assessment – Illustrative examples on how to apply ISO 14040 to impact assessment situations.

[19] BUWAL, *Bundesamt für Umwelt, Wald und Landschaft* (Federal Office for the Environment, Forests and Landscape)
[20] EMPA, *Eidg. Materialprüfungs- und Forschungsanstalt* (Swiss Federal Laboratories for Materials Testing and Research Institute)
[21] CML, *Centrum voor Milieukunde,* Leiden (Institute of Environmental Sciences, Leiden University, Netherlands)

- ISO/TR 14049:2012. Environmental Management – Life Cycle Assessment – Illustrative examples on how to apply ISO 14044 to goal and scope definition and inventory analysis.
- ISO/TS 14071:2014. Environmental Management – Life Cycle Assessment – Critical review process and reviewer competencies: Additional requirements and guidelines to ISO 14044:2006.

(a) Revised by ISO 14040:2006
(b) Revised by ISO 14040:2006
(c) Revised by ISO/TR 14049:2012
(d) Reviewed and confirmed in 2008
(e) Revised by ISO/TR 14047:2012
(f) Repealing the old ISO 14.041, 14.042 and 14.043. Reviewed and confirmed in 2010.
(g) Reviewed and confirmed in 2010.

While during the nineties the discussion was focused on the development of LCA methodology rather than its application, from the year 2000, the activity began to focus on promoting its use. In 2002 the United Nations Environment Programme (UNEP) and SETAC launched the *Life Cycle Initiative* in order to «develop and disseminate practical tools for evaluating the opportunities, risks and trade-offs, associated with products and services over their whole life cycle» [UNEP, 2014]. Within the main objectives of the initiative are to identify best practices using the ISO standards, putting much focus on Life Cycle Management, and to provide data and methodological development to conduct LCAs that can help decision-making both at political, business and practitioner level worldwide. On its behalf, the European Union recognized LCA as an essential tool for achieving a pattern of sustainable production and consumption [EC, 2005]. It also began to foster a number of initiatives to promote the use of LCA.

- In 2005 the European Union adopted the *Thematic Strategy on the Prevention and Recycling of Waste and Sustainable Resource Consumption* and established the *European Platform on LCA*. This platform gathers methodologies, databases and methodological recommendations to perform LCA studies.
- In 2006, the first online version of the European Life Cycle Database appears. This project, within the European Platform on LCA, aims at building a database with a common language in Europe that can be used by different software and ensure data quality for LCA.
- In 2008 the Action Plan on Sustainable Consumption and Production and Sustainable Industrial Policy is adopted
- In 2009, the project CALCAS22 (2006-2009) ends up. This is a project under the Sixth Framework Programme (Coordinated Actions) with the aim of looking for Innovation in Life Cycle Analysis for Sustainability, going beyond the current LCA boundaries of the ISO approach.

[22] CALCAS: *"Co-ordination Action for innovation in Life-Cycle Analysis for Sustainability"*

- In 2010 the ILCD Handbook, a guide that covers the latest methodological developments on LCA methodology, was published. After that, sector guidelines adapting to the ILCD Handbook are being produced.
- In 2013 the European Commission launched the Innitiative Single Market for Green Products [EC, 2013] which includes two main actions: the Product Environmental Footprint (PEF) and the Organization Environmental Footpring (POF). The aim of PEF and POF is to harmonize LCA methodology for the calculation of the environmental footprint of products and organizations in the European Market. These actions are currently under a pilot test process. First Product Category Rules for conducting LCA are expected to be available by December 2016 [Imola Bedo, 2014].

Future trends

In recent years there seems to be an opening debate on the need to expand the scope of LCA, restricted to the analysis of environmental issues, to incorporate the social and economic dimensions, allowing to perform more complete sustainability assessments, what has been called *Life Cycle Sustainability Analysis* (LCSA) [Klöpffer, 2008; Heijungs et al, 2009]. The proposed strategies are basically two: either expanding the scope of the LCA to include these aspects or integrating LCA results with the results from other analytical tools from other disciplines such as *Environmental Input-Output Analysis*, *Material Flow Analysis*, *Life Cycle Costing*, *Social LCA*, *Environmental Impact Assessment* or *Strategic Environmental Assessment*, especially in the context of decision-making.

References

- Assies J.A., 1992. *State of art.* In: Report of Workshop on Life-Cycle Assessment organized by SETAC-rEurope, December 2-3, 1991, Leiden, The Netherlands, pp. 1-20.
- Baumann H. and Tillman A.M., 2004. The Hitch Hiker's Guide to LCA. An orientation in life cycle assessment methodology and application. Studentlitteratur. Malmö (Sweden).
- Bider W., Seitter L., Bertuglia L., Hunt R., 1980. Estimates of air and water pollutants and energy consumption from the production of raw materials used in wind energy systems. Solar Energy Research Institute. Golden Colorado.
- Bundesamt für Umweltschutz, 1984. Ökobilazen von Packstoffen (Análisis de Ciclo de Vida de materiales de packaging, en alemán). Bern, Switzerland. (Schriftenreihe Umweltschutz Nº. 24)
- Consoli F., Allen D., Boustead I., Fava J., Franklin W., Jensen A., de Oude N., Parrish R., Perriman R., Postlewaite D., Quay B., Séguin J., Bigon. B. (Eds), 1993. Guidelines for life cycle assessment: A code of practice. Society of Environmental Toxicology and Chemistry (SETAC). Brussels, Belgium.

- Cross J., Welch R., Hunt R., Oark W., 1974. Plastics: Resources and environment profile analysis. Manufacturing Chemist Association. Washington, D. C.
- Druijff E., 1984. Milieurelevannte produktinformatie (Environmental information on products. CML Report nº15. Centre of Environmental Science, University of Leiden, Leiden.
- Elkington J., 1993. The LCA sourcebook: an European business guide to life cycleassessment. PCM Printers. London.
- European Commission, 2005. Communication from the Commission to the Council, the European Parliament, the European Economic and Social Committee and The Committee of the Regions - Taking sustainable use of resources forward - A Thematic Strategy on the prevention and recycling of waste {SEC(2005) 1681} {SEC(2005) 1682} COM/2005/0666 final.
- European Comission, 2013. Communication from the Commission to the European Parliament and the Council. Building the Single Market for Green Products. Facilitating better information on the environmental performance of products and organizations. Brussels, 9.4.2013. COM(2013) 196 final.
- Fava J., Jensen A., Lindfors L., Pomper S., Warren B.D., Vigon B., 1994. Life-cycle assessment data quality: a conceptual framework. SETAC Press. Pensacola, Florida, USA.
- Franke M., 1984. Umweltauswirkungen durch Getränkeverpackungen – Systematik zur Ermittlung der Umweltauswirkungen von Komplexen Prozessen am Biespiel von Einwegund Mehrweg-Getränkebehältern (Environmental impact of beverage containers - Nomenclature for the measurement of environmental impacts of complex processes at Biespiel of Einwegund returnable beverage containers) EF- Verlag für Energie – u. Umwelttechnik, Berlín, Germany.
- Franklin W. and Hunt R., 1972. Environmental impacts of Polyestyrene Foam and Molded Pulp Meat Trays: a Summary Report. Midwest Research Institute (Kansas City, Mo.).
- Fullana P. and Puig R., 1997. El análisis de ciclo de vida. Ed. Rubes. Barcelona.
- Hauschild M. and Barlaz M.A., 2011. LCA in waste management: Introduction to principle and method. In Christensen, T. H. (Ed). Solid Waste Technology and Management, Chapter 3.1. John Willey & Sons, Ltd, Chichester. ISBN: 978-1-405-17517-3.
- Heijungs R., Huppes G., Guinée J., 2009. A scientific framework for LCA. Deliverable (D15) of work package 2 (WP2) CALCAS Project.
- Hunt R. and Franklin W., 1996. LCA- How it came about. Personal reflections on the origin and development of LCA in the USA. The International Journal of Life Cycle Assessment 1 (1), 4-7. DOI: 10.10007/BF02978943.
- Hunt R. and Franklin W., 1973. Environmental effects of recycling paper. In W. Kaghan (Ed.). Forest Products and the Environment 69 (133), 67-78. American Institute of Chemical Engineers, New York.
- Hunt R. and Welch R., 1974. Resource and environmental profile analysis of plastics and non-plastic containers. The Society of the Plastic Industry. New York.

- Huppes G., 1996. LCA yesterday, today and tomorrow. Centre for Environmental Science (CML). Leiden University. RITE International Workshop on Total Ecobalance. February, 1996, 29-51.
- Imola Bedo, 2014. Production Coordinator at European Commission DG Environment.
- Jensen A.A. and Postlethwaite D., 2008. SETAC Europe LCA Steering Commitee the early years. The International Journal of Life Cycle Assessment 13 (1), 1-6.
- Klöpffer W., 2008. Life Cycle Sustainability Assessment of Products. The International Journal of Life Cycle Assessment 13 (2), 89-94.
- Lundholm M.P. and Sundström G., 1985. Resource and environmental impact of Tetra Brik carton and refillable and non-refillable glass bottles. Tetra Brik Aseptic environmental profile. AB Tetra Pak. Malmö, Sweden.
- Mekel O.C.L. and Huppes G., 1990. Environmental effects of different package systems for fresh milk. Centrum voor Milieukinde. Rijksunivesiteit, Leiden, The Netherlands. (CML-Mededelingen no 70).
- Milà i Canals LL., 2003. Contributions to LCA methodology for agricultural systems. Site-dependency and soil degradations impact assessment. PhD. Universitat Autònoma de Barcelona.
- Muñoz I., 2006. Life Cycle Assessment as a Tool for Green Chemistry: Application to Different Advanced Oxidation Processes for Wastewater Treatment. PhD. Universitat Autònoma de Barcelona.
- Pommer K., Hillesborg A.S., Hirsbak S., Hoffman L., Schmidt A., 1991. Miljomaessig vurdering af maekeemballage. Returflasker af polycarbonate sammenlignet med engagspapkartoner (Análisis ambiental de envases para leche, en Danés). Danish Environmental Protection Agency. Copenhagen, Denmark. (Environmental Project no 168).
- SETAC, 1990. Life Cycle Assessment. Workshop Report. August 1990. Brussels, Belgium.
- Udo de Haes H.A. (Ed.), 1996. Towards a methodology for life-cycle impact assessement. SETAC-Europe. Brussels, Belgium.
- Udo de Haes H.A., 1993. Applications of life cycle assessment. Expectations, drawbacks and perspectives. Journal of Cleaner Production 1 (3-4), 131-137.
- Udo de Haes H.A., Finnveden G., Goedkoop M., Hauschild M., Hertwich E., Hofstetter P., Jolliet O., Klöpffer W., Krewitt W., Lindeijer E., Müller-Wenk R., Olsen S., Pennington D., Potting J., Steen B. (Eds.), 2002. Life-cycle Impact Assessment: Striving towards best practice. SETAC Press. Pensacola, Florida, USA.
- UNEP 2014: Life-Cycle Initiative homepage. http://www.uneptie.org/pc/sustain/lcinitiative/home.htm (consulted on 14/06/2014)
- Vigon B.W., Tolle D.A., Cornaby B.W., Latham H.C., Harrison C.L., Boguski T.L., Hunt R.G., Sellers J.D., 1993. Life-cycle assessment: inventory guidelines and principles. Risk Reduction Engineering Laboratory Office of Research and Development. U. S. Environmental Protection Agency. Cincinnati, Ohio. (EPA/600/R-92/245)

B. Brief description of the LCA methodology

Life Cycle Assessment (LCA) is a methodology for assessing the environmental aspects and potential impacts associated with a product or service over its entire life cycle, from 'cradle to grave'. This means taking into account all processes occurring from the extraction of raw materials up to the final disposal of waste, including manufacturing, transport, use and recycling phases as well. It is standardized by ISO 14040 and 14044.

Essentially, LCA can be described as a balance of materials and energy of the analyzed product-system, combined with an assessment of the potential environmental impacts associated with the inputs (consumption of materials and energy) and outputs (emissions to water, soil and air) of the same system. All together, it provides a comprehensive and holistic view of the environmental loads of the products or services under study, covering a wide set of environmental performance indicators such as Global Warming Potential, Acidification Potential, Eutrophication Potential, Ozone Layer Depletion Potential, Human Toxicity Potential or Ecotoxicity Potential.

According to ISO, an LCA has to be performed following the 4 phases included in Figure A. 0.1:

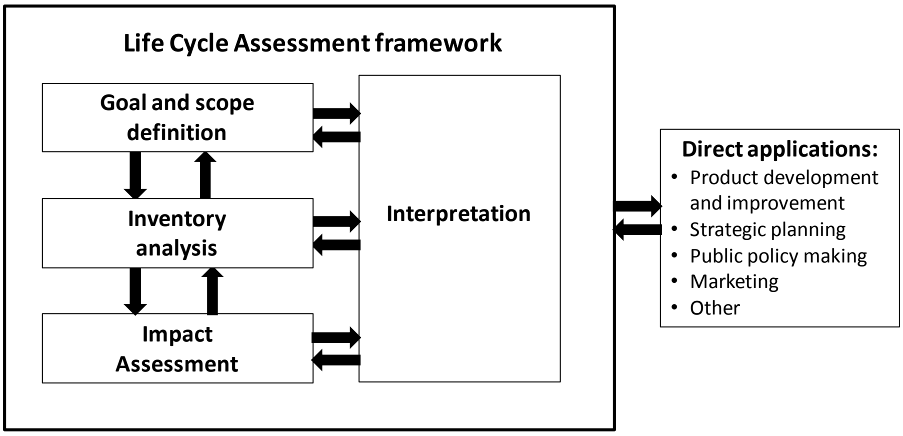

Figure A. 0.1: LCA methodology according to ISO 14040: 2006.

LCA is an iterative process. This means that in each of the phases a review of the objectives and scope of the study must be done; if the data obtained in the inventory are not sufficient or if the results of the impact assessment indicate that the objectives cannot be achieved, they should be reconsidered accordingly.

A.1. Goal and scope definition

According to ISO standards, this phase *«shall unambiguously state the intended application, the reason for carrying out the study and the intended audience».* All these aspects have to do with the context of the study, such as why it is done and how and by whom the results are going to be used [Bauman & Tillman, 2004].

This phase is, without any doubt, the most important one of an LCA. The decisions taken in it will determine choices regarding the methodology that will be necessary to use in the subsequent phases.

In this phase, among other issues, the system under study and its boundaries (conceptual, geographical and temporal) must be clearly defined. The quality of the data used, the main assumptions taken, the chosen impact categories, the allocation rules, as well as the limitations of the study must also be stated. Another key issue within the scope is the definition of the *functional unit*. This is the unit of the product or service whose environmental impacts will be assessed or compared. It is often expressed in terms of amount of product, but should really be related to the amount of product needed to perform a given function [Muñoz, 2006].

A.2. Inventory analysis

This phase consists of a compilation and quantification of inputs (consumption of materials and energy resources) and outputs (emissions to water, air and soil) that occur throughout the system under study. All inputs and outputs must be referred to the selected functional unit.

To facilitate the analysis, the system is divided into different subsystems, stages or processing units and the data are grouped into different categories in life cycle inventory (LCI) tables (Figure A. 0.2).

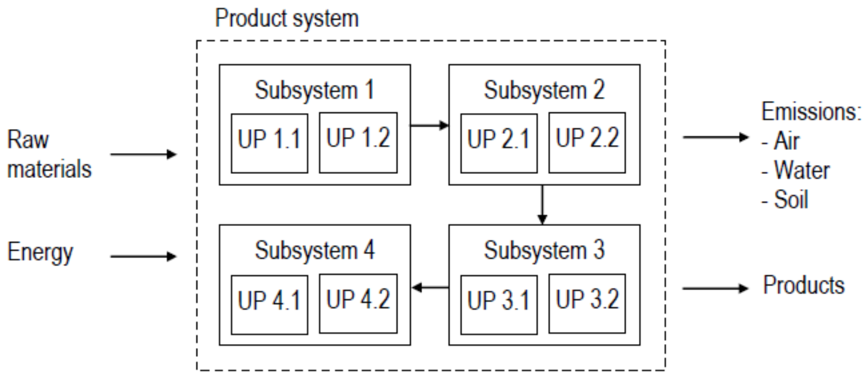

Figure A. 0.2: System definition in inventory analysis. UP: unit process.
Source: Muñoz, 2006.

The data routinely collected at the time of drawing up the inventory tables are inputs of materials, water consumption, energy consumption, fuel usage, electricity consumption, material outputs (products / co-products), air emissions, emissions to water (sewerage/ surface water), solid waste, emissions of hazardous compounds and energy outputs.

Conducting this phase seems very straightforward but, usually, it is complicated by the fact that in many cases the system under study produces more than one product (co-products). In those cases, a methodological problem arises, because it is necessary to allocate all inputs and outputs of the system to each co-product leaving the system. ISOs 10440 and 14044 give some recommendations on how to address this issue. The first choice is to try to avoid allocation, whenever possible, by means of increasing the level of detail of the model or extending the scope of the study in order to include all co-products. If this is not possible, then, the second option suggested is what is called *system expansion*, which means to find out an independent process that produces only one of the co-products – the ones that are not related to the system under study – and then, subtract their environmental load from the analysed system. Finally, if none of the previously mentioned options is applicable, allocation of inputs and output flows, by means of using one or more properties of the co-products (mass, energy content or economic value) is recommended.

A.3. Impact assessment

Life Cycle Impact Assessment (LCIA) aims to evaluate how significant are the potential environmental impacts related to the environmental loads quantified in the inventory analysis. Basically, the purpose of LCIA is to turn the inventory results into easier to understand environmental information, by means of converting the former into a reduced number of impact category indicators (such as the ones for global warming, acidification, ozone layer depletion or ecotoxicity).

This process is carried out through successive steps, which are briefly described below. Classification and characterization are compulsory in LCA according to the ISO standards, whereas normalization, grouping and weighting are optional:

- **Classification**. This simply means sorting the inventory flows or substances according to the impact categories they contribute to. For example, CO_2 and CH_4 emissions are classified into "*Global Warming Potential*" category.

- **Characterization**. This step consists of quantifying the contribution of the substances classified in each impact category, expressed in a common unit. This is done by using *"characterization factors";* factors that show the relative contribution of one singular emission at a given impact category using a reference unit. Global Warming Potential, for instance, is calculated in kg of CO_2 equivalent. Such *characterization factors* are based on scientific models of cause-effect chains in the natural systems. ISO standards (14040 and 14044) do not specify any particular set of characterization factors to be used, thus, different existing approaches developed by different research centers can be employed. At this point, the so-called *"environmental profile"*, consisting of a set of impact categories expressed in their relative units, is obtained.

- **Normalization**. This step consists of dividing the characterized results by the real or the expected total amount of pollutants emitted in a geographical area at a given moment in time (for instance, the total emissions that affect Global Warming Potential in Europe in 2014). This gives the "*relative importance*" of the environmental effects caused by the system under study in a given area.

- **Grouping**. This step consists of classifying the environmental categories quantified in the environmental profile into one or more "*groups*" (such as global/regional/local impacts or impacts with high/medium/low priority). This classification is also applicable when presenting the results of the LCI.

- **Weighting**. This step consists of converting the environmental profile to a single impact score, by means of using *"weighting factors"* based on subjective value judgments. The advantage of doing so is that the different impact categories (measured in different units) are transformed into a numerical score of environmental impact, thus facilitating decision-making. At the same time, the biggest drawback is that those weighting factors are mainly based on political criteria with hardly any scientific basis (as also mentioned in ISO 14044). Moreover, a lot of information is lost, and reality is oversimplified.

A.4. Interpretation

This is the last phase of an LCA (although, as explained above, it runs in parallel with the others). Here, the results obtained in the inventory and in the impact assessment are interpreted, while considering the defined goal and the limitations defined by the scope of the study, and recommendations for reducing the environmental impact of the analyzed system are formulated. Interpretation involves a review of all the previous phases of the analysis, in order to assess the consistency of the assumptions taken and the data quality. In the interpretation of the results, techniques such as sensitivity analysis, assessment of the limitations of the study and an external review must also be taken into account.

References

- Baumann H. and Tillman A.M., 2004. *The Hitch Hiker's Guide to LCA. An orientation in life cycle assessment methodology and application.* Studentlitteratur. Malmö (Sweden).
- ISO 14040:2006. Environmental Management – Life Cycle Assessment – Principles and framework.
- ISO 14044: 2006. Environmental Management – Life Cycle Assessment – Requirements and Guidelines.
- Muñoz I., 2006. *Life Cycle Assessment as a Tool for Green Chemistry: Application to Different Advanced Oxidation Processes for Wastewater Treatment*. PhD. Universitat Autònoma de Barcelona.

C. Important methodological aspects of LCA applied to WMS

General LCA methodology has been described in Appendix B. In this Appendix, certain methodological issues that come into focus when LCA is applied to WMS are presented and described.

C.1. General approach and system boundaries

While in LCA of products the *waste management stage* is one stage more in the complete life cycle of a product, in LCA of WMS the focus is put on analysing this particular stage of different product systems, regardless of the previous stages of each individual product (raw materials extraction, manufacture, distribution and use) (see Figure A. 03). This leads to what is called "the zero burden approach" in LCA of WMS, which means that waste do not carry any environmental burden caused by upstream stages (extraction, manufacture, distribution and use) or that wastes entering the system have "0" environmental impact. The zero burden approach is compatible with LCA methodology, which allows disregarding parts that are identical among all systems that are compared (in that case the waste generated entering the system, which is independent of the waste management design).

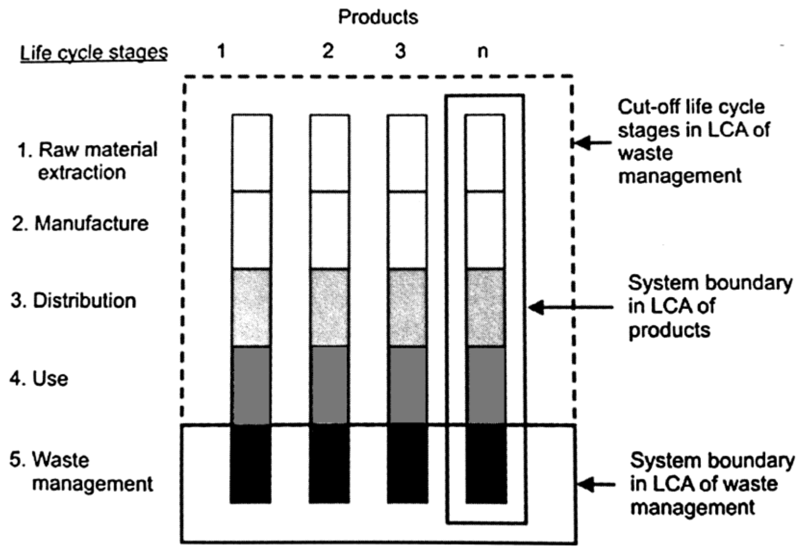

Figure A. 0.3: Boundaries of LCI of product (vertical axis) vesus LCI of solid waste (horizontal axis).

Source: Hauschild & Barlaz, 2011. Based on White et al., 1995. Reprinted with permission from Integrated Solid Waste Management – A life Cycle Inventory 2 E by F. McDougal, P. White, M. Franke and P. Hindle, 2001. Wiley.

The LCA of a WMS should take into account that resources recovered from waste (heat, electricity, materials or fertilizers) add additional functions to the system. For that reason, the system must be expanded in order to include the environmental burdens that are avoided due to the recovery of such resources. As a consequence of this expansion of the system and the zero burden approach, the results of LCAs of waste management systems are usually negative – contrary to what happens in LCA of products – meaning that the credits associated to the recovery of goods and energy are greather than the environmental burdens associated with collection and treatment of waste.

C.2. Functional Unit

In LCA of products the functional unit is defined by the outputs of the system (for instance the amount of goods produced by an industry). On the contrary, for WMS, the service function provided by the system is to collect, treat and dispose a certain amount of waste being, consequently, related to the input of the system.

If the purpose of the study is to analyse a specific waste management system, then, the amount and composition of waste of the region and the period of time under study must be considered. However, if the purpose is to compare different treatment options for a specific waste flow, the amount can be chosen arbitrarily.

Table A. 0.2: Aspects to be considered to define the functional unit of an LCA of WMS

- Amount of waste to be managed - Composition of waste - Duration of the system or the service upon which the environmental impacts will be quantified - Quality of the management system (emission boundaries, requirements for the recovered materials...)

C.3. Life cycle stages to be considered

The most important stages to be considered in LCA of WMS are summarized in Table A.0.3.

Table A. 0.3: Life cycle stages and unit processes to be taken into consideration in LCA of WMS

Life cycle stage/unit process	Comments/recommendations for studies
Household and/or industry distribution of waste on reception facilities	Waste bins where the waste has different destinations and/or treatment. Can be excluded from the system if common for all treatment alternatives under study.
Collection and transport	Processes for transporting waste to treatment facilities and waste treatment products to final consumption should be included. As transport processes usually give small contributions to the total life cycle impacts, they can be excluded for ancillary materials, if not already included in ready-made cradle-to-gate inventory data for the ancillary material. Transportation for collection of the waste will normally be important.
Production and use of fuel, electricity and heat	Important to include. See comments in the next row.
Manufacture of ancillary materials	Flows are divided into *primary flows* and *secondary flows*. The primary flows are the materials that the product is built up from. The secondary flows are auxiliary materials and energy that enables an activity to be performed. Several tiers of auxiliary flows may extend further and further from the main sequence. The analyst should set criteria on how many tiers of auxiliary flows will be included. The criterion is typically set from 0-2 tiers of auxiliary flows. 0 tier means that a material flow is only identified by the input amount and not by the upstream life cycle. 1 tier means that the material flow used in a process unit is included by its upstream life cycle, but the materials used in the upstream life cycle flow are not. It is common to use ready-made cradle-to-gate data for secondary flows (cradle-to-gate is the part of the life cycle including everything from resource extraction to ready-made product, but not use and disposal). The selection of tiers is then not a relevant issue.
Waste treatment processes	Waste treatment systems consist of the degradation system and other processes like pumps, cutting equipment, preheating etc. It is important to include both environmental impacts related to the degradation process itself and supporting processes.
Recycling/recovery of materials and/or energy	Important to include. 1. Energy recovery from incineration. 2. Energy recovery of bio-gas from anaerobic digestion. 3. Energy recovery of landfill gas. 4. Recovery of soil improvement material from composting and anaerobic digestion. 5. Recovery of materials from recycling processes.
Manufacture, maintenance and decommissioning of capital equipment	Usually of little importance. Should only be included on request, or if the capital equipment itself is the product subject to an LCA.
Additional operations such as building lighting and heating	Usually of little importance. Should only be included on request.

Source: Bjarnadóttir et al., 2002.

C.4. Process versus product approach

Unlike in LCA of products, when making LCI of WMS, there are two main approaches for modeling processes and estimating emissions that can be applied (see Figure A. 0.4):

- **Process approach:** it uses real data from emissions and resource consumption for different waste treatment facilities, including technological variations. If the composition of the waste entering the system is well known as well as the specific technology, this approach can be useful.
- **Product approach:** it uses mathematical models to calculate the emissions and resource consumption of different treatment facilities depending on the composition of the waste. For applying this approach, it is necessary to know the basic composition of waste (waste fraction: paper, different types of plastics, aluminium, steel, organc waste…) as well as the elementary composition of the waste entering the system (carbon content (C) from fossil or organic origin, nitrogen (N), sulfur (S), chlorine (Cl) or metals). This approach is useful when the composition of the specific waste entering the system is not fixed.

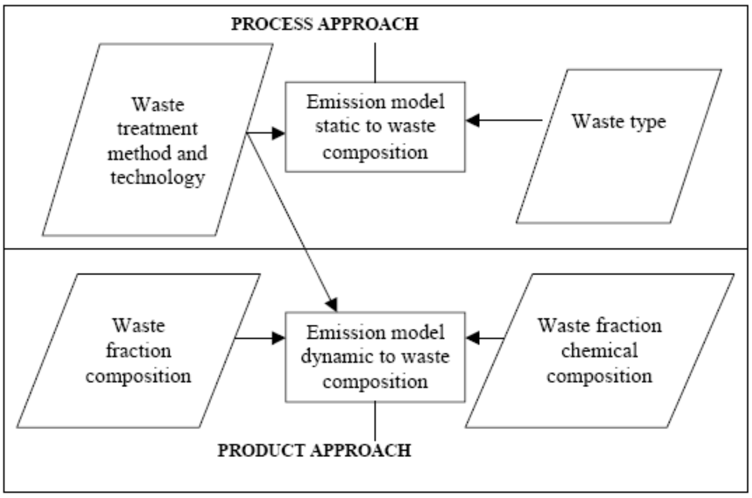

Figure A. 0.4: Main differences between process and product approaches for LCI of WMS.
Source: Bjarnadóttir et al., 2002.

C.5. Time perspective and landfills

Defining the time scale of an LCA is something required in the goal and scope definition phase. This is important for knowing for how long the conclusions of the study will be valid, as well as for selecting the most appropriate data in the inventory analysis.

However, in LCA of WMS there is a special problem regarding time that is related to the emissions from landfills. These emissions may continue long after the time period defined in the scope of the study, as opposed to instantaneously as is the case of the other waste treatment facilities (e.g. incineration) or life cycle stages of waste management (e.g. collection and transport or recycling) [Hauschild & Barlaz, 2011].

Thus, waste in landfills and products used to improve the quality of the soil once the landfill is closed will have a long duration environmental impact. This is the case, for instance, of metal leaching and gas emissions resulting from the degradation of organic matter such as methane. The challenge to be addressed in implementing an LCA is selecting an appropriate time interval and integrating functions of time-dependent emissions during this period. About this specific issue, Kendall et al. (2009) proposed a time correction factor to address CO2 emissions that occurs over time from biofuel production and a similar approach may be used in landfills.

The ISO 14040 and 14044 standards provide no specific recommendation regarding time horizon. SETAC, however, recommends that "E emission" (i.e. total aggregate emissions over time), is integrated over a period of infinite time (being T1 = 0 and T2 = infinity). If this is not possible, a time interval of 100 years is suggested. The third option would be any other time interval. Generally, LCAs of WMS tends to use a time horizon of 100 years.

Apart from the selection of the time horizon, there is an additional problem related to the time period in landfills that merits special attention. This is related to the fact that life cycle inventories only report the quantities emitted per functional unit, and do not consider the rate of emission. For certain impact categories this rate of emission is essential, because the effects of releasing one substance over the time in small quantities are not the same than releasing all this substance at once. This is the reason why, for instance, the «total cooper emitted from an electronic product deposited in a landfill may thus be so high that the associated ecotoxicity impact potential completely dominates the impact assessment results» [Hauschild & Barlaz, 2010]. For this problem, Hauschild and colleagues (2008) proposed an interim solution to treat toxic long emissions by means of creating a new impact category representing the “stored” human and ecotoxicity in landfills.

C.6. Open-loop recycling

Open-loop recycling takes place when materials recovered from waste are reintroduced into the market. As stated in section C.1., the recovery of materials adds new functions to our system (in this case producing materials apart from managing waste). To be able to analyse the environmental impacts of a WMS it is necessary to allocate the impacts of the system among the multiple functions performed. ISO standards for LCA describe procedures for allocation but recommend avoiding it by means of expanding the system boundaries to take into account the avoided burdens or to substract the avoided burden due to material (or energy) recovery (see Figure A. 0.5).

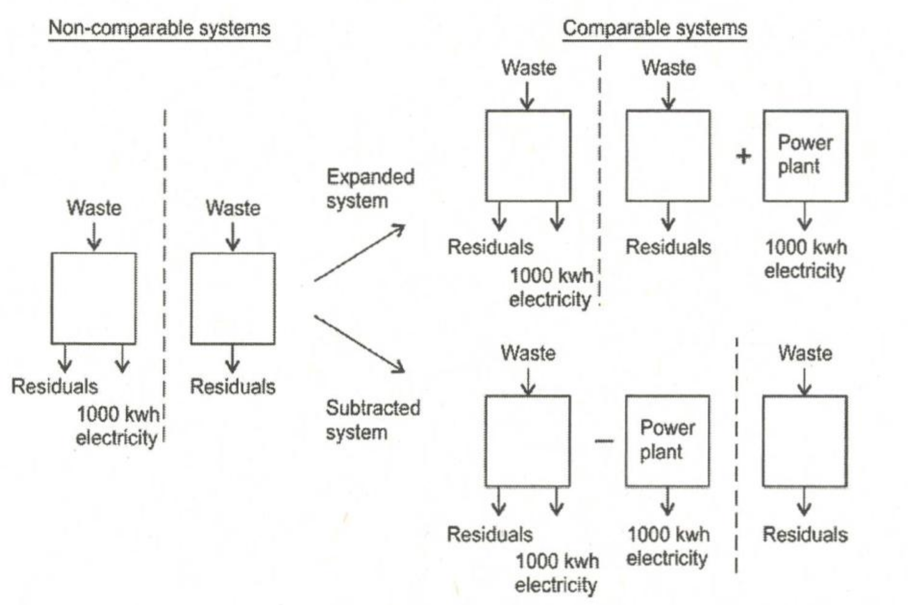

Figure A. 0.5: The principle of system expansion and substraction to obtain functional equivalence between different systems in LCA of waste management.
Source: Christensen & Barlaz, 2010.

Determining what avoided burdens have to be considered is not easy. Understanding what products are replaced when resources are recovered from waste requires insight into market mechanisms. Another issue to take into account is the purpose of the study. If it is a change-oriented (or consequential) analysis, the LCA shoud reflect the consequences of a choice and ideally marginal data should be used. In contrast, a descriptive or attributional LCA should reflect what is actually happening in the system. In this case the use of average data may be more appropriate [Björklund et al., 2011].

C.7. Multi-input allocation

A similar methodological problem to that of open-loop recycling occurs in the majority of processes related to waste management. They have multiple waste inputs, and if one wants to determine what emissions, resource use, or recovered products should be allocated to each individual waste flow, then, is is inevitable to have to use allocation procedures. ISO 14044 standard recommends the stepwise procedure presented below:

«a) ***Step 1:*** *Wherever possible, allocation should be avoided by:*

1) *dividing the unit process to be allocated into two or more sub-processes and collecting the input and output data related to these sub-processes, or*
2) *expanding the product system to include the additional functions related to the co-products, taking into account the requirements of 4.2.3.3.*

b) ***Step 2:*** *Where allocation cannot be avoided, the inputs and outputs of the system should be partitioned between its different products or functions in a way that reflects the underlying physical relationships between them; i.e. they should reflect the way in which the inputs and outputs are changed by quantitative changes in the products or functions delivered by the system.*

c) ***Step 3:*** *Where physical relationship alone cannot be established or used as the basis for allocation, the inputs should be allocated between the products and functions in a way that reflects other relationships between them. For example, input and output data might be allocated between co-products in proportion to the economic value of the products».*

An example of allocation in LCA of WMS is the allocation of emissions, consumption of resources, residues generated (ashes and slag) and energy recovery in an incineration plant. In this case, carbon dioxide, methane or NMVOC emissions must be allocated based on the carbon content of each material, metals emissions based on the metal content, fluoridric acid emissions based on the fluor content, electricity produced based on the heating value of each material, and fuels, ancilliary materials, emissions of PCDD/F, NO_x, N_2O, NH_3, dust, and ashes and slag produced must be allocated by mass (because they depend more on the technology and operational parameters than in the composition of the waste entering the system) [Margallo, 2014].

C.8. Biotic carbon and carbon sinks

In waste management, CO_2 can be emitted as a consequence of the incineration of materials, or by degradation in landfills or in organic treatments such as composting. When the CO_2 comes from organic material such as paper or food the CO_2 emitted is called biotic CO_2. It is common practice in LCA to account the global warming impact of biotic CO_2 as zero. In reality, biotic CO_2 has the same impact on global warming that fossil CO_2, but this simplification of considering a zero impact for the biotic CO_2 is based on the premise that incineration releasing X tons of biotic carbon, for instance, is balanced by its corresponding uptake in forests (see Figure A. 0.6).

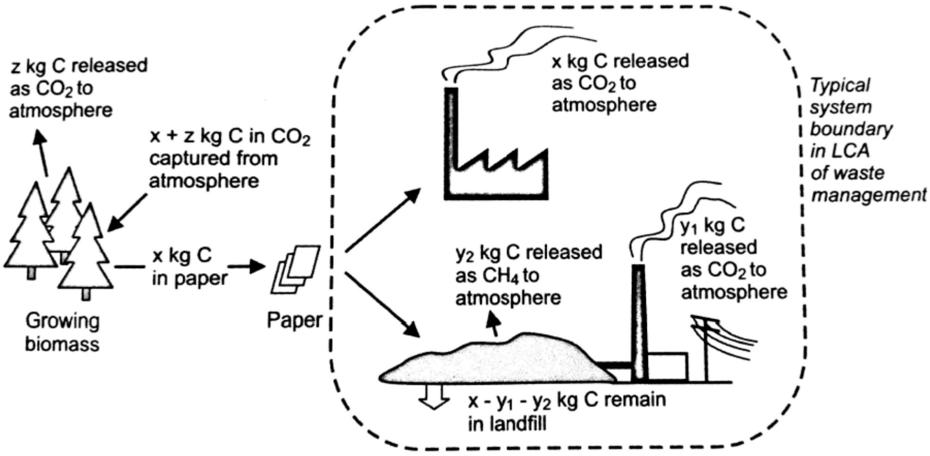

Figure A. 0.6: Uptake and release of biotic carbon.
Source: Björklund et al., 2011.

However, in the case of landfills, compost used in agriculture or compost substituting peat-based soils improvers this common practice may be misleading. When we select a time horizon for landfills different to infinite, what occurs is that after the time period selected, some carbon in organic materials remains in landfills, or what is the same, that landfills serve as carbon sinks. To take into account this fact in LCA of WMS, two alternatives can be chosen. The first one is to consider the uptake of carbon in growing biomass and accounting the biotic CO_2 as having the same impact as fossil CO_2 (1), and treat the emissions of biotic CO_2 in the same way. The second one is simply accounting the amount of biotic CO_2 retained in landfills and applying a negative contribution (-1) to global warming.

References

- Bjarnadóttir H.J., Fridriksson G.B., Johnsen T., Sletsen H. Guideline for the use of LCA in the waste management sector, Nordtest, Espoo, Finland (Nordtest Report 517), 2002.
- Björklund A., Finnveden G., Roth L., 2011. 3.2. Application of LCA in Waste Management. Christensen, T.H. (Ed). Solid Waste Technology and Management. Blackwell Publishing.
- Hauschild M. and Barlaz M.A., 2011. LCA in waste management: Introduction to principle and method. Christensen, T. H. (Ed). Solid Waste Technology and Management. Blackwell Publishing.
- Hauschild M.Z., Olsen S.I., Hansen E., Schmidt A., 2008. Gone…but not away – addressing the problem of long-term impacts from landfills in LCA. The International Journal of Life Cycle Assessment 13 (7), 547-554.
- ISO 14044: 2006. Environmental Management – Life Cycle Assessment – Requirements and Guidelines.
- Kendall A., Chang C., Sharpe B., 2009. Accounting for time-dependent effects in biofuel life cycle greenhoyuse gas emissions calculations. Environmental Science and Technology 43 (18), 7141-7147.
- Margallo M., Life cycle model of waste to energy technologies in Spain and Portugal. PhD in Chemical and Biomolecular Engineering. University of Cantabria. Department of Chemical and Biomolecular Engineering. Santander, 2014.
- White P.R., Franke M., Hindle P., 1995. Integrated Solid Waste Management: a Life cycle Inventory. London, UK: Blackie Academic and Professional.

D. Brief description of Emergy Accounting methodology

The conceptual and theoretical framework of Emergy Accounting methodology is grounded in thermodynamics and systems ecology. The evolution of the theory over the last 30 years was documented by Odum (1995, 1996).

Emergy can be defined as the amount of available energy of one form (usually solar) that is required, directly or indirectly, to provide a product or service expressed in one type of energy, usually solar emergy. The ratio of emergy required to make a product to the energy of the product is called transformity. Solar emergy is expressed in solar emergy joules (called solar emjoules and abbreviated sej), while solar transformity is a ratio of solar emergy joules per Joule of output flow (sej/J). Materials are expressed as emergy per mass (sej/g).

In the most general sense, the total emergy driving a process is a measure of the activity required and converged to make that process possible. It is a measure of the work (in both the past and present) necessary to provide a given resource or service, be it the present stock of iron ore or oil deep in the planet or services provided by labor. Emergy content of major raw material resources of the earth are evaluated using the total emergy driving the biosphere and total quantities of global resources (Odum, 1996, 2000). Emergy of human services is evaluated using the total emergy required by workers for their support. The emergy of any product or process is the sum of the emergies used in both the past and present to make it.

Transformities of the main natural flows in the biosphere (wind, rain, ocean currents, geological cycles, etc.) are calculated as the ratio of total emergy driving the biosphere as a whole to the actual energy of the flow under consideration. The transformity of solar radiation is assumed equal to one (1). Transformities have been calculated for a wide variety of energies, materials, and services (Odum, 1996, 2000; Brown and Ulgiati, 1999a; Ulgiati and Brown, 1999; Ulgiati et al., 1994).

Emergy quantifies energy and material resources as well as environmental and human services within a common framework. It reflects differences in the quality of energy and resources. Embodied in the emergy of products are the services provided by the environment, which are free and outside the money economy (Brown and Ulgiati, 2001; Ulgiati and Brown, 2001). By accounting for quality and free environmental services, resources are not valued by their money cost or society’s willingness to pay, which are often very misleading, especially if decisions need be made regarding sustainability or environmental costs. They are misleading because in no way does society’s willingness to pay reflect environmental costs or services. For a more full treatment of this topic see Brown and Ulgiati (1999b).

Emergy algebra rules are summarized below:

Table A. 0.4: Emergy algebra rules

- **Rule number 1**: all source emergy to a process is assigned to the process output.
- **Rule number 2:** co-products [(1)] from a multi-output process have the total emergy assigned to each pathway.
- **Rule number 3:** when a pathway splits [(2)], the emergy is divided among each "leg" of the split based on its percentage of the total energy flow on the pathway.
- **Rule number 4:** emergy cannot be counted twice within a system: (a) emergy in feedbacks cannot be double counted, and (b) co-products, when reunited, cannot be added to equal a sum greater than the emergy source from which they were derived.

(1) Co-products are "product items showing different physico-chemical characteristics, but which can only be produced jointly" (Sciubba and Ulgiati, 2005).

(2) Splits are "originating flows showing the same physical-chemical characteristics" (Sciubba and Ulgiati, 2005). Therefore, emergy of split products will be different, proportionally assigned on the basis of their quantity, while their transformity is the same.

References

- Brown M.T. and Ulgiati S., 1999a. Emergy based indices and ratios to evaluate sustainability: monitoring technology and economies toward environmentally sound innovation. Ecological Engineering 9, 51-69.
- Brown M.T. and Ulgiati S., 1999b. Emergy evaluation of natural capital and biosphere services. AMBIO September 28 (6), 486-93.
- Brown M.T. and Ulgiati S., 2001. The role of environmental services in electricity production processes. Journal of Cleaner Production 10, 321-34.
- Odum H.T., 2000. Emergy of Global processes. Handbook of Emergy Evaluations, Folio #2. Center for Environmental Policy, P.O. Box 116450, University of Florida, Gainesville, FL 32611.
- Odum H.T., 1995. Self-Organization and Maximum Empower', in C.A.S.Hall (ed.) Maximum Power: The Ideas and Applications of H.T.Odum, Colorado University Press, Colorado.
- Odum H.T., 1996. Environmental Accounting. Emergy and Environmental Decision Making. John Wiley and Sons, N.Y.
- Sciubba E. and Ulgiati S., 2005. Emergy and exergy analyses: Complementary methods or irreducible ideological options? Energy 30, 1953–1988.
- Ulgiati S. and Brown M.T., 1999. Emergy accounting of human-dominated, large scale ecosystems. In: Jorgensen, Kay, editors. Thermodynamics and Ecology. Elsevier.

- Ulgiati S. and Brown M.T., 2001. Emergy evaluations and environmental loading of alternative electricity production systems. Journal of Cleaner Production 10, 335-348.
- Ulgiati S., Odum H.T., Bastianoni S., 1994. Emergy use, environmental loading and sustainability. An emergy analysis of Italy. Ecological Modeling 73, 215 -68.