# Sustainability assessment of municipal compost use in horticulture using a life cycle approach

Julia Martínez Blanco

## Doctoral thesis

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A thesis submitted in fulfilment of the requirements for the Doctoral degree in Environmental Sciences and Technology

Sostenipra research group
Institut de Ciència i Tecnologia Ambientals (ICTA)
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"Not everything that counts can be counted,
and not everything that can be counted counts"

Albert Einstein

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Bellaterra (Cerdanyola del Vallès), May 2012

Joan Rieradevall Pons | Pere Muñoz Odina | Assumpció Antón Vallejo

# Contents

# Figures

# Tables

# List of acronyms, abbreviations and notation

| | |
|---|---|
| 1.4 DB eq. | 1.4 dichlorobenzene equivalent emissions |
| ADP | Abiotic depletion potential |
| AP | Acidification potential |
| C- | Cauliflower case study |
| $C_2H_4$ eq. | Ethylene equivalent emissions |
| CED | Cumulative energy demand |
| CF | Carbon Footprinting |
| CFC-11 eq. | Trichlorofluoromethane equivalent emissions |
| $C_H$ | Cultivation option with high-dose of compost and mineral fertilizers. |
| $C_L$ | Cultivation option with low-dose of compost and mineral fertilizers. |
| CML | Institute of Environmental Sciences (Leiden) |
| $CO_2$ | Carbon dioxide |
| $CO_2$ eq. | Carbon dioxide equivalent emissions |
| CP | Stage compost production |
| CT | Stage compost transport |
| Cu | Stage cultivation |
| CuE | Sub-stage post-application emissions |
| CuF | Sub-stage fertirrigation infrastructure |
| CuI | Sub-stage irrigation |
| CuM | Sub-stage machinery and tools |
| CuN | Sub-stage nursery plant |
| CuO | Sub-stages of field operations |
| CuP | Sub-stage phytosanitary substances |
| DTU | Technical University of Denmark |
| ELCD | European Reference Life Cycle Database |
| EP | Eutrophication potential |
| ESAB | Escola Superior d'Agricultura de Barcelona (UPC) |
| FP | Stage mineral fertilizers production |
| FT | Stage mineral fertilizers transport |
| FU | Functional unit |
| FWAE | Fresh water aquatic ecotox. |
| G | Stage greenhouse |

| | |
|---|---|
| GH | Cultivation option under greenhouse |
| GHG | Greenhouse gas emissions |
| GICOM | Grup d’Investigació en Compostatge (UAB) |
| GIS | Geographic Information System |
| GM | Sub-stage greenhouse management |
| GS | Sub-stage greenhouse structure |
| GW | Green waste |
| GWP | Global warming potential |
| HC | Home composting |
| $HNO_3$ | Nitric acid |
| HT | Human toxicity |
| IC | Industrial composting |
| ICTA | Institute of Environmental Science and Technology (UAB) |
| ILCD | International Reference Life Cycle Data System |
| ILO | International Labour Office |
| IPCC | Intergovernmental Panel on Climate Change |
| IRTA | Institute of Agriculture and Food Research and Technology |
| ISO | International Organization for Standardization |
| $K_2SO_4$ | Potassium sulfate |
| $KNO_3$ | Potassium nitrate |
| $KPO_4H_2$ | Monopotassium phosphate |
| LCA | Life Cycle Assessment |
| LCC | Life Cycle Costing |
| LCI | Life Cycle Inventory |
| LCIA | Life Cycle Impact Assessment |
| LCSA | Life Cycle Sustainability Assessment |
| LCSD | Life Cycle Sustainability Dashboard |
| LCWE | Life Cycle Working Environment |
| M | Cultivation option with mineral fertilizers. |
| MAE | Marine aquatic ecotoxicity |
| MAL | Maximum authorised load |
| MBT | mechanical biological treatment |
| MJ eq. | Mega joules equivalent |
| MSW | Municipal solid waste |
| ODP | Ozone layer depletion potential |

| | |
|---|---|
| OECD | Organisation for Economic Co-operation and Development |
| OF | Cultivation option in open-field |
| OFMSW | Organic fraction of the municipal solid waste |
| OM | Organic matter |
| PAW | Plant available water |
| $PO_4^{3-}$ eq. | Phosphate equivalent emissions |
| POP | Photochemical oxidation potential |
| Sb eq. | Antimony equivalent emissions |
| SETAC | Society of Environmental Toxicology and Chemistry |
| SHDB | Social Hotspot Database |
| SLCA | Social Life Cycle Assessment |
| SLCIA | Social Life Cycle Impact Assessment |
| $SO_2$ eq. | Sulphur dioxide equivalent emissions |
| SOC | Soil organic carbon |
| SOM | Soil organic matter |
| Sostenipra | Sustainability and Environmental Prevention |
| T- | Tomato case study |
| TE | Terrestrial ecotoxicity |
| TUB | Technical University of Berlin |
| UAB | Universitat Autònoma de Barcelona |
| UNEP | United Nations Environmental Program |
| VOC | Volatile organic compounds |
| WHC | water holding capacity |

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# Summary

Waste management is a global problem in developed countries due to the rapid collapse of landfills and the high impacts related to biodegradable waste dumping. On the other side, many agricultural soils present low organic matter contents, especially in Mediterranean regions. Therefore, facing these two facts, composting arises as a relevant option to close the organic matter and nutrient cycles, interconnecting both city and country systems. Nevertheless, specific, real and quantitative data about composting technologies and compost application, including all the aspects of the life cycle and from several perspectives, are necessary.

Life-cycle-thinking tools are methodologies to construct either environmental or social or economic profiles of production systems along their life cycle. Although a considerable effort has been made in recent years to adapt Life Cycle Assessment (LCA) to agricultural systems, more research is needed to evaluate its suitability for the environmental quantification of compost production, from the organic fraction of the municipal solid waste (OFMSW), and application in soils. Regarding, social and economic assessment tools, they are still on their early stages to be adequate enough to asses sustainability.

This dissertation presents a holistic approach of compost, including all the stages of its life cycle and assessing environmental performance, agricultural viability, as well as consumer, territorial and other sustainability perspectives. In addition, methodological improvements for the different assessments of compost production and application are proposed.

Real data from two composting sites, at home and full scale, as well as real data from horticultural fields have been collected within the framework of the thesis. Subsequently, these data are used on the environmental and agricultural assessment of compost production and application, taking into account two horticultural crops (tomato and cauliflower), three fertilization options and two location sites (open field and greenhouse). The outcomes state industrial composting production as the main source of environmental impact in the entire life cycle, especially due to electricity consumption, waste collection and volatile organic compounds. This result implies that cultivation options with compost are more impacting than options only using mineral fertilizers for all the impact categories (when avoided burdens from not landfilling the OFMSW are not subtracted); whereas, when these avoided burdens are considered, they are more environmentally friendly for the impact categories EP, GWP and OLDP and have similar impact for CED and ADP. Home composting seems to be an environmentally better alternative than industrial composting for resources consumptions, even though it has major gaseous emissions.

Regarding agronomic performance, sufficient yields and quality of the crops are only assured when compost is accompanied by a source of rapid nitrogen release. In addition, resource investments on greenhouse are only justified when sufficiently larger yields are obtained than in open field.

Two of the methodological issues regarding compost life cycle environmental assessment that are discussed in the dissertation are: functional unit definition and compost benefits inclusion. Because higher bioactive compounds content is detected in cauliflower when high dosages of compost are applied, functional units related to quality, apart from production ones, are considered and then the environmental differences among fertilization options are tempered or even reversed. Furthermore,

the dissertation provides a description of the magnitude, dynamics and limiting factors for each of the considered benefits of compost application and discusses the key methodological issues for their quantification within LCA.

Carbon Footprinting (CF), which has been increasingly used for the environmental quantification and communication of products, is compared with LCA. According to the results, PAS2050 standard underestimates the environmental figures communicated to consumers, mainly due to capital goods. The dissertation also includes one of the few attempts to apply Social Life Cycle Assessment (SLCA) and Life Cycle Sustainability Assessment (LCSA). It consists on the assessment of nitrogenated mineral fertilizers and compost along their entire production life cycle. Nitric acid is identified as the best fertilizing alternative for the social, economic and environmental dimensions, compost is the worst for the two latter, and potassium nitrate is the worst for social dimension. Finally, territorial planning perspective regarding compost use in horticulture is overviewed. In those areas with nitrate polluted aquifers and nutrient demand from either horticulture or general agriculture, the combined use of compost from municipal waste and nitrate polluted water seems to be a potential fate for both flows, as well as a way to reduce non-renewable resources consumption.

In conclusion, this thesis is a relevant step towards sustainability taken by compost production and application and offers a multidisciplinary approach of the system. Among other important follow-up lines of research, future work should focus on the optimization of compost production process at several scales, the assessment of compost application in the long-term, the study of the alternatives for the allocation of compost burdens among crops in a rotation, a better understanding of mineralization rates and their key affecting factors, the inclusion of potential drawbacks of compost application in LCA, a deeply assessment of compost application from the regional perspective, the development of several environmental impact categories necessary for the correct assessment of compost application, and further development of sustainability tools.

# Preface

The present doctoral thesis was developed within the research group on Sustainability and Environmental Prevention (Sostenipra) at the Institute of Environmental Science and Technology (ICTA) of the Universitat Autònoma de Barcelona (UAB) from October 2007 to May 2012 (including the period of the Master in Environmental Studies).

This dissertation analyses the municipal compost application and production in the Mediterranean region from a sustainable perspective. This dissertation is the result of a multidisciplinary approach that aims to highlight the environmental performance of municipal compost as a fertilizer; but goes further and seeks to quantify and analyze the social, economic, technological and agronomic results and to propose strategies to improve production from a sustainability point of view. Methodological proposals are also presented.

The dissertation is mainly based on the following book chapters and papers either published or under review in peer-reviewed indexed journals:

- Martínez-Blanco, J., Muñoz, P., Antón, A., & Rieradevall, J., (2009) Life Cycle Assessment of the use of compost from municipal organic waste for fertilization of tomato crops. *Resources Conservation & Recycling*, 53 (6), 340-351.
- Martínez-Blanco, J., Colón, J., Gabarrell, X., Font, X., Sánchez, A., Artola, A., & Rieradevall, J. (2010) The use of life cycle assessment for the comparison of bio-waste composting at home and full scale. *Waste Management*, 20, 983-994.
- Martínez-Blanco, J., Anton, A., Rieradevall, J., Castellari, M., & Munoz, P. (2011) Comparing nutritional value and yield as functional units in the environmental assessment of horticultural production with organic or mineral fertilization: The case of Mediterranean cauliflower production. *International Journal of Life Cycle Assessment*, 16, 12-26.
- Martínez-Blanco, J., Muñoz, P., Antón, A., & Rieradevall, R. (2011) Assessment of tomato Mediterranean production in open-field and in standard multi-tunnel greenhouse with compost or mineral fertilizers from an agricultural and environmental standpoint. *Journal of Cleaner Production*, 19, 985-997.
- Martínez-Blanco, J., Muñoz, P., Rieradevall, J., Montero, J.I., & Antón, A. (2011) Regional assessment of waste flow eco-synergy in food production: using compost and polluted ground water in Mediterranean horticulture crops. In: Matthias Finkbeiner (ed.) *Towards life cycle sustainable management*. Springer, New York, 318-330. ISBN-978-94-007-1898-2.
- Martínez-Blanco, J., Rieradevall, J., Muñoz, P., Pascual, P., Oliver-Solà, J., Gabarrell, X., & Gasol, C.M. (2012) Carbon Footprinting and Life Cycle Assessment for greenhouse gas impact quantification in horticulture. Under publication in the *International Journal of Life Cycle Assessment*.
- Martínez-Blanco, J., Lazcano, C., Boldrin, A., Muñoz, P., Rieradevall, J., Möller, J., Antón, A., & Christensen, T.H. Assessing the environmental benefits of compost use-on-land through an LCA perspective: a review. Accepted in May 2012 for publication in: Eric Lichtfouse (ed.) *Sustainable Agriculture Reviews*, Volume 11. ISSN-2210-4410.

- Martínez-Blanco, J., Lazcano, C., Boldrin, A., Muñoz, P., Rieradevall, J., Möller, J., Antón, A., & Christensen, T.H. State of the art and future challenges for quantification of compost use-on-land through an LCA perspective. Accepted in May 2012 for publication in the international journal *Agronomie for Sustainable Development*.
- Martínez-Blanco, J., Lehmann, A., Muñoz, P., Antón, A., Traverso, M., Rieradevall, R., & Finkbeiner, M. Life Cycle Sustainability Assessment of compost and mineral fertilizers production for agriculture - application challenges for social LCA. Accepted with major revisions in May 2012 for publication in the Special Issue on Life Cycle Sustainability Assessment in the *International Journal of Life Cycle Assessment*.

In addition, during the dissertation period the opportunity has been given to work in other papers, which were published in peer-reviewed journals and are also related with the goals of the dissertation:

- Colón, J., Martínez-Blanco, J., Gabarrell, X., Rieradevall, J., Font, X., Artola, A., & Sánchez, A. (2009) Performance of an industrial biofilter from a composting plant in the removal of ammonia and VOC after medium replacement. *Journal of Chemical Technology & Biotechnology* 84(8), 1111-1117.
- Ruggieri, L., Cadena, E., Martínez-Blanco, J., Martínez, C.M., Rieradevall, J., Gabarrell, X., Gea, T., Sort, X., & Sánchez, A. (2010). Recovery of organic wastes in the wine industry. Technical, economic and environmental analysis of wastewater sludge and stalk composting. *Journal of cleaner production* 17, 830-838.
- Colón, J., Martínez-Blanco, J., Gabarrell, X., Artola, A., Sánchez, A., Rieradevall, J., & Font, X. (2010) Environmental assessment of home composting. *Resources Conservation & Recycling* 54, 893-904.

The following oral communications and posters presented to congresses and conferences also form part of this doctoral thesis:

- Nuñez, M., Martínez-Blanco, J., Muñoz, P., Antón, A., & Rieradevall, J. (2008) Estudios preliminares de evaluación de impacto ambiental global en la aplicación de compost como fertilizante en cultivos de tomate al aire libre y en invernadero. Oral communication. *I Jornadas de la Red Española de Compostaje*, 6-9 February 2008, Barcelona (Spain).
- Martínez-Blanco, J., Muñoz, P., Antón, A., & Rieradevall, J. (2008) LCA of the application of compost from organic municipal solid waste in culture fertilization. Poster. *6th International Conference on Life Cycle Assessment in the Agri-Food Sector*, 12-14 November 2008, Zurich (Switzerland).
- Martínez-Blanco, J., Colón, J., Gabarrell, X., Font, X., Artola, A., Rieradevall, J., & Sánchez, A. (2009) Emissions and environmental assessment of the home composting of two domestic wastes. Poster. CILCA 2009, *III International Conference of LCA in Latin America*, 27-29 April 2009, Pucón (Chile).
- Martínez-Blanco, J. (2010). ACV de l'aplicació del compost. Oral communication. *Zero Waste Workshop* Barcelona, 10 February 2010, Barcelona (Spain).
- Martínez-Blanco, J., Muñoz, P., Antón, A., Castellari, M., & Rieradevall, J. (2010) Integrated assessment of the nutritional and environmental performance of horticulture in Mediterranean area with compost or mineral fertilizers. Poster. *7th*

*International Conference on Life Cycle Assessment in the Agri-Food Sector*, 22-24 September 2010, Bari (Italy).

- Pascual, P., Gasol, C.M., Martínez-Blanco, J., Oliver-Solà, J., Muñoz, P., Rieradevall, J., & Gabarrell, X. (2010) Calculation of $CO_2$ equivalent emissions in agri-food sector applying different methodologies. Oral communication. *7th International Conference on Life Cycle Assessment in the Agri-Food Sector*, 22-24 September 2010, Bari (Italy).
- Martínez-Blanco, J., Rives, J., Colón, J., Muñoz, P., Antón, A., Artola, A., Font, X., Sánchez, A., Gabarrell, G., & Rieradevall, J. (2010) Agronomic and Environmental Viability of Compost Application in Mediterranean horticultural Crops. Poster. *International Conference on EcoBalance 2010. Towards & Beyond 2020*, 9-12 November 2010, Tokyo (Japan).
- Martínez-Blanco, J., Antón, A., Muñoz, P., Ceron, I., & Rieradevall, J. (2011) The use of Carbon footprint in horticulture: Environmental performance of two horticultural products in the Mediterranean region. Oral communication. *CILCA 2011, VI International Life Cycle Assessment Conference in Latin America*, 27-29 April 2011, Veracruz (Mexic).
- Martínez-Blanco, J., Antón, A., Muñoz, P., & Rieradevall, J. (2011) Huella de carbono y productos hortícolas del Maresme. Oral communication. *Jornadas técnicas sobre la Huella de carbono e impacto en el sector frutícola* organizadas por Departament d'Agricultura, Ramaderia Pesca, Aliemntació i Medi natural, 29 September 2011, Lleida (Spain).

The doctoral thesis is based on the methodology and results developed under the framework of the following projects:

- *Aplicación de Compost de Fracción Orgánica de Residuos Sólidos Municipales en la fertilización de cultivos hortícolas en la comarca del Maresme* (461/2006/3-2.3 and A246/2007/2-02.3). 2006-2009. A Project funded by the Dirección General de Calidad y Evaluación Ambiental, part of the Spanish Ministerio de Medio Ambiente Medio Rural y Marino. The Project was coordinated by the Institute of Agriculture and Food Research and Technology (IRTA) and had the participation of the Universitat Autònoma de Barcelona (ICTA - Institute of Environmental Science and Technology), Universitat Politècnica de Catalunya (ESAB - Escola Superior d'Agricultura de Barcelona) and the Federació Selmar.
- *Avaluació de les emissions gasoses d'un compostador casolà i els impactes ambientals associates*. 2008. A project funded by the Diputació de Barcelona. The project had the participation of the ICTA and the Grup d'Investigació en Compostatge (GICOM) from the Department of Chemical Engineering, both groups at the Universitat Autònoma de Barcelona.

As well as, during the following research stays:

- Two-month stay (October – December 2010) at the Department of Environmental Engineering of the *Danmarks Tekniske Universitet* (DTU), in Lyngby, Denmark. The hosting researcher was the Professor Thomas H. Christensen.
- Two-month stay (October – December 2011) at the Chair of Sustainable Engineering of the *Technische Universität Berlin (TUB)*, in Berlin, Germany. The hosting researcher was the Professor Matthias Finkbeiner.

# Structure of the dissertation

The structure of the dissertation is organised into five main parts and twelve chapters. For clarity, the structure of the doctoral thesis is further outlined in Figure 0. This flow chart can be used throughout the reading of this manuscript as a *dissertation map*.

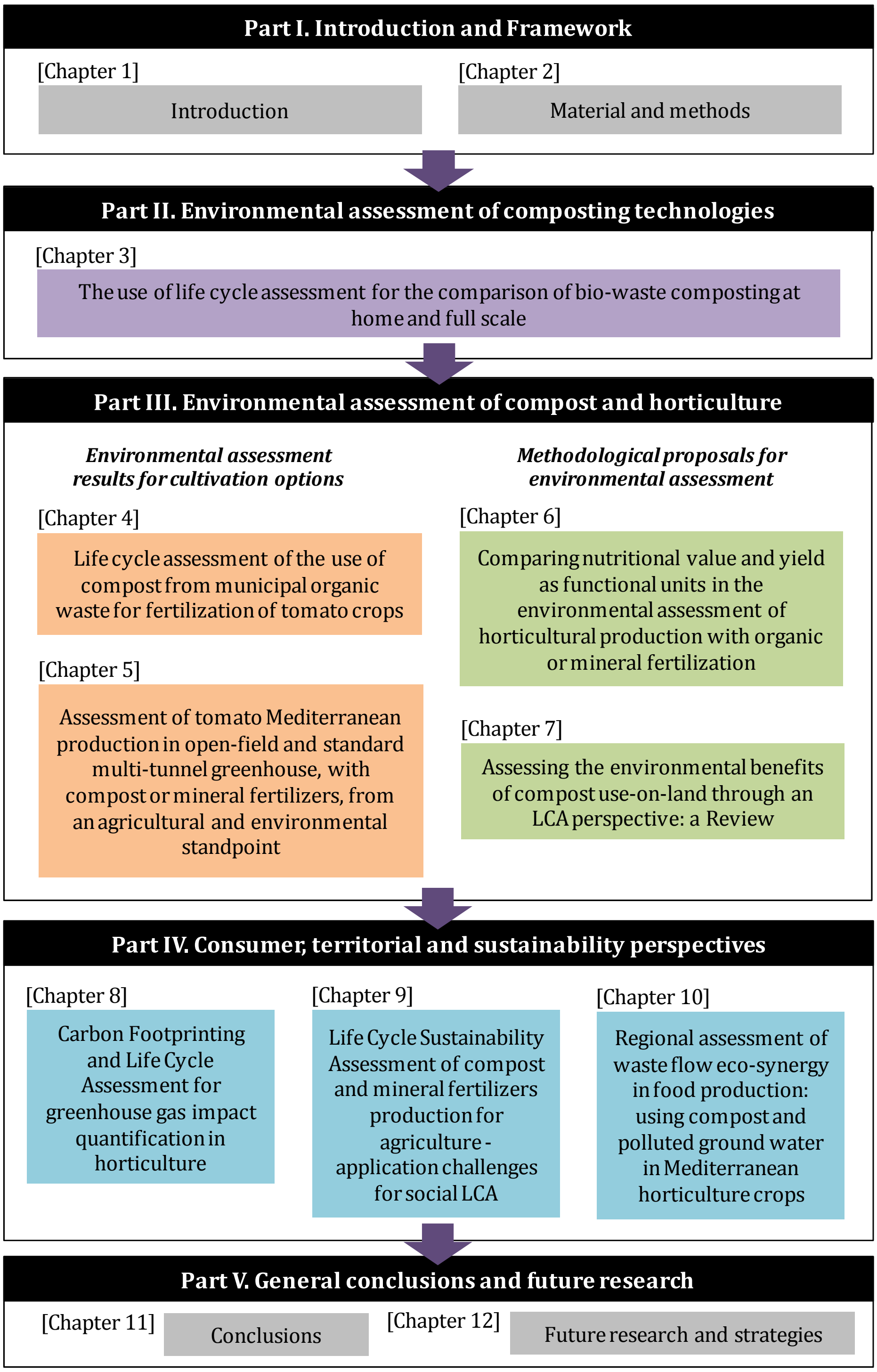



**Figure 0. Map structure of the dissertation.**

## Part I. Introduction and framework

**Part I** is composed of two chapters. **Chapter 1** *[Introduction]* presents an overview of the topic of sustainability of agriculture and soils, focusing on the increasing demand of sustainable products, the current situation of fertilization, and the shortage of nutrients in soils. Furthermore, it sets out an introduction to municipal bio-waste current generation and management, including the environmental implications of waste management options, as well as the main European and Catalan legislation trends. Subsequently, the introduction focus on composting process and available technologies and then potential compost applications and effects in soils are presented. The fourth section outlines the particularities of life cycle approach when applied to waste management and agricultural systems. Finally, the motivation of this dissertation is presented, and the objectives are enumerated. **Chapter 2** *[Material and methods]* describes the methodology applied, which is divided into sustainability assessment tools and field and laboratory works, plus systems of study considered throughout the research.

## Part II. Environmental assessment of composting technologies

**Part II** is composed of one only **Chapter 3** [*The use of life cycle assessment for the comparison of bio-waste composting at home and full scale*]. This chapter presents inventory data and the environmental assessment of two composting technologies, home composting and industrial composting in-vessel. Real data was used throughout the chapter and gaseous emissions and quality parameter of the composts obtained were measured for both case studies.

## Part III. Environmental assessment of compost and horticulture

**Part III** is including four chapters dealing with two main issues: the environmental assessment of several cultivation options and the methodological proposals for carrying on the environmental assessment of compost application.

### *Environmental assessment results for cultivation options*

**Chapter 4** [*Life Cycle Assessment of the use of compost from municipal organic waste for fertilization of tomato crops*] presents a broad description of the inventory data for the production of tomato in open field using three different fertilization options, combining compost and mineral fertilizers and including the entire life cycle. Furthermore, the environmental impacts and the agricultural performance are assessed for the three options. **Chapter 5** [*Assessment of tomato Mediterranean production in open-field and standard multi-tunnel greenhouse, with compost or mineral fertilizers, from an agricultural and environmental standpoint*], apart from fertilizing alternatives, compares open field and greenhouse production from agricultural and environmental perspectives. Additionally, the chapter sets out a detailed explanation of the Mediterranean greenhouse and the main contributions to its environmental impacts.

### *Methodological proposals for environmental assessment*

In **Chapter 6** [*Comparing nutritional value and yield as functional units in the environmental assessment of horticultural production with organic or mineral fertilization: The case of Mediterranean cauliflower production*] yield and the quality of the harvest are stated for three options of fertilization, as well as the bioactive compounds content. Subsequently, in order to face the common practice in agricultural Life Cycle

Assessment (LCA) of using tons produced, several functional units linked with either the quantity produced or with the quality of the product are used here for the quantification of the environmental impacts and then the results for the several functional unit options are compared among them. **Chapter 7** [*Assessing the environmental benefits of compost use-on-land through an LCA perspective: a review*] deals with nine of the environmental benefits of compost application typically reflected in the scientific literature but often not or never included in the environmental assessments. These benefits are quantified through literature revision and the existing impact methodologies that are related to them are presented. Finally, the future challenges for assessing the benefits under a LCA perspective are discussed.

## Part IV. Consumer, territorial and sustainability perspectives

The three chapters of **Part IV** are attending to other perspectives of compost production and use, apart from the environmental one. **Chapter 8** [*Carbon Footprinting and Life Cycle Assessment for greenhouse gas impact quantification in horticulture*] faces the consumer perspective, with regard of labelling products with information on environmental performance. In this chapter, the increasingly used Carbon Footprinting methodology is applied to several horticultural case studies and their results compared with LCA ones, with special focus on the effects of capital goods and carbon sequestration. **Chapter 9** [*Life Cycle Sustainability Assessment of compost and mineral fertilizers production for agriculture - application challenges for social LCA*] overviews the social, economic and environmental impacts related to mineral fertilizers and compost, from a life cycle perspective and considering a real case study. The chapter deeply explains the approach to perform the Social Life Cycle Assessment, including the country, sector and company levels of assessment. Additionally, it integrates the three results by means of the dashboard of sustainability. **Chapter 10** [*Regional assessment of waste flow eco-synergy in food production: using compost and polluted ground water in Mediterranean horticulture crops*] points out some ideas about territorial planning aspects regarding compost use in horticulture. Here, the potential eco-synergetic effects of using compost complemented by nitrate polluted water for the substitution of mineral fertilizers are assessed, using macro-data at county level for the calculations and Geographic Information System, for the illustrations.

## Part V. General conclusions and future research

**Part V** includes **Chapter 11** [*Conclusions*] and **Chapter 12** [*Future research and strategies*] and provides the general conclusions of the dissertation and proposes future fields of research associated with the cork sector.

[*Note:* Each chapter from 3 to 10 presents an article or book chapter –either published or under review. For this reason, an abstract and a list of keywords are presented at the beginning of the chapter, followed by the main body of the article].

# Part I

## Introduction and framework

# Chapter 1. Introduction

**Chapter 1** introduces some relevant circumstances influencing the evolution of sustainable agriculture. Furthermore, the current situation and the intergovernmental efforts on efficient bio-waste management are briefly presented. Then, it presents the framework of compost production and applications and a succinct outline of life cycle approach when it is applied to waste management and agriculture sectors, which are the cornerstones of this dissertation. Finally, it presents the justification of the dissertation and enumerates its objectives. Because the case studies included in the dissertation and explained in Chapter 2 are located in Catalonia, the Chapter 1 is focused on European, Catalan, and in some cases Spanish information.

This chapter is structured as follows:

- Sustainability of agriculture and soils.
- Municipal bio-waste generation and management in Europe.
- Compost production and application.
- Outline of life cycle approach in the assessed sectors.
- Motivation of the dissertation.
- Objectives of the dissertation.

# 1.1. Sustainability of agriculture and soils

In the last century, production techniques in the agri-food sector, and particularly horticulture, have intensified. This trend is a response to the growing demand of an increasing population, technological changes, year-round production, and human diet modifications, as well as due to economic interests (Smith et al. 2001; Heinberg and Bomford 2009).

The loss of traditional farming practices to spread intensive agriculture has led to many environmental problems, of which the European Environment Agency (EEA 2012) highlights soil erosion, water pollution, over-exploitation of water resources, loss of biodiversity, pesticide-born damage and risk for human health. Consequently, food production has become a relevant contributor to the depletion of natural resources, acidification, eutrophication and climate change impacts (Nonhebel 2004; Tukker et al. 2005). It is, therefore, of major interest to quantify the environmental behavior of products, processes and services.

The global anthropogenic GHG emissions were 24% higher in 2004 compared to 1990 and 70% higher than in 1970. Smith (2007) reported that agriculture currently accounts for 10-12% of the total global anthropogenic greenhouse gas (GHG) emissions generated worldwide, it is very close to the 13.5% considered in the IPCC (2007). There are GHG emissions not only in the agricultural and farming stages of the life cycle of a food product, but also during food packaging, transport, marketing and consumption (Kramer et al. 1999; Heinberg and Bomford 2009; Roy et al. 2009).

The adverse impacts of inappropriate agricultural land management affect the capacity of the land to function effectively within an ecosystem (Núñez 2011). Soil degradation affects around 30% of the world’s irrigated lands and 40% of rainfed agricultural lands (Watson et al. 1998).

## 1.1.1. Increasing demand of more sustainable agriculture

Facing the impacting potentiality of agriculture, most developed countries have been tracking the environmental performance of agriculture, which is informing policy makers and society on the state and trends in agri-environmental conditions, and can provide a valuable aid to policy analysis.

Due to the increasing consumer demand for more environmentally friendly and healthy food products (Antón 2004; Podsedek 2007; Montero et al. 2009b; Romero-Gámez et al. 2009; Lairon 2010), real alternatives to current intensive production methods need to be supported by scientific research (Blengini 2008; Schau and Fet 2008).

The increasing demand of this type of products is pointed, for example, by the raise on the area under organic farming in the EU-27, that increased by 57% between 2002 and 2009 (Eurostat 2012), and also by the emergence of specific eco-friendly and healthy initiatives, labels and marketing.

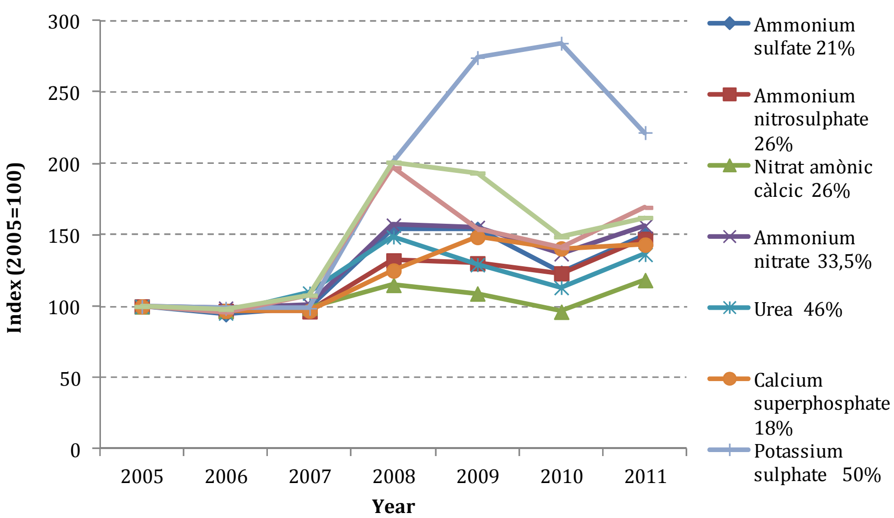
Agricultural production systems include organic production systems, integrated production systems, and conventional production systems. To sum up, conventional farming means the traditional, most common system of production; in organic farming the use of synthetic chemicals (both fertilisers and pesticides) is virtually excluded (EC2007; 2008a) and integrated farming, where the use of chemicals is limited although not strictly forbidden and production systems with a low environmental impact are employed (EISA 2010).

Slow food initiative is an example of the increasing concern for the quality, the security and the consequences of the food that is being consumed. It is an international grassroots membership organization promoting good, clean and fair food for all (Slow Food 2012). Under the Slow Food framework, it was created the Guide of KM0 Restaurants, which are using local or regional products and looking for a direct trade with the farmer. Moreover, many labels have been created the last years by many institutions in order to differentiate organic products, to inform the consumer about the ameliorated healthy quality of products or their Carbon Footprinting.

### 1.1.2. Some concerns related to conventional fertilization

As was aforementioned, agricultural expansion and intensification are needed to meet the increasing global demand of food. Agricultural intensification involves increased fertilization; in most cases there is a large response to nitrogen fertilization measured as crop yield. As the cost of fertilizers is often small compared to the cost of lost yield, farmers prefer over-fertilization of crops with nitrogen rather than risking under-fertilization and consequent loss of revenue (Del Amor 2007). However, excess nitrogen may result in lodging, greater weed competition and pest attacks, with substantial losses of production.

Furthermore, the nitrogen not taken up by the crop is likely to be lost to the environment, which potentially contributes to groundwater and atmospheric pollution (Weinbaum et al. 1992; FAO 1998; Roy et al. 2006).




*The corresponding Producer Prices Index was disccounted from the average price of each year.*

**Figure 1.1. Evolution of the index of prices of fertilizers in Catalonia 2005-2011.**

*Source: Own elaboration from DARPAMN* (2012) *and Idescat* (2012).

Besides, due to several reasons – such as, oil prices increase, the spread of biofuels production, and enhance on fertilizers consumption of China and India –, a continuous rise in fertilizer prices was observed the last years and mineral fertilization costs are not so small nowadays. For instance, fertilizer prices have increased 91% on average between 2005 and 2011 in Catalonia (Figure 1.1). The marked increase on the prices would favour the use of other types of fertilization, such as organic fertilizers.

### 1.1.3. Nutrients and organic matter shortage in soils

There is increasing concern about soil interrelated environmental problems such as soil degradation, desertification, erosion, and loss of fertility (European Commission 2006c). These problems are partially consequence of the decline in organic matter content in soils. A level of 2% of soil organic carbon (SOC) is commonly considered desirable for maintaining good soil structure for agricultural activities (Van-Camp et al. 2004).

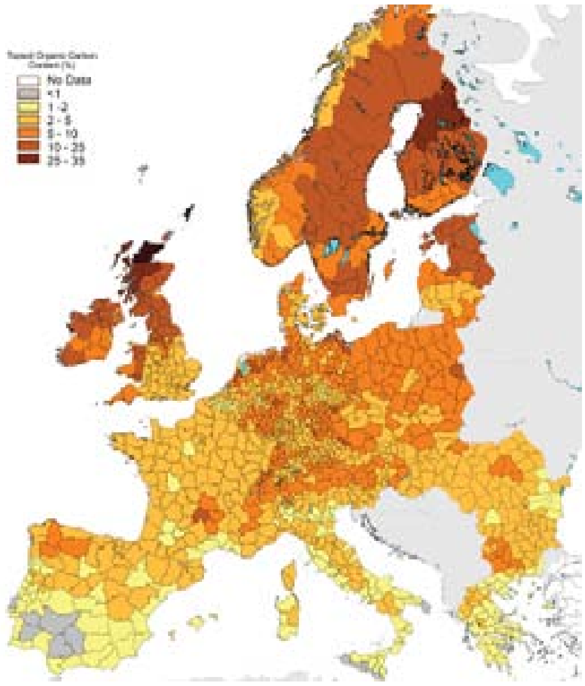



**Figure 1.2. Organic carbon content in European soils.**
*Source: European Soil Bureau Network* (2012).

An estimated 45% of European soils have low (<2%) soil organic matter content, as it is shown in Figure 1.2 principally in southern Europe, i.e. in the Mediterranean regions, but also in areas of France, the UK, and Germany and Sweden (European Commission 2006c).

The 50% of agricultural land and pastures in Spain have less than 1.7% of organic matter in soil (1% C). Therefore, in accordance with the *Thematic Strategy for Soil Protection* (European Commission 2006a), there is a real risk of desertification by 50% of agricultural land and pastures in Spain. The European Commission adopted the *Soil Thematic Strategy* (European Commission 2006a) and a proposal for a *Soil Framework Directive* (European Commission 2006b) with the objective to protect soils across the EU. The draft *Soil Framework Directive* imposes the obligation for member States to design programmes of measures to combat organic matter decline. The resolution also required "the Commission shall present a proposal for a bio-waste Directive setting quality standard for the use of bio-waste as a soil improver" (European Commission 2010).

### 1.1.4. Current situation of horticultural production

The agricultural case studies considered in the dissertation are all referred to horticultural crops. Fruit and vegetables are generally more profitable than cereals and arable crops. Fresh vegetable production accounted for 17% of the EU-27 overall crop output value at producer prices of 2009 but only 2.1% of the total cultivated area (Eurostat 2011; 2012). The annual horticultural production in Spain exceeded the figure of five thousand millions Euros, nearly 1% of the Spanish GDP, during the period 1995-2007 (MMAMRM 2010; INE 2012).

Figure 1.3 shows the share of horticultural cultivated area devoted to each crop in Europe, Spain and Catalonia. In the EU-15 tomato, lettuce, cauliflower and broccoli, and onions account for nearly 40% of the total area. Apart from these four crops, melon and artichokes production are also being a relevant share of the total area for Spain and Catalonia, respectively.

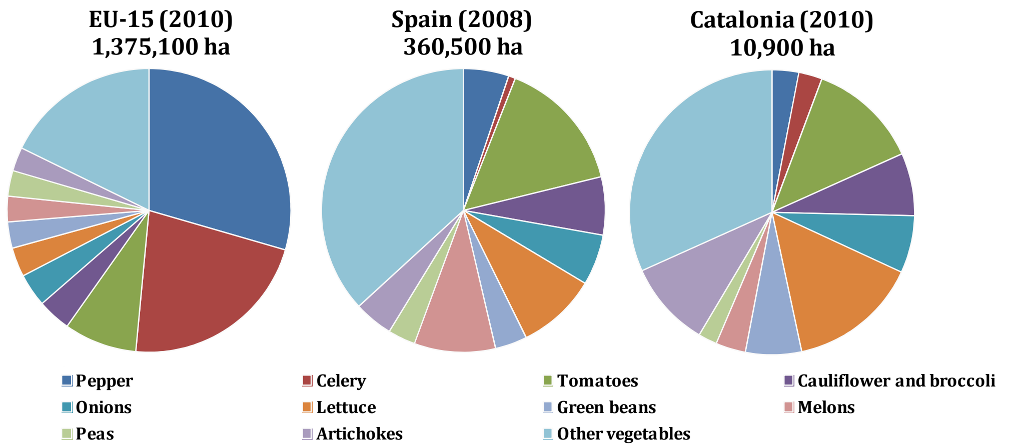



**Figure 1.3. Main vegetable products in European Union (EU-15), Spain and Catalonia according to the cultivated area.**

*Source: Own elaboration from MMAMRM* (MMAMRM 2010)*, DARPAMN* (DARPAMN 2012)*, and Eurostat* (2012).

Most vegetable crops are relatively concentrated in areas with a mild winter climate where typical weather is generally more favorable (Montero et al. 2009b), and even more under greenhouse protection. According to Eurostat (2012), Spain and Italy are the main producers from the EU, with a total vegetable production of about 12 millions of tonnes each one, additionally France and Greece are also among the major producers.

Currently, the total greenhouse protected area in the southern European Union member states is about 90,000 ha, with Spain at the top of the list with more than 80,000 ha (Sigrimis et al. 2009). In Spain, protected vegetable production represents 36% of the total vegetable production (MMAMRM 2010).

In Catalonia, greenhouses type Mediterranean are used; as described by Antón (2004), Mediterranean greenhouses is common on temperate regions, characterized by a plastic cover and usually without heating, or just for exceptional situations. It is called Mediterranean to differentiate it from the typical greenhouses from colder European regions. The latter often uses glass as a cover material and is heated and lighted. The most often used types of Mediterranean compost are multi-tunnel, parral, and wooden greenhouse.

(a) Mediterranean Multi-tunnel

(b) Northern Europe Venlo

**Figure 1.4. Mediterranean and Northern Europe greenhouses.**
*Source: Montero et al.* (2009a).

To sum up, the shortage of organic matter on soils, mainly on Southern Europe, as well as the increasing price of fertilizers and the growing environmental concern by authorities and consumers, involves that farmer would increase their interest in compost as a potential environmentally friendly way of fertilization.

# 1.2. Municipal bio-waste generation and management in Europe

Waste includes all the items that people no longer have any use for, which they either intend to get rid of or they have already discarded. From extraction, production and distribution, and final consumption of goods and services, as well as during waste collection and treatment (e.g. sorting residues in recycling facilities, incinerator slag); all human activities are potential sources of waste.

## 1.2.1. Waste and environment protection. Focus on bio-waste and composting

The massive growth of industrial activities, population and urban planning has lead to an increase in waste generation. At present, it is becoming a global problem in developed countries due to the rapid collapse of landfills and the high impacts related to bio-waste dumping. In view of these problems, society is being increasingly conscience of the importance of reducing the impacts of human activities. This is reflected in current legislation, about the importance of achieving adequate waste management and treatment in order to ensure the protection of human health and the prevention of environmental impacts (Eriksson et al. 2005).

### *(i) Environmental impacts of waste management options*

The nature and dimension of waste impacts on the environment depend upon the amount and composition of waste streams as well as on the method adopted for treating them. Improper management of waste has already caused numerous cases of contamination of soil and groundwater, threatening the natural functioning of ecosystems and the health of the exposed population. The generation of waste represents also an inefficient use of valuable resources (Eurostat 2010).

According to the EU greenhouse gas inventory reports (EEA 2011), the waste sector contributes around 3% to all GHG emissions in the EU-27. Since 1990, it has reduced its emissions by 31%, mainly via reduced methane emissions from landfills (EEA 2011). Nevertheless, these accounts are not including emissions from recycling, waste transport, and waste incineration with energy recovery, which are reported under other IPCC sectors.

GHG emissions from landfills are mainly attributed to methane generated by anaerobic degradation of biodegradable wastes – i.e. putrescible materials as well as paper and cardboards, textile and fibres materials (Manfredi et al. 2009). On the contrary, recycling of resources serves both in reducing the consumption of virgin supplies and in preventing the discharge of associated residuals back into the natural environment. However, even if the potential to recycle material from waste is high, it may not be appropriate in all cases (Georgakellos 2006; JRC and IES 2011b). One of the reasons is that, in general, the additional energy consumed mainly during transportation in the recycling steps (and the subsequent relevant emissions) may be significant when compared to the overall energy consumption of the system (Iriarte et al. 2009; Rives et al. 2010), apart from the airborne emissions, when biodegradable wastes are involved.

Generally, composting has been considered to reduce some environmental impacts. First of all, it potentially avoids methane production from degradation of organic waste in landfills (as degradation is aerobic). Secondly, improvements in soil fertility and soil organic matter content leading to possible downstream benefits from reduced need for inorganic fertilisers and peat, reduced need for irrigation and lower soil erosion rates (see section 1.3.3). Furthermore, compost application could lead to carbon sequestration through increasing the store of soil organic matter (Smith et al. 2001). However, it needs careful control of the composting process to avoid bioaerosols and gaseous emissions (Smith et al. 2001; Cadena 2009; Andersen et al. 2010; Colón et al. 2012).

### *(ii) Waste legislation affecting bio-waste*

The treatment and disposal of waste is one of the central topics of sustainable development.

**Main European legislation**

The *Landfill Directive*, Council Directive 1999/31/EC (EC1999), with the aim to reduce the emission of greenhouse gases from landfills, requires Member States to progressively reduce landfilling of biodegradable waste to 35% of the total municipal waste produced by 2016 (compared to 1995). Member States that in 1995 relied heavily on landfilling for biodegradable waste management are given a 4-year extension period (i.e. until 2020).

The *Waste Framework Directive*, Council Directive 2008/98/EC (EC2008b), proposes a hierarchical system based on four subsequent levels – prevention recycling, energy recovery and, the last option, disposal – for waste management (Figure 1.5).

The *Directive on Renewable Energy Sources*, Council Directive 2009/28/EC (EC2009), and the Working document on *Directive of biological treatment of bio-waste* (European Commission2001) are also relevant for European bio-waste management.

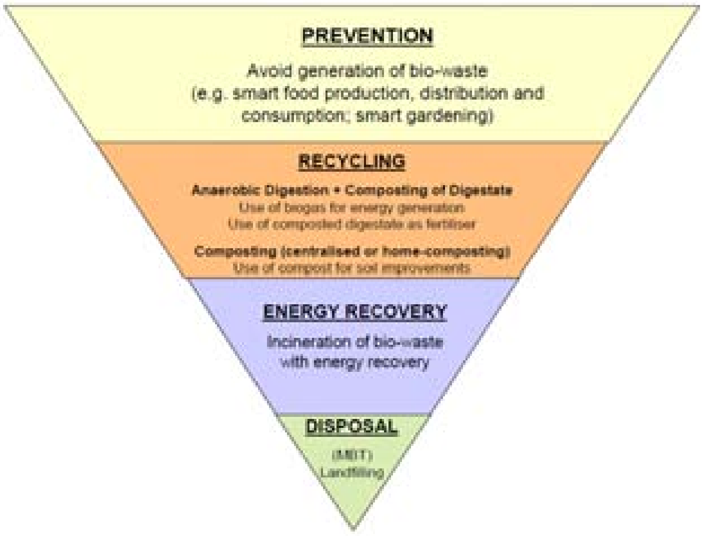



**Figure 1.5. Waste hierarchy applied to bio-waste.**
*Source: JRC and IES (2011b).*

**Main Spanish legislation**

The Council Directive 2008/98/EC (EC2008b) was transposed into Spanish Law 22/2011 of waste and polluted soil (Jefatura del Estado2011) and the PNIR, Integrated National Plan of Waste 2008-2015 (MMA 2008). Moreover, the Council Directive 1999/31/EC (EC1999) was transposed by the Spanish Royal Decree 1481/2001 (MMA2001), which was later modified by the Royal Decree 367/2010 (Ministerio de la Presidencia2010).

**Main Catalan legislation**

According to the article 149.1.23 of the Spanish Constitution, Catalan Autonomous Community has competences to legislate on environmental issues and propose additional regulations, always fulfillling Spanish and European laws. Because, Catalonia has a more restrictive legislation and is responsible for managing the municipal waste generated throughout Catalonia, the introduction chapter is focusing mainly on Catalan legislation and data.

In Catalonia there are two main legislative instruments that specify the waste framework model that must operate in Catalonia, in accordance with European regulations and the current socioeconomic circumstances. Catalan Low 9/2008 (Departament de la Presidència2008b) is the reference low for waste regulation, whereas Catalan Low 8/2008 (Departament de la Presidència2008a), deals with financing infrastructure and waste management fees on the disposal of waste.

### 1.2.2. Municipal bio-waste and management options

Definitions of municipal solid waste vary from country to country. Municipal waste is defined by Article 2 (b) of the Landfill Directive (EC1999) as *the waste from households, as well as other waste, which because of its nature and composition is similar to waste from households*. This is compatible with the IPCC definition, which includes household waste, yard/garden waste and commercial/market waste. Municipal solid waste is including plastic and metals from packaging, glass, paper and cardboard, and organic waste. Furthermore many other minor fractions are found, such as used oils, fabrics and fibres, garden wastes and wood, sanitary wastes, bulky waste, etc.

### *(i) Bio-waste definitions*

**Biodegradable municipal waste**

There is no specific definition provided for biodegradable municipal waste in the Landfill Directive (EC1999), however, combining its biodegradable waste and municipal waste definitions, they provide the following definition: *Biodegradable municipal waste means biodegradable waste from households, as well as other biodegradable waste, which because of its nature and composition is similar to biodegradable waste from households* (EC1999). It is including bio-waste as well as paper and cardboard and textiles.

**Bio-waste**

Waste Framework Directive (EC2008b) defines bio-waste as biodegradable garden and park waste, food and kitchen waste from households, restaurants, caterers and retail premises and comparable waste from food processing plants. The concept of bio-waste used in the dissertation is according to the latter definition, as paper and cardboard are not included. This definition is also in accordance with bio-waste definition from Catalan Law 15/2003 (Departament de la Presidència2003). Figure 1.6 shows the potential sources of bio-waste.

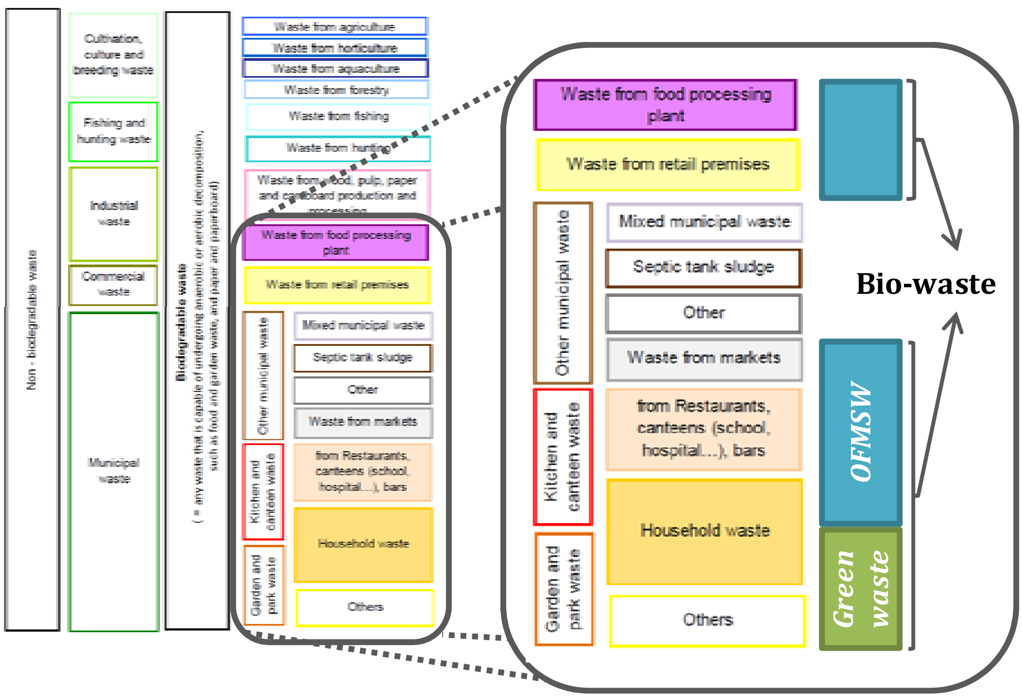



**Figure 1.6. Potential sources of biodegradable waste and bio-waste.**
*Source: Adapted from JRC and IES* (2011b).

Bio-waste includes the organic fraction of the municipal solid waste (OFMSW) and the garden and park (or green) waste. On the contrary, it does not include forestry or agricultural residue and, thus, should not be confused with the wider term "biodegradable waste" (EC1999), which also includes wood, paper, cardboard, sewage sludge, natural textiles.

**OFMSW and green waste**

The materials recommended by Agència Catalana de Residus (ARC 2012) to be included in the organic fraction of the municipal solid waste (OFMSW), i.e. the food and kitchen waste collected from households, are: peel and rejected parts of fruit and vegetables, fish and meat bones and waste, shellfish and mollusc shells, leftover bread, coffee grounds and tea waste, dirty kitchen paper and paper napkins, dry flowers and leaves, fallen leaves, corks, and sawdust.

Nevertheless, the specific organic materials to include in domestic composting that are recommended by local authorities are sometimes differing from the aforementioned ones – e.g. there is controversy about including fish and meat leftovers due to flies, rodents and odors problems.

Green waste is including biodegradable organic waste of gardens and parks origin. It can be sub-divided into two specific streams: small-sized, non-woody plant fraction (grass cuttings, fallen leaves, bunches of flowers, etc.) which can be assimilated into the OFMSW; and pruning waste, which is the large-sized, woody plant fraction requiring grinding prior to recovery (ARC 2012).

Typical characteristics of the two parts of the bio-waste according to ARC (2012) are briefly summarized in Table 1.1.

**Table 1.1. Basic characteristics of OFMSW and green waste in Catalonia.**

| | **OFMSW** | **Green waste** |
|---|---|---|
| **Moisture** | High (75 - 85%) | Low (20 - 40%) |
| **Organic matter** | 75 - 85% | 80% |
| **Organic nitrogen** | 2.5% | 1.2% |
| **C/N** | 17 | 32 |
| **Density** | 0,6 a 0,8 T $m^{-3}$ | 0,3 a 0,4 T $m^{-3}$ |
| **Odors** | yes | no |
| **Generation** | Constant | Seasonal |

*Source: ARC* (2012).

Apart from bio-waste, other biodegradable wastes are produced thorough human activities – e.g. cattle manure, sludge from waste water treatment plants, and animal by-products not intended for human consumption – and in some cases they could be processed with the same waste management options as bio-waste.

### *(ii) Management options*

Several treatment options are available for diverting biodegradable wastes from landfilling: composting, anaerobic digestion and gasification for source separated organic waste, and mechanical biological treatment (MBT), incineration and pyrolisis for all wastes. Regardless the waste management option, all the management systems have in common the need for collection, sorting, processing and transport from the source of the waste to the waste treatment/disposal facilities and end markets for recovered materials, if any (Smith et al. 2001).

Apart from the waste management hierarchy proposed by the *Waste Framework Directive* (EC2008b), local considerations should be taken into account when choosing treatment optimal options, such as: infrastructural, economic, legal and environmental conditions; local availability of green waste; and demands for fertilisers, soil conditioners and/or energy (Crowe et al. 2002; Andersen 2010; JRC and IES 2011b).

The Thematic Strategy on the Prevention and Recycling of Waste added that management for this type of waste should be determined by the Member States using life-cycle thinking (European Commission 2005).

### 1.2.3. Catalan and European bio-waste generation and management

#### *(i) Current situation in Europe*

As a result of pressure from European Union legislation (EC1999), the overall amount of municipal solid waste disposed in landfills has gradually decreased in Europe from 295 kg per capita in 1995 to 186 kg per capita in 2010 (Eurostat 2012). Regarding the total amount of MSW generated per capita, it has increased since 1995, from 474 kg to 502 kg in 2010. In 2010, MSW accounted for 8% of all waste generated in EU-27 plus Croatia, Iceland, Norway and Turkey (Eurostat 2012).

**Bio-waste generation**

At the European Union level, a total of 76.5-102 Mt year$^{-1}$ of bio-waste is generated, which is the largest fraction of the total annual municipal waste generated and represents 30-40% of the total (European Commission 2008). Accordingly, Table 1.2 shows the European average composition of MSW. The potential of compost production from bio-waste is estimated at 35 to 40 Mt (European Commission 2008).

However, according to Eurostat (2010), there are large differences between Member States municipal waste, and particularly bio-waste, production and management, due to differences in population, urbanisation and affluence. Furthermore, as already noted above, this type of information tends to be compromised by the variance in definition of waste.

**Table 1.2.Estimate average composition of MSW (%) in several regions.**

| Material flow | Europe (2007) | Spain (2007) | Catalonia (2006) |
|---|---|---|---|
| **Bio-waste** | 39 | 44 | 36 |
| **Paper and cardboard** | 25 | 21 | 18 |
| **Plastic** | 7 | 11 | 12 |
| **Metals** | 5 | 4 | |
| **Glass** | 8 | 7 | 7 |
| **Textiles** | 1 | - | 4 |
| **Others** | 15 | 13 | 23 |

*Source: ARC* (2007), *MMA* (2008), *and ACR+* (2012).

**Bio-waste treatment**

There are large differences between Member States in MSW management. From Eurostat (2010) it can be stated than Northern countries are heavily relying on incineration accompanied by a high level of material recovery and often biological treatments; while Southern countries have relatively high material recovery rates but low incineration.

According to the calculations from European Commission (2010), 23% of the generated bio-waste in the EU-27 is currently composted, and nearly 40% is going to some sort of biological treatment (Figure 1.7).

It is expected that the total amount of bio-waste will increase by an average of 10% by 2020 (European Commission 2010). More waste generated, better collection schemes and less waste to landfills will result in larger amounts of organic waste that needs to be treated biologically (e.g. by composting) in the foreseeable future.

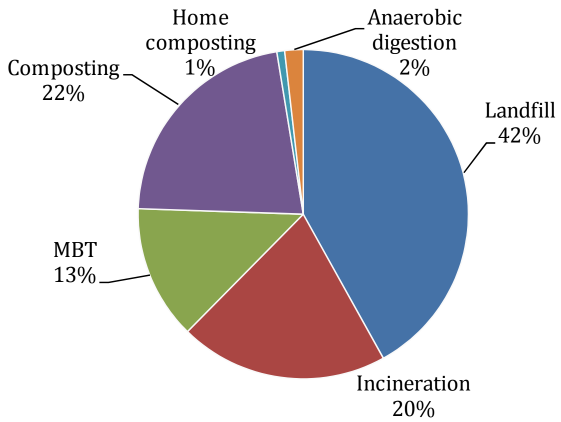



**Figure 1.7. Bio-waste treatment methods in the EU-27 (2008).**
*Source: Own elaboration from European Commission* (2010).

### *(ii) Current situation in Catalonia*

The situation of waste management, and particularly for bio-waste, is very different in Catalonia from the other regions of Spain. Currently, 21 out of the 34 Spanish composting plants treating source separated bio-waste are located in Catalonia, whereas other regions are more committed to MBT technologies. Because all the studies in the dissertation are located in Catalonia, this section is focused on Catalan waste management system rather than the Spanish one.

**Bio-waste generation**

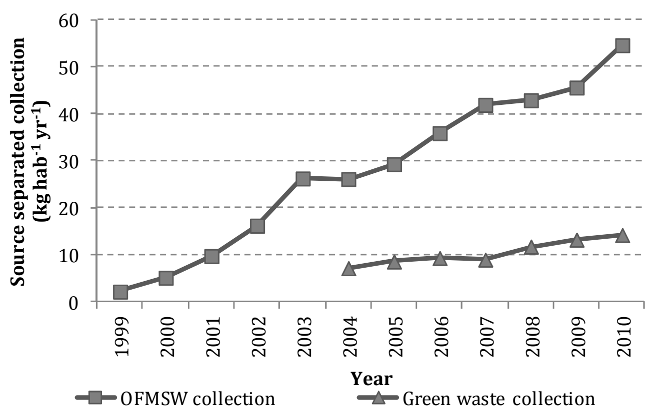



**Figure 1.8. Source separated collection of bio-wastes in Catalonia.**
*Source: Own elaboration from ARC* (2012), *Idescat* (2012), *and PROGREMIC* (2012).

In Catalonia, 558 kg per capita of municipal solid wastes was generated in 2010 (ARC 2012) – 11% above the European average – from which about 36% corresponded to bio-waste, the fraction with the higher percentage of the MSW (Table 1.2). Therefore, 201 kg of bio-waste were generated per Catalan citizen, 30% of which was separately collected in 2010 (Figure 1.8). According to the report from Huerta et al. (2010), the average content of improper materials in the OFMSW was 14-23% in the period 2004-2005.

**Bio-waste treatment**

According to ARC (2012) and PROGREMIC (2012), a 13% of municipal solid waste in Catalonia is currently going to anerobic digestion, 9% to composting and 2% to MBT. Regarding bio-waste, the goal of the PROGREMIC 2007-2012 (ARC 2007) was to separately collect and treat 55% of the waste generated by 2012 – unlikely to achieve, as only 30% was collected in 2010 – and to reduce the rate of improper materials below 15%.

As was aforementioned, Catalonia currently has 21 composting/anaerobic digestion plants. Table 1.3 presents the number of each type of plant and the total treatment capacity. In section 1.3.2 methodologies for composting are introduced.

**Table 1.3. Composting and anaerobic digestion plants in Catalonia.**

| Treatment option | Total capacity (OFMSW $yr^{-1}$) | Number of plants |
|---|---|---|
| Turned windrows | 74,500 | 8 |
| Turned aerated windrows | 85,000 | 2 |
| In-vessel + static piles | 75,000 | 7 |
| In-vessel + turned windrows | 32,000 | 2 |
| Anaerobic digestion + in-vessel | 155,000 | 2 |
| **TOTAL** | **421,500** | **21** |

*Source: Adapted from Huerta et al.* (2010) *and AMB* (March 2012).

To summarize, according to the figures presented in this section, MSW and bio-waste production is still increasing in Europe and Catalonia. On spite of accounting for the larger share of the total municipal waste production, bio-waste has been the last MSW fraction to be source-separated collected and treated, especially in Southern countries of the European Union, where major horticultural production is accounted, as was aforementioned. Environment protection involves reduction in the amount of biodegradable waste landfilled as well as specific studies assessing the impacts of each technology on each site and circumstances. Composting appears as an attractive and widely recommended option that additionally allows nutrients and organic matter recover on soils.

## 1.3. Compost production and application

History and experience on composting is wide and the pillars on which it is based are well defined (Haug 1993; Hoitink and Keener 1993). Biological decomposition is an age-old, natural process. As vegetation falls to the ground, it slowly decays, providing minerals and nutrients needed by plants, animals, and microorganisms.

Composting is often used synonymously with biological decomposition (Rynk and Sailus 1992).

### 1.3.1. Fundamentals of composting process

For this dissertation, composting refers to the purposeful and controlled decomposition of organic matter, as is defined by Haug (1993): *the biological decomposition of organic substrates, under conditions that allow thermophilic temperatures to be attained as a result of biologically produced heat, to generate a final product that is stable, free of pathogens and plant seeds, and can be beneficially applied to land*.

It is a dynamic process, biological, aerobic and therefore thermophilic, which for being conducted needs: organic matter, initial microbial population and the optimum conditions for it to develop with multiple functions and synergistic activities (Soliva et al. 2007). Outputs of the composting process are (mainly) $CO_2$ and other gaseous emissions (such as nitrous oxide, methane and volatile organic compounds), heat, water, minerals and biologically stabilised material, compost.

The composting process usually occurs in two main phases, a high-rate composting phase (decomposition stage) and a maturation phase (curing stage). The former, is called high-rate because during this stage the decomposition activity of the feedstock into simpler compounds by microorganisms is intense and, as a result of the metabolic activities, heat is produced. During the second stage the product is "cured" or finished and thus it is characterized by slow degradation because the nutrients available to microorganisms have been depleted (Haug 1993). The progress of temperature, microorganism types and pH during these two phases is presented in Figure 1.9.

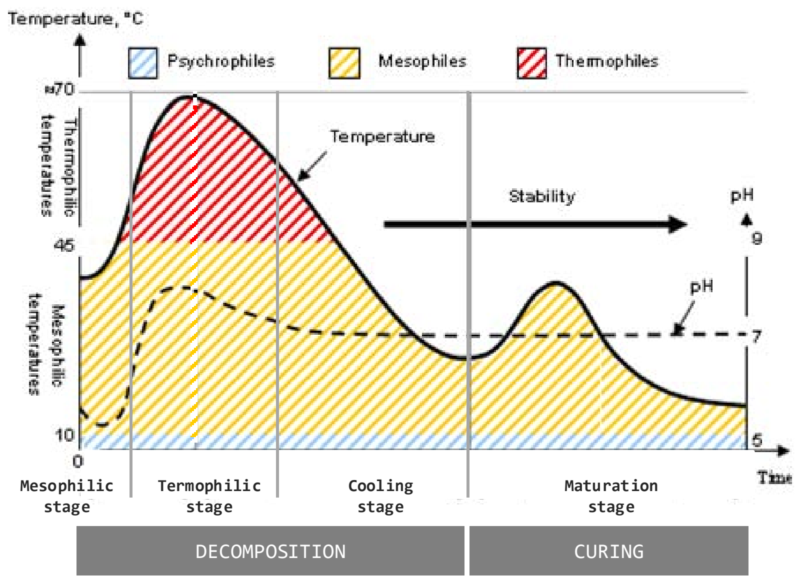



**Figure 1.9. Typical temperature, microorganisms and pH profiles through the stages of composting process.**
*Source: Adapted from Cadena* (2009).

The fundamentals of composting are: waste volume reduction (organic matter and water losses); waste stabilization (less C and N mineralizable); sanitation of the material (due to high temperatures); and adding-value to the final product (see section 1.3.3). Another key issue, for truly closing nutrient cycles and being an effective process, is ensuring that adequate and reliable markets are available for compost produced from bio-waste.

Several operations after and before main phases of composting – decomposition and curing – are necessary. Process operations prior to decomposition (grinding, mixing with the bulking agent, hand-storing, etc.) are named pre-processing and those at intermediate stages or after the curing phase (trommel screening, ballistic separators, etc.) are named post-processing operations (Cadena 2009).

**Table 1.4. Main process monitoring and organic feedstock parameters affecting composting and compost quality.**

| Parameter | Description / optimum range / effects of increasing and lowering |
|---|---|
| | ***Process monitoring parameters*** |
| **Moisture** | It is essential for physiological activity of microorganisms. Optimum: 40-60%. Controlled with: watering, aeration, initial waste mix. |
| | ↓ Less activity of microorganisms and the process is slowed down. |
| | ↑ Pores occupied by water. Anaerobic process. Putrefaction of OM. |
| **Oxygen** | For an aerobic process, oxygen is essential. Optimum concentration: 10-15%. Controlled with: aeration, turning and mixing. |
| | ↓ Less activity of microorganisms and the process is slowed down. |
| | ↑ Pores occupied by water. Anaerobic process. Putrefaction of OM. |
| **Temperature** | Determined by the level of activity of the heat-generating microorganisms. Optimum: 35-60 ºC. Controlled with: aeration. |
| | ↓ Insufficient elimination of pathogens, parasites and weeds. |
| | ↑ Not necessarily negative. Removes heat and excessive evaporation. |
| **pH** | Direct influence due to its effect on microbial processes. Bacteria tolerance: 6-7.5. Fungi tolerance: 5-8. Controlled with: aeration, watering. |
| | ↓ Microorganisms die off and decomposition slow down. |
| | ↑ Ammonium becomes ammonia that is toxic for microorganisms. |
| | ***Organic substrate parameters*** |
| **C/N ratio** | C and N are usually the limiting factors for efficient decomposition. Optimum: 25-35. Corrected with: initial waste mix. |
| | ↓ Initially accelerates microbial activity but ammoniac is produced. |
| | ↑ Inhibit the growth of decomposing microorganisms. |
| **Particle size and porosity** | Reactions and microbial activity occurs on the material surface exposed. Optimum Ø: 1-5 cm. Corrected with: initial grinding and waste mix. |
| | ↓ Higher contact surface but sealing problems and insufficient oxygen. |
| | ↑ Insufficient contact surface and incomplete decomposition process. |
| **Organic matter** | It is fundamental for the agronomic quality of compost. The OM content decreases throughout the process due to mineralization. |

↓ *Below optimum values;* ↑ *Above optimum values*

*Source: Elaborated from Rynk et al.* (1992), *Haug* (1993), *Bueno et al.* (2007), *and Cadena* (2009).

Moreover, previous to the composting plant stage, the bio-waste is generated by household, gardens and parks, or assimilated sources, and stored in containers and bins. Best option would be source separated waste for assuring a correct quality of the obtained compost. Later it must be collected with trucks and transported to the plant.

### *(i) Main factors affecting composting process and compost quality*

To fulfill the composting major aims – i.e. to obtain a final product useful as fertilizer and to reduce the environmental burdens of bio-waste treatment –, the process cannot be spontaneous, but variables have to be controlled to ensure a completed process, in a short time, and with minimum costs and emissions.

The most relevant variables affecting composting systems can be classified into two types: monitoring parameters (moisture, oxygen, temperature and pH) and related parameters to the nature of the substrate (C/N ratio, particle size, porosity and organic matter). Table 1.4 summarizes the main parameters and their effects, as well as their optimal values.

Regarding compost added-value, apart from feedstock quality and correctness of the composting process, other relevant issue is to have a "clean" waste stream (improving separate collection methods and public education).

## 1.3.2. Types of composting

Composting usually takes place in central facilities in the form of either open (e.g. windrow, static pile, mattress), enclosed (e.g. channel and cell, aerated pile) or reactor (e.g. tunnel, rotating drum) technologies (Andersen 2010). The latter is called industrial composting or full-scale composting.

There are different technologies used in composting to treat solid waste. The most commonly full-scale composting methods currently employed are summarized in Table 1.5. Furthermore, the four chosen methods are sorted according to the time and space necessities, the economical costs, the level of control of the process and the release of gaseous emissions – the latter is inversely proportionate to the level of gas captured and treated.

Exhaust gases from composting facilities are usually characterized by high flow rates and low pollutant concentrations. For some type of plants, for instance in-vessel and aerated static piles, gaseous emissions could easily be collected. These gases must be treated to avoid atmospheric pollution. Biofiltration is a suitable odor reduction technique for the treatment of gaseous emissions from composting processes. It is based on two phenomena: first, the pollutant is transferred to the supporting matter by absorption and adsorption; later, the pollutant is bioconverted into biomass, metabolic by-products, carbon dioxide, salts and water (Colón et al. 2009; 2012).

**Home composting**

Another option is the self-composting of the bio-waste as well as the use of the compost in a garden belonging to a private household. It is called home composting, backyard composting or domestic composting.

**Table 1.5. Most commonly used industrial composting methodologies in Europe and main differences.**

| | Composting methodologies | Space necessities | Time required | Costs | Process control | Released emissions |
|---|---|---|---|---|---|---|
|  | **Passive piles**<br>Bio-waste is placed in elongated heaps, heaps of triangle or trapezoid cross-section that remain static without alteration and may occasionally be turned during the process. | Higher | Higher | Lower | Lower | Higher |
|  | **Turned (aerated) windrows**<br>Periodically turned piles by mechanical means in order to increase the porosity of the heap to increase the active surface accessible to micro-organisms and increase the homogeneity of the waste. Optionally, they can be forcedly aerated. | | | | | |
|  | **Aerated static piles**<br>Forcing or pulling air through a trapezoidal compost pile, which minimizes the need for turning. It also can be conducted under semi-permeable covers or in enclosed buildings, for a better control of odors and other emissions. | | | | | |
|  | **In-vessel / tunnel composting**<br>Composting of bio-waste in a closed reactor where the composting process is accelerated by an intensified air exchange and an automated temperature control. Exhaust gases can be collected and treated; the most common treatment technology being biofilters. | Lower | Lower | Higher | Higher | Lower |

*Source: Own elaboration from Haug* (1993), *Crowe et al.* (2002), *Cadena* (2009), *and JRC and IES* (2011b).

The most common home composting techniques are: heap/piles (usually covered), composting bins, silos or open boxes, and in-house worm composter (JRC and IES 2011b). Industrial composting imply the consumption of energy for waste collection, transport and processing, the emission of odors and other contaminants, the proliferation of insects, birds and rodents and the mixture of different quality materials (Haug 1993). Non-biodegradable material is common problem in industrial composting plants. Notwithstanding these disadvantages, some of the methodologies for composting at industrial scale permit a higher control of composting process variables and the treatment of the exhaust gases.

In contrast, home composting considerably reduces the economic, material and energetic investments and it allows a direct control of the process and the organic feedstock by avoiding or reducing the inclusion of impurities (McGovern 1997; Ligon and Garland 1998; Jasim and Smith 2003; Boldrin et al. 2009). However, home composting also presents some problems: compost obtained often is not homogeneous; temperatures sometimes are not higher enough to ensure sanitation; and odors and other pollutants are emitted directly to the atmosphere (Amlinger et al. 2008; Ansorena 2008; Andersen 2010).

### 1.3.3. Potential benefits and drawbacks of compost production and application

In the following the positive and negative aspects traditionally related with composting process (Table 1.6) and compost application (Table 1.7) are detailed.

**Table 1.6. Potential benefits and drawbacks of composting process typically reflected in the literature.**

| Benefits | Drawbacks |
|---|---|
| Possible simple, durable and cheap technology (except some in-vessel facilities). | Requires source separation of bio-waste, including continuous information to waste generators. |
| It recovers approximately 40–50% of mass (weight) for plant growth. | A market for the compost products must be developed and maintained. |
| Maximum recovery of the nutrients required for low-input farming systems (i.e., P, K, Mg and micronutrients). | Produces the emission of odorous compounds and other pollutants: $CH_4$, $N_2O$, $CO_2$, $NH_3$, and VOC. |
| Produces humic substances, beneficial micro-organisms, and slow-release nitrogen required for landscape gardening and horticulture. | Potential vector-problems (seagulls, rats, flies) when treating bio-waste. |
| Eliminates weeds and pathogens in the waste material. | Requires skilled staff. |
| Facilitates the extraction of inorganic materials to be recycled and increases its calorific value. | It generates low economic returns due to the considerable investment in technology are required. |
| Reduces waste volume thus ease of handling. | The final product has a relatively low price, partly for the negative perception of the composting process. |

*Source: Own elaboration from Crowe et al.* (2002) *and Cadena* (2009).

The composting of bio-waste could lead to positive consequences, although several potential drawbacks were also reported. In addition to the potential benefits of the composting process, numerous studies indicate that the use of compost on land may improve several plant and soil parameters, which would make compost an interesting fertilizing option not only for agriculture but also for soil restoration purposes (Diacono and Montemurro 2010).

**Table 1.7. Potential benefits and drawbacks of compost application typically reflected in the literature.**

| Benefits | Drawbacks |
|---|---|
| Increases organic matter content. | Immobilizes soil mineral N of soil. |
| Enhances nutrient availability thus reducing mineral fertilizers necessities. | The nutrients are slowly released. |
| Provides a slow source of nutrients. | Induces accumulation of heavy metals. |
| Enhances aggregation and stability of soil thus makes easier workability. | Increases salinity. |
| Prevents surface sealing. | Increases fertirrigation gaseous emissions and leaching. |
| Improves water infiltration, and enhances water holding capacity. | Produces weed and pathogen contamination. |
| Reduces runoff generation and soil erosion. | |
| Promotes carbon sequestration. | |
| Improves biological activity and increases biodiversity. | |
| Induces suppression of soil borne diseases thus reducing pesticides application. | |
| Produces higher yields and better quality of the harvest. | |

*Source: Own elaboration from Giménez et al.* (2005)*, ROU* (2007)*, European Commission* (2008)*, Hargreaves et al.* (2008)*, Farrell and Jones* (2009)*, and Diacono and Montemurro* (2010).

### 1.3.4. Potential compost applications. Focus on agricultural use

As aforementioned, key issues for the feasibility of composting are: to have a market willing to use the produced compost and thus a suitable price for the product. Compost is used for a diversified number of uses, and the quality and characteristics of it must be according to the specific use (Giménez et al. 2005). Several potential end uses exist for MSW-derived composts: hobby gardening, nursery plant production, commercial horticulture, fruit growing and viticulture, agriculture, landscaping, land restoration, and building, extracting and landfilling industries (Hauke et al. 1996; Giménez et al. 2005; Farrell and Jones 2009). Table 1.8 shows the approximate fates of composts produced in Europe and Catalonia.

Agronomic and horticultural use represents a large potential market for MSW compost (Farrell and Jones 2009). According to Table 1.8, agriculture and horticulture are among the two most common uses in both Europe and Catalonia. Organic agriculture often pays more for compost products as compared with high-input agriculture, but it is also asking for high-rate quality products produced exclusively from source separated bio-wastes.

Even though the high-potential of compost production and agricultural consumption in Catalonia, farmers are reluctant to use it mainly due to ignorance of the dosages and application specificities, to distrust on heavy metals, impurities and other potential pollutants to their soils, and to economical factors (Giménez et al. 2005).

**Table 1.8. Uses of compost in Catalonia and Europe.**

| Region | Europe | Region | Catalonia |
|---|---|---|---|
| **Agriculture** | 33% | **Agriculture** | 27% |
| **Horticulture and green house production** | 8% | **Gardening and landscaping** | 53% |
| **Gardening and Landscaping** | 17% | **Land restoration** | 6% |
| **Blends** | 7% | **Public works** | 2% |
| **Soil mixing companies** | 7% | **Other uses** | 12% |
| **Wholesalers** | 7% | | |
| **Land restoration and landfill cover** | 18% | | |
| **Other uses** | 4% | | |

*Source: Giménez et al.* (2005) *and Barth et al.* (2008).

### *(i) Quality standards and minimum requirements*

Standards on the use and quality of compost exist in most Member States, but differ substantially, partly due to differences in soil policies, whereas there is no comprehensive Community legislation (Barth et al. 2008; European Commission 2008).

Table 1.9 includes the standards required in Spain by the RD 824/2005 (Ministerio de la Presidencia2005), on fertilizing products. Although it is not specified in the RD 824/2005, the Class A corresponds to the quality requirements of compost from selectively collected bio-waste suitable for organic agriculture, according to Commission Regulation 889/2008/EC (EC2008a). Table 1.9 also shows the minimum requirements for compost according to the Working document on *Directive of biological treatment of bio-waste* (European Commission2001).

Nevertheless, the two regulations in Table 1.9 are omitting two relevant parameters to prevent negative impacts to the environment and health – pathogens standards and list of allowed input materials. Moreover, Barth et al. (Barth et al. 2008) proposed other quality criteria to guarantee a minimum use performance and to prevent any deception of and misuse by the user or customer, i.e. weeds content, bioassay results, minimum organic matter and nutrients content, detectable impurities, odor, salinity and stability. The biological stability is defined as the measure of the degree of decomposition of biodegradable organic matter contained in a matrix and it can be directly measured by means of respirometric indices (Colón et al. 2012).

Additionally, apart from quality standard regulations, Council Directive 91/676/EEC (ECC1991), the *Nitrate Directive*, imposes limits on N loads on farmlands. Barth et al. (2008) pointed out that this, in general, may impose a constraint on the use of soil improvers, but may also trigger a greater application of compost as a replacement of mineral fertilisers, given the lower N availability and the fact that compost is a slow-release source of N.

**Table 1.9. Spanish and European regulations on compost quality standards.**

| Parameter | RD 824/2005 (Spain) | | | 2nd draft Directive bio-waste (Europe) | |
|---|---|---|---|---|---|
| **Moisture** | 30-40% | | | - | |
| **Organic matter** | > 35% | | | - | |
| **C/N** | < 20 | | | - | |
| **Particle sizes** | Ø > 5mm <5%<br>Ø > 2mm < 3%<br>Ø < 25mm 90% | | | ø > 5 mm < 5% | |
| **Impurities** | < 3% | | | < 0,5% | |
| **Metals content (mg kg⁻¹sms)** | **Class A** | **Class B** | **Class C** | **Class 1** | **Class 2** |
| **Cd** | <0.7 | <2 | <3 | < 0,7 | < 1,5 |
| **Pb** | <45 | <150 | <200 | < 100 | < 150 |
| **Cu** | <70 | <300 | <400 | < 100 | < 150 |
| **Zn** | <200 | <500 | <1000 | < 200 | < 400 |
| **Ni** | <25 | <90 | <<100 | < 50 | < 75 |
| **Cr** | <70 | <250 | <300 | < 100 | < 150 |
| **Hg** | <0.5 | <1.5 | <2.5 | < 0,5 | < 1 |

*Source: European Commission* (2001) *and Real Decreto 824/2005* (2005).

In conclusion, composting is a biological process that allows the obtaining, from bio-waste, of useful and potentially valuable product that improves organic matter content of soils and thus other parameters, as well as diverting bio-waste away from landfill. Nevertheless, several potential drawbacks are also detected for this bio-waste treatment at the production and application level, thus further research is needed at the environmental, agricultural and socio-economic levels. Several options exist for perform composting process: industrial composting – either in-vessel or in piles – and home composting. In addition, agricultural use is one of the main fates of compost produced from wastes, and it was the application studied in this dissertation.

## 1.4. Outline of life cycle approach in the assessed sectors

Life Cycle Assessment (LCA) was promoted in different European directives as a robust quantitative tool, and a keystone in decision making by producers and stakeholders. It has been widely used for the assessment of the waste and agricultural sectors.

LCA is a tool to evaluate the environmental performance of products (goods and services). LCA takes into account a product's full life cycle, from the extraction of resources and processing of raw material, through production, use and recycling, to the disposal of remaining waste. The standardised LCA framework consists of four phases (ISO 2006a): goal and scope definition, inventory analysis, impact assessment, and interpretation (a more detailed description of the methodology is provided in Chapter 2).

### 1.4.1. Life cycle tools and agricultural systems

LCA was initially developed for the assessment of industrial systems. Several groups began to apply LCA to agricultural systems in the 1990s and also launched the first related guidelines (e.g. Audsley (2003) and Wegener Sleeswijk et al. (1996)).

The first attempts of LCA implementation in crops were mainly focused on extensive agriculture, such as wheat (Weidema et al. 1996; Hansson and Mattsson 1999) and sugar (Brentrup et al. 2001). Jungbluth et al. (2000) aimed to help Swiss consumers to include environmental aspects when choosing products. The author identified transoceanic transport and heating of greenhouses as the factors with greater environmental burdens.

Regarding LCA application to horticultural systems, the first product to being assessed were tomatoes (Jolliet 1993; Nienhuis and Vreede 1996; Van Woerden 2001). Horticultural production studies in Spain were performed later (Antón 2004; 2005a; 2005b; 2007). One reference paper for fresh fruit production is Milà i Canals et al. (2006) assessing the environmental impact of apples production in New Zealand. Recently, the European project EUPHOROS assessed the environmental impacts of greenhouse production in both locations from South and Northern countries of Europe (Montero et al. 2009a).

The comparison between organic and conventional farming is one of the important themes in the LCA studies of agricultural production systems. Nemecek et al. (2011) a systematic analysis of organic farming compared with integrated production for Swiss conditions.

Last years, Carbon Footprinting (CF) methodology, which is also based on the life cycle approach, although confined to the analysis of GHG emissions (see Chapter 2), is gaining strength in the field of environmental assessment, mainly within the private sector. It has to be calculated through one of the current, at least 16, specifications from private initiatives or government initiatives, such as the PAS 2050:2011 (BSI 2011). Moreover, the BSI has recently launched the PAS 2050-1:2012 (BSI 2012), which is the first specific guideline for the GHG assessment of horticultural products.

Regarding the environmental performance of compost application in soils some previous attempts were made by several authors (Lundie and Peters 2005; Hansen et al. 2006a; ROU 2007; Blengini 2008; Meisterling et al. 2009; Boldrin et al. 2010). Data has been lacking on the potential savings from the use of compost on soil, which means that previous environmental assessments have not fully assessed the loads and savings from composting (Andersen 2010), as most of the potential benefits of compost on plant and soil have not been taken into account. Up-to-date and due to methodological difficulties, carbon sequestration and nutrient supply are the unique environmental benefits taken into account when evaluating the environmental impacts of compost application (Hansen et al. 2006a; Favoino and Hogg 2008; Boldrin et al. 2009). ROU (2007) is, to our knowledge, the only study where an attempt was made to include most of the abovementioned effects within LCA, although the study remains at the inventory stage.

### *(i) Main methodological concerns regarding LCA application to agricultural systems*

Results of the aforementioned studies, among others, showed that there are considerable differences between industrial and agricultural systems (Table 1.10), which hindered the applicability of the LCA method to agricultural products.

**Table 1.10. Differences between industrial and agricultural systems.**

| Characteristic | Industrial systems | Agricultural systems |
|---|---|---|
| **Dependency on location** | Independent. | Dependent. |
| **Functionality** | One or few functions. | Multifunction. |
| **Obtained products** | Typically one. | Typically more than one (co-production). |
| **System boundaries** | Clearly defined. | Unclear, both physically and temporally. |
| **Main sources of impacts** | Energy and materials consumption. | Land and water use, pesticide emissions, energy and materials consumption. |
| **Degree of knowledge** | High (simple and pre-designed processes). | Relatively low (complex, natural processes). |

*Source: Núñez* (2011).

Agricultural systems typically produce more than one output (co-products). For instance, wheat production provides grain for human consumption as well as straw for several uses. Therefore, the resolution of multi-functionality problems plays an important role in results. The functional unit is the basis for comparisons between different systems in LCA (ISO 2006a; 2006b). Adequately selecting a functional unit is of prime importance because different functional units can lead to different results for the same product system. In most articles on the environmental assessment of food production, the functional unit is based on mass or volume, omitting the other functions and co-products of agriculture systems (Hayashi et al. 2006; Mourad et al. 2007; Reap et al. 2008a; 2008b; Schau and Fet 2008).

Regarding the temporal system boundaries, they should cover the whole rotation since this is an efficient technique for internal nutrient recycling, maintaining the long-term productivity of the land, and thus reduction of the fertilizers requirement, as well as in the control of pest and diseases (Mourad et al. 2007; Zegada-Lizarazu and Monti 2011).

The boundary between the technosphere (i.e. the studied system) and the ecosphere (i.e. the environment) is much more difficult to define, especially because the soil can be seen as a part of both. Though materials and energy consumption are a source of global impacts both in industrial and agricultural systems, many key impacts of agriculture occur at the same agricultural field and are related to land and water use and pesticide emissions, involving very complex biological processes (Núñez 2011).

Moreover, according to the EULCIA project on the recommendation of methods and characterisation factors for a standardised Life Cycle Impact Assessment (LCIA) framework (JRC 2007), existing LCIA methods do not sufficiently address or even ignore several environmental impacts of agricultural systems (such as land use, soil erosion, water depletion, desertification, pesticides toxicity, etc.).

### 1.4.2. Life cycle tools and waste management

LCA thinking applied to waste management systems is receiving increasing attention. They have become important methods for waste management and policy and have been successfully applied to integrated waste management systems in a number of case studies (Finnveden et al. 2007). A common limitation in the majority of published works is a lack of field data corresponding to real waste treatment processes working under real conditions. EASEWASTE (Kirkeby et al. 2006), ORWARE (Dalemo et al. 1997) and WASTED (Diaz and Warith 2006), are simulation tools which include the environmental burdens associated with waste management. Similarly, Sevigné et al. (Sevigné Itoiz et al. submitted 2012) proposed a CF tool for municipal solid waste management for policy options in Europe.

Concerned with the environmental impact of composting technologies, some studies have mainly focused on atmospheric emissions (ammonia, methane, nitrous oxide and volatile organic compounds), most of them performed at pilot or laboratory scale and only a few at real scale (Eitzer 1995; Smet et al. 1999; Beck-Friis et al. 2000; Hellebrand and Kalk 2001; Komilis et al. 2004; Pagans et al. 2006b; Amlinger et al. 2008; Lou and Nair 2009; Andersen 2010; Scheutz et al. 2010), and just a few of them were studied by means of LCA (Güereca et al. 2006; Iriarte et al. 2009; Rives et al. 2010).

However, little literature can be found on the global impact of a specific technology or facility by using in situ measurements, especially with bio-wastes (Blengini and Busto 2009; Andersen et al. 2010; Boldrin et al. 2010; Scheutz et al. 2010), presented the outcome of the analysis of existing 86 studies on the management of municipal bio-waste using a life cycle perspective.

#### *(i) Main methodological concerns regarding LCA application to waste management systems*

More or less 'standard' LCA methodology can be used when applied to integrated waste management systems (Finnveden et al. 2007). However, the applicability of LCA for waste management assessment is restricted by certain limitations. Table 1.11 summarizes the main particularities of LCA applied to waste management systems, which are discussed in the following.

**Heterogeneous composition**

Very often in environmental assessment of waste management systems, it is not an analysis of a product, but a lot of them and usually we have limited knowledge about the content, i.e. the constituents of waste are several products: paper, plastics, organic matter, metals, chemicals, etc. Besides, even if we do know the content of the different waste materials, there may be limitations of our understanding of the processes and thus our possibilities to predict emissions (Finnveden et al. 2007).

**System boundaries**

When LCA are applied to waste management services, the assessment typically focus on a comparison of different waste management options, not covering the entire life cycle of the products which have become waste (JRC and IES 2011a). The typically adopted philosophy would be "from-gate-to-grave" or "from-gate-to-cradle" (Blengini 2008), as the analyzed product (the waste) comes from the technosphere.

Open-loop recycling is a common problem in LCA and is encountered where: recycled material or energy in cascade use is introduced to the product system from other applications; or material or energy is exported from the product system to other applications (Werner 2005). There are several approaches of how to solve the distribution of the burdens to the involved systems. The "cut-off" method used in this dissertation considers that each system is assigned the burdens for which it is directly responsible (Ekvall and Tillman 1997) - i.e., dumped materials are accounted for in the system generating the waste, while burdens for recycled or reused waste are attributed to the system using the waste as a raw material source.

**Table 1.11. Specific features of LCA application to waste management systems.**

| Issue | Specific features |
|---|---|
| **Heterogeneous composition** | Unknown waste composition.<br>Limitations of our understanding of the processes for all the components. |
| **System boundaries** | Usually "from-gate-to-grave" or "from-gate-to-cradle" perspectives.<br>Open-loop recycling. Cut-off methods.<br>Multi-functionality. Allocation methods.<br>Delayed emissions and carbon sequestration. |
| **Functional unit** | Functional unit as a waste flow rather than as a product. |
| **Carbon accounting** | Biogenic carbon.<br>Carbon sequestration. |

Delayed emissions in waste management occur because emissions from landfills can continue for thousands of years (Finnveden et al. 2007). Furthermore, carbon sequestration in soils, promoted with compost application, also continues a lot of years before its application (Smith et al. 2001). An important question then is how future emissions should be valued against current emissions.

**Functional unit**

In a LCA of a product, the functional unit is usually referred to the output stream of the production system or to the function performed by it. However when evaluating waste management systems, the considered functional unit is usually the amount of waste generated in a certain region or the flow entering into a given treatment. Colón et al. (2012) proposed a new functional unit for composting based on the degree of waste stabilization achieved by each treatment.

When conducting comparative LCAs of waste management systems it is usual to have co-functions in addition to the main function provided by the system considered (JRC and IES 2011a). For instance, in case of composting of OFMSW with compost production, in addition to the main function of providing treatment to the, the co-service "compost production" should be considered and properly accounted for.

**Carbon accounting**

For biodegradable materials the carbon will have been absorbed from the atmosphere by photosynthesis during plant growth relatively recently. If this carbon is released again as $CO_2$ during the treatment process then the carbon re-enters the natural carbon cycle. For this 'short-term' carbon cycle or biogenic $CO_2$, many studies consider that there is no net global warming impact, and no global warming potential is associated with the $CO_2$ emission, since the atmospheric concentration of short-cycle carbon dioxide is relatively constant from year to year. If the emission occurs in the form of $CH_4$, however, then this has a higher global warming potential than $CO_2$, so must be accounted for (Smith et al. 2001; UNEP 2010a).

However, if not all of the carbon is released and it is sequestered in a form which is unavailable to the natural carbon cycle over a sufficiently long time period, then it could be considered as a 'sink' for carbon (Christensen et al. 2009). The two main routes for carbon storage in waste management are in landfills and in compost applied to soil (where a proportion of the carbon becomes converted to very stable humic substances which can persist for hundreds of years). The permanency of such sinks is difficult to assess, and depends on the time scale used to define permanent, as aforementioned (Smith et al. 2001; Christensen et al. 2009).

## 1.5. Motivation of the dissertation

Facing the current necessity of bio-waste management and organic amendment product availability, and the claim for a sustainable and less polluting agriculture, composts arises as a relevant option to close the organic matter and nutrient cycles interconnecting both city and country systems.

This dissertation is motivated by the following realities:

- According to the European Commission (2006a), an estimated **45% of European soils** have **low soil organic matter** content. The *Thematic Strategy for Soil Protection* (European Commission 2006a) calls for the use of compost as one of the **best sources of stable organic matter**. Furthermore, the increasing price of mineral fertilizers influences farmers to look favourably on organic amendments.
- As a response of the political concern as well as the increasing consumer **demand for environmentally friendly food products,** as well as a growing concern about the **role of diet in human health** and in food safety and security, real alternatives to current intensive production methods need to be supported by scientific research.
- **Southern European countries are the major producers** of horticultural products, with Spain and Italy on the top of the list. Southern countries have also the **major covered surface under greenhouses** of the European Union.
- **Tomato and cauliflower** are some of the widely-cultivated horticultural crops in the European Union. They could be produced either in **open field or under greenhouse protection**.
- At present, **waste management is becoming a global problem** in developed countries due to the rapid collapse of landfills and the high impacts related to bio-waste dumping (EC1999). The overall annual food and garden waste included in mixed municipal solid waste in the European Union is within 76.5–102 Mt that represents **30–40% of the total annual municipal waste generation** (European Commission 2008).

- On spite of accounting for the larger share of the total municipal waste production, **bio-waste has been the last MSW fraction to be source-separated collected and treated**, especially in Southern countries of the European Union, where major horticultural production is accounted.
- Composting has been claimed as a **green and sustainable alternative** to manage and recycle organic solid wastes (EC2008b; European Commission 2008). There is an international concern, which is reflected in current legislation, about the importance of achieving adequate waste management and treatment in order to ensure the protection of human health and the prevention of environmental impacts (Eriksson et al. 2005).
- **However, specific, real and quantitative environmental data** about **composting technologies** for bio-waste is scarce. For this reasons, this dissertation would improve the knowledge about the environmental concerns, as well as other dimensions, of composting technologies (at full and home scale) and to help in the regional planning of organic waste management.
- Although there has been an increase in studies on composting, mostly focus only on a specific aspect of the composting life cycle, such as gas emissions or the use of compost as a fertilizer. A **holistic approach** including all **production, transport and compost application** in real horticultural systems, taking into account several aspects of **sustainability and management** is necessary.
- One of the major destinations for compost produced in Europe and Catalonia is being a **source of organic matter and slow-released nutrients** for agriculture. Mineral fertilizers substitution by compost may lead to **differences in the quantity and the quality** – including sizes and nutritional value – of the agricultural yield.
- Life Cycle Assessment (LCA) was promoted in different European directives as a **robust quantitative tool, and a keystone in decision making** by producers and stakeholders. Nevertheless, **several concerns regarding its use** have been also reported by the scientific sphere, for general use and specifically for waste management and agriculture, such as functional unit selection. Therefore, it is necessary to assess LCA suitability for the environmental quantification of the studied systems.
- Other life-cycle-thinking tools claimed to be more suitable for environmental communication to consumers, such as the **Carbon Footprinting**, are internationally proposed and used in the food industry.
- The assessment of the **positive and negative impacts** associated to **compost use on land** have not yet been fully quantified and introduced in LCA approach, apart from carbon sequestration and mineral fertilizers savings. Because of the modeling complexity, ROU (2007) is, to our knowledge, the only study where an attempt was made to include compost application impacts within LCA of two Australian case studies.
- Several studies dealing with methodological and practical questions for SLCA were published. However, an **implementation of SLCA to fertilizer** has not been carried out yet, and more importantly no case study so far considers the **upstream processes**. From the best of our knowledge, apart from the current paper, only two other SLCA studies are assessing agricultural systems (Andrews et al. 2009; Franze and Ciroth 2011), while only one assesses waste management

(Vinyes et al. Accepted 2012). Similarly, hardly any attempt is found for LCSA application.

- In **agricultural peri-urban areas,** such as the Maresme County, there is an added value through using compost from urban areas, and the proximity to the market brings the opportunity to sell the agricultural products and so **close metabolic cycles**. Moreover, **high-nitrate-polluted water** could be a very appropriate source of rapidly available nitrogen to complement compost. Those synergetic exchanges between cities (providing agriculture with a source of nutrients) and agriculture (providing cities with food) need to be further studied.
- The area of study, **Catalonia**, was selected as representative of other **Mediterranean regions** due to the high population density, and therefore, **large production of wastes**; because **it has firmly committed for composting** as the major treatment for bio-waste (ARC 2007); because it has an annual production of almost 330,000 tons of **horticultural products** (DARPAMN 2012); and because it presents relevant levels of **nitrates in ground water**.

# 1.6. Objectives of the dissertation

The main objective of this dissertation is to assess compost production from bio-waste and its application in Mediterranean horticultural crops from a sustainable perspective. Environmental, socio-economic, technological and technical, territorial planning, as well as methodological aspects of compost production and application assessment are included in the dissertation.

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

1. To assess the **environmental performance of composting technologies**:
   → To quantify material requirements, energy consumption and gaseous and waste emissions associated to home and industrial composting, and to compare the two systems.
   → To determinate and compare the environmental impacts of home and industrial composting using the environmental tool of Life Cycle Assessment and to identify the main sources of impact.
2. To assess the **environmental performance and other aspects of several horticultural cultivation options** using compost:
   → To compare the production and the harvest quality (both size parameters and, for cauliflower, antioxidant compounds) for the several cultivation options and the several crops.
   → To determine the critical environmental stages of compost life cycle (i.e., production and application).
3. To determine agricultural and environmental differences of cultivation options characterized by the divert dosages of compost and mineral fertilizers for tomato and cauliflower field experiments, and by greenhouse or open field cultivation for tomato. To propose and discuss **methodological improvements for the environmental assessment** of compost production and application:
   → To assess the different environmental performance of several cultivation options when functional units based either on production or quality are used.
   → To quantitatively address LCA modeling.

→ To explain some of the main **dynamics** determining the environmental and agronomic benefit of those **positive effects traditionally associated to land application** of compost originating from bio-waste and to provide with **quantitative data**.

→ To describe **existing impact methodologies and future challenges** for assessing those benefits under a LCA perspective.

4. To study **socio-economic, territorial, consumer and environmental aspects of compost production and application**:

→ To assesses the value of CF and LCA as tools to calculate GHG emissions from horticultural production systems and to communicate environmental performance.

→ To identify the critical issues in the choice of CF and LCA methodologies that can give misleading information for several horticultural cases.

→ To develop a theoretical proposal for SLCA application in a real agricultural case study.

→ To identify social hotspots of two fertilizing alternatives, revealing potential risks and improvements regarding its social performance.

→ To be one of the few real case application of the LCSA approach.

→ To study the potential eco-synergetic effects of using two waste flows, composted organic municipal waste (nutrients slow release) and nitrate polluted water (rapid release) in Catalonia.

# Chapter 2. Material and methods

**Chapter 2** presents an overview of the dissertation's methodological aspects. First, the several sustainable assessment tools applied in the dissertation, which are under a life-cycle-thinking approach, are presented: *Life Cycle Assessment (LCA)*, which is more broadly described, *Carbon Footprinting (CF),* L*ife Cycle Sustainable Assessment (LCSA), Social Life Cycle Assessment (SLCA),* and *Life Cycle Costing (LCC).* In addition two other territorial and economical tools are briefly explained. Subsequently, the systems under study will be presented schematically focusing on their interconnection and main aspects related to field and laboratory works and data collection processes will be detailed.

This chapter is structured as follows:

- Sustainability assessment tools.
- Systems of study and experimental set.

# 2.1. Sustainability assessment tools

According to the Bruntland Commission of the United Nations on 20 March 1987, sustainable development is "development that meets the needs of the present without compromising the ability of future generations to meet their own needs" (Bruntland 1987). Sustainable development is usually presented as the intersection between environment, society and economy, which are conceived of as being separate but connected entities.

A sustainable assessment tool is a systematic process designed to identify, analyse and evaluate the environmental, social and economic consequences of a product, service or system. Many tools and indicators for assessing and benchmarking the sustainability of systems, from different perspectives and with different scopes, have been developed (Finnveden and Moberg 2005; Ness et al. 2007; SustainabilityA-Test March 2012).

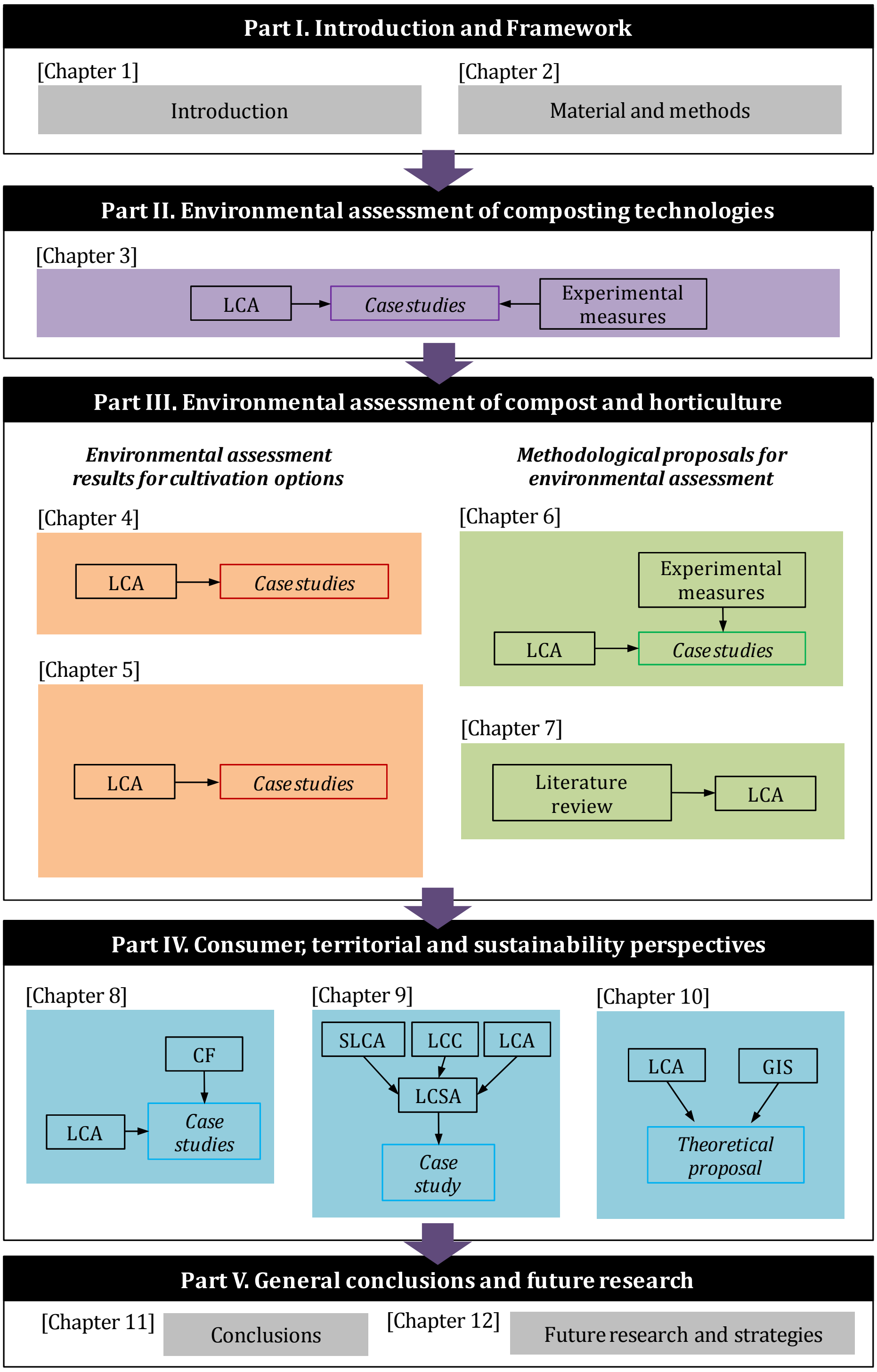
The entire dissertation is using sustainable assessment tools based on life-cycle-thinking. This concept implies to consider the whole product system life cycle from the "cradle to the grave". It aims to prevent individual parts of the life cycle from being addressed in a way that just results in the environmental burden being shifted to another part (Finkbeiner et al. 2010). Figure 2.1 shows the specific methods used in each chapter. All the chapters of the dissertation (from 3 to 10) are using environmental Life Cycle Assessment (LCA). It is an analytical tool which can usefully assess the potential environmental impacts and resources used throughout a product, service or system life cycle from raw material acquisition to production, use, and disposal (Finnveden and Moberg 2005). In Part IV additional life cycle methodologies are used: Chapter 8 is comparing LCA and Carbon Footprinting (CF), which is focused on greenhouse gases emission and has its own methodologies; Chapter 9 is applying the three life-cycle-thinking tools for sustainability – Life cycle costing (LCC) assessing the economic dimension, Social Life Cycle Assessment (SLCA) assessing the social dimension, and LCA assessing the environmental side – that are integrated with Life Cycle Sustainability Assessment (LCSA). Additionally, in Chapter 10, Geographic Information System (GIS) tool is used along with LCA for a territorial management proposal dealing with bio-waste.

## 2.1.1. Life Cycle Assessment

The concept that later became (environmental) LCA first emerged in the 1960s (UNEP 2011), but it was in the 80s, when the LCA awoke scientific interest and that its use became more widely extended. The institutions of reference that started to develop the methodology were the Swiss Federal Laboratories for Materials Testing and Research (EMPA), the Swiss Agency for the Environment, Forests and Landscape (BUWAL) and the Institute of Environmental Sciences (CML) in Leiden (Guinée et al. 1993).

However, until the early 1990s, studies that undertook an assessment of the material, energy and waste flows of a product's life cycle were conducted under a variety of names – including the resource and environmental profile analysis (REPA), ecobalance, integral environmental analyses and environmental profiles (UNEP 2011).

In 1990, SETAC hosted workshops with the aim of developing a standardized method of (environmental) LCA, which was to serve as the basis for the launch, seven years later, of the ISO 14040 series, the international reference guides for LCA – ISO 14040:1997, ISO 14041:1998, ISO 14042:2000 and ISO 14043:2000 (ISO 1997; 1998; 2000a; 2000b). In 2006, the ISO 14040:2006 (ISO 2006a) and the ISO 14044:2006 (ISO 2006b) replaced the previous four standards.




*CF, Carbon Footprint; GIS, Geographic Information System; LCA, environmental Life Cycle Assessment; LCC, Life Cycle Costing; LCSA, Life Cycle Sustainability Assessment; SLCA, Social Life Cycle Assessment.*

**Figure 2.1. Overview of the methods used in Parts II, III and IV of the dissertation.**

Many software packages, databases and impact methods have been created and extended to facilitate the realisation of LCA studies, even though the method is still under development. Nowadays, several international initiatives are ongoing to help build a consensus and provide recommendations, including the Life Cycle Initiative of the United Nations Environment Programme (UNEP) and the Society of Environmental Toxicology and Chemistry (SETAC) that would enable users around the world to put life-cycle-thinking into effective practice; or the European Reference Life Cycle Database (ELCD) and the International Reference Life Cycle Data System (ILCD) of the European Commission (Rives 2011).

### *(i) Definition*

LCA is a structured, comprehensive and internationally standardised method. It quantifies all relevant emissions and resources consumed and the related environmental and health impacts and resource depletion issues that are associated with any goods or services ("products"). LCA takes into account a products full life cycle: from the extraction of resources, through production, use, and recycling, up to the disposal of remaining waste. Critically, LCA studies thereby help to avoid resolving one environmental problem while creating others (JRC and IES 2010b).

The first official definition of the LCA methodology was provided by the SETAC (1991): "Life-Cycle Assessment is an objective process to evaluate the environmental burdens associated with a product, process or activity by identifying and quantifying energy and materials used and wastes released to the environment, to assess the impact of those energy and material uses and releases to the environment, and to evaluate and implement opportunities to affect environmental improvements. The assessment includes the entire life cycle of the product, process or activity, encompassing extracting and processing raw materials, manufacturing, transportation and distribution, use, re-use, maintenance, recycling, and final disposal."

The ISO has also provided highly relevant input to the process of defining LCA. According to the ISO 14040:2006 standard (ISO 2006a), *LCA is the compilation and evaluation of the inputs, outputs and the potential environmental impacts of a product system throughout its life cycle.*

### *(ii) Methodology*

The evaluation framework most commonly applied in LCA involves the following phases: (a) definition of goal and scope, (b) inventory analysis, (c) impact assessment, and (d) interpretation (Figure 2.2). To carry out an LCA study is almost always an iterative process: once the goal of the work is defined, the initial scope settings are derived that define the requirements on the subsequent work. However, as during the life cycle inventory phase of data collection and during the subsequent impact assessment and interpretation more information becomes available, the initial scope settings will typically need to be refined and sometimes also revised (JRC and IES 2010b).

**Goal and scope**

This is the first stage of the study and probably the most important, since the elements defined here, such as the purpose, scope, and main hypothesis considered are the keys to the study.

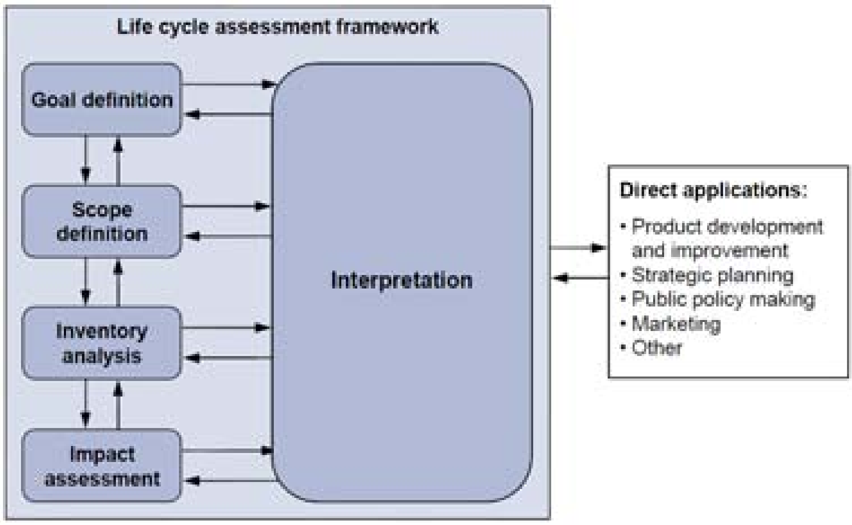



**Figure 2.2. Phases of the Life Cycle Assessment.**
*Source: JRC and IES* (2010b).

The goal of an LCA states the intended application, the reasons for carrying out the study, the intended audience – i.e. to whom the results of the study are intended to be communicated –, and whether the results are intended to be used in comparative assertions (ISO 2006a).

The decision-context is one key criterion for determining the most appropriate methods for the Life Cycle Inventory (LCI) model, i.e. the LCI modeling framework (i.e. "attributional" or "consequential"). The attributional life cycle model depicts its actual specific or average supply-chain plus its use and end-of-life value chain. The existing system is embedded into a static technosphere, i.e. the system is hence modelled as it is or was (or is forecasted to be). The consequential life cycle model depicts the generic supply-chain as it is theoretically expected in consequence of the analysed decision. The system interacts with the markets and those changes are depicted that an additional demand for the analysed system is expected to have in a dynamic technosphere that is reacting to this additional demand (JRC and IES 2010b).

The scope of an LCA should be sufficiently well defined to ensure that the breadth, depth and detail of the study are compatible and sufficient to address the stated goal. The scope includes (a) the description of the system under study, (b) its functions, (c) the functional unit, (d) the system boundaries, (e) the allocation procedure rules, (f) the methodology of impact assessment and the selected impact categories, (g) data requirements, (h) assumptions established and limitations, and other requirements (ISO 2006a). Functional unit and system boundaries are further explained as they are relevant issues for the current dissertation. Allocation alternatives are pointed out in the following.

A system may have a number of possible functions and the one(s) selected for a study depend(s) on the goal and scope of the LCA. The functional unit (FU) is a key element that defines the quantification of the identified functions (performance characteristics) of the product. The primary purpose of a functional unit is to provide a reference to which the inputs and outputs are related. This reference is necessary to ensure comparability of LCA results.

The system boundaries delimit the unit processes that are going to be included in the system. Defining system boundaries is partly based on choices that should be detailed and justified in order to provide confidence in the study. The system boundaries should notice which stages, unit processes and flows are to be considered in the study.

**Inventory analysis**

In the inventory analysis phase, or LCI, all emissions released into the environment and resources extracted from the environment along the whole life cycle of a product are grouped in an inventory. Energy and raw materials consumed, emissions to air, water, soil and solid waste produced by the system under study are split up into several subsystems and unit processes, and the data obtained is grouped in different categories in a LCI table. The main steps are data collection, the identification of the relevant and non-relevant elements, mass and energy balance, and allocation of the system burdens.

Many processes usually perform more than one function or output. The environmental load of that process needs to be allocated over the different functions and outputs. There are different ways to make such an allocation. According to ISO 14044:2006 (ISO 2006a), wherever possible, allocation should be avoided by either dividing the unit process to be allocated into two or more sub-processes or, in second place, by expanding the product system to include the additional functions related to the co-products. Where allocation cannot be avoided, the inputs and outputs of the system should be partitioned between its different products or functions according to physical relationships. The last option is to allocate the burdens according to other relationships, such as economical ones, between function or out-puts.

**Impact assessment**

In the third phase the LCI results or indicators of environmental interventions are translated, with the help of an impact assessment method, into environmental impacts at the midpoint and at the endpoint (UNEP 2011).

The midpoint and the endpoint approaches are two possible levels at which to quantify the environmental impacts. In the midpoint approach, the impact category indicator is defined close to the intervention (i.e. problem-oriented, such as global warming potential), while in the endpoint approach (i.e. damage-oriented) indicators are close to recognisable values for society, also called areas of protection – there are three human health, natural environment and natural resources (JRC and IES 2010a).

Issues such as choice, modeling and evaluation of impact categories can introduce subjectivity into the LCIA phase. Therefore, transparency is critical to the impact assessment to ensure that assumptions are clearly described and reported (ISO 2006a). During the initial goal and scope phase those issues must be selected and clearly indicated.

LCIA consists of four steps that are briefly described below. Classification and characterisation steps are mandatory, while normalisation is optional (Figure 2.3):

- Classification corresponds to a process in which all the environmental interventions identified in the inventory (inputs and outputs) are grouped in different impact categories or indicators, according to the environmental effects they are expected to contribute. Classification answers to the question *What does this emission contribute to?* For example, $CO_2$, $CH_4$ and $N_2O$ emissions are classified in the global warming potential category.

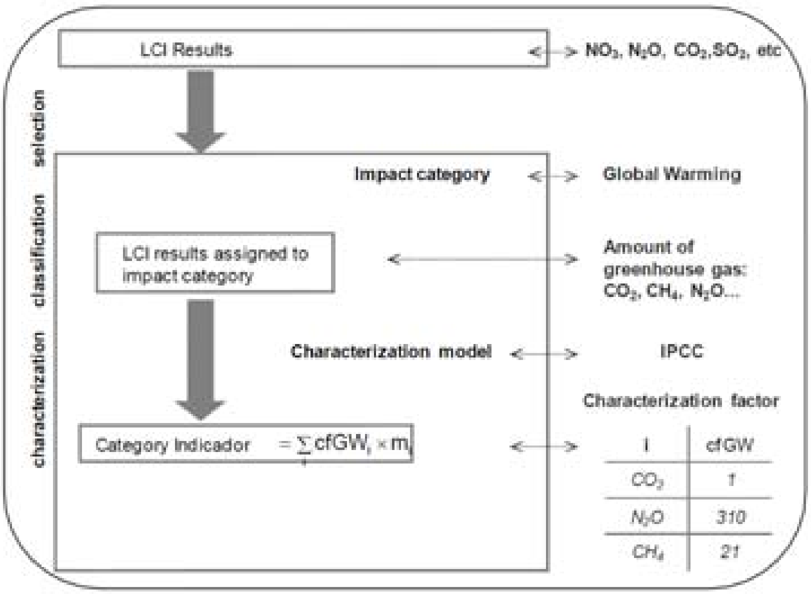



**Figure 2.3. Elements of the LCIA phases of an LCA.**
*Source: Montero et al.* (2009a).

- Characterisation is the calculation of impact category indicators using specific characterisation factors that are available to practitioners in literature, databases, and LCA support tools (JRC and IES 2010a). Characterisation factors are factors derived from characterisation model which allows all substances that contribute to this category to be reduced to a single reference substance. This step answers to the question *How much may it contribute?* For example, for global warming potential, the classified $CO_2$, $CH_4$, $N_2O$, etc. emissions are reduced to an equivalent substance: $CO_2$ equivalent.
- Normalization, aggregation and weighting are optional LCIA steps, according to ISO 14040:2006 and ISO 14044:2006 (ISO 2006a; 2006b). Limitations and subjectivity are important in normalization, and especially in grouping and weighting, and for this reason this practices are optional and often LCA studies avoid carrying out them.
  - → Normalization provides the contribution of each impact category in comparison to a reference by converting differing units into a common and dimensionless format, for example, relating the system's global warming potential to a country's yearly global warming potential. Normalization answers the question *Is that much?*
  - → Grouping is a semi-quantitative process that involves sorting and/or ranking results across impact categories. Ranking is based on value-choices. In our example, global warming potential is considered a burden of high priority by the international community.
  - → Weighting is the process of converting indicator results of different impact categories by using numerical factors based on value-choices. It may include aggregation of the weighted indicator results. Grouping and weighting answer to the question *Is it important?*

Issues such as choice, modeling and evaluation of impact categories can introduce subjectivity into the LCIA phase. Therefore, transparency is critical to the impact assessment to ensure that assumptions are clearly described and reported (ISO 2006a).

**Table 2.1. Environmental impact categories considered during the dissertation.**

| Acronym and full name | Description and unit |
|---|---|
| **ADP** **Abiotic depletion potential** | It is concerned with the protection of human welfare, human health and ecosystem health. It is related to the extraction of minerals and fossil fuels due to inputs into the system. The Abiotic Depletion Factor (ADF) is determined for each extraction of minerals and fossil fuels based on concentration reserves and the rate of de-accumulation. Unit: kg Sb eq. |
| **AP** **Acidification potential** | Acidifying substances cause a wide range of impacts on soil, groundwater, surface water, organisms, ecosystems and materials (buildings). AP factor emissions into the air are calculated with the adapted RAINS 10 model, describing the fate and deposition of acidifying substances. Unit: kg $SO_2$ eq. |
| **EP** **Eutrophication potential** | It includes all impacts due to excessive levels of macro-nutrients in the environment caused by emissions of nutrients into the air, water and soil. It is based on the stoichiometric procedure of Heijungs and Guinée (1992). Fate and exposure is not included. Unit: kg $PO_4^{3-}$ eq. |
| **GWP** **Global warming potential** | It can result in adverse affects upon ecosystem health, human health and material welfare. Climate change is related to emissions of greenhouse gases into air. The characterisation model as developed by the Intergovernmental Panel on Climate Change (IPCC) is selected for development of characterisation factors. Factors are expressed as for time horizon of 100 years. Unit: kg $CO_2$ eq. |
| **OLDP** **Ozone layer depletion potential** | Because of stratospheric ozone depletion, a larger fraction of UV-B radiation reaches the earth's surface. This can have harmful effects upon human health, animal health, terrestrial and aquatic ecosystems, biochemical cycles and on materials. This category is output-related and at global scale. The characterisation model is developed by the World Meteorological Organisation (WMO) and defines ozone depletion potential of different gasses. Unit: kg CFC eq. |
| **POP** **Photochemical oxidation potential** | Photo-oxidant formation is the formation of reactive substances (mainly ozone) which are injurious to human health and ecosystems and which also may damage crops. Photochemical Ozone characterization factor for emission of substances to air is calculated with the UNECE Trajectory model (including fate), and expressed in kg ethylene equivalents. Units: kg $C_2H_4$ eq. |
| **HT Human toxicity potential** | It concerns the effects of toxic substances on the human environment. Health risks of exposure in the working environment are not included. Characterisation factors are calculated with USES-LCA, describing fate, exposure and effects of toxic substances for an infinite time horizon. Units: kg 1.4 DB eq. |
| **TE Terrestric ecotoxicity potential** | It refers to impacts of toxic substances on terrestrial ecosystems, as a result of emissions of toxic substances into the air, water and soil. It is calculated with USES-LCA, describing fate, exposure and effects of toxic substances. Units: kg 1.4 DB eq. |
| **FWAE Fresh water aquatic ecotoxicity potential** | It refers to impact on fresh water ecosystems, as a result of emissions of toxic substances into to the air, water and soil. It is calculated with USES-LCA, describing fate, exposure and effects of toxic substances. Units: kg 1.4 DB eq. |
| **MAE Marine aquatic ecotoxicity potential** | It refers to impacts of toxic substances on marine ecosystems, as a result of emissions of toxic substances into the air, water and soil. It is calculated with USES-LCA, describing fate, exposure and effects of toxic substances. Units: kg 1.4 DB eq. |
| **CED Cumulative energy demand** | It aims to investigate the energy use throughout the life cycle of a good or a service. This includes the direct as well as the indirect uses. Characterization factors were given for the energy resources divided in: non renewable, fossil and nuclear, renewable, biomass, wind, solar, geothermal and water. Unit: MJ eq. |

*Source: Guinée et al.* (2001) *and Frischknecht and Jungbluth* (2003).

In the present dissertation, six mid-point impact categories – abiotic depletion, acidification, eutrophication, global warming, ozone layer depletion, and photochemical oxidation potentials – which had been defined by the CML 2001

(Guinée 2001), and cumulative energy demand (Frischknecht and Jungbluth 2003), as an energy flow indicator, are considered for the chapters applying LCA – Chapter 10 is additionally including environment and human ecotoxicity impact categories. The main environmental impact categories used throughout the dissertation are briefly presented in Table 2.1.

**Interpretation**

Interpretation is the phase of LCA in which the findings from the inventory analysis and the impact assessment are considered together. The interpretation phase should indicate the consistency of the results according to all the aspects defined during the goal definition and scope phase. In this last phase of LCA, it is necessary to outline conclusions, explain limitations that have occurred, and provide recommendations. The interpretation phase may involve the iterative process of reviewing and revising the scope of the LCA, as well as the quality of the data collected.

### *(iii) Strengthens and limitations of the LCA*

**Table 2.2. Main LCA problems along the stages "Goal and scope definition" and "Life cycle inventory analysis".**

| Phase | Problem |
|---|---|
| | ***Goal and scope definition*** |
| **Functional unit definition** | Multiple potential sources of error: (1) inaccurate reflection of the product system reality; (2) assigning functional units to multiple functions; (3) carrying out strict, functionally equivalent comparisons; and (4) when handling non-quantifiable or difficult-to-quantify functions. |
| **Boundary selection** | Considering the right amount of breadth and depth in one's boundary selection to inspire enough confidence in the interpretation of the LCA results. If not: (1) lead to incorrect interpretations and comparisons; or (2) lower decision maker confidence in making decisions based on the results. One of the main issues is the cut-off criteria chosen and the reasons. |
| **Social and economic impacts** | Recommendations based on LCA fail to address possible trade-offs between environmental protection and both social and economic concerns in the product life cycle. LCSA is being currently developed (section 2.1.3). |
| | ***Life cycle inventory analysis*** |
| **Allocation** | Many solutions can be suggested to allocate the environmental burdens of a multi-functional process. Hierarchy from the ISO 14044:2006 says division > expansion > physical allocation > economic allocation. |
| **Negligible contribution** | Truncating models of physical flows (cut-off criteria) in a system threatens to omit burdens with the potential of generating decision-altering impacts. |
| **Local technical uniqueness** | Product's life cycle can vary with location: (1) types and amounts of resources demanded and wastes produced; and (2) type of technology. |

*Source: Own elaboration from Reap et al.* (2008a).

LCA is a holistic, system analytic tool and is now an established and integral part of the environmental management tools. LCA is distinguished from other environmental assessment tools by two main features (Finkbeiner et al. 2010). First, the life cycle perspective: all phases ("from the cradle to the grave") of the life cycle of a product (good or service) have to be assessed with regard to all relevant material and energy flows. Second, cross-media environmental approach: all relevant environmental impacts are taken into account – both on the input side (use of resources) and on the output side (emissions to air, water and soil, including waste).

**Table 2.3. Main LCA problems along the stages "Life cycle impact assessment" and "Life cycle interpretation", as well as a general problem.**

| Phase | Problem |
|---|---|
| | *Life cycle impact assessment* |
| **Impact category and methodology selection** | Several issues: (1) lack of current standardization in the impact modeling approach and the categories selected; (2) certain impact categories such as land use, impacts on biodiversity, human toxicity and ecotoxicity suffer from significant data gaps; (3) difficulties with selecting impact category indicators and models; and (4) difficulties with assigning LCI results to impact categories. |
| **Spatial variation** | Those impact categories affecting local, regional and continental scales should use spatial information in order to accurately associate sources with receiving environments of variable sensitivity. Most assessments continue to ignore it. |
| **Local environmental uniqueness** | Each location has it owns geological, topographic and meteorological geometry and is affected by resource extraction or pollution unique. As a result, each local environment is uniquely sensitive to the stresses placed upon it by a particular product system's life cycle. |
| **Dynamics of the environment** | Issues related with ignoring system dynamics: (1) temporal factors such as timing of emissions, rate of release, and time-dependent environmental processes affect the impact of pollution; and (2) temporal patterns in product production, use, and disposal. |
| **Time horizons** | LCA integrates environmental impacts over time and is necessary to define whether finite or infinite limits are considered. Two options: implicit valuation (i.e., to decide a period or the infinite limit) or explicit valuation (i.e., to use a discount rate to the future emissions). |
| | *Life cycle interpretation* |
| **Weighting and valuation** | The use of weights can pose a challenge for two general reasons: (1) it may be difficult to assure that an elicited weight accurately reflects a decision maker's value; and (2) weights derived through different value (or preference) elicitation methods may not be comparable. |
| **Uncertainty in the decision process** | Problems associated with evaluating uncertainty in LCA fall into four categories: (1) modeling of uncertainty; (2) incorporation of multiple uncertainties; (3) completeness of analysis; and (4) cost of analysis. |
| | *All* |
| **Data availability and quality** | The main types of uncertainty due to data quality: (1) badly measured data ('data inaccuracy'); (2) data gaps; (3) unrepresentative (proxy) data; (4) model uncertainty; and (5) uncertainty about LCA methodological choices. |

*Source: Own elaboration from Reap et al.* (2008b).

According to ISO 14040: 2006 (ISO 2006a), LCA can support in:

- Identifying opportunities to improve the environmental performance of products at various points in their life cycle.
- Informing decision-makers in industry, government or NGOs.

- Selecting of relevant indicators of environmental performance, including measurement techniques.
- Marketing: e.g. implementing an ecolabelling scheme, making an environmental claim, or producing an environmental product declaration.

LCA exists in multiple forms, claims a growing list of practitioners, and remains a focus of continuing research. Despite its popularity and codification by international organizations, LCA is a tool in need of improvement (Reap et al. 2008a; Bare 2010). Table 2.2 and Table 2.3 explain technical problems and problematic decisions in LCA, according to the literature review by Reap et al. (2008a; 2008b).

#### *(iv) Software and databases of the dissertation*

Nowadays, there are many computer programs that help with the realisation of LCA dissertations and which incorporate databases of various processes – e.g. GaBi (IKP), Simapro (Pré Consultants), TEAM (ecobilan group), Umberto (IFEU), etc. Software packages introduce inventory data and model the stages of the product or service. In this dissertation Simapro software was used, always in the newest version available for each paper when it was prepared. It was developed by the company Pré Consultants in the Netherlands (PRé Consultants 2012).

The database used in this dissertation with regard to the secondary data was the ecoinvent database, also in its newest version available. It first was released in 2003 and was the result of a very large effort by Swiss institutes to update and integrate the well-known ETH-ESU 96, BUWAL250 and several other databases (SCLCI 2012).

### 2.1.2. Carbon Footprinting

Within the framework of LCA, this "new" GHG-gas-focused tool has recently become widely used and many guides have been developed. Its popularity is thanks to keeping things simple, to the many online tools for Carbon Footprinting, and because the calculated value can easily be grasped (Johnson 2008; Weidema et al. 2008).

In contrast with LCA, Carbon Footprinting (CF) methodology is confined to the analysis of GHG emissions. This indicator is calculated in a similar way to the global warming potential (GWP) used in LCA. Because PAS 2050:2008 (BSI 2008a), which was repealed by the currently PAS 2050:2011 (BSI 2011), is used in this dissertation, methodology explanations are focused on these specifications. Nevertheless, currently at least 16 other specifications exist from private or government initiatives.

Besides, despite the fact that some months ago the PAS 2050:2012-1 (BSI 2012), which provides with specific requirements for the cradle-to-gate stages of GHG assessments of horticultural products, was launched, it is not either used or explained here, as it was later to the chapters involved.

**Definition**

In accordance with PAS 2050 (BSI 2008b) 'product carbon footprint' refers to the GHG emissions of a product across its life cycle, from raw materials through production (or service provision), distribution, consumer use and disposal/recycling. It includes the greenhouse gases carbon dioxide ($CO_2$), methane ($CH_4$) and nitrous oxide ($N_2O$), together with families of gases including hydrofluorocarbons (HFCs) and perfluorocarbons (PFCs).

**Methodology**

The quantification of life cycle GHG emissions and removals for products shall be identified as either (BSI 2011):

- The cradle-to-grave quantification, which includes the emissions and removals arising from the full life cycle of the product. Business-to-consumer (B2C) approach.
- The cradle-to-gate quantification, which includes the GHG emissions and removals arising up to the point at which the product leaves the organization undertaking the assessment for transfer to another party. Business-to-business (B2B) approach.

According to BSI (2008b), there are four (five) steps within the CF, which are shown in Figure 2.4:

- Step 1: Building a process map. The goal of this step is to identify all materials, activities and processes that contribute to the chosen product's life cycle. Initial brainstorming helps to build a high-level process map that can then be refined through desktop research and supply chain interviews.
- Step 2: Checking boundaries and prioritisation. In this step system boundaries must be defined – i.e. the scope for the product carbon footprint (which life cycle stages, inputs and outputs should be included in the assessment) – and non-relevant contributions, identified. Any single source resulting in less than 1% of total emissions can be excluded from the analysis, with a maximum of 5% of the full product CF omitted. Furthermore, human inputs to processes, transport of consumers to retail outlets and animals providing transport are not included either.

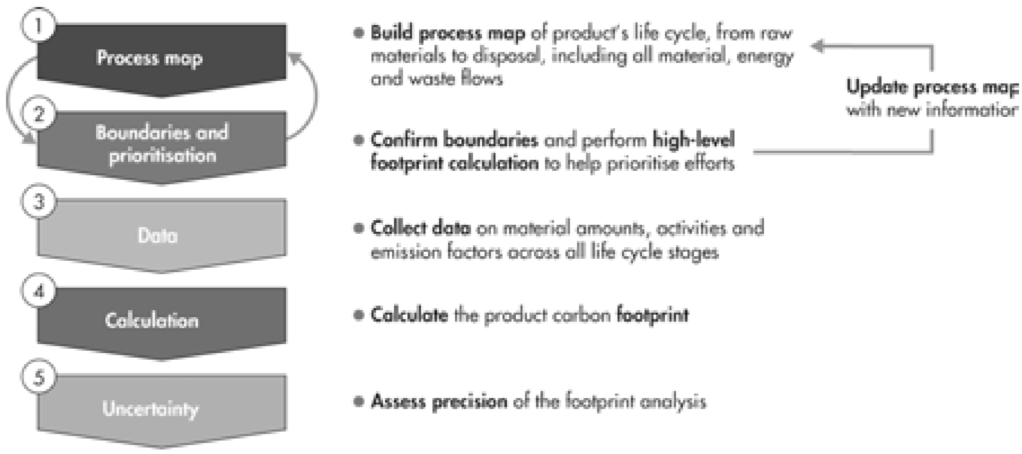



**Figure 2.4. Five steps to calculating the Carbon Footprinting.**
*Source: BSI* (2008b).

- Step 3: Collecting data. Guided by the initial calculations in Step 2, begin collecting more specific data following the requirements and recommendations of PAS 2050 (2008a; 2011), which will enable assessment of the Carbon Footprinting in more detail. Two types of data are necessary to calculate a carbon footprint: activity data and emission factors. Activity data refers to all the material and energy amounts involved in the product's life cycle; while emission factors provide the link that converts these quantities into the resulting GHG emissions: the amount of greenhouse gases emitted per 'unit' of activity. Both can come from either primary, preferably, or secondary sources.

- Step 4: Calculating the footprint. The equation for product Carbon Footprinting is the sum of all materials, energy and waste across all activities in a product's life cycle multiplied by their emission factors, which requires a mass balance to ensure all input, output and waste streams are accounted for. Once GHG emissions are calculated for each activity, it should be converted to $CO_2$ equivalent emissions using the relevant global warming potential in PAS 2050 (2008a; BSI 2011). Only the GHG emissions arising from the life cycle of products over a 100-year period are included.
- Step 5: Checking uncertainty (optional). There is a further optional step where technical uncertainty is checked in order to improve confidence in the results and any decisions made that are based on them.

The PAS 2050 reference documents (BSI 2008a; 2008b; 2011) provide with further information to deal with allocation, land use, recycling, delayed emissions, carbon storage, offsetting, etc.

### 2.1.3. Life Cycle Sustainability Assessment

The first conceptual ideas leading to the LCSA approaches of today can be attributed to the German Oeko-Institut with their method called "Product Line Analysis" (Oeko-Institut 1987) and later O'Brien et al. (1996). Last year, the UNEP/SETAC Life Cycle Initiative (UNEP 2011) published the report *Towards a Life Cycle Sustainability Assessment* that starts the first steps towards the standardisation of the LCSA. Nevertheless, the LCSA still remains in the theoretical level, as hardly any attempt is found for its application (Lindner et al. 2010; Capitano et al. 2011; Traverso et al. Accepted 2011; Vinyes et al. Accepted 2012).

#### *(i) Definition*

According to UNEP (2011), LCSA refers to the evaluation of all environmental, social and economic negative impacts and benefits in decision-making processes towards more sustainable products throughout their life cycle. Klöepffer (2007; 2008) and Finkbeiner et al. (2010) proposed a conceptual formula for LCSA framework [1], which was slightly modified by UNEP (2011). Combining LCA, Social Life Cycle Assessment (SLCA) together with LCA-type Life Cycle Costing (LCC) contributes to an assessment of products, providing more relevant results in the context of sustainability.

$$LCSA = LCA \cup LCC \cup SLCA \qquad [1]$$

#### *(ii) Methodology*

UNEP (2011) gives general indications and recommendations on how to start a life cycle sustainability assessment (LCSA) and effectively combine the three approaches:

- Step 1: Goal and scope definition. It describes the purpose, delimitation and the target audience of the study. LCA, LCC and SLCA have different aims and this must be understood clearly when working towards a combined approach. While taking into account these differences, a common goal and scope are strongly recommended when undertaking a combined LCSA.

- Step 2: LCSA inventory. In LCSA, the inventory compiles exchanges between unit processes and organizations of the product system and the external environment which lead to environmental, economic and social impacts. Because of the importance of achieving consistency with the three techniques, it is recommended that data is collected at the unit process and organizational level and that quantitative, but also qualitative and semi-quantitative information is used, mainly for SLCA.
- Step 3: Impact assessment. UNEP (2011) recommends that only the classification and characterization steps are implemented. Considering that characterization models are not available for all impact categories – including social, economic as well as environmental ones –, it may neither be possible to convert all LCSA inventory data into common units nor to aggregate them within each impact category. In addition, any aggregation and weighting of results of the three techniques used are not recommended because of the early stage of LCSA research and implementation and because the individual goals of each of the techniques applied are not directly comparable to the other.
- Step 4: LCSA interpretation. The evaluation results may help to clarify: if there are trade-offs between economic benefits and environmental or social burdens; which life cycle stages and impact subcategories are critical; and if the product is socially and environmentally friendly by understanding the impacts of the products and materials on society. It is recommended that the results are read in a combined fashion based on the goal and scope definition.

### *(iii) Presentation and integration of LCSA results*

Though partially in contradiction with the step 3, Finkbeiner et al. (2010) states that weighting happens in real world decision making anyway—at least implicitly— and thus ignoring or neglecting them is not a feasible option for developing evaluation schemes. Consequently, they propose two schemes that still allow to do weight in a transparent rather than an implicit way, if the decision-makers decide to apply quantitative weighting, and to graphically and in an easily understandable manner present results. Moreover, the chosen utility analysis approaches have the advantage that in addition to quantitative criteria also qualitative assessments can be considered and presented in the same assessment scheme. The two proposals are: the Life Cycle Sustainability Triangle (LCST) by Hofstetter et al. (1999) and the Life Cycle Sustainability Dashboard (LCSD) proposed by Traverso and Finkbeiner (2009; Accepted 2011). An example of application of each one is shown in Figure 2.5.

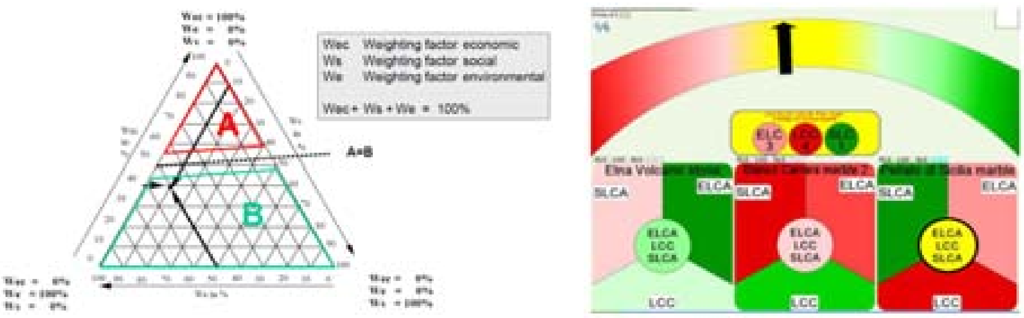



**Figure 2.5. LCSA graphical schemes: LCST of two alternatives A and B (left) and LCSD of three types of marble (right).**

*Source: Finkbeiner et al.* (2010).

### 2.1.4. Social Life Cycle Assessment

The discussion on how to deal with social and socio-economic criteria in LCA started in a SETAC workshop that took place in 1993 (Fava et al. 1993). Since then, many studies and research groups have contributed to SLCA development, dealing with important issues that still needed further research (O'Brien et al. 1996; Norris 2003; Hunkeler 2006; Norris 2006; Weidema 2006; Jørgensen et al. 2008; Klöepffer 2008).

In 2004, the UNEP/SETAC life cycle initiative recognized a need for a task force on the integration of social criteria into LCA. As a result, the *Guidelines for Social life cycle assessment of products* (UNEP 2009) were launched in 2009. These Guidelines ground the assessment of the social and socioeconomic aspects — referred simply as social aspects — into the LCA framework. The proposed framework is in line with the ISO 14040 and 14044 standards (ISO 2006a; 2006b) for LCA but adapted for the social aspects (Parent et al. 2010). A year later, the *Methodological Sheets for the Subcategories of SLCA* (UNEP 2010b), "the Methodological Sheets", were made available to support practitioners engaging in the field (Benoît-Norris et al. 2011).

Last years, several studies dealing with methodological and practical questions for SLCA were published (Jørgensen 2010; Lehmann et al. 2011), as well as few case studies partially performing SLCA (Andrews et al. 2009; Kruse et al. 2009; Dreyer et al. 2010; Ciroth and Franze 2011; Franze and Ciroth 2011; Peri et al. 2011; Vinyes et al. Accepted 2012).

#### *(i) Definition*

UNEP (2009) defines SLCA as a social impact (and potential impact) assessment technique that aims to assess the social and socio-economic aspects of products and their potential positive and negative impacts along their life cycle encompassing extraction and processing of raw materials; manufacturing; distribution; use; re-use; maintenance; recycling; and final disposal.

The ultimate objective for conducting a SLCA is to promote improvement of social conditions and of the overall socio-economic performance of a product throughout its life cycle for all of its stakeholders (UNEP 2009). Additionally, it allows identification of key issues, assessing, and telling the story of social conditions in the production, use, and disposal of products (Benoît-Norris et al. 2011).

#### *(ii) Methodology*

As was aforementioned, SLCA follows the same four main iterative steps as those used in LCA – i.e. goal and scope definition, life cycle inventory analysis, impact assessment and interpretation –, consequently only relevant issues and main differences are developed in this section.

*Functional unit*

SLCA often works with semi-quantitative or qualitative information, which are not relevant to directly express per unit of process output. Several authors recommended the aggregation to be carried out by the use of the activity variable "working time" (Hunkeler 2006; UNEP 2009).

*Stakeholders, subcategories and indicators*

The UNEP (2009) methodology has listed 31 subcategories according to stakeholder throughout the life cycle of a product, which are listed in Table 2.4. A stakeholder category is a cluster of stakeholders that are expected to have shared interests due to their similar relationship to the investigated product systems. Additional categories of stakeholders or further differentiations or subgroups can be defined and used by the practitioners (Benoît et al. 2010b).

A Methodological Sheet was developed for each of the 31 subcategories of assessment outlined in the Guidelines. Each sheet includes a subcategory definition tailored to SLCA, an explanation of how the subcategory relates to sustainable development, information on data assessment, including examples of inventory indicators, units of measurement, and data sources, along with a reference section that points the user to further information (UNEP 2010b).

**Table 2.4. Stakeholder categories and subcategories for SLCA.**

| Stakeholder categories | Subcategories | |
|---|---|---|
| **Stakeholder "worker"** | Freedom of Association and Collective Bargaining<br>Child Labour<br>Fair Salary<br>Working Hours | Forced Labour<br>Equal opportunities/Discrimination<br>Health and Safety<br>Social Benefits/Social Security |
| **Stakeholder "consumer"** | Health & Safety<br>Feedback Mechanism<br>Consumer Privacy | Transparency<br>End of life responsibility |
| **Stakeholder "local community"** | Access to material resources<br>Access to immaterial resources<br>Delocalization and Migration<br>Cultural Heritage<br>Safe & healthy living conditions | Respect of indigenous rights<br>Community engagement<br>Local employment<br>Secure living conditions |
| **Stakeholder "society"** | Public commitments to sustainability issues<br>Contribution to economic development | Prevention & mitigation of armed conflicts<br>Technology development<br>Corruption |
| **Value chain actors** | Fair competition<br>Promoting social responsibility | Supplier relationships<br>Respect of intellectual property rights |

*Source: UNEP* (2009).

*Inventory and data*

Following the definition of stakeholder categories and subcategories of impact, indicators need to be defined in order to conduct the inventory. More than 200 indicators are proposed in the methodological Sheets (UNEP 2010b), and other can be added by the user.

It is not necessary to collect primary data at every unit process across a product life cycle. Instead, there is a need to combine the approaches of prioritizing data gathering and making use of average or proxy data where feasible. First, data on unit process activity variables should be collected to provide a first set of information on the relative importance of the unit processes, for example using working time.

Second step is a hotspot assessment, which provides more information on where the most important potential social impacts may be located within the product life cycle (Benoît et al. 2010b).

Social hotspots are unit processes that are within a sector and region that has high risks of negative impact or high opportunities for positive impact. Recently, the Social Hotspot Database, developed by Benoît and Norris among others (SHDB 2011), was released. It is an international meta-analysis assessing social themes and social categories that states the level of risk or opportunity that they exist at the country or sector-country-specific level.

Regarding SLCA, not only quantitative data is used, as is the case in LCA and LCC. Semi-quantitative and qualitative data that conforms to the ISO 14040 standard (ISO 2006a) also have to be integrated, according to the Guidelines (UNEP 2009).

*Impact assessment*

Social impacts have no relation to the processes themselves but rather to the conduct of the companies performing the processes (Jørgensen 2010). The social impacts of a product are not well addressed by the existing SLCA methods and the Guidelines stress the need for research in this field (UNEP 2009). Some methods have been developed to provide estimates of social impacts directly at the unit process activity (Hunkeler 2006); whereas others provide estimates of how unit processes can lead to potential human health consequences through socio-economic pathways (Weidema 2006). This and other method proposals are better explained in Parent (2010).

#### *(iii) Need of further research*

The authors of the guidelines recognized that the SLCA methodology is still in its early days, and that the technique will be further refined in the coming years (Benoît et al. 2010b). The main concerns regarding SLCA are the criteria for the selection of social indicators and the valuation method, as well as the difficulties to relate the social results for each indicator to the functional unit of the product-system. Besides, the existence of social databases is limited and on-site company data collection for processes along the life cycle of a product is highly time demanding and not always feasible (Klöepffer 2008; Swarr 2009; Finkbeiner et al. 2010; Jørgensen 2010; Ciroth and Franze 2011).

### 2.1.5. Life cycle costing

LCC is the oldest of the three life cycle techniques. Developed originally from a strict financial cost accounting perspective, in recent years LCC has gained importance. The motivation is that, for many products, the purchase price reflects only a minority of the costs that will be caused by the product. The origins of LCC go back to 1933. Today there are a variety of approaches for the calculation of cost and performance along life cycle (UNEP 2011).

#### *(i) Definition*

Among them, a working group of SETAC defined the term Environmental Life cycle costing (LCC) in the senses of "…an assessment of all costs associated with the life cycle of a product that are directly covered by any one or more of the actors in the product life cycle (…) with complimentary inclusion of externalities that are anticipated to be internalized in the decision-relevant future" (Hunkeler et al. 2008).

This work was the basis for the published code of practice (Swarr et al. 2011). According to Finkbeiner et al. (2010), approaches that fall under this definition are particularly meaningful with regard to LCSA, because they are consistent with the environmental dimension and avoid double-counting.

***(ii) Methodology***

It is usually carried out in the well-known four phases of life-cycle-thinking tools: define a goal, scope and functional unit; inventory costs; aggregate costs by cost categories; and interpret results. Some several issues are merely highlighted by UNEP (2011), the discount rate, the viewpoint of the life cycle actor (whether supplier, manufacturer, user or consumer), the definition of cost categories, data availability, and data quality assessment and assurance.

### 2.1.6. Territorial Planning Tool: Geographic Information Systems

Geographic Information Systems, abbreviated as GIS, are special-purpose digital databases in which a common spatial coordinate system is the primary means of reference. The term GIS refers both to the specific software and to the data sets to be used with the software. GIS software contains subsystems for: (1) data input; (2) data storage, retrieval, and representation; (3) data management, transformation, and analysis; and (4) data reporting and product generation. A GIS supports spatial data collection, analysis, and decision making (Núñez 2011).

The release of free and accurate GIS software and data sets provides a relatively novel spatial and temporal resolved data source which can be applied together with other environmental assessment tools such as LCA. GIS and LCA are different and complementary. The former is specifically designed to organise and analyse spatial data, while the latter inventories and assesses product system data without providing geospatial information. Although the lack of spatial differentiation of LCA has been frequently criticised, it was not until recently that LCA began to take advantage of GIS properties (Geyer et al. 2010; Gasol et al. 2011; Núñez 2011).

According to Gasol (2009), GIS provides reliable territorial scenarios based on the scientific assessment of geographical, climate or other environmental variables. These scenarios can supply huge quantities of data that is usable to make LCA inventories. At the same time, the application of LCA provides environmental impact results on the performance of the proposal for territorial organization and new environmental data to be taken into account in territorial planning. The final planning obtained should be that of less impact and will be the result of different interactions between both tools, LCA and GIS.

In this dissertation, the software MiraMon was used (CREAF 2012). The MiraMon is developed cooperatively by various members of the Consolidated Research Group GRUMETS belonging to the Centre for Ecological Research and Forestry Applications (CREAF) and to the Autonomous University of Barcelona (UAB), and seeks to provide low cost, powerful and rigorous software.

## 2.2. Systems of study and experimental set

Two groups of case studies were studied in this dissertation: the ones related with compost production and those related with horticultural fields. This section describes the case studies considered in each chapter and the fieldworks included. All the case studies were located in Catalonia (Figure 2.6)

Catalonia is a nationality, constituted as one of the seventeen autonomous regions (known as communities) of Spain. It is situated in the north-eastern part of the Iberian Peninsula bordering France and Andorra in the north, and occupying about 32,000 km$^2$. Currently, it has more than 7.5 million of inhabitants. The climate of Catalonia is Mediterranean due to its location, even though diverse due to its orography. The areas lying by the cost, where the case studies of the dissertation are located, have coastal and pre-coastal climates with precipitations between 500-1000 mm and average temperatures between 14-15°C.

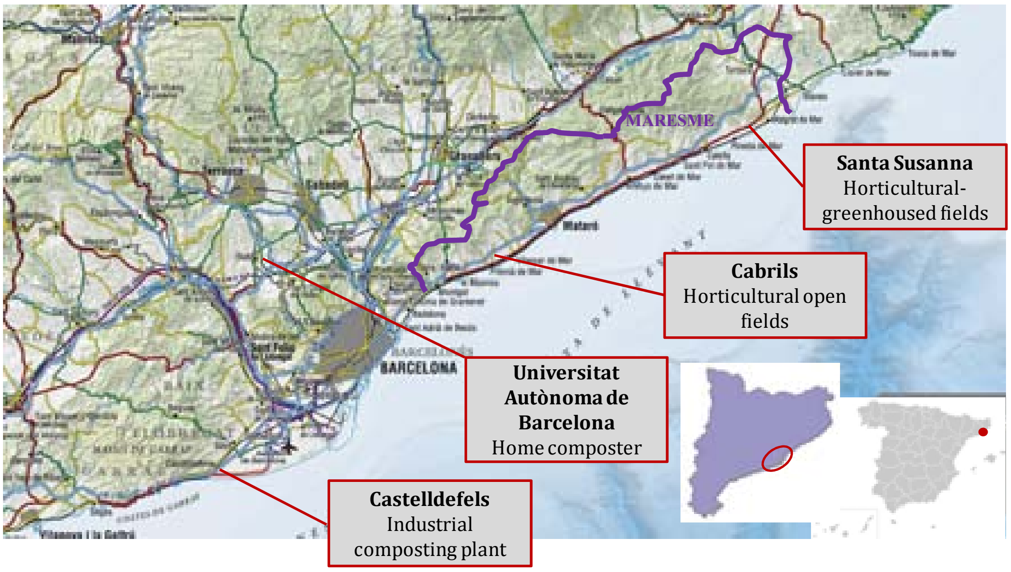



**Figure 2.6. Case studies location in the surroundings of Barcelona, Catalonia, Spain.** *Source: Adapted from ICC* (2012).

## 2.2.1. Composting at home and full scale

In Chapter 3 of the dissertation, two types of composting – at full and home scale – are studied, including inventory flows analysis, gaseous emissions quantification, feedstock and final product characterization and environmental impacts assessment. The monitoring of these two case studies and the results obtained were the outcome of close collaboration between the Grup d'Investigació en Compostatge (GICOM) from the Universitat Autònoma de Barcelona and Sostenipra research group (including ICTA, IRTA and Inèdit Innovació SL). Moreover, the Diputació de Barcelona partially funded the study. The environmental results obtained are used in all the Chapters apart from 7 and 10, as the source for compost production data.

### *(i) Industrial composting plant*

The composting plant analysed is located in the town of Castelldefels (Figure 2.6), 15 kilometres to the south of Barcelona (41°17'19″N, 1°58'20″E, Spain). The Castelldefels composting plant began operating in 1998 and its management depended on METROCOMPOST SA. It closed down at the beginning of 2009 due to the reiterated complains by the neighborhood that had considerable increased the previous years.

The composting plant had a treatment capacity of 15,000 tons. Three types of biodegradable wastes were treated – household wastes and organic waste from markets (thereafter OFMSW), and green waste – which were obtained by source-separated collection systems from six nearby municipalities and the main Barcelona market suppliers. Figure 2.7 shows the several steps and processes involved in the operation of the plant. It used decomposition in tunnels with forced aeration and maturation in turned windrows (see section 1.3.2). Two different lines were set for household (3 tunnels) and for market wastes (2 tunnels), i.e. they were not mixed. At the end of the chain, the processed material was screened to separate the mature compost from the pieces of pruning waste that were not totally decomposed (which re-enter the process) and other impurities (which are dumped in a sanitary landfill). All the stages lasted approximately 10 weeks. Some pictures of the composting plants are shown in Figure 2.8. More details about the composting plant operation are presented in Martínez-Blanco et al. (2009b; 2010).

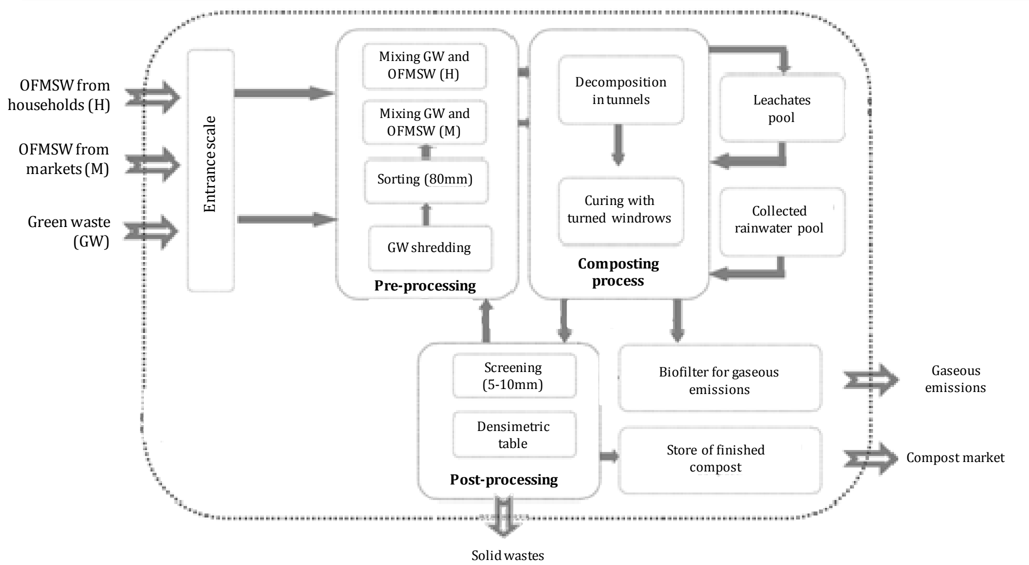



**Figure 2.7. Diagram of the composting stages in the Castelldefels composting plant.**
*Source: Own elaboration from Cisneros* (2006) *and Huerta et al.* (2010).

All the stages took place on an enclosed building and the gaseous emissions were collected and treated in biofilters. They treat the exhaust gases from composting and have the entire surface open to the atmosphere so that the outlet gases become atmospheric pollutants. The biofilters were originally filled with wood chips previously used as bulking agent in the composting process (Colón et al. 2009). Regarding leachates and rainwater, they were collected and (re-)introduced for watering of windrows and tunnels.

**Data obtaining**

A combination of a questionnaire addressed to plant managers and systematic sampling work was used to obtain the necessary data from the composting plant. Data on amounts of treated OFMSW and green waste, refuse and compost production, electricity and water consumption, etc., were obtained through the questionnaire.

Regarding gaseous emissions, as biofilters are area emission sources, several sampling points were established on the surface of each biofilter, to ensure the representativeness of the measure as the variability of air velocity and pollutant concentration was considered (Colón et al. 2009). Air velocity and pollutant concentration were sampled once a week, when feasible, during some periods of 2008. The emissions of $NH_3$ were determined in situ, while air samples were taken for the quantification in laboratory of $N_2O$, $CH_4$ and VOC (see section 2.2.1.iii). Two samples of initial feedstock materials and two samples of final compost were also taken and characterized (see section 2.2.1.iii).

*(a) Pre-processing*

*(b) Decomposition tunnels*

*(c) Maturation stage*

*(d) Gaseous treatment biofilters*

**Figure 2.8. Four stages at the industrial composting plant (Castelldefels, Catalonia).**
*Source: Julia Martínez Blanco.*

### *(ii) Home composter*

Data from home composting system were obtained from an experimental composter controlled and managed by the authors and placed outdoors of the School of Engineering of the Universitat Autònoma de Barcelona in Cerdanyola del Vallès (41˚29'55″N, 2˚6'46″E, Spain), in shadow conditions. The experiment took from November 2008 to February 2009.

OFMSW and green waste, previously shredded, were used to feed the composter. The composter had a lateral system of natural ventilation to guarantee aerobic conditions. The organic waste was poured to the upper part of the composter and was extracted through the lower panels. The composter was weekly fed, mixed and watered, if necessary. All the process lasted approximately 12 weeks. Some pictures of the composter operation are shown in Figure 2.9. More details about the composting plant operation are presented in Martínez-Blanco et al. (2010) and Colón et al. (2010).

(a) Feedstock before shredding (b) Recently fed composter (c) Composter

**Figure 2.9. Home composter at the UAB (Cerdanyola del Vallès, Catalonia).**
*Source: Julia Martínez Blanco.*

**Data obtaining**

A systematic sampling work was used to obtain the necessary data from the composter. Amounts of OFMSW and green waste shredded and poured were periodically recorded during the 100 days of the experiment.

Regarding gaseous emissions, samples were taken in only one point on the upper surface area of the composter assuming a homogeneous emission in all the composter small emission surface area. Air velocity and pollutant concentration were sampled once a week during the entire experiment. The emissions of $NH_3$ were determined in situ, while air samples were taken for the quantification in laboratory of $N_2O$, $CH_4$ and VOC (see section 2.2.1.iii). Several samples of initial feedstock materials and final compost were also taken and characterized (see section 2.2.1.iii).

### *(iii) Field and laboratory works*

As was aforementioned, gaseous emissions were measured at either biofilter or composter surface and organic feedstock and final compost were characterised. The analysed parameters are summarised in Table 2.5. Field and laboratory works were mainly performed by Joan Colón from GICOM research group.

**Determination of gaseous emissions**

The methodology developed by Colón et al. (2009) and Cadena et al. (2009) for the sampling and determination of gaseous emissions in industrial composting facilities was used to calculate gaseous emissions from the composting plant. This methodology was also adapted to determine the home composting emissions, as was done by Colón et al. (2010).

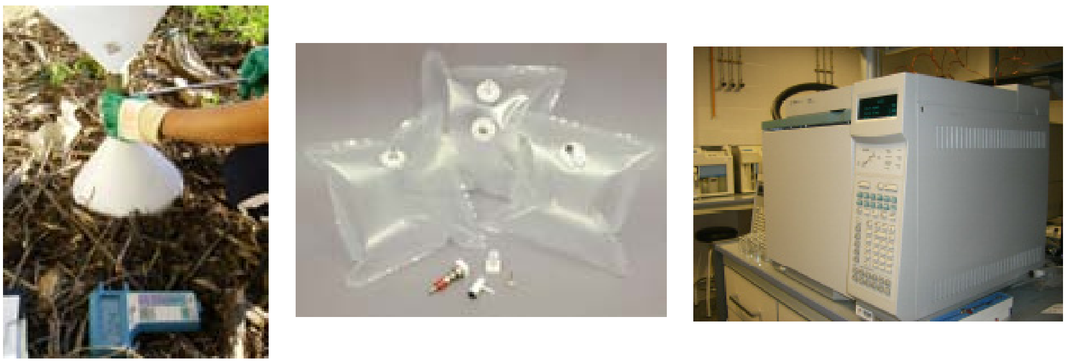
Air velocity on the emission surface was determined using a thermo-anemometer and a specially designed Venturi tube to increase airflow velocity (Figure 2.10a). Ammonia concentration in gaseous emissions was determined in situ using an ammonia sensor; whereas gaseous samples were also taken in Tedlar bags (Figure 2.10b) for the laboratory determination of VOC, methane and nitrous oxide. The three later gases were determined by gas chromatography (Figure 2.10c) – using a Flame Ionization Detector for VOC and methane and an Electron Capture Detector, for nitrous oxide – from the gas samples taken in the composting plant. More details of the methodology are available in Martínez-Blanco et al. (2010) and Colón et al. (2009; 2010; 2012).

**Table 2.5. Analysed parameters for industrial and home composting systems.**

| Parameter | Units | Parameter | Units |
|---|---|---|---|
| *Gaseous emissions* | | *Parameters analysed for solid materials (feedstocks and final compost)* | |
| Methane ($CH_4$) | g $m^{-3}$ | Moisture | %, wb |
| Ammonia ($NH_3$) | g $m^{-3}$ | Organic matter | %, db |
| Nitrous oxide ($N_2O$) | g $m^{-3}$ | pH (extract 1:5 w:v) | - |
| Volatile Organic Compounds (VOC) | g $m^{-3}$ | Electrical conductivity (extract 1:5 w:v) | mS $cm^{-1}$ |
| Air velocity | $m^3$ $h^{-1}$ | N-Kjeldhal | %, db |
| | | Static respirometric index | mg $O_2$ $g^{-1}$ OM $h^{-1}$ |
| | | Zinc (Zn) | mg $kg^{-1}$ |
| | | Copper (Cu) | mg $kg^{-1}$ |
| | | Nickel (Ni) | mg $kg^{-1}$ |
| | | Chrome (Cr) | mg $kg^{-1}$ |
| | | Lead (Pb) | mg $kg^{-1}$ |
| | | Cadmium (Cd) | mg $kg^{-1}$ |

**Solid material analytical methods**

For solid samples – either feedstock material or final compost – moisture and organic matter content, pH, electrical conductivity, N-Kjeldhal, heavy metals content and bulk density were determined following the standard methodology proposed by the US Department of Agriculture and the US Composting Council (2001). Porosity was assessed by air pycnometry (Ruggieri et al. 2009b).



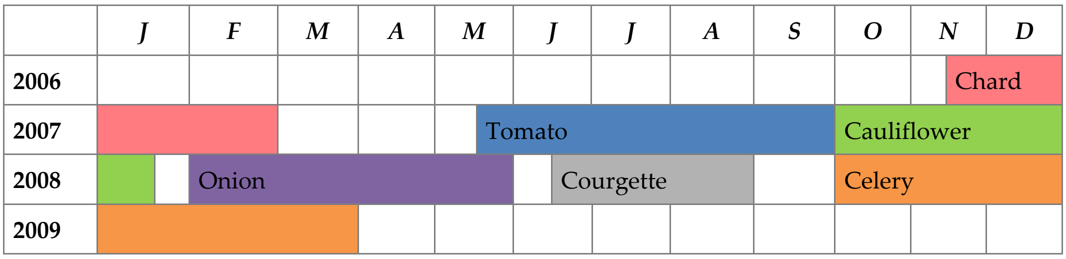
*(a) Air velocity measure* *(b) Tedlar bags* *(c) Gas chromatographer*

**Figure 2.10. Gaseous emissions sampling in industrial and home composting.**
*Source: Julia Martínez Blanco and Internet.*

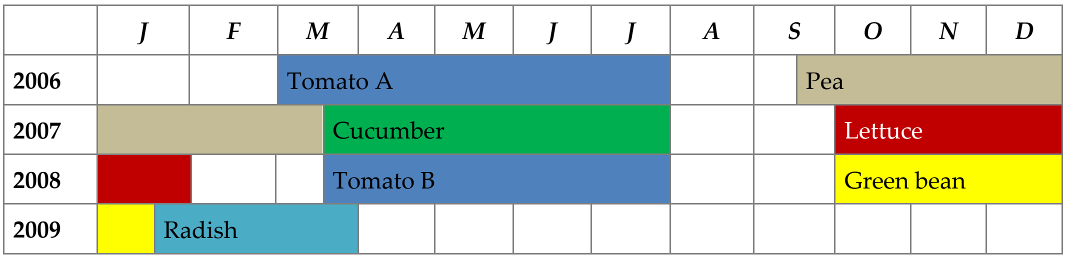
The static respirometric index (SRI), which was used as a measure of the biological stability of the material, was determined following the methodology proposed by Barrena et al. (2005). Apart from informing about the quality of the applied compost, these values were worthy to refer the several inputs and outputs of the systems (e.g. energy, water, emissions) to the functional unit and using real data.

### 2.2.2. Mediterranean horticultural crops

In Chapters 4-6 and 8 and 9 of the dissertation, real data on compost application to several horticultural crops is used. The experimental fields were operated within the project *Aplicación de Compost de Fracción Orgánica de Residuos Sólidos Municipales en la fertilización de cultivos hortícolas en la comarca del Maresme* (461/2006/3-2.3 and A246/2007/2-02.3) during the period 2006-2009 (henceforth in this section, Framework Project). The main goal of the Framework Project was to disseminate and demonstrate the technical feasibility of the use of compost produced from the organic fraction of the municipal solid waste as a fertilizer and/or organic amendment, determining the dose and criteria for application in a vegetable crop rotation. The crops included are shown in Figure 2.11.




*(a) Crop rotation open field*




*(b) Crop rotation greenhouse*

**Figure 2.11. Crops cultivated in the open field (a) and greenhouse (b) experimental rotations of the Framework Project.**
*Source: Muñoz et al.* (2009).

The Framework Project was coordinated by the Institute of Agriculture and Food Research and Technology (IRTA-Cabrils), in charge of the field experimental design (i.e. fertilizers dosages, irrigation program, etc.) and monitoring (i.e. water and fertilizers balances, bio-productivity, nitrogen analysis, etc.) and was developed with three partnerships. Universitat Autònoma de Barcelona (ICTA - Institute of Environmental Science and Technology) made the environmental impact assessment; Julia Martínez Blanco was the researcher who mainly works on this task. Finally, Universitat Politècnica de Catalunya (ESAB - Escola Superior d'Agricultura de Barcelona) was responsible for the soil and compost analysis and the Federació Selmar was in charge of the operation of the experimental fields and the results dissemination.

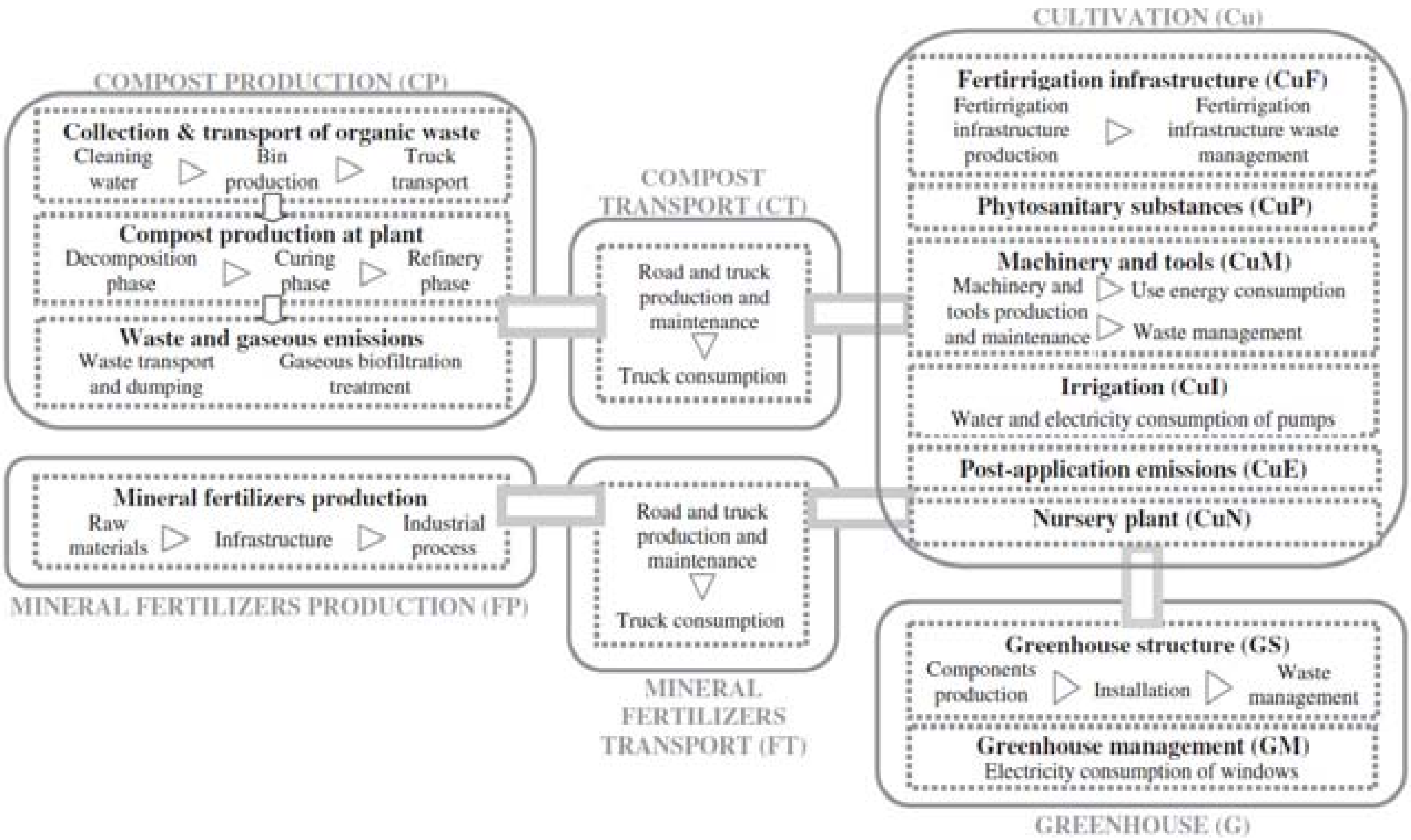



**Figure 2.12. Processes included in the stages and sub-stages considered in the system.**
*Source: Martínez-Blanco et al.* (2011b).

Apart from agricultural and analytical tasks, environmental assessments for all the crops included in the Framework Project were conducted, using the LCA methodology and were presented in detail in the Final Report of the project (Muñoz et al. 2009). For each crop, three different cultivation options were assessed, according to the type of fertilization applied. The system boundaries for the environmental impact assessment were including compost and mineral fertilizers production, compost and mineral fertilizers transport and the cultivation stage (Figure 2.12). The compost used in the crops was produced in the Castelldefels composting plant (section 2.2.1).

Nevertheless, the Framework Project included more crops than assessed in this dissertation. The results of four crops for open field and one for greenhouse were effectively included in the chapters of the dissertation and discussed from several approaches (i.e. bioactive compounds, fertilizing, functional units, etc.). Table 2.6 shows, for each chapter, the crops assessed, the number of blocks considered and some other interesting information.

**Table 2.6. Crops and options assessed in each chapter of the dissertation.**

| Chapter | Crop/s | | Plot | | Fertilizing | | | Block | Additionally |
|---|---|---|---|---|---|---|---|---|---|
| | T | C | OF | GH | $C_H$ | $C_L$ | M | | |
| **4** | x | | x | | x | x | x | 4 | Detailed inventories. |
| **5** | x | | x | x | | x | x | 3 | Detailed study of greenhouse structure and components. |
| **6** | | x | x | | x | x | x | 3 | Bioactive compounds analysis. New functional units. |
| **8** | x | x | x | x | | x | x | 3 | LCA and CF comparison. |
| **9** | x | | x | | - | - | - | - | Only fertilizers characteristics and post-application emissions. |

*Crop: T, tomato; C, cauliflower. Plot options: OF, open field; GH, greenhouse. Cultivation options regarding fertilization: M, mineral fertilizers; $C_L$, mineral fertilizers and compost in a low dosage; and $C_H$, mineral fertilizers and compost in a high dosage.*

### *(i) Experimental fields*

**Location and characteristics**

As was shown in Figure 2.6, agricultural practices data were obtained in two experimental plots located in the county of Maresme. The plots were divided in three sub-plots, for the three cultivation options (Figure 2.13). Similarly, each sub-plot was divided in a block design with four replicates for each cultivation option (a total of 12 blocks). However, for most of the chapters, only 3 blocks (1, 2 and 3) were considered for the field data as block 4 obtained bad results for all the cultivation options almost certainly due to the edge effect (Table 2.6).

The open field experimental plot is located in Santa Susana (41°38′27″N, 2°43′00″E, Spain) and has an area of 414 $m^2$, divided in three sub-plots of 138 $m^2$, one for each cultivation option. The soil was a Typic Xerothent with a loamy sand texture in the first 20 cm and sandy loam at greater depth. The site has a Mediterranean climate with an average annual evapotranspiration (ET0) of 782 mm and rainfall 641 mm for the period 1990-2008.

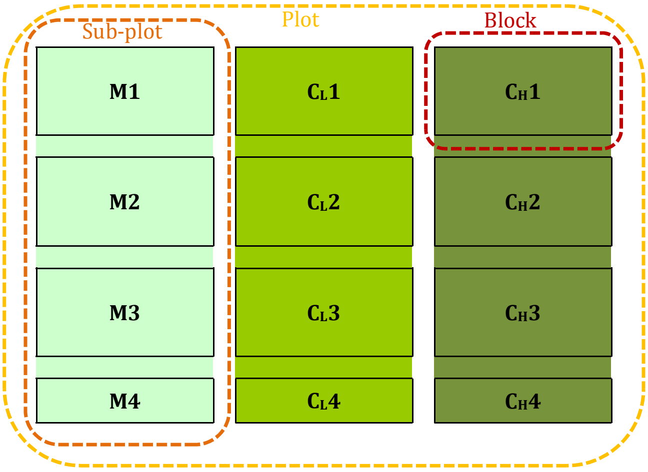



**Figure 2.13. Diagram of the plots and subsequent divisions.**
*Source: Julia Martínez Blanco.*

Regarding the plot protected under a greenhouse, it was located in Cabrils (41˚31′31″N, 2˚22′07″E, Spain) and has an area of 105 m², divided in three sub-plots of 35 m², one for each cultivation option. The soil was a Typic Xerothent with a sandy-loam texture in the first 20 cm and at greater depth. The site has a Mediterranean climate with an average annual evapotranspiration (ET0) of 900 mm and rainfall 625 mm for the period 1995-2009. Figure 2.14 shows a picture for the two plots.

*(a) Open field experimental plot*

*(b) Greenhouse experimental plot*

**Figure 2.14. General view of the two experimental plots.**
*Source: Muñoz et al.* (2009).

A Mediterranean round arched greenhouse with vertical sidewalls was considered in the greenhouse experimental plot (Figure 2.15). It was made up of several spans, with steel frames, concrete foundations and covered by low-density polyethylene film. The dimensions of the greenhouse were extrapolated to a plot of 5,000 m² per cultivation option, to avoid scale problems. It was made up of six spans, with steel frames, concrete foundations and covered by low-density polyethylene film (thickness 0.2 mm). The multi-tunnel greenhouse was 104 m long, 8 m wide per span, 4m high at the gutter and 5.5 m high at the ridge. Chapter 5 includes a 3D diagram of the greenhouse considered and a list of all its components.

*(c) Greenhouses. Exterior view* *(d) Greenhouses. Inside view* *(e) Greenhouses. Lateral view*

**Figure 2.15. Several views of commercial greenhouses of the type multi-tunnel.**
*Source: Antón* (2004).

**Cultivation options**

For all the crops of the Framework Project, and thus for tomato and cauliflower, three cultivation options were considered with regard of the fertilization:

- Option $C_H$: a high-dose of compost (and the rest of nitrogen necessities supplied by mineral fertilizers).
- Option $C_L$: a low-dose of compost and the rest of nitrogen necessities supplied by mineral fertilizers.
- Option M: only mineral fertilizers (including $HNO_3$, $KNO_3$, $KPO_4H_2$ and $K_2SO_4$).

Compost alone without mineral nitrogen added was not an option in open field, due to the particular characteristics of the irrigation water used in the field, with a high concentration of nitrates. Therefore, all the options were actually supplied with mineral fertilizers through irrigation water. Nevertheless, in Chapter 4, this fact was not taken into account for the environmental assessment- but it was for the compost dosage application – and option $C_H$ is not including mineral fertilizers.

According to the Catalan Decree 136/2009 (Catalan Parlament2009), the maximum amount of compost per county is restricted to do not apply more than 170 kg N ha$^{-1}$ year$^{-1}$ from organic sources. Nevertheless, this regulation was launched after the horticultural experiments involved in this dissertation, thus it was not taken into account for the compost dosages applied and option $C_H$ is, in some cases, exceeding the abovementioned limit.

Regarding greenhouse field, in spite of irrigation water having lower nitrate concentration – or probably due to this fact –, option with high-dose of compost was not included in the Chapter 5, as yield was much lower than acceptable by the farmer.

**Plots management**

Cultivation for the two plots, the several crops and cultivation options, were following the best available techniques for integrated cultivation management (MAPA 2002) to compare efficient systems in resources, energy and emissions.

The doses of compost and mineral fertilizers were calculated depending on the expected crop production and the nitrogen concentrations of both fertilizers, for each treatment. Compost is not normally applied to every crop, but in cycles of 1–2 years, to distribute environmental burdens associated with the transport and machinery used.

Irrigation water was supplied from close wells and, for greenhouse plots, rain water was collected and also used for irrigation. Crops were – micro-sprinkler or drip – irrigated depending on the tensiometer reading that determined the matric water potential evapotranspiration demands of the soil.

Pesticide application was based on the regulations for integrated agriculture (MAPA 2002), at the minimums specified by the Spanish registry of phytosanitary products (MAAMA 2010).

Greenhouse management only involved the opening and closing of the roof and the side-wall openings. As was aforementioned (section 1.1.4), Mediterranean greenhouses do not use lighting or heating the crops.

### *(iii) Field and laboratory works*

For all the crops studied among the Framework Project, yield and harvest quality were assessed at the block level and later average values per cultivation option were provided. All these analyses were mainly carried out by IRTA-Cabrils research group. Bioactive compounds were additionally measured for cauliflower (Chapter 6) by Massimo Castellari research group (Food Technology group from the Institute of Research and Technology in Agri-food Sector, Monells, Catalonia). Other related analyses, as soil and compost characterization, which were mainly carried out by ESAB and IRTA-Cabrils, are briefly presented at the end of the section. Table 2.7 summarises the analysed parameters.

**Table 2.7. Analysed parameters for the horticultural crops.**

| Parameter | Units | Parameter | Units |
|---|---|---|---|
| *Harvest and product quality analysis* | | *Soil analysis at three different depths* | |
| Total yield | t $ha^{-1}$ | Moisture | % |
| Commercial yield | t $ha^{-1}$ | Mineral nitrogen content | mg N-$NO_3^-$ $kg^{-1}$ dw |
| Non-commercial yield | t $ha^{-1}$ | Organic nitrogen content | % |
| Marketable part weight | g | Other parameters of soil fertility | - |
| Marketable part diameter | cm | *Bioactive compounds analysis** | |
| Marketable part length | cm | Total sinapic acid derivatives (5 compounds) | mg SAE $kg^{-1}$ |
| *Nitrogen and other nutrients content in leaves, stems and/or fruits* | | Total phenols | mg CAE $kg^{-1}$ |
| Wet weight | % | Total glucosinolates (11 compounds) | mg $kg^{-1}$ |
| Nitrogen content | % | Total flavonoids (3 compounds) | mg Q3RE $kg^{-1}$ |
| Other micro and macro-nutrients content | ppm or % | | |

*SAE, sinapic acid equivalents; CAE, caffeic acid equivalents; GTPE, glucotropaeolin equivalents; Q3RE, rutin equivalents; dw, dry weight; ppm, parts per million.*
** These parameters are analysed only for cauliflower (Chapter 6).*

All these data was analyzed with the Enterprise Guide software package (SAS institute Inc. 2006). Analysis of variance was conducted using the General Linear Models procedure, and the least significant difference test (LSD, $p<0.05$) was used to establish differences between cultivation options.

**Harvest and product quality analysis**

Harvest and the quality of the marketable parts of the plant were assessed, in order to assess not only de quantity produced, but also its value. Total yield, marketable yield and non-marketable yield were determined per block and per cultivation option during harvest time. Furthermore, one or more size parameters (weight, diameter and/or length) of the marketable part of the plant were measured from a representative sample of each block.

In addition, at several points during cultivation, aerial samples of the plants (distinguishing between leaves, stems and/or fruits, according to the type of crop) were taken from each block. The wet weight was measured for each sample, and the moisture was determined with a drying temperature of 65°C until constant weight. Two plants from each block (with a fruit, leaf, and stem sample), at each of the several sampling points, were dried and the N content analyzed by the Kjeldahl method (Doltra and Muñoz 2010). Micro and macro-nutrient content was also measured for these samples using ICPOES spectrometry.

**Table 2.8. Characteristics of the applied compost.**

| Parameter | | | Value |
|---|---|---|---|
| **pH (1/5)** | | | 8.13 |
| **Electrical conductivity (dS $m^{-1}$) (1/5)** | | | 2.62 |
| **% Moisture** | | | 26.79 |
| **mg $kg^{-1}$ N-$NH^{+4}$ soluble (1/5) db** | | | 552 |
| **mg $kg^{-1}$ N-$NO^{-3}$ (1/5) db** | | | nm |
| **% Organic matter db (OM)** | | | 52.56 |
| **% Organic nitrogen db (ON)** | | | 1.29 |
| **C/N db** | | | 20 |
| **% Resistant organic matter db (ROM)** | | | 26.68 |
| **% SD db** | | | 50.75 |
| **% Non easily hydrolysable organic nitrogen (NH-N) db** | | | 0.74 |
| **% NH-N/ON db** | | | 57.23 |
| **Parameter** | **Value** | **Parameter** | **Value** |
| **% P db** | 0.2 | **mg $kg^{-1}$ Zn db** | 124 |
| **% K db** | 0.86 | **mg $kg^{-1}$ Mn db** | 120 |
| **% Na db** | 0.21 | **mg $kg^{-1}$ Cu db** | 34 |
| **% Ca db** | 2.98 | **mg $kg^{-1}$ Ni db** | 6 |
| **% Mg db** | 0.55 | **mg $kg^{-1}$ Cr db** | 7 |
| **% Fe db** | 1.51 | **mg $kg^{-1}$ Pb db** | 26 |
| | | **mg $kg^{-1}$ Cd db** | 0.13 |

*Source: Muñoz et al.* (2008a).

**Bioactive compounds analysis**

Sinapic acid derivatives, flavonoids and glucosinolates, the most important nutritional compounds of cauliflower (Nilsson et al. 2006; Gratacós-Cubarsí et al. 2010), and total phenols were assessed for cauliflower. More detailed explanation of the methods used is provided in Chapter 6.

**Other related analysis**

At the beginning of the project a very complete analysis of the plot soils were carried out. Later, several soil samples were taken during the length of the project to measure the evolution of moisture and mineral and organic nitrogen content at three different depths. In addition, organic and mineral fertility was measured at the end of each crop.

Finally, the quality and characteristics of the compost batches applied to the fields, which were produced in Castelldefels composting plant, were measured. The studied parameters were related to its fertilizing value, organic matter content and quality, and heavy metal content. Table 2.8 shows the parameters for the compost applied to the crops considered in the dissertation; it was applied the 13$^{th}$ of November 2006 for open field plot and the 27$^{th}$ of March 2007 for greenhouse plot.

The moisture content of the compost applied to the experimental plots was 27%, slightly lower than the 30-40% interval established by RD 824/2005 (Ministerio de la Presidencia2005), and the organic matter content was 53% for dry material. In relation to heavy metals, this was class A compost (section 1.3.4).

# Part II

## Environmental assessment of composting technologies

# Chapter 3. The use of Life Cycle Assessment for the comparison of bio-waste composting at home and full scale

**Chapter 3** is based on the following paper:

Martínez-Blanco, J., Colón, J., Gabarrell, X., Font, X., Sánchez, A., Artola, A., & Rieradevall, J. (2010) The use of Life Cycle Assessment for the comparison of bio-waste composting at home and full scale. Waste Management, 20, 983-994.

# Abstract

Environmental impacts and gaseous emissions associated to home and industrial composting of the source-separated organic fraction of municipal solid waste have been evaluated using the environmental tool of Life Cycle Assessment (LCA). Experimental data of both scenarios were experimentally collected. The functional unit used was one ton of organic waste. Ammonia, methane and nitrous oxide released from home composting (HC) were more than five times higher than those of industrial composting (IC) but the latter involved within 2 and 53 times more consumption or generation of transport, energy, water, infrastructures, waste and volatile organic compounds (VOC) emissions than HC. Therefore, results indicated that IC was more impacting than HC for four of the impact categories considered (abiotic depletion, ozone layer depletion, photochemical oxidation potentials and cumulative energy demand) and less impacting for the other three (acidification, eutrophication and global warming potentials). Production of composting bin and gaseous emissions are the main responsible for the HC impacts, whereas for IC the main contributions come from collection and transportation of organic waste, electricity consumption, dumped waste and VOC emission. These results suggest that HC may be an interesting alternative or complement to IC in low density areas of population.

# 3.1. Introduction

At present, waste management is becoming a global problem in developed countries due to the rapid collapse of landfills and the high impacts related to bio-waste dumping. In view of these problems the European Union (EU) published in 1999 the Landfill Directive (EC1999), which requires the member states to reduce the amount of biodegradable waste being dumped by promoting the adoption of measures to increase and improve sorting activities at the origin, recovery and recycling. The overall annual food and garden waste included in mixed municipal solid waste in the European Union is within 76.5–102 Mt that represents 30–40% of the total annual municipal waste generation (European Commission 2008)().

Composting, which can be defined as the aerobic biological degradation and stabilization of organic substrates under controlled, thermophilic and aerobic conditions (Haug 1993), has been presented as an environmental friendly and sustainable alternative to manage and recycle organic solid wastes, with the aim of producing a quality product known as compost, to be used as organic amendment in agriculture (European Commission 2008). For these reasons, exhaustive and systematic evaluations about its environmental performance are necessary. Potential environmental impacts, positive and negative, of municipal waste treatments should be considered including their potential pollution and their contributions to climate change, among other environmental impacts.

Industrial composting, in-vessel composting or windrow composting, imply the consumption of energy for waste transport and processing, the emission of odors and other contaminants, the proliferation of insects, birds and rodents and the mixture of different quality materials (Haug 1993). The organic fraction of municipal solid waste (OFMSW) or bio-waste usually has a percentage of non-biodegradable materials, which in Catalonia (northeast of Spain) can account for 1–30% (ARC 2009). Collection and separation of waste according to its different fractions (organics, metals, glass, papers, aluminium, fabrics, wood) is one of the best ways to obtain a good quality final compost product that may be used without great concerns (Barreira et al. 2008). Therefore the non-biodegradable material in the initial OFMSW affects the normal composting process by reducing the available capacity of composting plants and increasing the metal contamination of compost. Notwithstanding these disadvantages, bio-waste composting at industrial scale presents great benefits such as a proper control of composting process variables (temperature, moisture, oxygen content, etc.) or the treatment of the exhaust gases.

Home composting or backyard composting, which means the self-composting of the bio-waste as well as the use of the compost in a garden belonging to a private household (European Commission 2008), presents some potential benefits when compared to the industrial process: it avoids the collection of the OFMSW; it considerably reduces the economic, material and energetic investments; and finally, it allows a direct control of the process and the organic materials input by avoiding or reducing the inclusion of impurities (McGovern 1997; Ligon and Garland 1998; Jasim and Smith 2003; Boldrin et al. 2009; Martínez-Blanco et al. 2009a). However, home composting also presents some problems: compost obtained often is not homogeneous; odors and other pollutants such as methane, ammonia or nitrous oxide are emitted directly to the atmosphere during the decomposition process (Amlinger et al. 2008; Ansorena 2008), etc.

As observed, these two composting technologies present important differences and each one can be appropriate for different situations. For instance, home composting can be a good alternative to industrial composting in low density urban areas where a large investment in transport is required for the separate collection of the OFMSW. On the contrary, it is difficult to substitute industrial composting in high density urban areas, because of site, hygienic and monitoring requirements. The present study is intended to improve the knowledge about the environmental concerns of composting facilities and to help in the regional planning of organic waste management, since specific real and quantitative environmental data about the different alternatives available for the management of the OFMSW is scarce. Hence, the aim of this study is to quantify material requirements, energy consumption and gaseous and waste emissions associated to home and industrial composting and to determinate their environmental impacts using the environmental tool of Life Cycle Assessment, LCA (Rebitzer et al. 2004; ISO 2006a; 2006b).

# 3.2. Materials and methods

## 3.2.1. Data origin

Data from home composting system were obtained from an experimental composter controlled and managed by the authors and located at the Universitat Autònoma de Barcelona (Barcelona, Spain) following the practices recommended by some local Catalan municipalities (ARC 2010). In relation to industrial composting, data were obtained in an industrial composting facility located in the Barcelona province (Spain): gaseous emissions were experimentally determined by the authors, whereas the rest of the inventory data were supplied by plant managers. This composting facility was selected because its steady state, its adequate technical and environmental characteristics and because it generates a compost of an agronomic quality within the legal regulations in Spain (Martínez-Blanco et al. 2009b).

## 3.2.2. Organic fraction of municipal solid waste

The organic material treated in both composting processes was the source-separated organic fraction of the municipal solid waste (OFMSW) that is defined as household waste that because of their nature or composition is capable of undergoing anaerobic or aerobic decomposition (leftovers of raw fruits and vegetables, food scraps and raw fish or meat and other similar waste), excluding green waste from gardens and parks. In fact, green waste (GW), which includes tree cuttings, branches, grass and wood (European Commission 2008), was used as bulking agent in home and industrial composting to provide enough porosity and to prevent leachate generation.

## 3.2.3. Home composting experimental set-up

The organic waste was poured to the upper part of the composter and was extracted through the lower panels. The composter has a lateral system of natural ventilation to guarantee an aerobic process. The composter (70 x 70 x 103 cm) was made of recycled plastic from source-separated municipal collection.

The composter was fed once a week following the next methodology, similar to that reported by Colón et al. (2010). First, the GW was shredded by means of an electric garden chipper (BOSCH AXT 2500 HP, Barcelona, Spain). Then, the OFMSW and GW were mixed in an average volume ratio of 0.8:1 (OFMSW:GW), the mixing ratio was slightly variable at each load depending on the moisture content of the material in the composter – the moisture content was determined using the fist test according to the US Department of Agriculture and the US Composting Council (2001). An average of 11.4 kg of mixture (8.3 kg of OFMSW and 3.1 kg of GW) were added to the composter each week. This amount corresponds to the average quantity of the OFMSW produced by a Spanish two-member family (MMA 2008). In order to aerate the mixture, upper layers of the composter were weekly mixed with a commercial tool specially designed for this purpose (mixing tool (Compostadores SL 2010), Barcelona, Spain). Also, the moisture content of the material was adjusted by direct tap water addition (to increase moisture) or by adding green waste (to reduce moisture), when necessary. The organic material was composted for approximately 12 weeks and finally the compost was extracted (0.05 $m^3$) through the lower panel and it was immediately analyzed. No sieving was necessary since it was considered that the entire obtained product could be directly applied to soil.

The composter was placed outdoors in the Escola d'Enginyeria (Barcelona, Spain) of the Universitat Autònoma de Barcelona in shady conditions on a paved surface. The experiment was carried out during 100 days from November 2008 to March 2009. This corresponds to typical winter Mediterranean mild conditions (temperature from 5 to 20$^{o}$C and scarce rainfall).

### 3.2.4. Industrial composting operation

The composting plant studied has the capacity to treat 15,000 ton of organic waste a year (OFMSW and GW) using the in-vessel (''tunnel'') decomposition technology with a curing phase in turned windrows in an enclosed building. The plant has different biofilters for the treatment of the exhaust gases. Such characteristics are the most widely established in closed industrial composting facilities in Spanish populated areas.

The industrial composting process in the studied plant lasts for 10 weeks and includes four main steps. During the pre-treatment step, OFMSW and GW are prepared and mixed, at a volume ratio of 1:1 (OFMSW:GW). The decomposition phase takes place in-vessel (composting tunnels) with forced aeration and irrigation systems. The decomposed material is then disposed in composting windrows for the curing phase as third step. Windrows are weekly turned. Finally, during the post-treatment final step, the processed material is screened to separate the mature compost from the pieces of green waste that are not totally decomposed and other impurities. Recovered green waste is used again in the composting process and the impurities (mainly glass, metals and plastics) are dumped in a sanitary landfill. A more comprehensive description of the process was previously described by Martínez-Blanco et al. (2009b).

### 3.2.5. Determination of gaseous emissions

Although during the composting process more than 100 types of gaseous compounds can be emitted (Chung 2007), only $NH_3$, $CH_4$, $N_2O$ and volatile organic compounds (VOC) have been considered in this study as they represent together with $CO_2$, 99%

of the total emission (Beck-Friis et al. 2000; Pagans et al. 2006a; Amlinger et al. 2008). $CO_2$ emitted from composting is not fossil-derived, and therefore, it was not considered as a greenhouse gas emission (Amlinger et al. 2008), in accordance to the European Commission (Smith et al. 2001).

The methodology developed by Colón et al. (2009) and Cadena et al. (2009) for the sampling and determination of gaseous emissions in industrial composting facilities was used to calculate gaseous emissions from the composting plant. This methodology was also adapted to determine the home composting emissions. In this case, measures were taken in only one point on the upper surface area of the composter assuming a homogeneous emission in all the composter small emission surface area (0.31 $m^2$) at each sampling time. The covered area for this sampling point corresponds to 25% of the total home composting emission area, and it was considered representative of the whole system after checking that the emission velocity was highly uniform in the entire surface as described by Colón et al. (2009) and Cadena et al. (2009).

Air velocity on the emission surface was determined using a thermo-anemometer (VelociCalc Plus mod. 8386, TSIAirflow Instruments, Buckinghamshire, UK) and a specially designed Venturi tube to increase airflow velocity (Veeken et al. 2002). Ammonia was determined on site using a multigas sensor (model iTXT82, Industrial Scientific, Vertex, Barcelona, Spain) with an ammonia detection range 0–200 mL $m^{-3}$. Gaseous samples were taken in Tedlar® bags for the laboratory determination of VOC, methane and nitrous oxide. Total VOC were analyzed as stated in Colón et al. (2009). Methane was analyzed by gas chromatography using a Flame ionization detector (FID) and a HP-Plot Q column with a detection limit of 1 ppmv (home composting) and by gas chromatography in a certified external laboratory with a detection limit of 10 ppmv (industrial composting). Nitrous oxide was analyzed by gas chromatography in a certified external certified laboratory with a detection limit of 50 ppbv (home composting) and by gas chromatography using a Thermal Conductivity Detector (TCD) and a GS-CarbonPlot column with a detection limit of 10 ppmv (industrial composting).

### 3.2.6. Organic waste and compost analytical methods for both systems

Moisture and organic matter content, pH, electrical conductivity, N-Kjeldhal, heavy metals content and bulk density of input materials and compost were determined following the standard methodology proposed by the US Department of Agriculture and the US Composting Council (2001). Porosity was assessed by air pycnometry (Ruggieri et al. 2009b). The static respirometric index (SRI), which was used as a measure of the biological stability of the material, was determined following the methodology proposed by Barrena et al. (2005).

## 3.3. Life Cycle Assessment

### 3.3.1. General methodology

LCA is a methodology for the determination of environmental impacts associated to a product, process or service from cradle to grave, in other words, from production of the raw materials to ultimate disposal of waste. According to ISO 14040 and 14044

(ISO 2006a; 2006b), there are four main steps in a LCA study: the goal and scope definition, the inventory analysis, the impact assessment and the interpretation.

In this study, the software SimaPro v7.1.8 (PRé Consultants 2012) was used to evaluate the environmental impacts of home and industrial composting. Only the obligatory phases defined by the ISO 14040 and 14044 regulation for the impact assessment (ISO 2006a; 2006b), namely classification and characterization, were performed as they are the more objective ones (Martínez-Blanco et al. 2009b).

The impact assessment method used was CML 2001, which was based on the CML Leiden 2000 method developed by the Centre of Environmental Science of Leiden University (Guinée 2001). The impact categories considered were: abiotic depletion potential (ADP), acidification potential (AP), eutrophication potential (EP), global warming potential (GWP), ozone layer depletion potential (OLDP), photochemical oxidation potential (POP) and a flow indicator, the cumulative energy demand (CED).

## 3.3.2. Goal and scope definition

### *(i) Goal of the study*

There were three main objectives in this environmental study: firstly, to evaluate the overall environmental impacts of home and industrial composting; secondly, to detect the environmental critical phases of each composting system; and finally, to compare home and industrial composting environmental performances.

### *(ii) Functional unit*

The key functions for the two composting technologies considered were the management of the OFMSW and the production of an organic fertilizer or amendment, the compost. The functional unit (FU) in LCA provides a reference to which the inputs and outputs of the inventory are related and allows the comparison between systems (ISO 2006a; ISO 2006b). In this study the functional unit selected was the management by composting of one ton of OFMSW.

### *(iii) Composting system burdens*

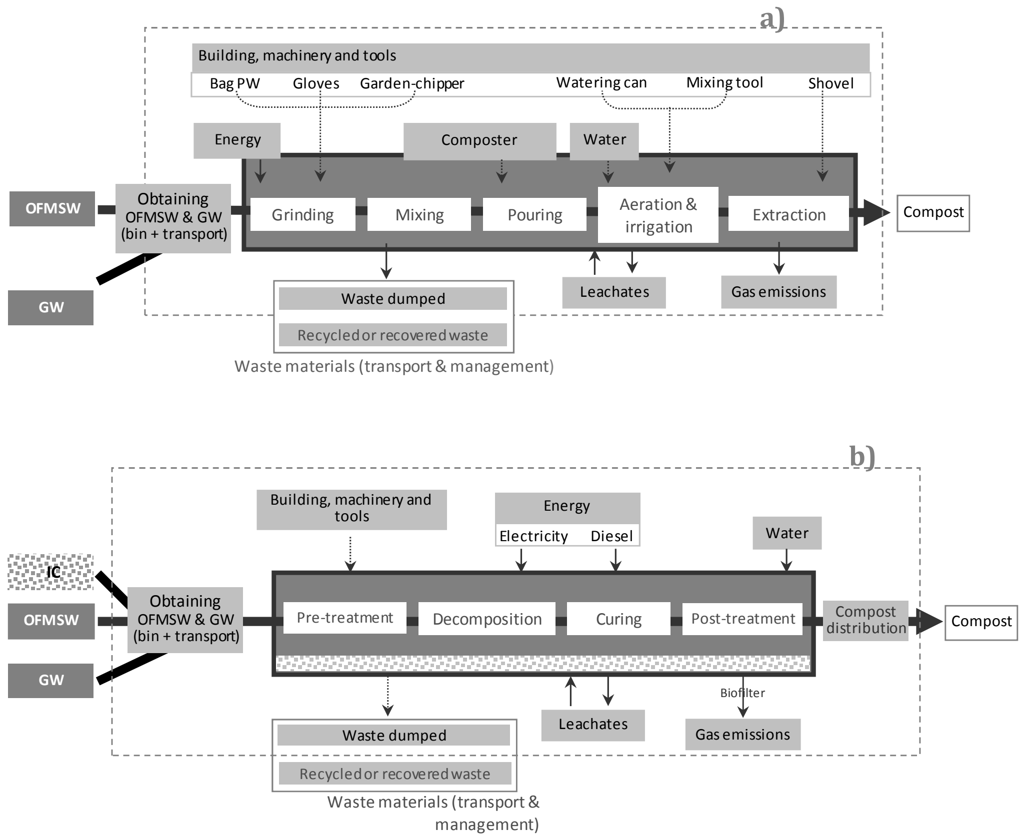
The two systems considered are home composting (HC) and industrial composting (IC). Both systems include the steps from the OFMSW and GW collection to the final compost use, for instance in decorative gardening and in productive courtyards (Figure 3.1).

**Home composting system**

The home composting system, which is represented in Figure 3.1a, includes: (i) the kitchen bin for the OFMSW storage; (ii) the tools and infrastructures used during the process; (iii) electricity and water consumption; (iv) transport and management of the waste dumped generated by the system (bags, gloves, etc.); and (v) gaseous emissions and leachate generated during home composting.

**Industrial composting system.**

The industrial composting system, represented in Figure 3.1b included (i) the OFMSW and GW generation that includes their transport and kitchen bin production; (ii) the building and other infrastructures of the facility; (iii) diesel, electricity and water consumption; (iv) the transport and management of the waste generated by the system that was dumped (solid waste fraction and building waste); (v) gaseous emissions and leachate generated during industrial composting; and (vi) transport of compost from the plant to the final user.




*OFMSW, organic fraction of municipal solid waste; GW, green waste; IC, impurities material content.*

**Figure 3.1. Definition and boundaries of the composting systems studied, including the main composting stages and the input and output flows considered. (a) Home composting system. (b) Industrial composting system.**

### *(iv) Main hypothesis*

An average house of two people was considered as OFMSW and GW producer, compost consumer and, additionally for home composting system, as compost producer. A composter made of municipal collected and recycled plastic (plastic mix) was considered, according to the actual industrial manufacture of this product. In home composting, no transport was considered for organic waste collection and compost distribution from and to the household. The aforementioned standard house was supposed to be located at 10 km from the industrial composting facility, which was nearly the average distance from the collection areas to the composting facilities in the Barcelona metropolitan area. In relation to the presence of impurities, it has been considered that a family spending time and efforts on home composting and using the compost obtained is presumably concerned for a well sorted organic waste. Therefore for home composting this material was not considered.

In the OFMSW input stream of the industrial composting facility, 8% of impurities were measured. However, with the aim of studying comparable composting systems, the burdens of the treatment of these impurities (by recycling, landfilling, incineration or other methods) were not considered, although the losses of the treatment capacity in industrial facility due the presence of this material were considered.

Throughout the inventory three average distances were considered for the transportation required: local distance (10 km), with a van of 3.5 ton MAL (maximum authorised load); regional distance (50 km), with a van of 3.5 ton MAL or a lorry of 3.5–16 ton MAL; and national distance (500 km), with a lorry of 3.5–16 ton MAL. In addition, for the municipal solid waste collection a specific lorry of 21 ton MAL was taken into account.

#### *(v) Allocation procedure*

When LCA is applied to complex systems (i.e. involving multiple products and recycling systems) the burden allocation procedures must be defined. For the system studied, the ''cut-off'' methodology defined by Ekvall and Tillman (1997) was used. Accordingly, environmental burdens should be assigned to the system that it is directly responsible for them. Thus, impacts of waste dumping, for example, are fully attributable to the systems being studied; whereas burdens of recycled or reused waste should not be accounted since it is considered that they should be attributed to the system that uses such waste as a material source (Ekvall and Tillman 1997; Finnveden 1999; Johns et al. 2008).

#### *(vi) Quality and origin of the data in the inventory*

In this study, data inventories for home and industrial systems were elaborated including energy, water and material resources, emissions and waste. The majority of these figures were experimentally obtained from both systems. Regarding the industrial composting, data previously collected on energy and materials inputs and outputs in the composting plant were used (Martínez-Blanco et al. 2009b). These data correspond to average amounts calculated during the 2003–2006 period. Data on gaseous emissions obtained in the same industrial composting plant were also considered (Colón et al. 2009). Regarding the home composting system, the data were collected specifically for the present study, which is based on Colón et al. (2010). To complete the life cycle inventories (LCI), bibliographical sources and the Ecoinvent database (SCLCI 2012) were used. The specific data sources used are compiled in Tables 3.1 and 3.2 (last column).

### 3.3.3. Life Cycle Inventory (LCI)

In this section the two composting systems are widely described and quantified with regard to the functional unit.

#### *(i) LCI of home composting system*

The home composting process has been previously described. In order to express the results based on the functional unit, the flows were related to one ton of OFMSW (Table 3.1).

**Generation of OFMSW and GW**

It was considered that both the OFMSW and the GW were generated in the same household where the composter was set up and collected in a polypropylene kitchen bin (8 L). No transport or collectively collection infrastructure were considered neither for OFMSW nor GW. In addition no shredding was considered for the OFMSW, but it was necessary for GW in order to adjust the particle size (Ruggieri et al. 2009b).

**Table 3.1. Summary of compost production inventory for the home composting system. Values are related to 1 ton of OFMSW (functional unit).**

| Stages | Element | Flow | Amount | Units ($t^{-1}$ OFMSW) | Lifespan (yr) |
|---|---|---|---|---|---|
| | | *Inputs* | | | |
| **Collection of OFMSW and GW** [1-3] | OFMSW kitchen bin | PP | 0.082 | kg | 7 |
| **Building, machinery and tools (composter)** [1-5] | Composter | Plastic mix | 4.380 | kg | 12 |
| | Plastic collection. Container | HDPE | 0.004 | kg | - |
| | Plastic collection. Cleaning water | Tap water | 0.006 | L | - |
| | Plastic collection. Transport | Transport | 0.146 | tkm | - |
| | Transport national | Transport | 2.190 | tkm | - |
| **Building, machinery and tools (others)** [1-4,6] | Garden chipper | Steel | 0.297 | kg | 10 |
| | | HDPE | 0.297 | kg | |
| | Bag for GW collection | PP | 0.076 | kg | 3 |
| | Shovel | Steel | 0.029 | kg | 12 |
| | | Wood | 0.016 | kg | |
| | Mixing tool | Iron | 0.133 | kg | 12 |
| | Watering can | PP | 0.003 | kg | 12 |
| | Gloves | Cotton | 0.011 | kg | 6 |
| | Transport national [a] | Transport | 0.364 | tkm | - |
| | Transport regional [b] | Transport | 0.013 | tkm | - |
| **Water consumption** [6] | Moistening water | Tap water | 50.870 | L | - |
| **Energy consumption** [4,6] | Electricity consumption (garden-chipper) | Electricity | 9.381 | kWh | - |
| | | *Outputs* | | | |
| **Gaseous emissions** [6] | Methane | $CH_4$ | 0.158 | kg | - |
| | Volatile organic compounds | VOC | 0.559 | kg | - |
| | Nitrous oxide | $N_2O$ | 0.676 | kg | - |
| | Ammonia | $NH_3$ | 0.842 | kg | - |
| **Waste Dumped** [1,2,5,6] | Waste management in landfill | Wood | 0.016 | kg | - |
| | | Cotton | 0.011 | kg | - |
| | | Plastic mix | 4.380 | kg | - |
| | Transport to landfill [c] | Transport | 0.441 | tkm | - |

*OFMSW, Organic fraction of municipal solid waste; GW, green waste; PP, polypropylene; HDPE, high density polyethylene.*

[a] *It includes garden chipper and mixing tool that were transported from a distance of 500 km by lorry of 3.5-16 ton MAL.*

[b] *It includes bag for GW collection, shovel, watering can and gloves that were transported from a distance of 50 km by van of 3.5 ton MAL.*

[c] *A transport of 50 km with a municipal solid waste collection lorry (21 ton MAL) was considered.*

*Sources:* [1] *SCLCI* (2012); [2] *WSOFM* (2010); [3] *Colón et al.* (2010); [4] *Compostadores SL* (2010); [5] *Iriarte et al.* (2009); [6] *own measurements.*

**Building, machinery and tools**

Environmental loads associated to the procurement of primary materials, the manufacture and the transport of the tools and the composter were considered. The data needed for the environmental inventory were obtained from local producers and distributors of the devices and from the database ecoinvent (SCLCI 2012), considering their lifespan. The composter and the rest of building, machinery and tools were represented as separate stages in the inventory and in the results.

Altogether, six tools were currently used for the home composting process: a polyethylene bag to collect the shredded GW; a shovel of steel and wood; an iron-made mixing tool; a watering can made of polypropylene used to maintain the moisture of the composting material; a pair of cotton gloves and an electric garden chipper (2500 W), which was supposed to be used collectively (i.e. by a community of 10 neighbours). It was considered that the shovel, the watering can and the cotton gloves were also used in the garden for other purposes (only a 15% of their total burdens where finally allocated to the home composting process).

The composter used was made of plastic mix, which is a mixture of several recycled plastics, such as HDPE (high density polyethylene) or PP (polypropylene), with a capacity of 0.4 $m^3$ and a weight of 28 kg. Collection and transport of the household source-separated plastic waste (Iriarte et al. 2009) and the fabrication process (melting and moulding) were accounted.

For the garden chipper, the mixing tool and the composter, transport in a lorry of 3.5–16 ton MAL and a distance of 500 km from the producer to the house were considered (national distance), since they were specific tools. The rest of the tools (bag GW, shovel, watering can and gloves) were transported by van of 3.5 ton MAL at a distance of 50 km from the producer (regional distance).

**Energy consumption**

Green waste shredding was the only stage where electricity was necessary. Technical characteristics of the equipment indicated that the garden chipper consumes 28 kWh per ton of GW, corresponding to 4.2 h $t^{-1}$ OFMSW.

**Water consumption and leachate production**

Occasionally, watering of the organic material in the composter was necessary due to its low moisture content. Overall, 51 L of water were used during the experimental period per ton of OFMSW. Leachate was not generated during the experiment.

**Gaseous emissions**

Gaseous compounds generated during the home composting process were directly emitted to the atmosphere as it typically occurs in home composting. $NH_3$, VOC, $CH_4$ and $N_2O$ emissions were considered and measured as previously described.

**Waste dumped**

According to the cut-off methodology (Ekvall and Tillman 1997) and considering that HDPE, polypropylene, steel and iron should be recycled or recovered, only the dumping of the wooden shovel, the gloves and the plastic mix composter were considered. The landfill was placed at a regional distance from the home composter (50 km) and the waste was transported with a municipal solid waste collection lorry (21 ton MAL).

### *(ii) LCI of industrial composting system*

The inventory flows and values for the industrial composting plant are summarized in Table 3.2.

**Table 3.2. Summary of compost production inventory for the industrial composting system. Values are related to 1 ton of OFMSW (functional unit).**

| Stages | Element | Flow | Amount | Units ($t^{-1}$ OFMSW) |
|---|---|---|---|---|
| | | *Inputs* | | |
| **Collection of OFMSW and GW** [1,2] | OFMSW kitchen bin | PP | 0.089 | kg |
| | Container | HDPE | 0.124 | kg |
| | Container cleaning water | Tap water | 10.604 | L |
| | OFMSW & GW collection [a] | Transport | 30.634 | tkm |
| **Building, machinery and tools** [b,3-6] | Building materials [c] | Building materials | 14.171 | kg |
| | Machinery production | Machinery | 0.149 | kg |
| | Diesel oil consumption | Diesel oil | 0.001 | kg |
| | Transport | Transport | 0.057 | tkm |
| **Energy consumption** [1] | Diesel oil consumption | Diesel oil | 4.743 | kg |
| | Electricity consumption | Electricity | 50.531 | kWh |
| **Water consumption** [1] | Tap water and rainwater | Water | 426.778 | L |
| | | *Outputs* | | |
| **Gaseous emissions** [7,8] | Methane | $CH_4$ | 0.034 [d] | kg |
| | Volatile organic compounds | VOC | 1.210 | kg |
| | Nitrous oxide | $N_2O$ | 0.092 [d] | kg |
| | Ammonia | $NH_3$ | 0.110 | kg |
| **Waste dumped** [1,3,5] | Waste management in landfill | Solid waste | 0.219 | t |
| | | Building waste | 0.014 | t |
| | Transport to landfill [e] | Waste | 23.488 | tkm |
| **Compost distribution** [8] | Compost distribution | Transport | 4.543 | tkm |

*OFMSW, Organic fraction of municipal solid waste; GW, green waste; PP, polypropylene; HDPE, high density polyethylene.*

[a] *Urban transport collection considered by Iriarte et al* (2009) *and transport intercity to the plant (10 km), with a municipal solid waste collection lorry (21 ton MAL).*

[b] *Resources entailed in the construction of the compost plant and its infrastructures. Lifespan of the composting plant is the 25 years* (WSOFM 2010).

[c] *The building materials required were cement, concrete, steel, gravel and polyester.*

[d] *Null emissions of $CH_4$ and $N_2O$ were detected according to the analytical methods applied which have a detection limit of 10 ppmv in both cases. For such compounds we considered an average emission value of 5 ppmv.*

[e] *It includes solid waste and building waste transport from the plant to the landfill considering a transport of 50 km with a lorry of 3.5-16 ton MAL.*

*Sources:* [1] *Martínez-Blanco et al.* (2009b); [2] *Iriarte et al.* (2009); [3] *Althaus et al.* (2004); [4] *SCLCI* (2012); [5] *ITeC* (2010); [6] *WSOFM* (2010); [7] *Colón et al.* (2009); [8] *own measurements.*

### Generation of OFMSW and GW

OFMSW and GW were obtained by source-separated collection systems from six nearby municipalities and the main Barcelona market suppliers, both located at 10 km of the composting plant (local distance). Transport of the organic waste, production and cleaning of the street collection containers and production of the kitchen bin were considered following the studies by Iriarte et al. (2009) and Martínez-Blanco et al. (2009b). The average content of non-organic impurities (plastics, glass, fabrics and metals, among others) in the OFMSW was 8% (ARC 2009). Such materials were not sorted at the beginning of the composting process and they implied a reduction in the plant treatment capacity, as commented previously.

**Building, machinery and tools**

The building and the machinery of the facility entailed materials production, transport and waste management that were accounted by Martínez-Blanco et al. (2009b). According to Washington State Office of Financial Management (2010) a lifespan of 25 years was accounted for the plant.

**Energy consumption**

The types of energy consumed by the composting plant were electricity, used in the aeration system, plant lighting and some machineries, and diesel oil, used by tractors and trucks. The average energetic consumption was of 383.5 MJ per ton of OFMSW.

**Water consumption and leachate production**

About 426.8 L of water were consumed per ton of OFMSW. The water was used for cleaning and for irrigating the organic material in active decomposition during the composting process. Approximately 20% of this water was rainfall water collected in biofilters and reused. Leachates generated in the composting process were also completely reused in the composting process.

**Gaseous emissions**

The exhaust gases generated in the composting tunnels and in the pre-treatment and curing areas of the composting plant were treated using biofilters before being released to the atmosphere. Biofilters consist of a 1 m layer of vegetal material previously used as bulking agent in the composting process, in which the gaseous contaminants are biodegraded by microorganisms. The emissions were determined according to the methodology previously explained (Colón et al. 2009). Null emissions of $N_2O$ and $CH_4$ were detected according to the analytical methods applied that have a detection limit of 10 ppmv in both cases. For such compounds we considered an average emission value of 5 ppmv.

**Waste dumped**

Two types of waste were generated in the composting plant: the solid waste fraction that was dumped and the machinery and building waste at the end of their useful life. Regarding the former, an amount equivalent to the weight of the improper material content (8% of the OFMSW treated) was not considered as previously mentioned and according to the allocation procedure. The transport and landfill disposal of the rest of solid waste fraction was included in the inventory. For machinery and building waste, only management of dumped waste was considered in the case of concrete and cement, whereas steel recycling was not accounted.

**Compost distribution**

The transport of the compost produced to the households, where it is used, was considered in the industrial system. A distance of 10 km was covered with a van of 3.5 ton MAL.

## 3.4. Results and discussion

This section has been divided into three parts: first, analytical results of the gaseous emissions and compost are shown; next, the input and output flow inventories for the two composting systems are compared; and finally, the impact assessment is presented.

### 3.4.1. Experimental results

#### *(i) Physicochemical characterization of compost in home and industrial systems*

The most important properties for compost obtained in the industrial composting facility were found within the quality proposed limits (Giró 1994; CCQC 2001; Giró 2001) as shown in Table 3.3. Compost obtained from home composting was near or within the legislation limits, and only moisture and nitrogen content were slightly above and below these limits, respectively. The compost obtained in home composting was not sieved for the analysis presented in Table 3.3 because it was considered that the obtained compost is directly used as soil amendment. However, if home compost had been sieved at 10 mm (as in the industrial plant), the nitrogen content of the home compost would increase to 2.25%.

**Table 3.3. Physicochemical properties of final compost obtained from the home composter and from the industrial composting facility. Compost quality standards are also reported for comparison.**

| Properties | Units | Compost | | Compost quality standards |
|---|---|---|---|---|
| | | Home | Industrial [a] | |
| **Moisture** | %, wb | 43.63 | 31.85 | 25-40 [b] |
| **Organic matter** | %, db | 47.96 | 55.33 | ≥40 [b] |
| **pH (extract 1:5 w:v)** | - | 7.83 | 7.88 | 6.5-8 [b] |
| **Electrical conductivity (extract 1:5 w:v)** | mS $cm^{-1}$ | 4.30 | 4.90 | ≤6 [b] |
| **N-Kjeldhal** | %, db | 1.71 | 2.04 | ≥2 [b] |
| **Respiration index** | mg $O_2$ $g^{-1}$ OM $h^{-1}$ | 1.13 | 0.89 | 0.5-1.5 [c] |
| **Zn** | mg $kg^{-1}$ | 156 | 150 | 200 [d] |
| **Cu** | mg $kg^{-1}$ | 44 | 47 | 100 [d] |
| **Ni** | mg $kg^{-1}$ | 9 | 9 | 50 [d] |
| **Cr** | mg $kg^{-1}$ | 9 | 8 | 100 [d] |
| **Pb** | mg $kg^{-1}$ | 28 | 32 | 100 [d] |
| **Cd** | mg $kg^{-1}$ | 0.30 | 0.24 | 0.7 [d] |

*wb: wet basis; db: dry basis; w: weight; v: volume; OM: organic matter; na: not analyzed.*
[a] *Data supplied by plant managers. Average values of the period 2001-2006.*
[b] *Regulation proposal for municipal solid waste compost in Spain* (Giró 1994).
[c] *Range for stable compost according to CCQC* (2001).
[d] *Regulation proposal for municipal solid waste compost in Europe* (European Commission2001).

Regarding the respirometric index, values between 0.5 and 1.5 mg $O_2$ g $OM^{-1}$ $h^{-1}$ were obtained in both technologies. These values correspond to a high level of stability according to CCQC (2001) and the European Commission (2001).

Metal concentrations in the final compost obtained from industrial composting and home composting were lower than the limits proposed by the European Commission (2001) (Table 3.3). Such concentrations were in agreement to the careful selection of input materials for home composting and the relative low presence of impurities in the OFMSW used for industrial composting.

***(ii) Gaseous emissions and leachate in home and industrial systems***

Ammonia, VOC, methane and nitrous oxide were measured during the studied period for both composting systems. According to Tables 3.1 and 3.2, the VOC emissions of home composting process were lower than industrial composting ones (0.559 and 1.210 kg C-VOC $t^{-1}$ OFMSW, respectively). Regarding ammonia, emissions in home composting were eight times higher than in industrial composting (0.842 kg and 0.110 kg $NH_3$ $t^{-1}$ OFMSW, respectively). This difference was obviously related to the high ammonia removal efficiency during the biofiltration process in the composting facility with removal efficiencies close to 90% (Colón et al. 2009). In the case of VOC the biofilter removal efficiency was close to 70%. On the other hand, methane and nitrous oxide were only detected in home composting. The emissions of these compounds were 0.158 kg $CH_4$ $t^{-1}$ OFMSW and 0.676 kg $N_2O$ $t^{-1}$ OFMSW, respectively. These higher emissions of methane and nitrous oxide during home composting can be explained by the lower oxygen availability due to passive aeration.

Amlinger et al. (2008) reported similar values of ammonia (0.474–0.972 kg $NH_3$ $t^{-1}$ OFMSW) and nitrous oxide (0.192–0.454 kg $N_2O$ $t^{-1}$ OFMSW) during home composting of OFMSW. On the contrary methane emissions detected in this study were between 5 and 14 times lower than those reported by these authors. In relation to VOC emissions, several authors presented emissions ranging from 0.590 to 0.430 kg C-VOC $t^{-1}$ OFMSW for IC (Smet et al. 1999; Muñoz et al. 2002; Diggelman and Ham 2003; Diaz and Warith 2006), which are similar to the values found in this work. Recently, Boldrin et al. (2009) has collected the most significant works published on IC emissions, showing a high level of dispersion. Nevertheless, it is important to highlight that the level of emissions will be highly dependent on the aeration system used, that is, natural convective aeration for home composting and forced aeration for the case analyzed of industrial composting (Barrington et al. 2003; Amlinger et al. 2008).

During the experimental period of the home composting, no leachate generation was observed. It must be pointed that other authors reported leachate generation from 0.01 to 0.07 $m^3$ $t^{-1}$ of bio-waste with home composting technology (Amlinger et al. 2008). In the industrial composting plant the leachates were totally reused (Martínez-Blanco et al. 2009b). Generation of leachate is a possible source of nitrogen losses, normally in the form of ammonia and an environmental load associated to eutrophication.

## 3.4.2. Main input and outputs flows

Table 3.4 presents a summary of the main input and output flows of energy, materials and emissions for the two composting systems. Data corresponds to the functional unit (1 ton of OFMSW). Unfortunately, no other data have been found in literature to compare both composting systems.

**Table 3.4. Life cycle inventory summary for the two composting systems: home (HC) and industrial composting (IC), including the management of all the OFMSW.**

| Flow | Units ($t^{-1}$ OFMSW) | Industrial composting | Home composting | Ratio (IC/HC) |
|---|---|---|---|---|
| **Water** | L | 437.383 | 50.876 | 8.60 |
| **Material resources** | kg | 13.386 | 5.327 | 2.51 |
| **Electricity** | kWh | 50.531 | 9.381 | 5.39 |
| **Diesel oil** | L | 4.744 | 0.000 | - |
| **VOC** | kg | 1.210 | 0.559 | 2.16 |
| **$NH_3$** | kg | 0.110 | 0.842 | 0.13 |
| **$CH_4$** | kg | 0.034 | 0.158 | 0.21 |
| **$N_2O$** | kg | 0.092 | 0.676 | 0.14 |
| **Waste dumped** | kg | 232.960 | 4.407 | 52.86 |
| **Transport** | tkm | 58.718 | 3.153 | 18.62 |
| **Land surface** | $m^2$ | 0.042 | 0.093 | 0.45 |

Values for industrial composting are generally higher than those of home composting apart from ammonia, methane and nitrous oxide emissions, and land required. The differences were especially significant for waste dumped and transport, with ratios of 53 and 19, respectively. Such high difference in waste production was mainly due to the solid waste rejects produced during the industrial composting process. Regarding transport, industrial composting included the transport of the OFMSW and the GW collection, the transport of the solid waste rejected, the transport of the plastic waste for the composter production, the transport of the materials and the waste of the building and the transport of the compost distribution, having the two first the major contribution to the total transport burdens. For home composting the transport flows included were only the transport of the composter and the tools from the producer and the transport of waste derived from some tools to landfill.

Home composting land requirements were nearly twice than that of industrial composting. As previously mentioned in relation to gaseous emissions, ammonia, methane and nitrous oxide emissions for home composting were 8, 5 and 7 times higher than for industrial composting, respectively.

## 3.4.3. Environmental impacts assessment

### *(i) Environmental assessment of the home composting process*

The environmental impacts attributable to each stage considered for the home composting are summarized in Table 3.5. In general, the composter and the $NH_3$, VOC and $N_2O$ emissions were the four main impacting items.

Particularly, for abiotic depletion potential (ADP), ozone layer depletion potential (OLDP) and cumulative energy demand (CED), the composter presented the highest impacts (40%, 79% and 41% of the total impact, respectively). As shown in Table 3.6, the main responsible for the impact produced by the composter was the manufacturing process, which includes between 63% and 95% to the total composter impact for all the categories studied.

**Table 3.5. Contribution to total environmental impact of the items considered in the home composting process (in percentages).**

| % | ADP | AP | EP | GWP | OLDP | POP | CED |
|---|---|---|---|---|---|---|---|
| **OFMSW kitchen bin** | 2.60 | 0.06 | 0.03 | 0.12 | 1.41 | 0.02 | 2.43 |
| **Composter** | 39.90 | 1.86 | 0.85 | 2.95 | 79.21 | 0.52 | 40.54 |
| **Building, machinery & tools** | 24.28 | 1.18 | 0.61 | 1.55 | 8.16 | 0.46 | 21.48 |
| **Water consumption** | 0.08 | 0.00 | 0.00 | 0.01 | 0.02 | 0.00 | 0.09 |
| **Electricity consumption** | 30.01 | 3.71 | 0.85 | 2.55 | 8.44 | 0.84 | 32.61 |
| **$CH_4$** | 0.00 | 0.00 | 0.00 | 1.65 | 0.00 | 0.40 | 0.00 |
| **VOC** | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 97.69 | 0.00 |
| **$N_20$** | 0.00 | 0.00 | 0.00 | 90.74 | 0.00 | 0.00 | 0.00 |
| **$NH_3$** | 0.00 | 92.98 | 91.02 | 0.00 | 0.00 | 0.00 | 0.00 |
| **Waste dumped** | 3.13 | 0.20 | 6.64 | 0.43 | 2.77 | 0.07 | 2.85 |
| **Total impact** | 100.00 | 100.00 | 100.00 | 100.00 | 100.00 | 100.00 | 100.00 |

*11-40%* *41-70%* *71-100%*

Returning to Table 3.5, the ammonia emissions contributed in a percentage of 93% and 91% to the acidification potential (AP) and the eutrophication potentials (EP), respectively. Nitrous oxide emissions were the main contributor (91%) to global warming potential (GWP), while VOC were the major contributors (98%) to photochemical oxidation potential (POP). It is important to highlight that the release of gaseous contaminants to atmosphere is an important concern in the environmental impact assessment of home composting, as it has been reported by other authors (Amlinger et al. 2008).

Regarding the rest of elements or stages considered, the electricity consumption represented 33% and 30% of the impact for CED and ADP, respectively; whereas for these categories the contribution of the building, machinery and tools was also relevant, 21% and 24%, respectively. For the rest of elements and categories the contributions were lower than 8%.

**Table 3.6. Contribution of the recyclable plastic collection, manufacturing process and distribution transport of the home composter (in % of total composter burdens).**

| Impact category | Unit | Plastic collection (%) | Manufacturing process (%) | Transport (%) [a] |
|---|---|---|---|---|
| **Abiotic depletion potential** | kg Sb eq. | 2.56 | 88.15 | 9.29 |
| **Acidification potential** | kg $SO_2$ eq. | 3.29 | 82.18 | 14.53 |
| **Eutrophication potential** | kg $PO_4^{3-}$ eq. | 6.41 | 62.83 | 30.75 |
| **Global warming potential** | kg $CO_2$ eq. | 3.13 | 85.70 | 11.17 |
| **Ozone layer depletion potential** | kg $CFC^{-11}$ eq. | 1.09 | 94.96 | 3.96 |
| **Photochemical oxidation potential** | kg $C_2H_4$ eq. | 2.92 | 86.52 | 10.56 |
| **Cumulative energy demand** | MJ eq. | 2.29 | 89.11 | 8.60 |

[a] *National transport was considered (500 km).*

### *(ii) Environmental assessment of the industrial composting process*

The contributions of the industrial composting elements or stages to the total environmental impact of the process are presented in Table 3.7. The three stages that presented a major impact were the collection of OFMSW and GW, the electricity supply and the solid dumped wastes, for all the categories considered except for photochemical oxidation potential. For POP, VOC emissions presented the highest contribution (94%). The collection of the OFMSW and GW was the most impacting stage for ADP, GWP and OLDP categories, contributing between 27–46%, and it also had a relevant contribution for AP and EP (23% and 16%, respectively), being the transport the main responsible. For CED the electricity consumption together with the collection of the OFMSW and GW had the higher contributions (32% each item). Electricity consumption was also the most impacting stage (37%) for AP and it contributed to a 29%, 20% and 12% for ADP, GWP and OLDP, respectively. In EP category, the solid waste dumped was the most impacting item that represented 55% of the total impact, while for GWP, OLDP and CED their contribution were higher than 11%. The high amount of solid wastes that were landfilled was related to the difficulty of a perfect separation of the compost obtained from the impurities, which is usual in industrial composting (Ruggieri et al. 2008).

Ammonia emissions and nitrous oxide emissions were responsible for more than 17% of the impact in the categories of AP, EP and GWP. For ADP, OLDP and CED, diesel consumption contributed to more than 14% to the total impacts. For the rest of elements and categories the contributions were lower than 11%.

**Table 3.7. Contribution to total environmental impact of the items considered in the industrial composting process.**

| % | ADP | AP | EP | GWP | OLDP | POP | CED |
|---|---|---|---|---|---|---|---|
| **Collection of OFMSW and GW** | 34.62 | 23.05 | 16.38 | 26.70 | 45.99 | 1.32 | 32.26 |
| **Water supply** | 0.12 | 0.08 | 0.02 | 0.09 | 0.05 | 0.01 | 0.14 |
| **Building and machinery** | 2.48 | 1.25 | 0.74 | 2.19 | 1.41 | 0.18 | 1.99 |
| **Diesel consumption** | 14.76 | 3.73 | 1.27 | 1.58 | 16.43 | 0.31 | 13.57 |
| **Electricity supply** | 28.84 | 37.24 | 6.62 | 19.85 | 12.32 | 2.00 | 32.29 |
| **CH4** | 0.00 | 0.00 | 0.00 | 0.51 | 0.00 | 0.04 | 0.00 |
| **VOC** | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 94.09 | 0.00 |
| **$N_20$** | 0.00 | 0.00 | 0.00 | 17.84 | 0.00 | 0.00 | 0.00 |
| **$NH_3$** | 0.00 | 22.66 | 17.26 | 0.00 | 0.00 | 0.00 | 0.00 |
| **Solid dumped wastes** | 10.76 | 7.42 | 55.01 | 25.19 | 14.07 | 1.44 | 11.23 |
| **Building dumped wastes** | 0.43 | 0.26 | 0.22 | 0.24 | 0.56 | 0.01 | 0.41 |
| **Compost distribution** | 7.98 | 4.32 | 2.48 | 5.82 | 9.17 | 0.61 | 8.10 |
| **Total impact** | 100.00 | 100.00 | 100.00 | 100.00 | 100.00 | 100.00 | 100.00 |

*11-40%* *41-70%* *71-100%*

### *(iii) Comparison of the environmental impacts between home and industrial composting*

In general, the industrial composting system implied higher consumption of energy during the process, larger transport requirements for the bio-waste collection and higher generation of waste compared to the home composting system, as it was previously discussed (Table 3.4). As a result of these environmental items the former was more impacting than the latter (between 4 and 6 times higher) for ADP, OLDP and CED categories (Figure 3.2). Additionally, industrial composting doubled POP impacts, whose main contributor was the VOC emissions, in comparison to home composting.

For the categories of AP, EP and GWP, in spite of the higher consumptions of energy and materials and the waste generated in industrial composting system, the significantly higher emissions of nitrous oxide and ammonia in home composting resulted in higher impacts in this system (between 31% and 46% higher).

The home composting system entailed the consumption of 351 MJ eq. and the emission of 220 kg $CO_2$ eq. $t^{-1}$ OFMSW, whereas for industrial composting, the energy consumption was 1908 MJ eq. and 153 kg $CO_2$ eq. were emitted per ton of OFMSW. In addition, it has to be pointed out that in the global warming context, composting contributes to produce emissions as well as to avoid emissions. For instance, Boldrin et al. (2009) show that when the final use of compost is also taken into account the overall emission factor for composting can vary between significant savings (-900 kg $CO_2$ eq. $t^{-1}$ OFMSW) to a net load (300 kg $CO_2$ eq. $t^{-1}$ OFMSW). It is evident that the benefits of compost use from the environmental point of view should be the focus of future studies.

### *(iv) Sensitivity analysis of the results*

As the results obtained correspond to a quite particular situation, environmental impacts for several hypothetical scenarios obtained by modifying relevant assumptions of the industrial composting system were assessed and compared with initial scenario studied (IC1) to perform the sensitivity analysis of the results. Three new assumptions were assessed: the distance between the composting facility and the household, the emissions of nitrous oxide and methane and the impurities content in the OFMSW. Results are illustrated in Table 3.8. Sensitivity analysis for home composting was performed by Colón et al. (2010) reporting as sensitive assumptions: whether recycled or raw plastic was considered as composter material and variations in nitrous oxide and ammonia emissions.

A distance of 10 km between the composting facility and the standard house has been considered in the inventory (IC1). Such situation may change as a function of the local distribution and restrictions of this kind of facilities. Impacts in scenario IC2 considered half reduced distance (5 km) and were significantly lower for all the impact categories (10–30%), apart from POP. For scenario IC3 that doubled the initial distance (20 km) impacts were significantly higher (15–38%), apart from POP. Such distance modifications presented more important effects on ADP, OLDP and CED because, as it is shown in Table 3.7, the collection of the OFMSW and GW supposed the maximum contribution of these categories.

Since the emissions of $N_2O$, VOC and $NH_3$ are reported to have a relevant contribution in some of the impact categories considered and nitrous oxide is not detected within the detection limits of the analytical methods used, a sensitivity analysis was performed for these emissions. To study this point, in IC4 scenario minimum emissions of $N_2O$ (0 ppmv) were considered, whereas in IC5 scenario maximum emissions of $N_2O$ (10 ppmv) were considered. As seen in Table 3.8, the only impact category affected relevantly was GWP (increasing or decreasing 18%), according to the results of Table 3.7.

In relation to methane, although the low impact of its emissions in our study, considerable higher emissions were reported in Amlinger et al. (2008) than in our experimental results (26 times higher). Therefore in the IC6 scenario an average emission of 900 g of $CH_4$ per ton of OFMSW was considered for industrial composting. Even though the high difference between the emission factors considered, an increase of only 13% was measured in IC6 with respect to IC1 for GWP (Table 3.8).

**Table 3.8. Comparison of the environmental impacts for the seven scenarios considered for industrial composting (IC2–IC8). Initial scenario (IC1) is considered as the base scenario (100% of contribution of each category), whereas the rest of scenarios are normalized to this base scenario.**

| Impact category | Units ($t^{-1}$ OFMSW) | Initial scenario (IC1) | Sensitivity analysis of other scenarios (%) | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | | | IC2 | IC3 | IC4 | IC5 | IC6 | IC7 | IC8 |
| **ADP** | kg Sb eq. | 7.68E-01 | 77 | 127 | 100 | 100 | 100 | 94 | 133 |
| **AP** | kg $SO_2$ eq. | 7.77E-01 | 85 | 120 | 100 | 100 | 100 | 95 | 126 |
| **EP** | kg $PO_4^{-3}$ eq. | 2.23E-01 | 90 | 115 | 100 | 100 | 100 | 95 | 128 |
| **GWP** | kg $CO_2$ eq. | 1.53E+02 | 82 | 122 | 82 | 118 | 113 | 95 | 126 |
| **OLDP** | kg $CFC^{-11}$ eq. | 1.33E-05 | 70 | 138 | 100 | 100 | 100 | 95 | 133 |
| **POP** | kg $C_2H_4$ eq. | 5.35E-01 | 99 | 101 | 100 | 100 | 101 | 100 | 102 |
| **CED** | MJ eq. | 1.91E+03 | 78 | 125 | 100 | 100 | 100 | 94 | 133 |

*Scenarios:*
*IC2: Distance from the household to the composting facility is 5 km.*
*IC3: Distance from the household to the composting facility is 20 km.*
*IC4: Emission of $N_2O$ is 0 ppmv.*
*IC5: Emission of $N_2O$ is 10 ppmv.*
*IC6: Emission of $CH_4$ reported by Amlinger et al.* (2008).
*IC7: Impurities content in the OFMSW is 1%.*
*IC8: Impurities content in the OFMSW is 30%.*

Finally, scenarios IC7 and IC8 considered high and low values of the impurities content of the OFMSW that the composting facility received (0–30%), according to the characterizations reported for the OFMSW in Catalonia (ARC 2009). Such modification had moderate effects on all the impact categories apart from POP that was mainly affected by VOC emissions. For IC7 scenario impacts were reduced between 5% and 6% depending on the category. For IC8 scenario impact, an increase between 26% and 33% was measured for the categories studied except for POP. The main reason for this behavior is that high impurities content reduced the treatment capacity of the industrial composting plant and therefore the energy, materials, transport and waste required to process 1 ton of OFMSW are increased. To our knowledge, no studies have been reported on the environmental impact that the impurities present in the OFMSW in source-separated collection systems produces in the full scale composting process.

# 3.5. Conclusions

A life cycle analysis was carried out in order to compare two types of composting systems: industrial and home composting. The physicochemical properties of the final composts obtained indicate that they were stable and according to the European standards.

$NH_3$, $N_2O$ and $CH_4$ emitted were between 5 and 8 times higher in home composting than in the industrial process. Biofiltration of exhaust gases and forced aeration during the composting process in industrial composting could be the main reasons to explain these differences. On the contrary, volatile organic compounds (VOC) emissions in the industrial process doubled that of the home composting.

The composter production, and particularly the manufacturing process, and the ammonia, VOC and nitrous oxide emissions were the four main impacting elements in the home composting system. In the industrial composting, the major impact contributions were related to obtaining of OFMSW and GW (mainly due to the requirements of the collection transport), electricity consumption and solid waste management for all the categories except for the photochemical oxidation potential. For such category VOC emissions presented the higher contribution. As a result of the relevant consumptions, industrial composting was more impacting (between 2 and 6 times) than the home composting process for ADP, OLDP and POP and CED. However, the significantly higher emissions of nitrous oxide and ammonia in the home system meant higher impacts for AP, EP and GWP categories for this system.

In reference to the sensitivity test performed, the impacts were proportional to the distance from the composting facility to the household being especially dependent in the case of ADP, OLDP and CED. In relation to the impurities content, their modification had moderately proportional effects on all the impact categories.

In conclusion, the incorporation of gas treatment systems for home composting and the use of low impact materials in the composter construction appears to be the main issues to minimize the environmental impact of this system, whereas the improvement in the biofiltration of VOC, jointly with the minimization of the energy use and the presence of impurities in the source-separated collection systems of the OFMSW, should be the main focus of research for the implementation of industrial composting programmes. Another important point of future research should be the comparison of home composting from the environmental point of view in different seasons of the year and, if possible, under very different climate conditions.

# Part III

## Environmental assessment of compost and agriculture

# Chapter 4. Life Cycle Assessment of the use of compost from municipal organic waste for fertilization of tomato crops

**Chapter 4** is based on the following paper:

Martínez-Blanco, J., Muñoz, P., Antón, A., & Rieradevall, J., (2009) Life Cycle Assessment of the use of compost from municipal organic waste for fertilization of tomato crops. Resources Conservation & Recycling, 53 (6), 340-351.

# Abstract

Several authors have assessed the positive repercussions of compost application in soil and the benefits of composting process, although most previous works focused only on a specific aspect of the whole life cycle of compost. The aim of this paper was to determine the environmental impacts associated to the use of compost, from the collection of organic municipal solid waste to its application to tomato crops, and to compare these results with mineral fertilizer application, using the environmental tool of Life Cycle Assessment. Three fertilizing systems were defined, arising from the dosages of mineral and organic fertilizers applied. The environmental performance of the pilot fields and the industrial composting were based on experimental measured data. The use of compost in horticulture demonstrated to be a treatment with fewer impacts than mineral fertilizer, if the avoided loads were considered, although compost production was a critical stage which needs to be optimised. No differences were observed in terms of agricultural production and quality.

## Keywords

Organic fertilizer; Industrial composting facility; OFMSW; Composting emissions; Environmental assessment; Waste management.

# 4.1. Introduction

In 1999, the European Union Landfill Directive (EC1999) required member states to reduce the amount of biodegradable waste being dumped (Antón et al. 2005b), in order to minimise environmental impacts and the loss of organic resources. This directive promoted the adoption of measures to increase and improve sorting at the origin, recuperation and recycling, including composting of organic fraction of the municipal solid waste (OFMSW) and green waste.

The composting of OFMSW, as well as reducing the total amount of landfill waste, yields a product that can be used in agricultural and destroys many of the pathogens and odor compounds (Jakobsen 1995). Several authors have also assessed the positive repercussions of the application of such compost in soil. These includes the reduction of weed germination and the number of plant pathogens; the increase in yield, quality and development of edaphic microorganism communities; and the increase in organic matter content, a parameter that has gradually decreased over recent years in the Mediterranean region, but which decreases erosion rate and improves moisture content and soil bulk density (Iglesias-Jimenez and Alvarez 1993; McConnell et al. 1993; Raviv 1998; European Commission2001; Elherradi et al. 2005; ROU 2007; Silva et al. 2007). In agricultural peri-urban areas there is an added value through using compost from urban areas, and the proximity to the market brings the opportunity to sell the agricultural products and so close metabolic cycles.

McConnell et al. (1993) found that there are local limitations on the use of compost in horticulture, for instance, the increase in salt content, the toxicity of heavy metals and the difference in the NPK ratio of the compost and that required by crops. However, most of the limitations can be reduced if the organic material is of the right composition. This may be achieved by proper sorting at origin (Hargreaves et al. 2008), followed by composting in controlled conditions with the necessary time for decomposition.

Although there has been an increase in studies on composting (Dalemo et al. 1997; Elherradi et al. 2005; Bruun et al. 2006; Cisneros 2006; Guerini et al. 2006; Güereca et al. 2006; Hansen et al. 2006a; Güereca et al. 2007; ROU 2007; Silva et al. 2007; Amlinger et al. 2008; Blengini 2008; Favoino and Hogg 2008; Hargreaves et al. 2008; Tsai 2008) most focus only on a specific aspect of the composting life cycle, such as gas emissions or the use of compost as a fertilizer. Moreover there is a lack of environmental quality data relating to compost application in horticulture, which could have an impact on transport, generation of indirect waste or changes in the nutrient supply to the plant that have not been sufficiently quantified (Sonesson et al. 2000).

Here we present details of the complete composting life cycle, providing information on the environmental impacts and demonstrating the agronomic viability of compost as a fertilizer when it was used for horticultural crops comparing this compost with traditionally used mineral fertilizers.

The plots were located in the county of Maresme, by the Mediterranean in the north-east of Spain. This agricultural area close to the city of Barcelona has major urban pressure and is one of the most important tomato producing areas in Catalonia. Cultivation there is in soil or in an artificial substrate, in the field or in greenhouse conditions. Owing to the soil characteristics in this area (sandy or sandy loam), with high permeability and low cationic exchange capacity, for a long time soil was considered an inert substrate, and farmers applied excessive water and fertilizers. Consequently, there was nutrient loss and aquifer pollution (López et al. 1999) and the Maresme was declared a region vulnerable to nitrate contamination, according to the European Directive 91/676 (ECC1991). Among other good practices, maximum nitrogen dosage was established and a fertilization management plan has been required from existing and new producers since then.

In previous research, using lower nitrogen amounts and water supply than usual in the zone, we obtained nearly identical harvest productions and qualities of tomatoes (Antón 2004; Muñoz et al. 2008a; 2008c). Economic and environmental performances could be improved using compost from OFMSW, with a reduction in use of mineral fertilizers and improvement of soil quality.

The Life Cycle Assessment (LCA) environmental tool was applied to quantify the environmental impacts and energy consumption associated with composting. According to ISO 14040 (ISO 2006a), LCA assesses the environmental aspects and potential environmental impacts during the whole process, from raw material acquisition through production, use, end-of-life treatment, recycling and final disposal.

# 4.2. Methodology

There are three obligatory stages in a LCA study: the definition of goal and scope, which is described in this section, the inventory analysis and the impact assessment.

## 4.2.1. Objective of the study

Two main objectives were defined. The first, agricultural and environmental viability of using compost on a tomato crop in open field production in a Mediterranean area was assessed and critical phases of the system, from the environmental point of view, were detected using the LCA tool. The second, to compare these results with the burdens associated with mineral fertilization.

## 4.2.2. Description of the systems

Three fertilization treatments used for the production of tomatoes in the Mediterranean area have been considered: compost ($C_H$), compost and mineral fertilizer ($C_L$), and mineral fertilizer (M).

Different stages were defined within each system: compost production (CP) and mineral fertilizer production (FP); transport of compost (CT) and mineral fertilizer (FT) to the crops; and cultivation (Cu). This latter includes the sub-stages of field operations (CuO), phytosanitary substances (CuP) and the fertirrigation infrastructure (CuF). For treatment with compost the CP, CT and Cu stages were considered; for treatment with mineral fertilizer FP, FT and Cu; and in the case of treatment with compost and mineral fertilizer CP, FP, CT, FT and Cu were considered (Figure 4.1).

The whole process, from raw materials required for the manufacture of the different elements to management of the generated waste, is considered, but not post-cultivation, as tomatoes cultivated in this area are destined for the local market and therefore their commercialisation does not involve a significant environmental burden, and is the same for the three treatments.

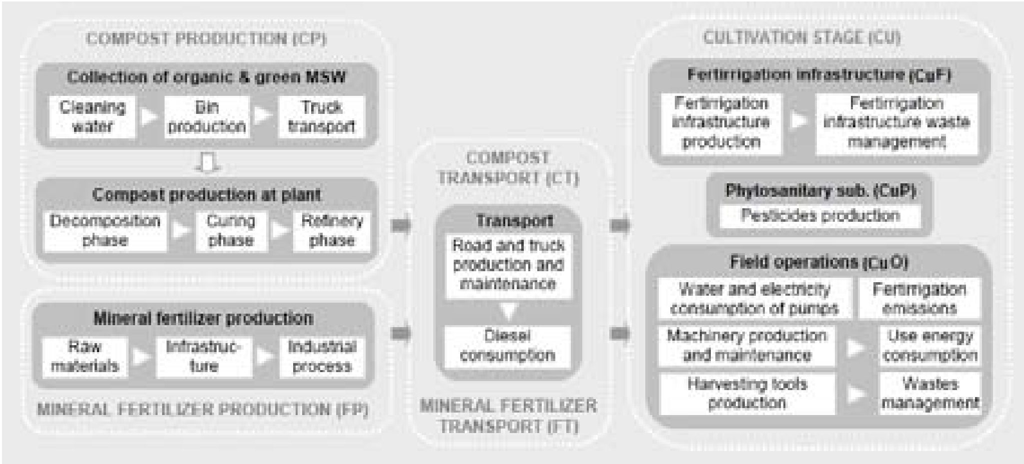



**Figure 4.1. Diagram of the systems studied, showing the mineral fertilizer and compost production, transport and cultivation stages considered.**

### 4.2.3. Functional unit

The functional unit chosen was the horticultural production of a ton of commercial tomatoes, to evaluate the quantity and quality of the product, as the tomatoes have to comply with certain minimum standards to be sold. This gave us a reference to normalize all the system's input and output flows.

### 4.2.4. Assigning burdens

Specification of assignment of environmental burdens was needed for two points of the systems: the management of waste generated in cultivation and the comparison between composting and the manufacture of mineral fertilizers. These "open-loop recycling allocations" come into play when comparing multifunctional systems, very common in the study of waste management that usually provides an extra function in addition to management itself (Ekvall and Tillman 1997; Finnveden 1999).

For the management of waste from cultivation, we used the cut-off method defined by Ekvall and Tillman (1997), by which each system is assigned the burdens for which it is directly responsible. With this method there is no uncertainty in the case of extraction of raw materials or the production processes and transport, as these are all directly assigned to the system. In the case of waste management, dumping of all materials is accounted for in the system being studied, while for recycled or reused waste, the burdens are attributed to the system using the waste as a raw material source.

When comparing the impacts of the three treatments ($C_H$, $C_L$ and M), it must be remembered that composting, as well as providing fertilizer, is a form of waste management of OFMSW, which is not the case in the production of mineral fertilizer. As proposed by Finnveden (1999) and Ekvall and Weidema (2004), to make these three systems comparable and to include the extra function of composting, the boundaries of the system should be expanded, considering a form of managing OFMSW alternative to composting. The method selected was dumping, with the environmental burdens subtracted from those treatments that include composting so that only the fertilizing function of the three treatments is compared.

### 4.2.5. Quality and origin of the data in the inventory

The systems of study defined in Section 4.2.2 required a detailed data-collection process, summarised in Table 4.1. Much of this, specifically in relation to CP and the sub-stages considered in Cu, was obtained experimentally. To complete the life cycle inventory, bibliographical sources and the ecoinvent database (SCLCI 2012) were used.

For the collection and transport of OFMSW, data from Bravo (2004) and Muñoz et al. (2002) were used, obtained on the basis of information provided by the local councils near to the industrial composting plants.

The CP used data from the industrial tunnel composting facility at Castelldefels, near Barcelona. This plant was selected because of its proximity to the experimental plots, its stationary state, its technical characteristics – such as its capacity for treatment, use of multi-tunnel technology and the type of bio-waste it treats – and because it generated a product of agronomic quality within the legal limits. These characteristics are similar to those of other plants, such that the results can be extrapolated to other Mediterranean areas.

Owing to the lack of standardised information on the emissions from biofilters in composting facilities, two of the gaseous compounds considered, ammonia ($NH_3$) and volatile organic compounds (VOC), were analysed between December 2007 and June 2008 at the composting facilities (Colón et al. 2009). For the other compounds considered, $CO_2$, $CH_4$ and $N_2O$, experimental data were taken from the literature (Soliva 1987; Sonesson et al. 1997; Beck-Friis et al. 2001; Muñoz and Rieradevall 2002; Cisneros 2006). Furthermore, experimental data from Muñoz et al. (2008a), on the physical and chemical characteristics of the compost obtained from the Castelldefels plant, were used.

The data relating to agricultural practices were obtained in the experimental fields located in Santa Susana (41˚38’27″N, 2˚43’00″E), a village close to Barcelona, between May and September 2007, using the best available techniques for integrated agriculture (MAPA 2002). The soil was sandy and the fertirrigation infrastructure was the standard for intensive horticulture on the Mediterranean coast, using water from wells.

Pesticide application was based on the regulations for integrated agriculture (MAPA 2002), at the minimums specified by the Spanish Ministry for Environment and Rural and Marine Affairs in the registry of phytosanitary products (MMAMRM 2008).

The doses of compost and mineral fertilizer were calculated depending on the expected tomato production and the nitrogen concentrations of both fertilizers, for each treatment.

Calculation of gas emissions from the diesel used by agricultural machinery was based on Gasola et al. (2007) and that from post-application emissions was based on Audsley (2003).

**Table 4.1. Quality and origin of data used in the life cycle inventory.**

| Stage | Type of data | Source |
|---|---|---|
| | *Compost production* | |
| **OFMSW and green waste collection and transport** | ERD | Bravo (2004), Muñoz et al. (2002) and data supplied by Castelldefels and Gavà municipalities. |
| **Composting process at plant: energy and water consumption, leachates compost obtained** | EOD | Data supplied by Castelldefels composting facility, managed by Metrocompost S.L. and Cisneros (2006). |
| **Composting emissions: $NH_3$ and VOC** | EOD | Experimental measurements (with Group Composting Research of UAB, GICOM) (2007-2008) and Colon et al. (2009). |
| **Composting emissions: $CO_2$, $CH_4$ and $N_2O$** | ERD | Beck-Friis et al. (2001), Cisneros (2006), Muñoz and Rieradevall (2002), Soliva (1987; 2001) , Sonesson et al. (1997) and EIPPC Bureau (2006). |
| **Compost characteristics** | EOD | Experimental measurements 2006 (with Barcelona School of Agriculture, ESAB), Muñoz et al. (2008a). |
| **All transports considered** | D | ecoinvent v1.2, 2005. |
| **Facility infrastructure and machinery** | D | Althaus et al. (2004), ITeC (2010), SCLCI (2012),WSOFM (2010). |
| | *Mineral fertilizer production* | |
| **Mineral fertilizer production: $KNO_3$** | D | ecoinvent v1.2, 2005. |
| | *Transports of mineral fertilizer and compost* | |
| **Mineral fertilizer transport and compost transport** | D | ecoinvent v1.2, 2005. |
| | *Fertirrigation infrastructure* | |
| **System design** | EOD | Experimental fields (2007) and MAPA (2002). |
| **Fertirrigation infrastructure production** | D | ecoinvent v1.2, 2005. |
| | *Phytosanitary substances* | |
| **Type of phytosanitary substances needed** | EOD | Experimental fields (2007) and MAPA (2002). |
| **Phytosanitary substances doses** | ERD | MAAMA (2010). |
| **Pesticide production** | D | ecoinvent v1.2, 2005. |
| | *Field operations* | |
| **Plot characteristics: type of soil, type of water supply, plant density** | EOD | Experimental fields (2007). |
| **Tomatoes harvest** | EOD | Experimental fields (2007). |
| **Average calibre and weight of the fruit** | EOD | Experimental fields (2007). |
| **Fertilizer consumption** | EOD | Experimental fields (2007). |
| **Machinery and tools: type of machinery used, fuel consumption operating rate** | EOD | Experimental fields (2007). |
| **Machinery and management tools production** | D | ecoinvent v1.2, 2005. |
| **Diesel emissions** | ERD | Gasola et al. (2007). |
| **Water consumption** | EOD | Experimental fields (2007). |
| **Post-application emissions** | ERD | Audsley (2003). |
| **Landfill waste disposal** | D | ecoinvent v1.2, 2005. |

*EOD, experimental own data; ERD, experimental reference data; D, database.*

The database ecoinvent data v1.2 (SCLCI 2012) database was used to calculate consumption associated with the production of mineral fertilizer, phytosanitary substances, machinery and tools, and the fertirrigation infrastructure, as well as the transport and the deposit of the waste in the landfill.

### 4.2.6. Categories of impact and LCA methodology

LCA methodology was initially developed for industrial products, so its agricultural application requires systematic application of existing methods with some adaptations (Audsley et al. 2003; Nemecek et al. 2004; Hansen et al. 2006b; ISO 2006a).

The computer tool used for impact analysis was the SimaPro v7.0 program (PRé Consultants 2012), only performing the obligatory classification and characterization phases of the impact assessment defined by the ISO 14040 regulation (ISO 2006a). In the classification, each burden is linked to one or more impact categories, while in the characterization stage the contribution of each burden to the impact categories is calculated by multiplying the burdens by a characterization factor (Guinée 2001).

The optional normalization and valorisation phases were excluded as they entail a high degree of subjectivity and reduce the information contributed with regard to environmental impacts. For normalization, the reference system is generally chosen using overall indicator results for a specific region and for a specific year (Pennington et al. 2004), as for weighting for valorisation which involves social, political and ethical value choices (Finnveden 1997; Blengini 2008). Moreover, there were no specific values for the region of study, not even for Spain.

The impact categories considered, defined by the CML (Guinée 2001), were: abiotic depletion potential (ADP), global warming potential (GWP), ozone layer depletion potential (OLDP), photochemical oxidation potential (POP), acidification potential (AP), eutrophication potential (EP) and a flow indicator, the cumulative energy demand (CED).

Fresh water, marine, terrestrial and human ecotoxicity were not considered as the few existing ecotoxicity effect indicators for chemicals are at a very early stage of development and significantly different methods have been suggested by the international scientific community. Furthermore, few attempts have been made to include ecotoxicity categories in current LCA methodology, so that considerable uncertainty could be introduced in impact assessment for several of the stages or processes considered in the study, such as heavy metal release and accumulation and pesticide use (Larsen and Hauschild 2007; Pettersen and Hertwich 2008).

The flow indicator for water consumption was also not considered, as the same quantities of irrigation water were applied to the fields regardless of the treatment.

## 4.3. Life cycle inventory

This section describes the inventory data and the methodology used to obtain it. To express energy and material flows in relation to the functional unit, the tomato harvest from each treatment was considered (Table 4.2).

All transports were considered to be by road in the different stages and sub-stages, doubling the distance to calculate the associated environmental burdens to take into account a return trip for unloading, except in the case of FT. We took into consideration the production, maintenance and disposition of the vehicle, the construction and maintenance of the roads and the diesel consumption (Spielmann et al. 2004).

**Table 4.2. Harvest production and parameters of agricultural quality.**

| Parameter | Units | $C_H$ | $C_L$ | M | LSD [a] |
|---|---|---|---|---|---|
| **Total harvest** [b] | t $ha^{-1}$ | 118b | 128a | 125a | - |
| **Commercial production** | t $ha^{-1}$ | 104 | 100 | 103 | ns |
| **Fruit average diameter** | mm | 79.1 | 78.2 | 78.7 | ns |
| **Fruit average weight** | g | 209.1 | 205.6 | 207.7 | ns |

[a] *Different letters mean significant effect and ns: not significant at P = 0.05.*
[b] *Total harvest is commercial production plus non-commercial production.*

## 4.3.1. Stage of compost production (CP)

The composting facility studied had the capacity to treat about 15,000 t of organic bio-waste a year – OFMSW and green waste – using multi-tunnel decomposition technology with the curing phase in turned windrows in an enclosed building. It also had different biofilters for the treatment of exhaust gases.

OFMSW is defined as household waste and other waste which, because of its nature or composition, is similar to household waste, capable of undergoing anaerobic or aerobic decomposition, excluding green waste from gardens and parks, which includes tree cuttings, branches, grass and wood (European Commission2001).

### *(i) Obtaining OFMSW*

The OFMSW for the plant was obtained by sorting at origin in six nearby municipalities and the main supply centre of fresh products to Barcelona. The green waste was from parks and gardens in these municipalities. OFMSW and green waste were considered a mixture of homogeneous organic material for collection and transport to the plant in 16 t maximum authorised load (MAL) trucks. The average distance between collection points and the composting facility was 14 km, with the impacts of return trips made by the trucks also attributed. The remaining considerations were described by Bravo (2004) and Muñoz et al. (2002). We added the energy needed for grinding green waste, which was done at a different facility to that studied.

### *(ii) Industrial composting process*

The industrial composting process in the studied plant included four main phases:

- Pre-treatment: reception of the material, grinding to a maximum of 80 mm and mixing at a ratio of 1:1 OFMSW and green waste.
- Composting or decomposition: this was in tunnels which had a forced aeration and irrigation system to optimise decomposition. The material remained there for a minimum of two weeks and the temperature must exceed 65$^{o}$C for two or three days to guarantee hygienization of the material. The temperature and oxygen concentration were automatically controlled.
- Curing: the decomposed material was put in piles periodically turned mechanically to enable internal aeration, and which were watered when there

was a need to increase humidity. The total maturation time was generally about eight weeks.

- Refining: screening to separate the mature compost, usually less than 15 mm, from the other fractions. Such were the green waste that had not totally decomposed and was put back into the composting process, and the solid waste fraction which was dumped. The latter included non-organic material such as plastics and tins – inappropriate items introduced in the process with OFMSW and green waste – and composted material not correctly separated.

The complete process taken around 10 weeks, and then it was considered that the material was stable and could be applied to the soil without any problems associated to the incomplete decomposition of easily degradable materials.

To consider the plant's flows, we used values for the 2003–2006 period (Table 4.3), provided by the managers of the plant and by Cisneros (2006). We also accounted for the impact of the building and the main machinery, including production of the necessary construction materials, their transport and waste management (Althaus et al. 2004; ITeC 2010; WSOFM 2010; SCLCI 2012).

**Table 4.3. Inventory of average input and output flows of materials and energy at the composting facility in Castelldefels (Barcelona) for the 2003–2006 period, and emission factors considered for the decomposition of organic waste (OFMSW and green waste).**

| Type of flow | Units | Annual flow | Type of flow | Units | Annual flow |
|---|---|---|---|---|---|
| *Input* | | | *Output* | | |
| **Electric supply** | MWh $year^{-1}$ | 465.9 | **Compost production** | t $year^{-1}$ | 2,094 |
| **Diesel oil consumption** [a] | $m^3$ $year^{-1}$ | 64.3 | **Solid waste dumped** | t $year^{-1}$ | 2,823 |
| **Water consumption** [b] | $m^3$ $year^{-1}$ | 3,935 | *Emissions* | | |
| **Total organic waste** | t $year^{-1}$ | 14,461 | **$CO_2$ biogenic** [c] | t year | 2,385.89 |
| **OFMSW** | t $year^{-1}$ | 10,022 | **$NH_3$** [d] | t year | 1.59 |
| **green waste** | t $year^{-1}$ | 4,439 | **$CH_4$** [c] | t year | 5.45 |
| **Inappropriate items** | % | 20 | **VOC** [d] | t year | 17.50 |
| | | | **$N_2O$** [c] | kg $year^{-1}$ | 0.30 |

[a] *Diesel consumption includes the consumption by the OFMSW treatment plant and the fuel required to grinding the green waste at an external plant.*
[b] *Water consumption includes the tap water supplied and rainwater collected in biofilters and reused.*
[c] *Bibliographic data on output from biofilters.*
[d] *Experimental data on output from biofilters.*

### (iii) Biofilter characteristics and gaseous emissions

The exhaust gases generated in the composting tunnels and in the pre-treatment and curing areas were treated using a biofilter before being released into the environment. This was a layer of organic material biologically enriched with ground pine, in which the contaminants were retained and metabolised by microorganisms.

Although during composting, more than 100 types of compound gases are emitted (Chung 2007), Table 4.3 shows those considered, representing 99% of the total emissions (Beck-Friis et al. 2001; Pagans et al. 2006a; 2006b; Amlinger et al. 2008).

Emissions of $NH_3$ and VOC were obtained using weekly experimental measures in the decomposition biofilters at the composting facility in Castelldefels (Colón et al. 2009). For the other gases considered, the emissions were estimated using theoretical factors and the following: (i) That 20% of the total waste was non-organic, so does not have associated emissions. (ii) We determined the efficiency of the biofilters, which was 80% for $CH_4$ and 85% for $N_2O$ (Sonesson et al. 1997; EIPPC Bureau 2006). (iii) In accordance with the European Commission (Smith et al. 2001) on the decomposition of biodegradable materials, the carbon cycle is short-term and is relatively constant from one year to another, so there is no net global warming and biogenic $CO_2$ was not included in the calculation of environmental impacts. Table 4.3 shows the annual factor applied for each substance.

### 4.3.2. Stage of mineral fertilizer production (FP)

The data on the manufacture of mineral fertilizer were from the processes inventoried in the database ecoinvent data v1.2 (SCLCI 2012). As described by Nemecek et al. (2004), this stage included the production infrastructure; transport of prime and intermediate materials to the plant; synthesis of the chemical components required, and deposition or treatment of the waste generated. Transport from the site of production to that of cultivation was not included as this was accounted for in FT.

### 4.3.3. Stages of mineral fertilizer (FT) and compost (CT) transport

In the European-Mediterranean area, the mineral fertilizer is produced in Germany and transported to the distributors in each country. Therefore, we considered transport over a distance of 1950 km in 16 MAL trucks. Due to the efficiency of international transport platforms, unlike other forms of transport included in the inventory, we considered that, for FT, the truck returned to Germany with another load and therefore only the outward journey was included.

The compost was transported from the composting facility in Castelldefels, a town located 88 km from the experimental plots. For CT we accounted for the environmental loads for return trips in a 16 MAL truck.

### 4.3.4. Cultivation stage (Cu)

#### *(i) Harvest production and quality*

Table 4.2 presents the commercial tomato production per surface unit and two commercial parameters, the average diameter and weight of the fruit. Analysis of variance was by means of the General Linear Model, and the least significant difference (LSD, $P = 0.05$) was used to establish differences between treatments using the Enterprise Guide software package (SAS institute Inc. 2006).

#### *(ii) Doses of fertilizing products applied*

Compost was not applied for each new crop, so reducing the burdens of the machinery used during compost application. We considered that the compost applied to the experimental fields had to supply nutrients to two other crops apart from tomatoes, a total period of 18 months. The environmental costs associated with the production, transport and application were distributed depending on the duration of each crop, 133 days in this case.

The contribution of N for each treatment was calculated by taking into account the following: (i) The high nitrogen content of the irrigation water used – 3 miliequivalents of nitrates – reduces the need for nitrogenated fertilizers. This was a normal value in vulnerable Mediterranean areas (ECC1991). (ii) The nitrogen absorbed by the crops was 39.1 g N $m^{-2}$, 35.8 g N $m^{-2}$ and 41.3 g N $m^{-2}$ for $C_H$, $C_L$ and M, respectively. (iii) The composition of the mineral fertilizer and the compost, taking into consideration the proportion of the total compost for the 133 days of the tomato crop cycle. The doses for each treatment are included in Table 4.4.

**Table 4.4. Types and quantities of fertilizers applied for each treatment.**

| Dose | $C_H$ | | $C_L$ | | M | |
|---|---|---|---|---|---|---|
| | Dose of fertilizer (g $m^{-2}$) | Dose of N (g N $m^{-2}$) | Dose of fertilizer (g $m^2$) | Dose of N (g N $m^{-2}$) | Dose of fertilizer (g $m^{-2}$) | Dose of N (g N $m^{-2}$) |
| **Compost** | 1,931.0 [a] | 23.4 [b] | 965.5 [a] | 11.7 [b] | 0.0 | 0.0 |
| **Irrigation water and rainfall** [c] | 23.9 | 23.9 | 23.9 | 23.9 | 23.9 | 23.9 |
| **$KNO_3$** | 0.0 | 0.0 | 90.2 | 12.5 | 146.2 | 20.2 |

[a] *Proportion of the total compost dose for the 133 days of the tomato crop. The total doses for the 18 months were calculated as 7,840 g $m^{-2}$ for $C_H$ and 3,924 g $m^{-2}$ for $C_L$.*

[b] *Nitrogen available the first year after spreading the compost.*

[c] *Irrigation water makes the largest contribution, with a concentration of 3 miliequivalents of $HNO_3$.*

The moisture content of the compost applied to the experimental plots was 27%, slightly lower than the 30–40% interval established by RD 824/2005 (Ministerio de la Presidencia2005), and the organic matter content was 53% for dry material. In relation to heavy metals, this was class A compost (European Commission2001). The complete analysis of the sample was included in Muñoz et al. (2008a).

Heavy metals released in field soil through compost application were not taken into account as their concentration were lower than regulation limits and because ecotoxicity categories, the only categories affected by heavy metal release, were not considered in this study (see Section 4.2.6).

### *(iii) Sub-stage of fertirrigation infrastructure (CuF)*

The system used in the experimental plots to supply water, fertilizers and phytosanitary products was a fertirrigation infrastructure with pumps and pipes for extracting and channelling the water, tanks, electrovalves for controlling dosage and a network of pipes supplying the plants (Table 4.5). T-TAPE® pipes were used as a drip irrigation system. We considered the transport of these elements from the distribution point, in 3.5 t MAL vans, and the waste management. The burdens attributed to the CuF, per functional unit, were assigned in accordance with the characteristics of the elements used and the duration of the crop with respect to their lifespan.

### *(iv) Phytosanitary substances sub-stage (CuP)*

The active substances considered in the CuP are chlorothalonil, sulphur, bacillus, miclobutanil, spinosad, indoxacarb, cymoxanil, famoxadone, bromopropylate, copper oxychloride, azoxystrobin and surfactant. The minimum doses of phytosanitary products were applied as recommended by the Spanish Ministry for Environment and Rural and Marine Affairs (MMAMRM 2008).

**Table 4.5. Manufactured materials and lifespan considered for the elements of fertirrigation infrastructure.**

| Element | Material | Amount (kg $m^2$) | Lifespan (years) |
|---|---|---|---|
| **Well pipes** [a] | Polyvinyl chloride | 8.949E-4 | 10 |
| **Primary distribution pipes** [b] | Polyethylene | 1.909E-3 | 10 |
| **Secondary distribution pipes** [c] | Polyethylene | 5.250E-2 | 1 |
| **Electrovalves** [d] | Polyvinyl chloride | 1.323E-2 | 10 |
| **Water tank** [e] | Concrete | 2.397E-2 | 50 |
| | Steel | 4.762E-2 | 50 |
| **Fertilizer tanks** [f] | Polyethylene | 4.247E-3 | 10 |
| **Water distribution pump** | Steel | 9.440E-4 | 20 |
| **Water extraction pump** | Steel | 9.440E-4 | 20 |

[a] *Pipes transporting water from the pumps to the tanks.*
[b] *Pipes transporting irrigation water from the tanks to each of the parcels.*
[c] *T-TAPE® with built-in drip irrigation systems.*
[d] *One electrovalve per treatment.*
[e] *One tank for the three treatments.*
[f] *500 L tanks are considered. Three tanks for treatments M and $C_L$ and one for treatment $C_H$.*

### *(v) Field management sub-stage (CuO)*

In CuO, we took into account the irrigation and post-application emissions as well as the tractor and other agricultural machinery (Table 4.6). The phases considered for the machinery were: obtaining the prime materials, manufacture, maintenance, fuel consumption during the field operations, and waste management. The operations were the same for the three treatments, except for those related to the application of compost, which were not considered in the case of M.

**Table 4.6. Characteristics of the machinery used during the yield stage.**

| Machinery | Weight (kg) | Life span (h) | Use of machinery (h) [a] | | |
|---|---|---|---|---|---|
| | | | $C_H$ | $C_L$ | M |
| **Tractor** | 3,900 | 7,200 | 6.80 | 6.80 | 3.59 |
| **Plough** | 300 | 3,000 | 3.75 | 3.75 | 1.25 |
| **Tow** | 1,490 | 6,000 | 0.58 | 0.58 | 0 |
| **Fertilizer spreader** | 1,000 | 800 | 0.12 | 0.12 | 0 |
| **Furrow opener** | 1,190 | 3,000 | 0.79 | 0.79 | 0.79 |
| **Spray bag** | 1,000 | 1,000 | 1.54 | 1.54 | 1.54 |

[a] *Hours of use the machinery in each treatment considering an area of 5,000 $m^2$.*

We also considered the burdens associated with the production and transport of the harvesting elements: the high density polyethylene (HDPE) crates and steel handcarts.

Irrigation water was pumped from a nearby well (depth, 10–15 m) to the fields using two pumps, one to pump the water out of the wells (4 kW) and the other to spread it over the plots (2.7 kW). The supply of water depended on the evapotranspiration demands, and in each treatment a meter was installed to determine the final consumption of water, 566 L $m^{-2}$.

To calculate the post-application emissions, we considered the nitrogen contributions made by each fertilizer and the nitrogen concentration absorbed by the crops. We accounted for emissions of $NH_3$, $N_2O$, $NO_x$ and $N_2$ into the air and $NO_3$ into the water, in accordance with Audsley's hypothesis for sandy soils (Audsley et al. 2003).

#### *(vi) Management of waste generated during the cultivation stage*

During the Cu stage, three different types of waste were generated: green waste containing the non-commercialised parts of the plant and the non-commercial tomatoes; waste from the fertirrigation infrastructure; and waste from harvesting elements, which included crates and handcarts. As discussed in Section 4.2.4, management of green waste and steel of the harvesting handcarts were not included in the impact assessment, as a recycling or recuperation treatment was considered for them. The waste from the fertirrigation infrastructure and the fruit harvesting crates were deposited in a landfill 53km from the fields at the end of their useful life. The impact of their management entailed transporting to the landfill, in 16 t MAL trucks, plus depositing, which depended on the material (SCLCI 2012). The burdens of the fertirrigation infrastructure were attributed to the CuF, and those of the harvesting crates, to CuO.

### 4.3.5. Dumping of OFMSW and green waste

As mentioned in Section 4.2.4, the alternative OFMSW management method selected was dumping, as the two alternative methods, biomethanization and incineration, are associated with the extra function of generation of energy, and incineration is not suitable for organic waste sorted at the origin as it is excessively damp. Until recently, dumping of OFMSW and green waste was common, but was controlled by Directive 1999/31/CE (EC1999).

There was no specific data (Finnveden et al. 2007) on the impacts of depositing OFMSW and green waste in landfill. It had been assimilated in that considered by database ecoinvent (SCLCI 2012) for the management of municipal solid waste in landfill, and to this the collection (manufacture and maintenance of containers) and the transport to the dump, which was located about 57 km from the collection area, were added. The impact of land use for the dump was also considered, as well as the construction of the landfill and road accesses, the machinery and fuel used for operation, and the combustion of methane without making use of the energy (Doka 2007; SCLCI 2012). One hundred years were considered to be the time limit of impact (Doka 2007).

Environmental burdens of dumping the amount of composted OFMSW and green waste equal to that used to produce the compost for the $C_H$ and $C_L$ treatments were subtracted from the total impact of these treatments.

## 4.4. Results and discussion

Table 4.7 presents the environmental burdens associated with the three treatments, according to the considerations in the inventory.

**Table 4.7. Total environmental impacts and by stages for treatments $C_H$, $C_L$ and M.**

| Impact categories | Units | Total impact | | Production stage | | Transport stage | | Cultivation stage | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | Avoided burdens [a] | Total [b] | FP | CP | FT | CT | CuF | CuP | CuO |
| | | | | | | | | | | |
| *Compost treatment ($C_H$)* | | | | | | | | | | |
| **ADP** | kg Sb eq. | 5.23E-01 | 1.26E+00 | | 6.72E-01 | | 1.08E-01 | 1.11E-01 | 4.51E-02 | 3.20E-01 |
| **GWP** | kg $CO_2$ eq. | -9.80E+02 | 1.70E+02 | | 9.84E+01 | | 1.54E+01 | 1.10E+01 | 3.69E+00 | 4.10E+01 |
| **OLDP** | kg CFC-11 eq. | 1.88E-06 | 1.68E-05 | | 9.58E-06 | | 2.06E-06 | 6.92E-07 | 6.47E-07 | 3.86E-06 |
| **POP** | kg $C_2H_4$ eq. | 6.42E-01 | 8.83E-01 | | 8.62E-01 | | 4.31E-03 | 4.88E-03 | 1.10E-03 | 1.13E-02 |
| **AP** | kg $SO_2$ eq. | 8.95E-01 | 1.63E+00 | | 1.04E+00 | | 8.99E-02 | 6.39E-02 | 2.49E-02 | 4.14E-01 |
| **EP** | kg $PO_4^{3-}$ eq. | -4.06E+00 | 2.42E-01 | | 1.35E-01 | | 1.82E-02 | 1.73E-02 | 1.54E-03 | 6.98E-02 |
| **CED** | MJ eq. | 1.34E+03 | 3.22E+03 | | 1.73E+03 | | 2.58E+02 | 2.84E+02 | 1.12E+02 | 8.34E+02 |
| *Compost and mineral fertilizer treatment ($C_L$)* | | | | | | | | | | |
| **ADP** | kg Sb eq. | 6.29E-01 | 1.01E+00 | 7.67E-02 | 3.29E-01 | 5.59E-02 | 4.78E-02 | 1.17E-01 | 5.13E-02 | 3.33E-01 |
| **GWP** | kg $CO_2$ eq. | -4.51E+02 | 1.33E+02 | 2.28E+01 | 4.93E+01 | 8.02E+00 | 6.85E+00 | 1.16E+01 | 4.20E+00 | 3.07E+01 |
| **OLDP** | kg CFC-11 eq. | 5.37E-06 | 1.32E-05 | 1.16E-06 | 4.52E-06 | 1.07E-06 | 9.15E-07 | 7.38E-07 | 7.36E-07 | 4.02E-06 |
| **POP** | kg $C_2H_4$ eq. | 3.45E-01 | 4.71E-01 | 1.30E-03 | 4.47E-01 | 2.24E-03 | 1.91E-03 | 5.09E-03 | 1.25E-03 | 1.18E-02 |
| **AP** | kg $SO_2$ eq. | 8.34E-01 | 1.08E+00 | 6.08E-02 | 5.17E-01 | 4.67E-02 | 3.99E-02 | 6.70E-02 | 2.83E-02 | 3.17E-01 |
| **EP** | kg $PO_4^{3-}$ eq. | -2.05E+00 | 1.61E-01 | 1.17E-02 | 6.47E-02 | 9.45E-03 | 8.08E-03 | 1.82E-02 | 1.76E-03 | 4.76E-02 |
| **CED** | MJ eq. | 1.59E+03 | 2.57E+03 | 1.75E+02 | 8.52E+02 | 1.34E+02 | 1.15E+02 | 2.98E+02 | 1.28E+02 | 8.67E+02 |
| *Mineral fertilizer treatment (M)* | | | | | | | | | | |
| **ADP** | kg Sb eq. | 6.75E-01 | 6.75E-01 | 1.21E-01 | | 7.27E-02 | | 1.14E-01 | 4.55E-02 | 3.21E-01 |
| **GWP** | kg $CO_2$ eq. | 1.06E+02 | 9.09E+01 | 3.61E+01 | | 1.04E+01 | | 1.12E+01 | 3.72E+00 | 2.94E+01 |
| **OLDP** | kg CFC-11 eq. | 8.46E-06 | 8.46E-06 | 1.84E-06 | | 1.39E-06 | | 7.17E-07 | 6.53E-07 | 3.86E-06 |
| **POP** | kg $C_2H_4$ eq. | 2.24E-02 | 2.24E-02 | 2.06E-03 | | 2.91E-03 | | 4.95E-03 | 1.11E-03 | 1.14E-02 |
| **AP** | kg $SO_2$ eq. | 7.04E-01 | 5.52E-01 | 9.61E-02 | | 6.07E-02 | | 6.51E-02 | 2.51E-02 | 3.05E-01 |
| **EP** | kg $PO_4^{3-}$ eq. | 1.29E-01 | 9.57E-02 | 1.84E-02 | | 1.23E-02 | | 1.77E-02 | 1.56E-03 | 4.58E-02 |
| **CED** | MJ eq. | 1.69E+03 | 1.69E+03 | 2.77E+02 | | 1.74E+02 | | 2.90E+02 | 1.13E+02 | 8.37E+02 |

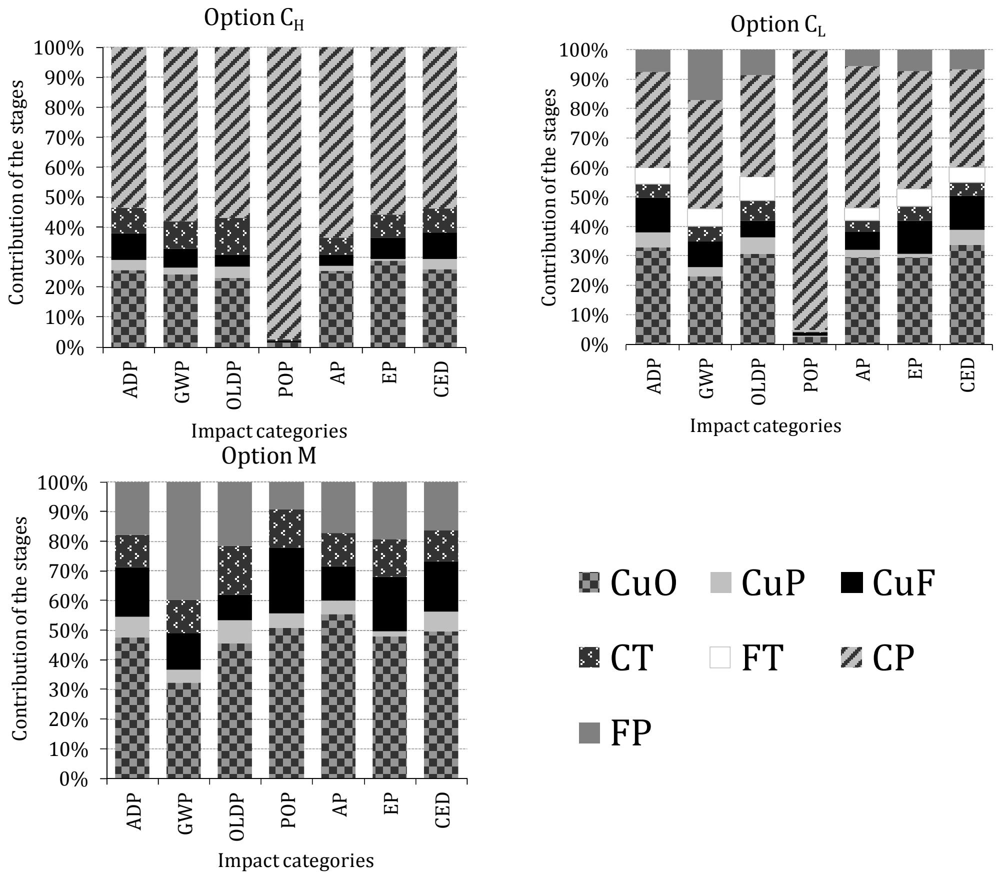
[a] *Total impact of the system considering avoided burdens through composting by not dumping OFMSW and green waste in treatments $C_H$ and $C_L$.*
[b] *Total impact of the system without considering avoided burdens (sum of the seven columns on the right).*

### 4.4.1. Harvest production and parameters of agricultural quality

No significant statistical differences were observed between treatments for commercial production or for the two quality parameters analysed, namely the fruit calibre and weight (Table 4.2). The harvests were slightly greater than normal values expected for the commercial production of tomatoes in the area (Muñoz et al. 2008b; 2008c). However, total tomato harvest, which is the adding up of non-commercial and commercial production, for $C_H$ was slightly lower than that ones for the other treatments because non-commercial production for $C_H$ was only 14% (in weight) of the commercial tomato production whereas it is 23% and 26% for treatment M and $C_L$.

### 4.4.2. Environmental assessment by stages and sub-stages

Figure 4.2 illustrates the results broken down by stages and sub-stages as defined in Section 4.2.2. These results do not take into consideration the avoided burdens for $C_H$ and $C_L$ on depositing composted OFMSW and green waste in landfill.




*Stages: CuO, sub-stage of field operations; CuP, sub-stage of phytosanitary substances; CuF, sub-stage of fertirrigation infrastructure; CT, transport of compost; FT, transport of mineral fertilizers; CP, production of compost; FP, production of mineral fertilizers.*

**Figure 4.2. Contribution to total environmental impacts of the stages for the three treatments.**

#### *(i) Treatment with compost*

For treatment $C_H$, the biggest impact for all the categories was the compost production stage, between 53% and 98% of the total impact, depending on the impact category considered. This is followed by field operations sub-stage, with an impact of between 23% and 29% of the total, for all categories except photochemical oxidation potential (POP). In the case of POP, the compost production stage is responsible for 98% of the total impact due to the emissions of volatile organic compounds. The cumulative energy demand (CED) for compost production is 1390 MJ eq. per ton of tomatoes for $C_H$, while for CuO it is 669 MJ eq.

Depending on the category, the cultivation stage, which is the sum of CuF, CuP and CuO, has an impact of between 2%, for POP, and 38%, for abiotic depletion potential (ADP) and cumulative energy demand. The main contributions to the impact of CuO are the water pumps and collection crates.

Figure 4.3 shows the contributions to the total impact of the composting production stage. The electricity consumption of the facility has the highest impact for ADP, GWP, CED and acidification potential (AP), representing 33–39% of the impact assigned to the CP stage for these categories. For the category POP, the emissions of volatile organic compounds contribute 96% of the impact. In the case of eutrophication (EP), ammonium emissions cause the greatest impact, 47%, with respect to the total impact associated to CP. For ozone layer depletion potential (OLDP), diesel consumption contributes 33% of the impact of CP also contributing considerably to the ADP, and CED categories. The management of solid waste generated at the composting facility contributes considerably to all categories apart from POP, between 12% and 29%; ammonium emissions to category AP and EP; and collection of OFMSW and green waste contributes more than a 15% to categories ADP, GWP, OLDP, EP and CED.

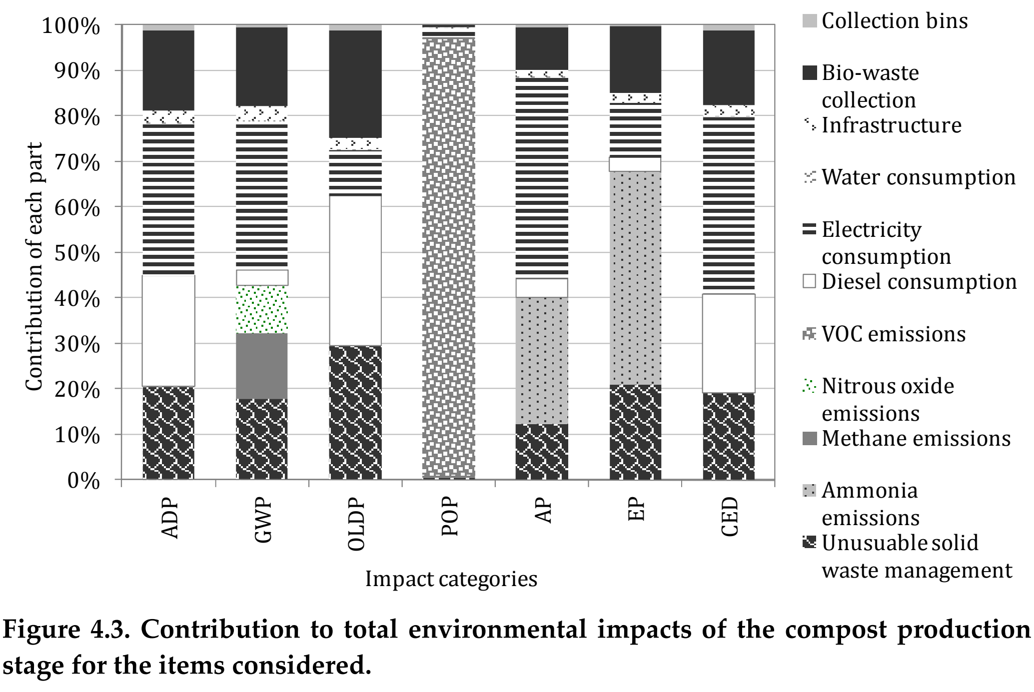



**Figure 4.3. Contribution to total environmental impacts of the compost production stage for the items considered.**

The higher impact of the CP stage with respect to the others can also be explained by the processes with which it is compared. The manufacture of mineral fertilizer is a highly efficient process resulting in a product with a high concentration of nutrients, so only small amounts are required to be transported, even though this involves long distances, while the transport of compost is local. As the main energetic source is solar energy (photosynthesis), agricultural activities generally have little environmental impacts in comparison to industrial activities.

#### *(ii) Treatment with compost and mineral fertilizer*

In the case of the $C_L$ treatment, compost production is the highest or one of the stages with the most impact for all the categories with a contribution to the total impact of between 33% and 95%, and an associated consumption of 682 MJ eq. $t^{-1}$ tomatoes (Figure 4.2).

Considering all the categories, fertilizer production accounts for between 0% and 15% of the total impact, compost production between 33% and 95%, and the cultivation stage, between 4% and 50%, in all cases depending on the category.

#### *(iii) Treatment with mineral fertilizer*

For treatment M, sub-stage field operations has the greatest impact for all categories, between 28% and 51% of the total, depending which is considered (Figure 4.2). The energy consumption for this sub-stage is 695 MJ eq. $t^{-1}$ tomatoes, whereas for the FP stage it is 229 MJ eq. $t^{-1}$ tomatoes. The burdens associated with the transport of the mineral fertilizer have an impact of between 0.3 and 1.4 times those of FP. The cultivation stage has between 42% and 78% of the total impact of M.

For the Cu stage, the differences between the percentages of the total impact for each treatment ($C_H$, $C_L$ and M), are the result of the combination of three factors. Firstly, the total impact is different for each treatment, and also the relative weight of the Cu stage, even though the absolute values may be similar (differences below 24%). Secondly, the higher the tomato production, the more evenly distributed the impacts among the functional units. Finally, there are differences between the inputs associated to the cultivation stage depending on the treatment, such as the number of deposits of manure, the post-application emissions and whether the machinery for compost spreading is taken into consideration.

### 4.4.3. Environmental assessment of the total impacts for each treatment

Figure 4.4 presents the total impacts associated to the three systems, i.e. the sum of the impacts for the considered stages for each of them, and the total impacts associated to the expanded systems. The latter consider the burdens avoided through composting by not dumping OFMSW and green waste, for treatments $C_H$ and $C_L$ that use compost. The $C_L$ system has intermediate values between $C_H$ and M, as it is a mixed fertilization option and the functional unit used, tomato production, is very similar for the three treatments.

The OFMSW and green waste treated by composting was 1.64 t $t^{-1}$ tomatoes for treatment $C_H$, 0.86 t $t^{-1}$ tomatoes for $C_L$, and 0 t $t^{-1}$ tomatoes for M, which does not use compost. We considered that these amounts were the OFMSW and green waste that was not managed by dumping to calculate the burdens avoided (see Section 4.3.5). The results with and without considering avoided burdens are included in Table 4.7.

#### *(i) Environmental assessment of the system*

As shown in Figure 4.4, for all the categories, and without considering subtracted burdens, the order of total environmental impacts is:

$C_H > C_L > M$

Treatment $C_H$ has 13–47% more impact than $C_L$ and between 33% and 95% more impact than M, depending on the impact category and without considering the POP. The biggest difference between treatments arises for POP.

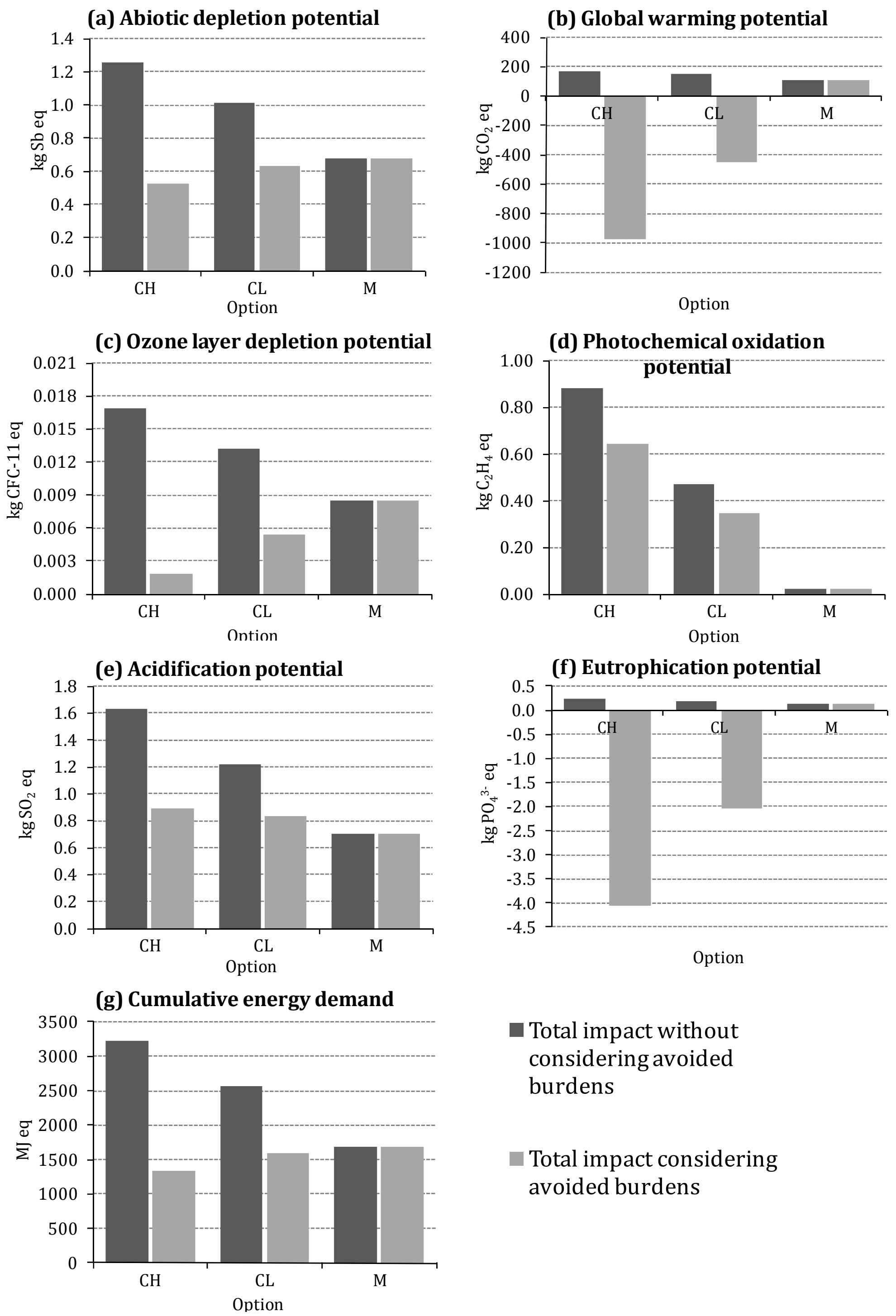



**Figure 4.4. Total impact for each category with and without additional burdens for the three treatments.**

Total energy consumption is 3,220 MJ eq. $t^{-1}$ tomatoes for treatment $C_H$ and 1,690 MJ eq. $t^{-1}$ tomatoes for M, with 170 kg $CO_2$ eq. $t^{-1}$ tomatoes and 106 kg $CO_2$ eq. $t^{-1}$ tomatoes emitted, respectively.

The compost production stage has the highest effect on the order of results, especially the gas emissions in those categories for which there is the greatest difference between treatments, POP and AP.

***(ii) Environmental assessment of the system considering avoided burdens***

If we consider the expanded system by integrating avoided burdens, the non-deposit in dump of composted OFMSW and green waste, the relative order of impacts is reversed for all impact categories except POP and AP (Figure 4.4):

$C_H^{exp} < C_L^{exp} < M^{exp}$

In the expanded case of treatment with mineral fertilizer ($M^{exp}$), the impact is the same whether or not we consider the expansion of the system, as there is no avoided impact to be subtracted ($M=M^{exp}$). Treatment $C^{exp}$ implies the consumption of 1,339 MJ eq. $t^{-1}$ tomatoes and avoids the emission of 980 kg $CO_2$ eq. $t^{-1}$ tomatoes.

For all the impact categories except POP and AP, treatments $C_H^{exp}$ and $C_L^{exp}$ have fewer impacts than $M^{exp}$, and, in the case of GWP and EP, have negative impacts, i.e. avoid impacts. For category AP the differences between treatments are small, M is slightly lower than $C_H$, with differences less than 21%; while for POP, $C_H^{exp}$ has 32 times more impact than $M^{exp}$.

For the categories GWP and EP, the impact of treatment $C_H^{exp}$ is 1.2 and 1.0 times lower, respectively, than $C_L^{exp}$ and 10.2, and 32.5 times lower than $M^{exp}$. The causes of these major differences are the impacts associated with the deposit of organic waste in landfill: the $CO_2$ and $CH_4$ emissions contribute to category GWP, and the emissions and lixiviates of nitrogen and phosphorous contribute to EP (Finnveden et al. 1995).

For categories ADP, ODP and CED the impact is positive for the three treatments: $C_H^{exp}$ has 16–65% less impact than $C_L^{exp}$ and 21–78% less impact than $M^{exp}$.

# 4.5. Conclusions and future perspectives

The results of the LCA indicate that the stage with the major impact is compost production, with between 53% and 98% of the total impact, depending on the impact category, mainly due to gas emissions generated and energy consumption at the composting facility. The cultivation stage also contributes substantially to the total impact.

Treatment $C_H$ has 33–95% more impact than treatment with mineral fertilizer (M), depending on the category of impact and excluding POP, as a consequence of compost production. We also considered an expanded system that integrates the burdens avoided by not depositing the composted OFMSW and green waste in landfill. For the expanded scenario, treatment with compost ($C_H^{exp}$) has similar or less impact than $M^{exp}$ for all the categories apart from POP, for which $C_H^{exp}$ has 32 times more impact than $M^{exp}$. In this case, compost can possibly be an environmentally better option than mineral fertilization for all categories except POP.

The application of compost as a fertilizer for tomato crops apparently not has a negative effect on harvest or product quality. Quite the opposite, non-commercial production is significantly lower for treatment $C_H$ although commercial production is similar between treatments.

To improve treatment with compost, efforts should be focus on the compost production stage, optimizing the exhaust gas treatment systems and minimizing energy consumption. Furthermore, future research must certainly provide new indicators to measure the potential local impacts in Mediterranean regions, such as erosion and water consumption, and specific data on the environmental impacts associated to depositing organic wastes in dump, for a more accurately subtraction of impacts.

# Chapter 5. Assessment of tomato Mediterranean production in open-field and standard multi-tunnel greenhouse, with compost or mineral fertilizers, from an agricultural and environmental standpoint

**Chapter 5** is based on the following paper:

Martínez-Blanco, J., Muñoz, P., Antón, A., & Rieradevall, R. (2011) Assessment of tomato Mediterranean production in open-field and in standard multi-tunnel greenhouse with compost or mineral fertilizers from an agricultural and environmental standpoint. Journal of Cleaner Production, 19, 985-997.

# Abstract

This study presents detailed and comprehensive inventories on the horticultural production of tomato using compost ($C_L$) or mineral fertilizers (M), in both open fields (OF) and greenhouses (GH), providing information on the environmental impacts and assessing the agronomic viability of the four cultivation options. Life Cycle Assessment (LCA) was used to calculate the potential environmental impacts of the tomato production cycle per ton of product. The stages in the assessment included: mineral and organic fertilizers production, fertilizers transport, cultivation stage and greenhouse stage. The data were obtained experimentally in pilot fields and in an industrial composting facility using municipal organic waste, both located in the Mediterranean area. The results indicate that replacing a fraction of the mineral fertilizers dosage with compost is a good option, as this did not alter yield or fruit size parameters. Greenhouse protection increased infrastructure materials requirements but enhanced harvest by almost 50% and reduced the water and pesticides requirement. Compost production and greenhouse stages were the most impacting stages. Without subtracting the avoided burdens by composting and not dumping organic waste, the cultivation option OF_M had the lowest and OF_$C_L$ the highest impact. When avoided burdens were taken into consideration, the environmental impacts of the four cultivation options varied, depending on the impact category, with bigger differences due to fertilization as a variable rather than the production system.

## Keywords

Organic fraction of municipal solid waste; Composting process; Environmental impacts; Horticulture technologies; Greenhouse; Food production.

# 5.1. Introduction

Most vegetable crops are relatively concentrated in areas with a mild winter climate where typical weather is generally more favorable (Montero et al. 2009b) and even more under greenhouse protection.

Currently, the total greenhouse protected area in the southern European Union member states is about 90,000 ha, with Spain at the top of the list with 54,000 ha (Sigrimis et al. 2009). In Spain, tomato (Lycopersicon esculentum) production represents 26% of the total area under greenhouses (MMAMRM 2008).

Greenhouse cultivation has been often perceived by consumers as an artificial technology, characterized by low nutritional quality of the final product, intensive use of chemicals and infrastructure and a major visual impact (Muñoz et al. 2008b). However, the vast majority of Mediterranean greenhouses are of low cost and low technology, having plastic as the covering material and few climate control systems (Antón et al. 2005a; 2007).

Traditionally, fertirrigation practices in the Mediterranean region involved excessive nutrient fertilization, with the consequent contribution to groundwater pollution and eutrophication (Muñoz et al. 2008c). Compost application appears to be a good complement as a fertilizer: it improves the physical, biological and chemical properties of the soil (Martínez-Blanco et al. 2009b) and the rate of nitrogen release in the soil is slower (Hargreaves et al. 2008). Furthermore, composting closes metabolic cycles by using the organic fraction of municipal solid waste (OFMSW) in food production. The potential of quality compost production in the European Union is estimated at 35–40 Mt year$^{-1}$ (European Commission 2008), equivalent to a minimum of 131,000 t of available organic nitrogen (Ministerio de la Presidencia2005).

Some studies have compared the agronomical (Poudel et al. 2001; Elherradi et al. 2005; Ghorbani et al. 2008; Hargreaves et al. 2008; Odlare et al. 2008) and the environmental (Lundie and Peters 2005; Hansen et al. 2006b; Tidaker et al. 2007; Blengini 2008; Martínez-Blanco et al. 2009b; Meisterling et al. 2009; Ruggieri et al. 2009a) performances of organic and mineral fertilizing solutions. Others have compared the environmental and agronomical performances of different production systems (such as greenhouse, shadow house and open field) in a Mediterranean climate (Antón 2004; 2005a; Muñoz et al. 2008b; Romero-Gámez et al. 2009). Antón et al. (2005b) studied several options for waste management in greenhouse horticultural production and found composting the best option for biomass waste. However, most of these papers focused on improving the production process.

The aim of this research was to investigate different scenarios considering two production systems, open field and greenhouse covered, and two fertilizing options, to find the best environmental option without loss of yield.

The environmental tool of Life Cycle Assessment (LCA) was used for assessing the potential environmental impacts of a ton of tomatoes, considering its entire life cycle from resources extraction to waste disposal, excluding commercialization (ISO 2006b), for the four cultivation options.

# 5.2. Materials and methods

Here we present the methodology of the study which is coincident with the goal and scope definition in the ISO 14044 (ISO 2006b).

## 5.2.1. Goal of the study

The aims of this LCA were to determine agricultural and environmental differences of four cultivation options characterized by greenhouse or open field cultivation using compost plus mineral fertilizers or only mineral fertilizers. Five main aspects were considered: the harvest; size and weight quality parameters; the environmental inventory that includes energy and resources consumption and emissions (see Annex I.I); the environmental impacts; and finally, the identification of the critical environmental stages.

## 5.2.2. Description of the system

Figure 5.1 shows the stages and sub-stages considered for LCA in the cultivation options. The whole system, from obtaining raw materials required for the manufacture of the different elements to the management of generated waste, was considered for each stage. Commercialization was excluded as it was a local market with common characteristics for the cultivation options considered.

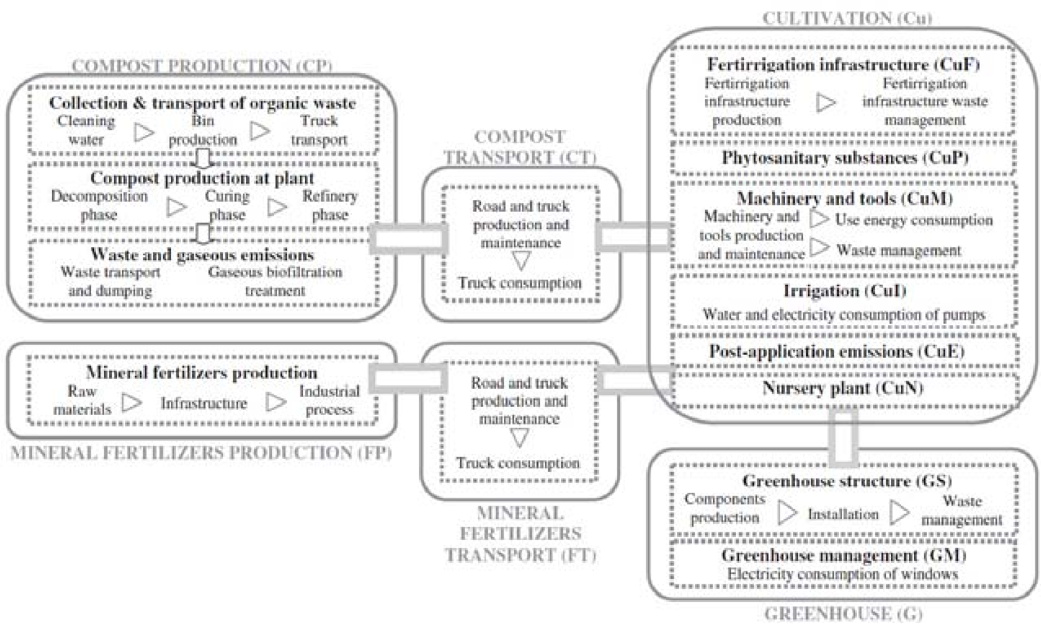
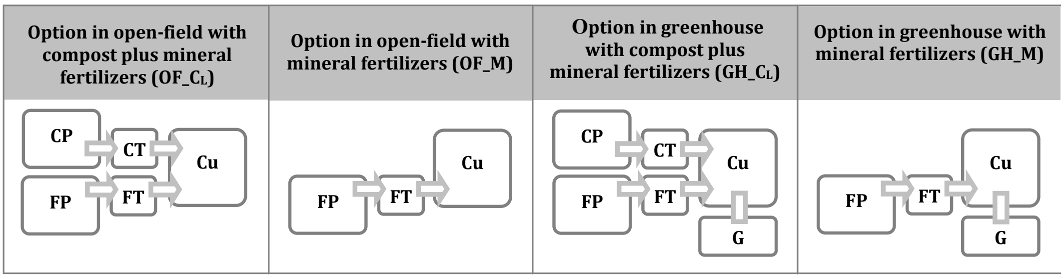



**Figure 5.1. Processes included in the stages and sub-stages considered in the system for the environmental impacts assessment of the four cultivation options.**

The four cultivation options were: open field cultivation with mineral fertilizers (OF_M) or compost plus mineral fertilizers (OF_ $C_L$); and greenhouse cultivation with mineral fertilizers (GH_M) or compost plus mineral fertilizers (GH_ $C_L$) (Figure 5.2).

Tomato yield using compost as the only fertilizer was much lower than the normal harvest levels in this area (Muñoz et al. 2008a; 2008b; 2008c; Martínez-Blanco et al. 2009b) so this option was not considered in this study (see Annex I.II).




*Stage acronyms: FP, Mineral fertilizers production stage; CP, Compost production stage; FT, Mineral fertilizers transport stage; CT, Compost transport stage; Cu, Cultivation stage; G, Greenhouse stage.*

**Figure 5.2. The four cultivation options considered and the stages included in each. The six stages are shown in Figure 5.1.**

## 5.2.3. Functional unit

Horticultural production of a ton of commercial tomato was chosen for the functional unit as a reference against which input and output flows were normalized (ISO 2006b), considering the main function of an agricultural system, the food production (Audsley et al. 2003). Different yields were taken into consideration for each cultivation option (Section 5.3.1.iii). Although tomato fruit characteristics were not the same between the cultivation options (Section 5.4.1), a standard ton of tomato was considered as the functional unit.

## 5.2.4. Quality and origin of the data in the inventory

The broad system of study (Figure 5.1) required a detailed data-collection process (Table 5.1). Most of this data were obtained experimentally in the fields and the composting facility by the authors or were local data previously collected by the research groups. When local information was not available, bibliographical sources and the database ecoinvent data v2.2 (SCLCI 2012) were used to complete the life cycle inventory.

### *(i) Data for compost production and transport*

The composting plant studied, located at Castelldefels (41°17′8″N, 1°58′16″E, Spain), in the Barcelona Metropolitan Area, had the capacity to treat 15,000 t of organic waste a year – organic fraction of municipal solid waste (OFMSW) and green waste (GW), mixed at a volume ratio of 1:1 – using the in-vessel decomposition technology with a curing phase in turned windrows in an enclosed building. The plant had different biofilters as exhaust gases treatment and leachates generated in the building and in the biofilters were collected and reintroduced. This composting technology is the most widely established in closed industrial composting facilities in Spain. The agronomic quality of the compost generated at the plant was within the established quality standards (Ministerio de la Presidencia2005; Martínez-Blanco et al. 2010).

The inventory of inputs and outputs of the industrial compost production was provided by the managers (Martínez-Blanco et al. 2009b), apart from collection and transport of OFMSW and GW, gas emissions (Martínez-Blanco et al. 2010) and characteristics of the compost used (Muñoz et al. 2008a).

**Table 5.1. Quality and origin of data used for the stages and sub-stages of the system and the field and cultivation characteristics. The data were split into experimental data (E), local data (L) and regional data (R).**

| Stage | Source | | |
|---|---|---|---|
| | E | L | R |
| | ***Mineral fertilizers production and transport*** | | |
| **Mineral fertilizers production** [1] | x | x | x |
| **Mineral fertilizers transport** [1] | | x | x |
| | ***Compost production and transport*** | | |
| **Compost production** [1,2,3,4] | x | x | x |
| **Compost transport** [1,3] | | x | x |
| | ***Cultivation stage*** | | |
| **Fertirrigation infrastructure** [3,5] | x | x | |
| **Phytosanitary substances** [5,6] | x | | x |
| **Machinery and tools** [1,3] | x | | x |
| **Irrigation** [7] | x | x | |
| **Post-application emissions** [8,9] | x | | x |
| **Nursery plants** [10] | x | x | |
| | ***Greenhouse stage*** | | |
| **Greenhouse structure** [1,10,11] | x | x | x |
| **Greenhouse management** [10] | | x | |

*Sources:* [1] *SCLCI* (2012); [2] *Martínez-Blanco et al.*(2010); [3] *Martínez-Blanco et al.*(2009b); [4] *Muñoz et al.* (2008a); [5] *MAPA* (2002); [6] *MMAMRM* (2008); [7] *RuralCat* (2009); [8] *Audsley* (2003); [9] *Brentrup and Küesters* (2000); [10] *Antón* (2004); [11] *Montero et al.*(2009a).

### *(ii) Data for mineral fertilizers production and transport*

Data on the manufacture and transport of mineral fertilizers and also the transport of compost were from the processes inventoried in the database ecoinvent data v2.2 (SCLCI 2012).

### *(iii) Data for agricultural practices*

Agricultural practices data were obtained in two experimental fields located in the county of Maresme, one of the major tomato producing areas in Catalonia and Spain, with a major urban pressure. Cultivation was following the best available techniques for integrated cultivation management, ICM (MAPA 2002) to compare efficient systems in resources, energy and emissions.

The tomato variety ElVirado® was cultivated during the spring and summer of 2007 in Santa Susana open fields (41˚38′27″N, 2˚43′00″E, Spain) and the variety Caramba® was cultivated under greenhouse protection during the spring and summer of 2008 in Cabrils (41˚31′31″N, 2˚22′07″E, Spain). These are common varieties for open field and greenhouse cultivation, respectively, in Spanish and French Mediterranean areas. Both types of tomatoes were grown in a Typic Xerothent soil (Soil Survey Staff 2006). Planting was at a plant density of 2.3 (open field) and 2.8 plants $m^{-2}$ (greenhouse), common densities for the two production systems in the area (Muñoz et al. 2008c). According to the annual series of meteorological data (RuralCat 2009), weather parameters in both locations were very similar during the cultivation period. The experiment had a block design with four replicates for each cultivation option.

#### *(iv) Data for greenhouse*

Data for the greenhouse structure and for greenhouse management were obtained from the experimental fields and from Antón (2004). Additionally, the database ecoinvent data v2.2 (SCLCI 2012) was used whenever it is necessary.

A round arched greenhouse with vertical sidewalls (Antón 2004; Montero et al. 2009b) of 5,000 m2 was considered. It was made up of six spans, with steel frames, concrete foundations and covered by low-density polyethylene film (thickness 0.2 mm). The multi-tunnel greenhouse was 104 m long, 8 m wide per span, 4m high at the gutter and 5.5 m high at the ridge. The paper includes a representation of the greenhouse considered and a list of all its components (Section 5.3.7).

### 5.2.5. Impact distribution procedure

Conflicts with distribution of environmental burdens are very common in waste treatment systems as these systems usually have one or more further functions in addition to waste management (Ekvall and Finnveden 2001), such as by-products or energy production, and are usually at the limit of the system boundaries defined for the LCA.

#### *(i) Open-loop recycling*

With regard to the management of several waste flows generated within the boundaries of the system (Figure 5.1), the "cut-off" method (Martínez-Blanco et al. 2009b) considered that each system was assigned the burdens for which it was directly responsible. In the case of waste management, dumped materials were accounted for in the system being studied, while burdens for recycled or reused waste were attributed to the potential system using the waste as a raw material source.

#### *(ii) Multi-functional processes*

When comparing the impacts of the four cultivation options, it must be remembered that composting, as well as providing fertilizer, is an option for OFMSW and GW treatment, which is not the case in the production of mineral fertilizers. To take this into account and trying to avoid allocation, as advised in the ISO 14044 (ISO 2006b), the boundaries of the system should be expanded (Ekvall and Finnveden 2001) to make the two fertilizer systems comparable. An alternative to composting, such as dumping, needs to be taken into consideration for management of OFMSW and GW. Although dumping of organic waste is theoretically very restricted in Europe (EC1999), it is still common practice in municipal waste management. Its environmental burdens were subtracted from those options that include composting, so that only the fertilizing function of the four cultivation options was compared.

### 5.2.6. Categories of impact and LCA methodology

Six impact categories which had been defined by the CML 2001 (Guinée 2001), and cumulative energy demand (Frischknecht and Jungbluth 2003), as an energy flow indicator, were considered. Owing to the lack of consensus in the international community, toxicity categories were not assessed (Martínez-Blanco et al. 2009b). The SimaPro v7.2.4 program (PRé Consultants 2012) was used for impact analysis, with the obligatory classification and characterization phases defined by the ISO 14044 regulation (ISO 2006b).

# 5.3. Life cycle inventory

The stages and sub-stages of the four cultivation options were according to Martínez-Blanco et al. (2009b) and are described below.

## 5.3.1. Preliminary considerations

Three main topics were considered before detailing the inventory stages.

### *(i) Doses of fertilizing products applied*

Because the study followed integrated crop management guidelines (MAPA 2002), the doses of fertilizers were calculated by taking into account the real agricultural necessities, considering the nutrient concentration present in the soils, the nitrogen content of the irrigation water, the nutrients extracted by the plants and the composition of the fertilizers. Due to the high content of potassium in the GH_$C_L$ experimental field soil, detected by routine analysis, potassium nitrate was not applied (Table 5.2). Similarly, potassium and other nutrients content in OF_M soil were sufficiently enough so that potassium phosphate and potassium sulfate were not applied.

The nitrogen content of the irrigation water was considerably higher in open field (192 g $m^{-3}$ of $NO_3^-$) than in greenhouse cultivation (41 g $m^{-3}$ of $NO_3^-$) as a result of the excessive use of mineral fertilizers in the region. This is a characteristic of local irrigation water and does not depend on the cultivation option. Accordingly, for the inventory, open field cultivation was given a virtual concentration of 41 g $m^{-3}$ of $NO_3^-$ as for greenhouse cultivation and the extra 151 g $m^{-3}$ of $NO_3^-$ was accounted as added mineral fertilizer ($HNO_3$), considering its production, transport and application.

Compost is normally applied every one or two years, considering the nutritional necessities of the annual or biannual crop cycle (Table 5.2), to avoid individual environmental burdens of transport and machinery for each crop. Consequently, only the proportion of the total compost application was taken into account for the tomato crops, with the allocation calculated according to nitrogen uptake by the crop.

### *(ii) Transport assumptions*

In the several stages, all transport was considered doubling the distances to take into account the return trip, except in the case of mineral fertilizers transport due to the efficiency of international transport platforms.

### *(iii) Yield as functional unit*

Section 5.4.1 shows the different tomato yields used to express the inventory results per functional unit.

**Table 5.2. Mineral fertilizers and compost, nitrogen and irrigation water applied for each cultivation option.**

| | Open field | | Greenhouse | |
|---|---|---|---|---|
| | OF_CL | OF_M | GH_CL | GH_M |
| | *Fertilizers application (g m⁻²)* | | | |
| **Compost** [a] | 1997.6 | 0 | 1743.4 | 0 |
| **$HNO_3$** [b] | 144 | 139.9 | 111.2 | 176.5 |
| **$KNO_3$** | 90.2 | 178.3 | 0 | 99.9 |
| **$KPO_4H_2$** | 0 | 0 | 0 | 73.6 |
| **$K_2SO_4$** | 0 | 0 | 0 | 117.6 |
| | *Nitrogen application (g N $m^{-2}$)* | | | |
| **Nitrogen organic** [c] | 8.11 | 0 | 7.08 | 0 |
| **Nitrogen mineral** [d] | 36.95 | 48.47 | 19.48 | 43.22 |
| **Nitrogen total** | 45.06 | 48.47 | 26.56 | 43.22 |
| | *Irrigation water* [e] | | | |
| **Per area (L $m^{-2}$)** | 571 | 555 | 490 | 622 |
| **Per ton tomato ($m^3$ $FU^{-1}$)** | 67.9 | 68 | 32.2 | 39.2 |

*FU, functional unit.*

[a] *The proportion of the total compost dose allocated to the two tomato crops taking into account the nitrogen uptake (the total compost cultivation cycle for open field and greenhouse was 17 and 24 months, respectively). The moisture content of the compost applied to the experimental plots was 27%, slightly lower than the 30-40% interval established by RD 824/2005* (Ministerio de la Presidencia2005), *and the organic matter content was 53% for dry material. In relation to heavy metals, this was class A compost* (European Commission2001). *The complete analysis of the sample has been published in* (Muñoz et al. 2008a).

[b] *Nitric acid 60%. In open field, high nitrogen water content was accounted as addition of synthetic nitric acid (section 5.3.1.i).*

[c] *Total nitrogen added by compost. Nitrogen available the first year after the spreading of compost is the easily hydrolysable organic nitrogen* (Pare et al. 1998; Saña 1999; Moral and Muro 2008).

[d] *Total nitrogen added by mineral fertilizers, water irrigation and rainfall.*

[e] *A rainfall of 105 L $m^{-2}$ must be added to open field options due to direct rainfall on soil for open fields whereas in greenhouse it was stored and distributed through the fertirrigation system.*

## *5.3.2. Stage of compost production (CP)*

This stage included the organic waste collection and the inputs and outputs of the composting process in the industrial composting plant of Castelldefels.

### *(i) Collection and transport of the organic waste*

OFMSW and GW were obtained by source-separated collection systems from nearby municipalities and the main Barcelona market suppliers, located, on average, 13 km from the composting plant. Transport of the organic waste with a municipal waste collection lorry of 21 t maximum authorized load (MAL) and production and cleaning of the street collection containers, were considered (Martínez-Blanco et al. 2009b; 2010).

### *(ii) Industrial composting process*

The data of Martínez-Blanco et al. (2009b) were used for the consumption of water, electricity and diesel, the generation of waste, the input of organic waste and the output of compost in the composting plant. The burdens of the building and the main machinery were also accounted for.

The industrial composting process in the studied plant took 10 weeks and was divided into four main stages: pre-treatment, when OFMSW and GW were shredded and mixed; decomposition phase, in tunnels, with forced aeration and irrigation systems; curing phase, when decomposed material was put in piles, periodically turned, mechanically; and finally, the sifting phase, when composted material was screened to separate the mature compost from the green waste (not totally decomposed and put back into the composting process) and the solid waste fraction (including non-organic material such as plastics and tins) which was dumped. The process of dumping the solid waste fraction was calculated with the Calculation Tool for waste disposal in Municipal Sanitary Waste Landfill (MSWLF) from the ecoinvent database (SCLCI 2012).

#### *(iii) Biofilter characteristics and gaseous emissions*

Before being released into the atmosphere, the exhaust gases generated in the composting tunnels and in the pre-treatment and curing areas were treated with a biofilter to retain contaminants. In previous research, we found that 0.08 kg of $NH_3$, 0.84 kg of volatile organic compounds (VOC), 0.024 kg of $CH_4$ and 0.064 kg of $N_2O$ were emitted per ton of organic waste (OFMSW and GW) after biofiltration (Martínez-Blanco et al. 2010).

### 5.3.3. Stage of mineral fertilizers production (FP)

Data on the manufacture of mineral fertilizers were from the database ecoinvent data v2.2 (SCLCI 2012), including the production infrastructure, transport of raw materials, synthesis of the chemical components required, and the deposition or treatment of waste generated (Nemecek et al. 2004).

### 5.3.4. Stage of compost transport (CT)

The compost was transported from the composting facility in Castelldefels (Barcelona, Spain), 66 km from the experimental plots, in 3.5–16 t MAL trucks.

### 5.3.5. Stage of mineral fertilizers transport (FT)

In the European-Mediterranean area, the nitric acid, potassium nitrate and potassium sulfate used are produced in Germany (1,950 km in a lorry of >16 t MAL), whereas potassium phosphate is produced in Israel (3,020 km in freight ship and a lorry of 3.5–16 t MAL). As mentioned above, only outward journeys were included in the inventory.

### 5.3.6. Stage of cultivation (Cu)

The management and design of the four cultivation options are described below.

#### *(i) Fertirrigation infrastructure sub-stage (CuF)*

The supply of water, fertilizers and phytosanitary substances to the experimental plots was using a standard fertirrigation infrastructure, with tanks, pumps and pipes for extracting and channelling the water, electrovalves for controlling dosage and a network of integrated drip pipes supplying the plants. We considered the production of these elements, their transport and their waste management.

### *(ii) Phytosanitary substances sub-stage (CuP)*

The type of pesticides applied was based on the regulations for integrated crop management (MAPA 2002). The doses applied were the minimums determined in the Registry of phytosanitary products (MAAMA 2010). Open field cultivation required 14 pesticide applications, with a mix of 1–4 types of pesticides, while only 2 of those applications were required for greenhouse cultivation. The production and transport of these pesticides were included.

### *(iii) Machinery and tools sub-stage (CuM)*

The production, maintenance, transport and waste management of the tractor, the agricultural machinery and the harvesting elements were considered for all four cultivation options, apart from those operations related to compost application, which were not considered for OF_M and GH_M.

### *(iv) Irrigation sub-stage (CuI)*

Pumps consumed electricity to pump irrigation water from nearby wells and to spread it over the plots.

The total supply of water to the soils included the rainfall (for open fields) and the irrigation water. For greenhouses, rain water collected from the covering was added to irrigation water. The irrigation water applied, shown in Table 5.2, depended on the matric water potential evapotranspiration demands of the soil.

### *(v) Post-application emissions sub-stage (CuE)*

Fertirrigation emissions and carbon sequestration were added in this sub-stage.

For the calculation of fertirrigation emissions, we considered the nitrogen contributions made by each fertilizer and the nitrogen concentration absorbed by the plants, in accordance with Audsley's (2003) and Brentrup and Küesters (2000) hypothesis. We accounted for air emissions of $NH_3$, $N_2O$, $NO_x$ and $N_2$, and water emissions of $NO_3$.

Carbon sequestration, produced when compost is applied to the experimental crops, has been recognized by the Intergovernmental Panel on Climate Change as one of the possible measures through which greenhouse gas emissions can be mitigated (Smith et al. 2001). It has been considered that the carbon which remains in the soil 100 years after compost application is 8% (Smith et al. 2001; Hansen et al. 2006b). It was accounted for as a negative contribution to the total greenhouse gas emissions.

### *(vi) Nursery plants sub-stage (CuN)*

Nursery tomato plant production was under greenhouse and using a heating system. Greenhouse, irrigation, fertilization, heating and transport burdens were included, according to Antón (2004). Nursery plants were grown to plant them out in the fields.

### *(vii) Management of waste generated in the cultivation stage*

As previously mentioned, generated waste that had a recycling or recuperation treatment (non-yield biomass, non-commercial tomatoes, steel of harvesting handcarts and plastic of harvesting crates) was not considered in the inventory (Martínez-Blanco et al. 2009b). Dumped waste (from the fertirrigation infrastructure) was deposited at the end of its useful life in a regional landfill 48 km from the fields. The impacts of the waste management were transport to the landfill, in 3.5–16 t MAL trucks, plus the burdens of deposition, which depended on the material (SCLCI 2012). The burdens of each waste were attributed to the appropriate sub-stage.

### 5.3.7. Greenhouse (G)

For the LCA, a multi-tunnel greenhouse (Antón 2004; Montero et al. 2009a; 2009b) was considered for GH_CL and GH_M cultivation options (Figure 5.3), with two main sub-stages, greenhouse structure and greenhouse management.

#### *(i) Greenhouse structure sub-stage (GS)*

The production, transportation, installation and waste management of the greenhouse components were included according to Antón (2004). Figure 5.3 shows the components of the greenhouse and their materials. One hundred percent of the steel used in the components production came from recycling, according to standard greenhouse components production. A lifespan of 20 years was considered for the steel framework. All the steel components had a protective galvanized layer. The plastic film was fastened by means of a system of omega clips attached to the structure itself, and was renewed every three years. Tomato plants were supported and guided by a plant staking system. The plastic used in the structural elements of the greenhouse was 100% from raw materials (Montero et al. 2009a).

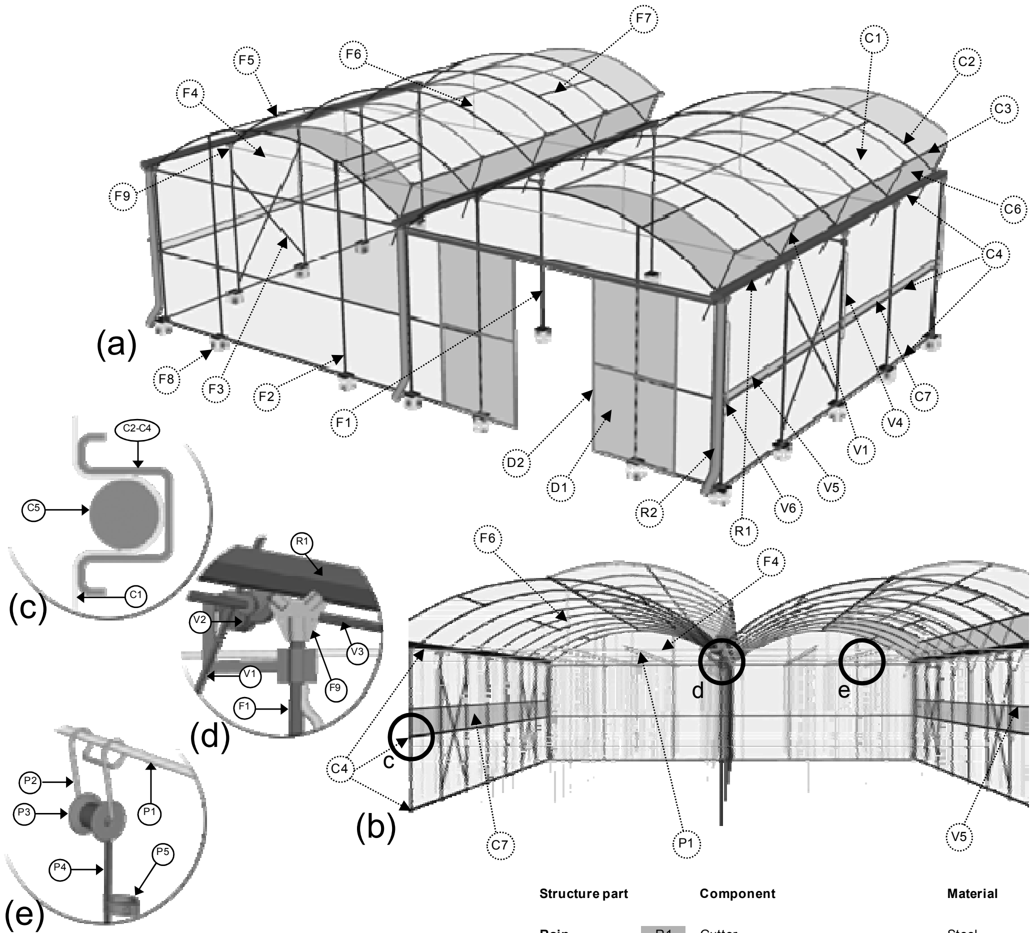
Regarding waste management, the plastics, the concrete and the steel of the greenhouse frame were totally or partially recycled, while the rest of the greenhouse structure was deposited in the landfill. Following Montero et al. (2009a), it was assumed that 50% of the mass of the structural plastics, apart from polycarbonate, was recycled and 50%, landfilled, due to the high solar radiation degradation and the dirt and contamination in the plastic waste. Recycling of the steel frame and the polycarbonate doors was of 100%, and 50% for the concrete.

#### *(ii) Greenhouse management sub-stage (GM)*

Greenhouse management only involved the opening and closing of the roof and the side-wall openings. The ventilation system had six roof openings and two side-wall openings controlled by 0.75 HP engines that allowed air renovation and therefore temperature control. The motors, with the associated electricity consumption, operated an average of 36 min per day (Antón 2004).

### 5.3.8. Avoided burdens of dumping OFMSW and GW in landfill

As previously mentioned, in the impact distribution procedure for the multi-functional processes (Section 5.2.5), dumping was the method selected as an alternative to composting for organic waste management. Environmental burdens for dumping the same amount of OFMSW and GW as used in the production of compost in OF_CL and GH_CL were subtracted from the total impacts of these options when the avoided burdens were considered.




| Structure part | | Component | Material |
|---|---|---|---|
| Frame structure | F1 | Inside post | Steel |
| | F2 | Outside post | Steel |
| | F3 | Side wall reinforcement | Steel |
| | F4 | Ridge board | Steel |
| | F5 | Roof arches | Steel |
| | F6 | Ridge board reinforcement | Steel |
| | F7 | Roof reinforcement | Steel |
| | F8 | Foundations | Concrete |
| | F9 | Capital | Steel |
| Cover | C1 | Covering film | LDPE |
| | C2 | Arch omega | Steel |
| | C3 | Ventilator omega | Steel |
| | C4 | Wall omega | Steel |
| | C5 | Plastic fixing bar | Recycled PE |
| | C6 | Roof ventilator insect proof screen | Polyester |
| | C7 | Side wall ventilator insect proof screen | Polyester |
| Rain drainage system | R1 | Gutter | Steel |
| | R2 | Rain drainage pipes | PVC |
| | * | Rain drainage tank | Concrete |
| Doors | D1 | Door sheet | Steel |
| | D2 | Door framework | PC |
| Ventilator openings | V1 | Side wall ventilator axle | Steel |
| | V2 | Side wall ventilator support bar | Steel |
| | V3 | Side wall hand crank | Steel |
| | V4 | Roof ventilator gear motor | Steel |
| | V5 | Roof ventilator rack | Steel |
| | V6 | Roof ventilator axle | Steel |
| Plant staking system | P1 | Staking wire | Steel |
| | P2 | Staking reel support | Steel |
| | P3 | Staking plastic support | HDPE |
| | P4 | Staking strip | HDPE |
| | P5 | Staking clip | HDPE |
| | * | Staking strainer | Steel |

*Materials: LDPE, Low-density polyethylene; PE, Polyethylene; PVC, Polyvinylchloride; PC, Polycarbonate; HDPE, High density polyethylene.*
*∗ Element not shown due to its size.*

**Figure 5.3. Illustration of standard Mediterranean multi-tunnel greenhouse. Structural parts and elements are indicated and also the main materials used for each element. (a) Greenhouse outside view. (b) Greenhouse inside view. (c) Covering film fixing system. (d) Capital and roof ventilator mechanism. (e) Plant staking system.**

*Source: The author of this illustration was Raúl Garcia Lozano (Institute of Environmental Science and Technology, UAB, and Inèdit Innovació Ltc., Barcelona, Spain).*

The process of dumping OFMSW or GW was calculated with the Calculation Tool for waste disposal in Municipal Sanitary Waste Landfill (MSWLF) from the ecoinvent database (SCLCI 2012). These organic wastes were assimilated into the waste flow “compostable material” in the calculation tool and the impurity materials were also included. To this was added the collection and transport of waste to the local landfill in a municipal waste collection lorry of 21 t MAL (a distance of 17 km), including the production and cleaning of the collection containers (Martínez-Blanco et al. 2009b; 2010). The construction of the landfill and road accesses, the machinery operation, the combustion of methane without energy recovery, and the land used, were all considered with a time limit of impact of 100 years (Doka 2007).

# 5.4. Results and discussion

We present the agronomical (harvest and size parameters) and environmental results (environmental contribution of stages and sub-stages to the total impact and comparison of the total environmental impacts). The environmental inventories for OF_CL and GH_CL are shown in the Annex I.I.

## 5.4.1. Yield and agricultural quality parameters

The yield and the fruit size parameters were compared using the Enterprise Guide software package (SAS institute Inc. 2006).

### *(i) Yields*

No significant statistical differences were observed for commercial and non-commercial yields between cultivation options in open field neither between cultivation options under greenhouse, with both fertilizing options (Table 5.3). Open field yield was within the upper average levels expected for this form of production of tomatoes in the area (Muñoz et al. 2008b), while the harvests for greenhouse were within the lower average levels expected for production under greenhouse in the area (see Annex I.IV). Nevertheless, Poudel et al. (2001) assessed a greater potential for nitrogen storage when organic amendment was applied in the long-term, which could considerably improve the harvest obtained for options with compost.

**Table 5.3. Parameters of harvest and fruit size.**

| Parameter | Units | Fertilizing options | | LSD [a] |
|---|---|---|---|---|
| | | Compost & mineral fertilizers (CL) | Mineral fertilizers (M) | |
| | | *Open field (OF)* | | |
| **Commercial yield** | t $ha^{-1}$ | 100 | 103 | ns |
| **Total yield** [b] | t $ha^{-1}$ | 127 | 127 | ns |
| **Tomato average diameter** | mm | 78.2 | 78.7 | ns |
| **Tomato average weight** | g | 205.6 | 207.7 | ns |
| | | *Greenhouse (GH)* | | |
| **Commercial yield** | t $ha^{-1}$ | 153 | 159 | ns |
| **Total yield** [b] | t $ha^{-1}$ | 158 | 166 | ns |
| **Tomato average diameter** | mm | 85.5 | 89.3 | ns |
| **Tomato average weight** | g | 246.1 [a] | 283.9 [b] | - |

[a] *Analysis of variance was by means of the General Linear Model, and the least significant difference test (LSD, p = 0.05) was used to establish differences between cultivation options* (SAS institute Inc. 2006). *Different letters indicate significant effect, "ns" no significant effect at p = 0.05.*

[b] *Total yield is the addition of commercial plus non-commercial tomato yield.*

Although the tomato varieties cultivated in open field and in greenhouse were different, and therefore care is needed with comparison, cultivations under greenhouse had 50% higher levels of commercial yield and considerably lower non-commercial yields than open field options (Table 5.3).

#### *(ii) Parameters of agricultural quality*

As Table 5.3 shows, there were no statistical differences between the open field fertilizing options for fruit average diameter and weight. With regard to the greenhouse options, there was no statistical difference for the diameter but the fruit average weight was significantly smaller for GH_CL than GH_M.

Both yield and the quality parameters studied performed better under greenhouses than in open fields.

### 5.4.2. Environmental contribution of stages and sub-stages

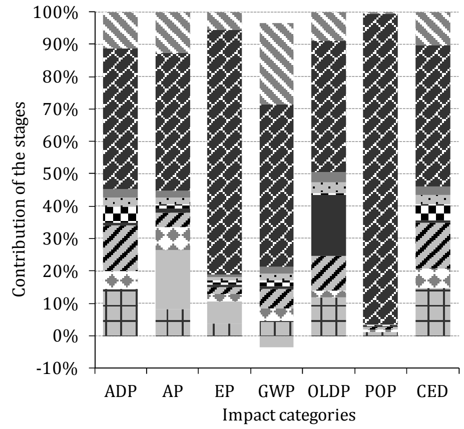
The environmental contribution of stages and sub-stages to the total impact of the four cultivation options and particular contributions to the critical stages were assessed. Fertilizer dosages were adjusted to the real needs of the fields and crops, thus the differences between the fertilizers applied to each cultivation option were affected, and therefore the environmental results.

#### *(i) Cultivation option in open field with compost plus mineral fertilizers*

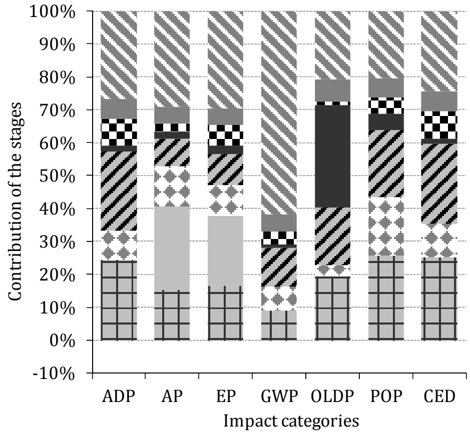
For OF_CL (Figure 5.4a) the biggest impact was for the compost production stage (CP), which represented 40–76% of the total impacts for all the impact categories considered except photochemical oxidation potential (POP). For this category, the compost production stage was responsible for 96% of the total impacts due to the emission of VOC. Later in this section there is a breakdown of the compost production stage. This stage was followed by: mineral fertilizers production stage (FP), with a contribution of 9–27% to the total impacts for all the impact categories considered, apart from POP; post-application emissions sub-stage (CuE) that contributed 18% to acidification potential (AP); machinery and tools (CuM) and nursery plants (CuN) sub-stages, with impacts between 11 and 14% for the former and between 12 and 15% for the latter, for abiotic depletion potential (ADP), ozone layer depletion potential (OLDP) and cumulative energy demand (CED); and finally, phytosanitary substances sub-stage (CuP) with a contribution of 19% to OLDP. The contributions to the remaining impact categories and stages were 8% or below this figure.

#### *(ii) Cultivation option in open field with mineral fertilizers*

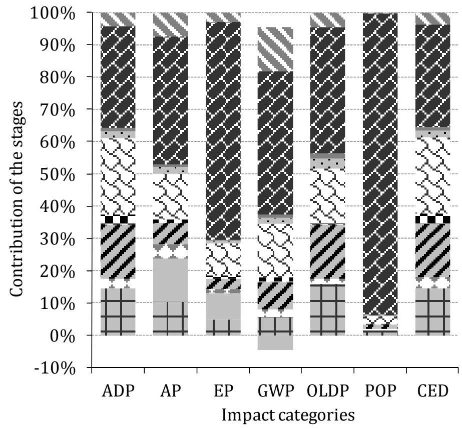
Considering the OF_M option (Figure 4b), the mineral fertilizers production stage (FP) had the biggest burdens, 27–62% of the total impact for ADP, AP, EP and global warming potential (GWP), and a relevant contribution, between 20 and 24%, for the remaining categories. All the cultivation sub-stages, apart from fertirrigation infrastructure (CuF), also had relevant impacts for all or some of the impact categories. The nursery plants (CuN) and the machinery and tools (CuM) sub-stages contributed 9–26% (the former) and 8–24% (the latter) depending on the impact category. The post-application emissions sub-stage (CuE) had 21 and 25% of the total impacts for AP and EP, respectively. For ADP, AP, EP, POP and CED, the irrigation sub-stage (CuI) contributed with 9–18%. Finally, the phytosanitary substances sub-stage (CuP) had a contribution of 31% to OLDP. The contributions to the remaining impact categories and stages were 8% or below this figure.



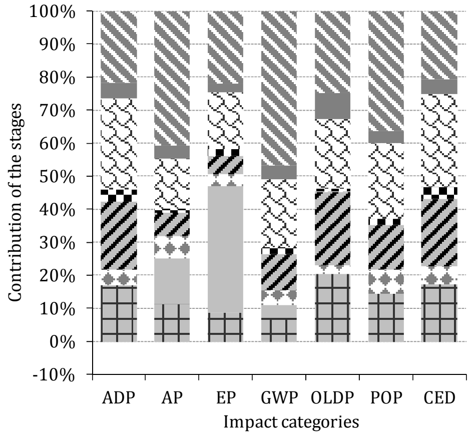

*(a) Cultivation option in open-field with compost plus mineral fertilizers (OF_CL)*




*(b) Cultivation option in open-field with mineral fertilizers (OF_M)*




*(c) Cultivation option in greenhouse with compost plus mineral fertilizers (GH_CL)*




*(d) Cultivation option in greenhouse with mineral fertilizers (GH_M)*

Mineral fertilizers production
Compost production
Mineral fertilizers transport
Compost transport
Greenhouse
Cultivation stage-fertirrigation infrastructure
Cultivation stage-phytosanitary substances
Cultivation stage-macninery and tools
Cultivation stage-irrigation
Cultivation stage-post-application emissions
Cultivation stage-nursery plants

**Figure 5.4. Contribution to total environmental impacts of the stages and sub-stages for the four cultivation options.**

### *(iii) Cultivation options under greenhouse with compost plus mineral fertilizers*

The compost production stage (CP) also had the major impacts on option GH_CL (Figure 5.4c) representing 32–67% of the total impacts for all the impact categories, apart from POP, for which the contribution was 93%. This was followed by greenhouse stage (G), which contributed 10–24% to the total impacts for all categories, apart from POP; machinery and tools sub-stage (CuM), contributing 9–17%

to ADP, GWP, OLDP and CED; nursery plants sub-stage (CuN) had 11–16% of the total impact for ADP, AP, OLDP and CED; and, post-application emissions sub-stage (CuE) contributed 13% to AP. The contributions to the remaining categories and stages were 8% or below this figure.

### *(iv) Cultivation options under greenhouse with mineral fertilizers*

For GH_M (Figure 5.4d) the main contributions were distributed among several stages and sub-stages. For all the categories, mineral fertilizers production (FP) represented 22–47% and the greenhouse stage (G) 15–28%. The machinery and tools sub-stage (CuM) contributed 13–22% to the total impacts for all the categories except AP and EP. The post-application emissions sub-stage (CuE) contributed 14% to AP and 38% to EP. Finally, the nursery plants sub-stage (CuN) had between 11 and 20% of the total impact for ADP, AP, OLDP, POP and CED. The contributions for the remaining impact categories and stages were below 9%.

### *(v) The critical stages*

Compost production and greenhouse were two of the stages with the highest impact contributions and also the stages which differentiated most between the four cultivation options. Both stages were studied in more depth.

**Compost production stage**

Figure 5.5 illustrates that energy consumption of the facility, mainly the electricity supply, and organic waste collection contributed with 19–49% (the former) and 30–58% (the latter) to the total impacts of the compost production stage (CP) for all the categories apart from EP and POP. Gas emissions, which were released during OFMSW and GW decomposition, contributed 17–23% to AP and GWP and 93% to POP; with ammonium and VOC emissions as the maximum contributors, respectively. Solid waste dumped represented 81% and 30% of the total impacts of the compost production stage for EP and GWP, respectively.

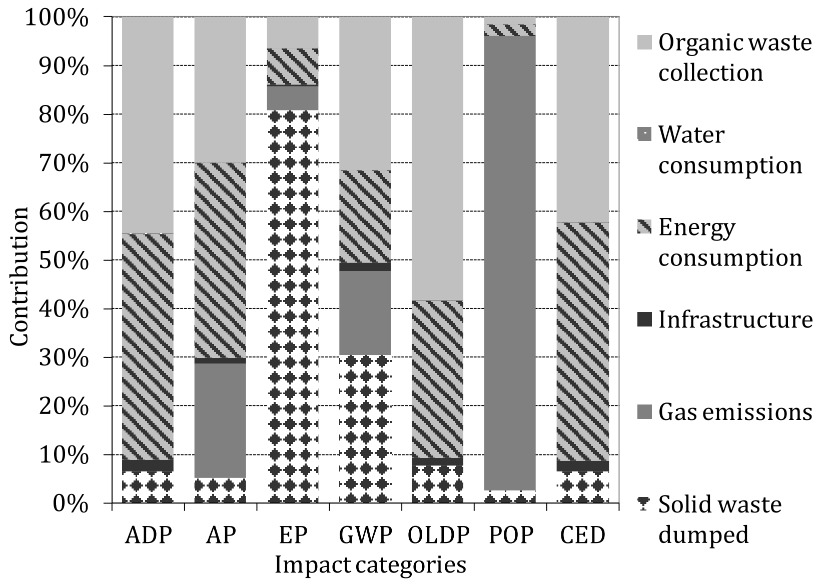
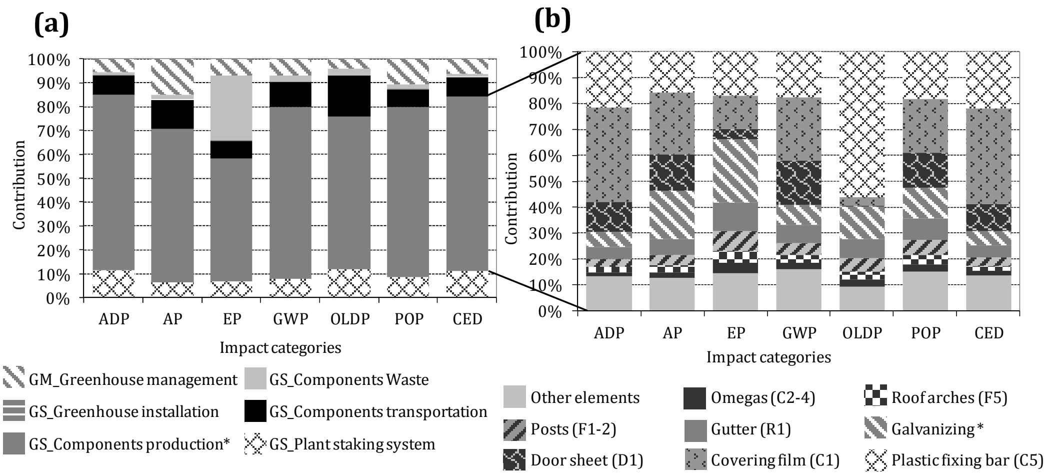



**Figure 5.5. Contribution to total environmental impacts of the Compost Production Stage for the items considered.**

Therefore, the environmental viability of cultivation with compost depends on reducing the high impacts of its production. This could be achieved by reducing energy use in the facility; improving or reducing the organic waste collection transport, such as promoting domestic composting where possible (Martínez-Blanco et al. 2010); guaranteeing good aeration during organic waste decomposition; using a more efficient gas emission treatment; and reducing the non-organic materials content of the municipal solid waste, which reduce the facility treatment capacity and increase the amount of dumped solid waste in the plant.

**Greenhouse stage**

As can be seen in Figure 5.6a, production of the greenhouse structure components entailed an impact contribution of 51–74% to the total greenhouse stage (G), for all the impact categories. Among the components considered in Figure 5.3, the plastic fixing bar, the covering film, the door sheet, the gutter, the roof arches and the zinc for galvanizing the steel frame structure were the components with the highest impacts (Figure 5.6b). The transport of all the elements from the production facility to the fields was also a relevant impact element for AP, GWP and OLDP categories (Figure 5.6a), contributing 10–17% to the G impact. The greenhouse management contributed with a 11 and 15% of the impact to AP and POP. The plant staking system represented 11–12% of the total impacts for ADP, OLDP and CED. The component waste had a contribution of 27% of the impact for EP.




*Sub-categories: GM, Greenhouse management; GS, Greenhouse structure.*
*The alphanumerical references correspond to the legend in Figure 5.3.*
** Component production includes all the components listed in Figure 5.3 except the plant staking system, and also includes the zinc for galvanizing the steel frame.*

**Figure 5.6. Contributions to total environmental impacts of Greenhouse Stage. (a) Contribution of greenhouse components production, transportation, installation and waste treatment and greenhouse management. (b) Main contributors to environmental impacts of greenhouse components production.**

The high impacts of the greenhouse could be reduced by focusing on reducing materials, the use of recycled and long-term materials (Montero et al. 2009b; Romero-Gámez et al. 2009), the reuse or recycling of structure component materials and local production of greenhouse components to reduce transport and promote local materials.

**Other stages**

Other relevant stages regarding impact contribution to total impacts (Figure 5.4) were the following: the nursery plants sub-stage (CuN), due to the high consumption of energy for the temperature regulation with heating; the machinery and tools sub-stage (CuM), as a result of harvesting crate production; and the post-application emissions sub-stage (CuE), with major impacts in AP and EP due to ammonia emissions to air and nitrate emissions to water, and with negative contribution to GWP due to carbon sequestration in soils throughout the compost application.

### 5.4.3. Environmental assessment of total impacts for the cultivation options

In this section, total impacts associated with the four cultivation options, considering and without considering expansion of the system boundaries are summarized (Section 5.2.5.ii). This expansion dealt with the subtraction of the burdens avoided through composting by not dumping organic waste (OFMSW and GW). For OF_$C_L$ and GH_$C_L$, 1.4 and 0.8 t of organic waste were used per ton of commercial tomato, respectively.

#### *(i) Comparison without considering avoided burdens*

Table 5.4 shows that, for most of the impact categories, cultivation option OF_M was the least impacting and OF_$C_L$, the most.

**Table 5.4. Total environmental impacts of the four cultivation options without considering avoided burdens through composting by not dumping OFMSW and GW.**

| Impact category | Unit (per FU$^{-1}$) | Open field | | Greenhouse | |
|---|---|---|---|---|---|
| | | OF_$C_L$ | OF_M | GH_$C_L$ | GH_M |
| **Abiotic depletion potential** | kg Sb eq. | 174% | 9.46E-01 | 136% | 112% |
| **Acidification potential** | kg $SO_2$ eq. | 190% | 8.88E-01 | 117% | 106% |
| **Eutrophication potential** | kg $PO_4$ e eq. | 435% | 2.34E-01 | 278% | 149% |
| **Global warming potential** | kg $CO_2$ eq. | 189% | 102% | 119% | 1.53E+02 |
| **Ozone layer depletion potential** | kg CFC-11 eq. | 226% | 134% | 133% | 1.39E-05 |
| **Photochemical oxidation potential** | kg $C_2H_4$ eq. | 2360% | 2.28E-02 | 1381% | 140% |
| **Cumulative energy demand** | MJ eq. | 176% | 2.26E+03 | 139% | 113% |

*100%* *101-150%* *151-200%* *>200%*

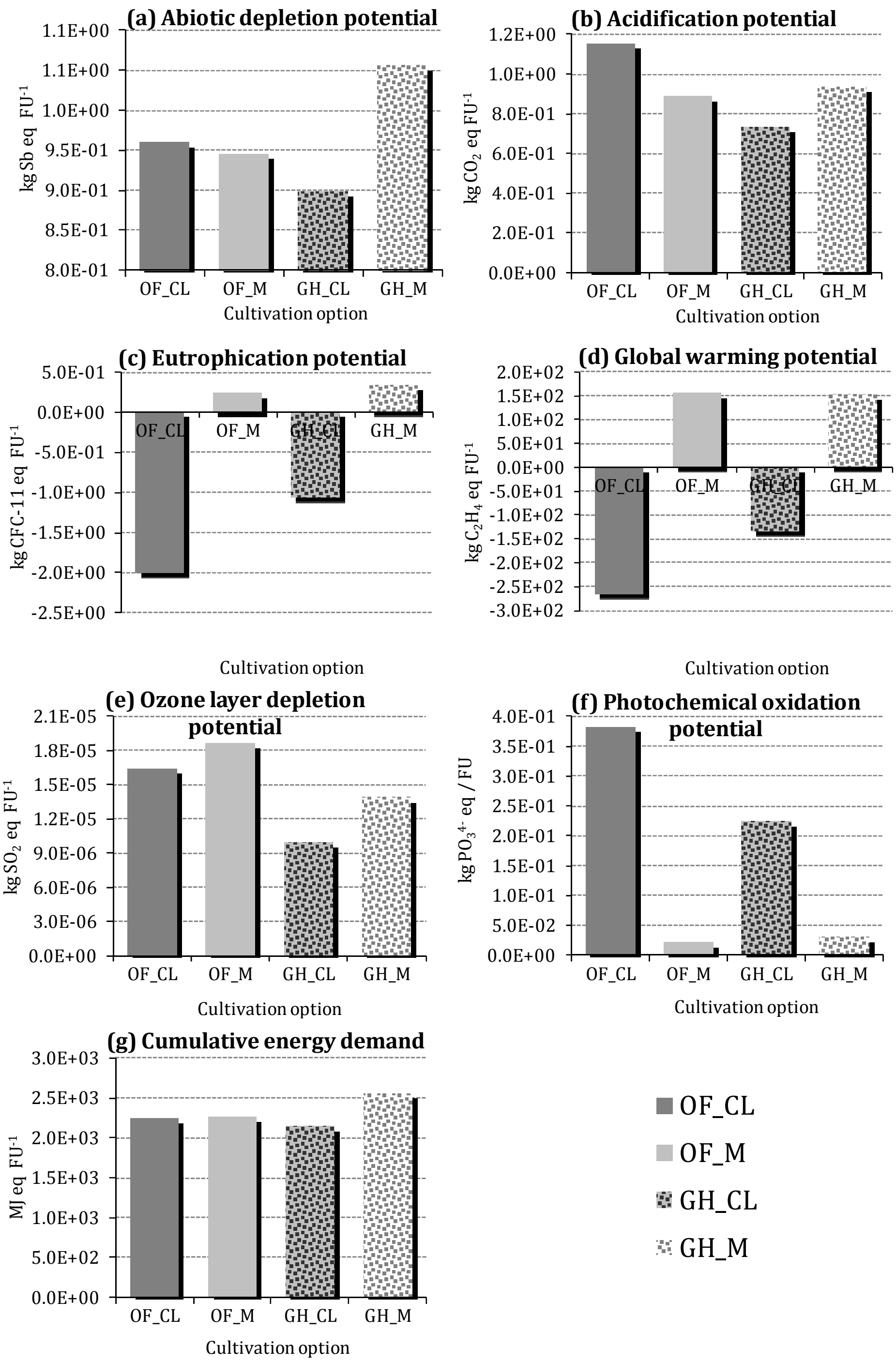
*FU, functional unit.*

*The shaded figures are percentages and represent the increase regarding the minimum impact for in each category (for each row). The figures in scientific notation are the minimum impact among the four cultivation options and represent the 100% basis. The absolute values are available in the Annex I.III.*

The compost plus mineral fertilizers ($C_L$) options had more impact than mineral fertilizers (M) options. For open field (OF), the differences were in the region of 74–90%, apart from EP, with a difference of 335%, and POP. For greenhouse (GH), the differences were of 10–33%, apart from POP. For the category POP, in both production systems, the differences were considerably higher due to VOC emissions.

With regard to the production system differences, the open field option OF_$C_L$ had larger impacts than GH_$C_L$, with differences below 70%, due to the higher tomato yield in the latter and the larger mineral fertilizers dosage in the former. For the M options, the major GH_M impacts (below 49%) compared to OF_M, apart from GWP and OLDP, were caused by the large amount of mineral fertilizers applied and the extra burdens related to the greenhouse technology.

### *(ii) Comparison between fertilizing options considering avoided burdens*




*The values are available in the Annex I.III.*

**Figure 5.7. Total environmental impacts of the four cultivation options considering avoided burdens through composting by not dumping OFMSW and BA.**

Considering the avoided burdens for open field (Figure 5.7), the open field options (OF_CL and OF_M) had differences below 2% for ADP and CED. The former had more impact than the latter for AP (30%), due to high impacts related to the composting plant (Figure 5.4 and Figure 5.5), and POP (16 times), as a consequence of VOC emissions in the plant. For OLDP, the impact of OF_CL was 12% lower than for OF_M. The impacts of EP and GWP in OF_CL were negative (meaning that impacts were saved, as organic waste was not deposited in landfill), and had 10 and 3 times less impact than OF_M, respectively.

As can be seen in Figure 5.7, considering the avoided burdens for greenhouse cultivation options, GH_M had 18–40% more impacts than GH_CL for the ADP, AP, OLDP and CED categories. For EP and GWP, GH_CL had negative, saved, impacts, with 4 and 2 times less impact than GH_M, respectively. The impact of POP was 6 times higher in the GH_CL option than the GH_M option.

### *(iii) Comparison between greenhouse and open field considering avoided burdens*

As Figure 5.7 shows, in reference to compost plus mineral fertilizers options, OF_CL had 5–69% higher impacts than GH_CL for all categories apart from EP and GWP. For these categories the difference was reversed and OF_CL had nearly 50% less impact than GH_CL given that the total amount of OFMSW and GW per functional unit considered was slightly higher for OF than for GH, so that the avoided impacts were also higher. Concerning the mineral fertilizers options, the impacts for GH_M were larger than for OF_M (6–49%), apart from OLDP, due to the higher quantity of mineral fertilizers applied and the greenhouse structure and management, and GWP, with quite similar impacts. For OLDP, the open field option (OF_M) had 34% higher impact than greenhouse (GH_M) due to the higher consumption of phytosanitary substances (Figure 5.4).

According to previous studies in this area (Muñoz et al. 2005; 2008a; 2008b), normal tomato yield in open field is around 85 t $ha^{-1}$ and that in greenhouses between 150 and 200 t $ha^{-1}$. As it is presented in Annex I.IV, if average tomato yield rates in the area for open field and greenhouse are considered, greenhouse cultivation becomes a less harmful environmental option than open field cultivation for all the impact categories apart from EP and GWP. Greenhouse structure must be associated with relevantly large yields to justify the expense in energy and resources.

## 5.5. Conclusions

Replacement of a fraction of the mineral fertilizers dosage with compost appears to be a good agronomical solution for tomato crops, in both open field and greenhouse. The greenhouse infrastructure consumed more materials per functional unit than the open field option, nevertheless, control of microclimatic conditions due to greenhouse protection resulted in a major increase in yield (50%), reduced harvest losses, and a major reduction in use of pesticides and irrigation water (see Annex I.I). For the cultivation options using compost its production was the major contributor to the total impacts mainly due to energy consumption, organic waste collection, gaseous emissions and solid waste dumping.

Without considering the subtraction of the burdens of avoided organic waste, the cultivation option OF_M was the least impacting and OF_CL the most. When avoided burdens were taken into consideration, the environmental impacts of the four cultivation options varied, depending on the impact category, with bigger differences due to fertilization as a variable rather than the production system for most of the impact categories.

Life Cycle Assessment was useful to assess the environmental performance of the cultivation options. However, the results were obtained with the currently available methodology and data and, therefore, some future research points should be considered. First of all, other functional units including economic value or quality aspects could be considered, as done by Martínez-Blanco et al. (2011a). Secondly, new indicators to measure potential local impacts in Mediterranean regions, such as erosion and water consumption, should be defined in future research. Lastly, the improvement of soil characteristics by compost application appears in the long-term, and, as this is an important factor to be considered, experiments should be carried out over a longer time period.

# Chapter 6. Comparing nutritional value and yield as functional units in the environmental assessment of horticultural production with organic or mineral fertilization: The case of Mediterranean cauliflower production

**Chapter 6** is based on the following paper:

Martínez-Blanco, J., Anton, A., Rieradevall, J., Castellari, M., & Munoz, P. (2011) Comparing nutritional value and yield as functional units in the environmental assessment of horticultural production with organic or mineral fertilization: The case of Mediterranean cauliflower production. International Journal of Life Cycle Assessment, 16, 12-26.

# Abstract

***Background, aim, and scope.*** We report the environmental assessment of the cultivation cycle of cauliflower (*Brassica oleracea* L. var. *botrytis*), chosen due to its high levels of natural bioactive compounds, using different fertilization practices. The functional units used during the impact assessment were linked with the quantity produced, considering different units of commercialization, or with the cauliflower quality, considering its antioxidant compounds content. Although nutrient content has been described and used as a possible functional unit, using antioxidant compounds as a functional unit has not previously been published

***Method.*** Three cultivation options with similar dosages of total nitrogen were considered: using mineral fertilizers (M) alone or mineral fertilizers plus compost, with a high ($C_H$) or a low ($C_L$) dosage. During the cultivation period, the soil characteristics and nitrogen and moisture content of the fruit were monitored, and the yield and the fruit size were analyzed. In addition, the glucosinolates and the phenolic compounds (sinapic acid, phenols, and flavonoids) content were assessed for the three options. Life Cycle Assessment (LCA) was used to determine the environmental impacts of the whole cauliflower production cycle, including production of mineral and organic fertilizers, fertilizers transport, and crop stage.

***Results and discussion***. Commercial yields were higher for cultivation options with M and $C_L$ than for option $C_H$, while higher levels of bioactive compounds were detected in the latter. For $C_H$ and $C_L$, eutrophication, global warming and ozone layer depletion potentials were generally lower and photochemical oxidation potential was always higher than for the M option, regardless of the functional unit. Regarding functional units involving production (yield, fruit and dry matter harvest), there were higher impacts with the $C_H$ cultivation option than with M for abiotic depletion, acidification, photochemical oxidation potentials and cumulative energy demand. When the differences in bioactive compounds content (total sinapic acid derivatives and total phenols) were sufficiently high, this was reversed, with $C_H$ having lower impacts for all the environmental categories apart from photochemical oxidation and abiotic depletion potentials.

***Conclusions and perspectives***. The differences in the magnitude of individual environmental impacts between cultivation options, and also the order, were highly dependant on the functional unit considered. When functional units associated with production and total phenols content were considered, the $C_H$ cultivation option had the highest impact in four out of seven categories, whereas for the functional unit involving sinapic acid content, this cultivation option had the least impact in five out of seven categories.

**Keywords**

Compost; Food products; Functional unit; Life Cycle Assessment; Mineral fertilizers; Nutritional compounds.

# 6.1. Introduction

The area under organic farming in the EU-27 increased by 7.4% between 2007 and 2008 (Rohner-Thielen 2010). Due to the increasing consumer demand for environmentally friendly food products (Antón 2004; Podsedek 2007; Montero et al. 2009b; Romero-Gámez et al. 2009; Lairon 2010), real alternatives to current intensive production methods need to be supported by scientific research (Schau and Fet 2008; Blengini and Busto 2009).

More attention is also being paid to the role of diet in human health and in food safety and security (Podsedek 2007; Lairon 2010), which means that secondary metabolites content is a factor that must also be considered during the assessment of agriculture systems. Antioxidants and the antioxidant capacity of food, nutraceuticals, botanicals, and other dietary supplements have recently attracted the attention of the industry, scientists and consumers, because of potential health benefits (Sun and Tanumihardjo 2007).

Antioxidant compounds in food, which include phenolic compounds and glucosinolates, are linked with cell maturation and therefore promote cardiovascular health, inhibit the growth of cancerous tumours and cell masses, slow the aging process in the brain and nervous systems, and lessen the risk and severity of neurodegenerative diseases (Pyo et al. 2004; Benbrook 2005; Podsedek 2007; Gratacós-Cubarsí et al. 2010). Although glucosinolates have rather low antioxidant activity, products of their hydrolysis can also protect against cancer (Podsedek 2007).

A wide range of factors can influence the mix of secondary metabolites that a plant manufactures, as they have direct roles in plant responses to stress (Benbrook et al. 2008; Lairon 2010). Gratacós-Cubarsí et al. (2010) indicated that these factors include genetic and agronomical factors (i.e. species, variety, crop management strategies and circumstances, postharvest storage, plant stage, cooking, etc.). In addition, the positive effects on health depend on the dosage ingested (Nilsson et al. 2006).

Despite the potential importance for human well-being, only a limited number of studies on the effects of these factors have been specifically carried out, as described by Benbrook et al. (2008) and Lairon (2010). Focusing on crop management strategies, which include soil type and chemistry, available nitrogen and levels of other nutrients, moisture levels, temperature, and pest pressure (Benbrook 2005; 2008), some case studies have compared organic and conventional agriculture. These studies indicate a tendency towards nutritional superiority but lower yields of organic products (Benbrook 2005; 2008; Lairon 2010). Wang and Lin (2003) presented the first study assessing the effect of compost application on flavonoid levels and total antioxidant capacity of strawberries. De Pascale et al. (2006) have demonstrated that both the farming system and N rate affect carotenoid content and antioxidant activity of tomato fruits. In fields where high levels of nitrogen are readily available, plants can, and normally do, grow rapidly and to a relatively large size, but concentrations of polyphenols and some vitamins are typically lower. Benbrook (2005) has referred to this phenomenon as the “dilution effect”.

Although compost application appears to be a good fertilizing complement (e.g., improving soil characteristics), as demonstrated in previous environmental studies, it has not performed as well as systems with mineral fertilizers (per ton or kilo of agricultural product) in most impact categories (Hansen et al. 2006a; ROU 2007; Blengini 2008; Blengini and Busto 2009; Martínez-Blanco et al. 2009b; 2010).

Members of the Brassicaceae family are widely consumed and considered to have an important role in human nutrition (Gratacós-Cubarsí et al. 2010) partly due to their high content of secondary metabolites. According to Podsedek (2007), the secondary metabolites in Brassica vegetables are basically vitamins, carotenoids, glucosinolates and phenolic compounds which have an important antioxidant activity. Glycosides of kaempferol and quercetin, and their derivatives in combination with hydrocinnamic acids as well as sinapic acid derivatives, have been found to be the major phenolic compounds in the Brassicaceae family (Gratacós-Cubarsí et al. 2010). These vegetables also have a large group of glucosinolates, which are sulphur-containing compounds (Nilsson et al. 2006) on which there has been considerable research.

Life Cycle Assessment (LCA) is a suitable methodology for the environmental evaluation of food systems (Audsley et al. 2003; Williams et al. 2006; Blengini and Busto 2009). Nevertheless, as pointed out in previous studies on the environmental performance of compost use in agriculture (ROU 2007; Martínez-Blanco et al. 2009b), the importance of introducing some positive aspects of compost, such as the improvement of soil characteristics and the increase in nutrient content, should not be underestimated.

The functional unit is the basis for comparisons between different systems in LCA (ISO 2006a). Adequately selecting a functional unit is of prime importance because different functional units can lead to different results for the same product system. Even though the primary function of food is nutrition, in most articles on the environmental assessment of food production, the functional unit is based on mass or volume. The use of 1 ton or kilo for normalization of the inventory and the environmental impacts is particularly common in agricultural production. In some studies, the functional units have been defined as other mass or volume parameters (e.g. dry mass), the economic value (e.g. the market price), the quality of the product (e.g. the nutrient, energy or protein content of the product or its keeping quality), the consumer's reaction to the product or land use (e.g. hectare), among others (Hayashi et al. 2006; Mourad et al. 2007; Reap et al. 2008a; Schau and Fet 2008).

Here, we varied the dosage of compost applied to quantify the potential effects on the content of the main nutritional compounds in cauliflower, and then carried out an environmental assessment of the whole system from a productivity and nutritional standpoint. As far as we know, although nutrient content has been described and used as a possible functional unit (Schau and Fet 2008), this is the first time glucosinolates or phenolic compounds have been used.

The first goal of this research was to assess and compare the yield and size of cauliflower using three cultivation options that combine mineral fertilizers and compost. The second was to establish the potential effects of the fertilization on the content of antioxidant compounds. The third goal was to assess the environmental performance of the three options considering five functional units. The first two functional units were linked with commercial yield and commercial fruit, representing the units of commercialization, per mass or per fruit unit, two common units of measure for cauliflower which depend on the market. The third unit used was a kilo of dry matter, which expresses the real content in proteins, minerals, vitamins, etc. but not the water content, a variable that depends highly depends on several parameters (such as irrigation and time of day, among others). The two last functional units were related with total sinapic acid derivatives content and total phenols content, two important nutritional quality parameters connected with the antioxidant activity, one of the most highly appreciated components of cauliflower. Bioactive compounds with non-significant differences between cultivation options were not considered as a potential functional unit.

# 6.2. Methodology

This section has been split into four parts as follows: the experimental methodology for the agricultural production; the laboratory measurement of the bioactive compounds; the statistical methods; and, finally, the LCA goal and scope.

## 6.2.1. Agricultural methodology

Cauliflower was chosen from the four crops cultivated using a horticultural Mediterranean rotation (chard, tomato, cauliflower, and onion) due to its high levels of natural bioactive compounds (Podsedek 2007; Gratacós-Cubarsí et al. 2010), so increasing the likelihood of finding significant differences in content of these compounds.

### *(i) Climate conditions, soil, and cultivation period*

The experimental plot was at the SELMAR research fields in Santa Susanna (Barcelona, NE Spain), with a Typic Xerothent soil and Mediterranean climate. The plants (*Brassica oleracea* L. var. *botrytis*, commercialized as Trevi®) were transplanted on 28 September 2007 at a plant density of 2.1 $m^{-2}$. The cauliflower was cultivated for 110 days and harvested from 11–18 January 2008. For the period 1990–2008, the average annual evapotranspiration was 771 mm, and rainfall, 649 mm. Climate data were obtained from a weather station next to the field. Cultivation followed the best available techniques for integrated crop management (Bradley et al. 2002; MAPA 2002), aiming to compare efficient systems for resources, energy, and emissions.

### *(ii) Water and fertilizers application*

Crops were micro-sprinkler irrigated three times a week depending on the tensiometer reading that determined the matric water potential evapotranspiration demands of the soil. The total irrigation and rainfall water are shown in Table 6.1.

Three cultivation options were used: only mineral fertilizers (M); mineral fertilizers plus low-dose compost ($C_L$) with a third of the total nitrogen needs for cultivation supplied by compost and the rest by mineral fertilizers; and mineral fertilizers plus high-dose compost ($C_H$) with two thirds of the total nitrogen needs supplied by compost. Compost alone without mineral nitrogen added was not an option, due to the particular characteristics of the irrigation water, with a high concentration of nitrates (Martínez-Blanco et al. 2009b; 2011b).

The experiment had a block design with three replicates for each cultivation option (a total of 9 blocks of 39 $m^2$). The doses of fertilizers were calculated by taking into account the soil nutrient content and the agricultural necessities with the aim of comparing cultivation options with similar available nutrient rates. The mineral fertilizers used for the $C_L$ and M cultivation options were potassium nitrate and nitric acid. The high nitrogen content of the irrigation water (192 g $m^{-3}$ of $NO_3^-$), a result of the excessive use of mineral fertilizers in the region (Muñoz et al. 2008c), was also considered a mineral source of nitrogen for the three options. Table 6.1 shows the fertilizers and N supplied to the crop in the three cultivation options.

Compost is not normally applied to every crop, but in cycles of 1–2 years, to distribute environmental burdens associated with the transport and machinery used. Consequently, only the corresponding proportion of the total compost for each crop was taken into account, with the allocation calculated according to total nitrogen uptake by the crop from the experimental data of the horticultural rotation. For this study, compost was applied once in a rotation of 17 months.

**Table 6.1. Mineral fertilizers and compost, nitrogen and irrigation water applied for each cultivation option.**

| Substance | Cultivation option | | |
|---|---|---|---|
| | $C_H$ | $C_L$ | M |
| | *Fertilizer dose (g $m^{-2}$)* | | |
| **Compost** [a] | 1,996.06 | 998.03 | 0.00 |
| **$HNO_3$** [b] | 32.80 | 32.50 | 33.80 |
| **$KNO_3$** | 0.00 | 34.30 | 71.40 |
| | *Nitrogen dose (g N $m^{-2}$)* | | |
| **Organic nitrogen** [c] | 8.11 | 4.15 | 0.00 |
| **Mineral nitrogen** [d] | 9.93 | 14.56 | 20.10 |
| **Total nitrogen** | 18.04 | 18.71 | 20.10 |
| | *Water dose (l $m^{-2}$)* | | |
| **Irrigation water** | 231 | 229 | 238 |
| **Rainfall water** | 80 | 80 | 80 |

*$C_H$, mineral fertilizers plus high-dose compost; $C_L$, mineral fertilizers plus low-dose compost; M, mineral fertilizers alone.*
[a] *The proportion of the total compost allocated to the cauliflower crops taking into account the nitrogen uptake (17-month rotation period). The moisture content of the compost applied to the experimental plots was 27% and the organic matter content was 53% for dry material. In relation to heavy metals, class A compost* (Martínez-Blanco et al. 2010; Martínez-Blanco et al. 2011b).
[b] *Nitric acid, 60%. The high nitrogen content in water was accounted as addition of synthetic nitric acid.*
[c] *Nitrogen from compost. Nitrogen available the first year after spreading of compost it is the easily hydrolysable organic nitrogen* (Pare et al. 1998; Saña 1999; Moral and Muro 2008).
[d] *Nitrogen from mineral fertilizers, irrigation water and rainfall.*

### *(iii) Soil and yield measurements*

Several soil samples were taken during the cultivation period of cauliflower to measure the evolution of moisture and nitrogen content ($NO_3$-N) at three different depths (Doltra and Muñoz 2010).

Total and marketable yield in the whole plot area were determined per block and per cultivation option during harvest time, and the diameter of a representative sample of 30 commercial cauliflowers per cultivation option (ten per replicate) was measured. Five cauliflowers per cultivation option were sampled and immediately within 2 h) frozen at −80°C, for bioactive compounds measurement described in the next section.

At four points during cultivation, aerial samples of the cauliflower (distinguishing between leaves, fruit and stems) were taken from each block. The wet weight was measured for each sample, and the moisture was determined with a drying temperature of 65°C until constant weight. Two plants from each block (with a fruit, leaf, and stem sample), at each of the four sampling points, were dried and the N content analyzed by the Kjeldahl method (Doltra and Muñoz 2010). Micro and macro-nutrient content was also measured for these samples using ICP-OES spectrometry.

## 6.2.2. Bioactive compounds analysis

Sinapic acid derivatives, flavonoids and glucosinolates, the most important nutritional compounds of cauliflower (Nilsson et al. 2006; Gratacós-Cubarsí et al. 2010), and total phenols were assessed.

### *(i) Chemicals*

Methanol and acetonitrile, HPLC gradient grade, were obtained from Baker (J.T. Baker, Deventer, The Netherlands). The commercial standards for the Folin–Ciocalteau 2 N reagent, $Na_2CO_3$ and formic acid, were purchased from Sigma-Aldrich-Fluka (Madrid, Spain). Glucosinolate and phenolic compound standards were obtained from Sigma-Aldrich-Fluka (Madrid, Spain), Phytolab GmbH & Co. KG (Vestenbergsgreuth, Germany) and ChromaDex Inc. (Santa Ana, CA, USA).

Standard stock solutions of the commercial standards, at 1 g $L^{-1}$, were prepared by dissolving in MeOH:Water (60:40, *v*/*v*).

### *(ii) Sample treatment*

Cauliflower samples were vacuum packed in PE/aluminium bags, frozen at −80°C and analyzed within 2 months. Frozen cauliflower florets were minced in a Robot-coupe Blixer 3. Samples were extracted and analyzed as described by Gratacós-Cubarsí et al. (2010).

### *(iii) UPLC-MS/MS analysis*

An aliquot of 1.5 g of frozen sample was mixed with 75 µl of internal standards (GTP and Q3R) and 7.5 ml of methanol, and maintained at 70°C for 15 min.

Extracts were refrigerated in an ice water bath and centrifuged at 10,000 rpm for 10 min at 4°C in a Beckman J2-MC centrifuge (Beckman Instruments INC., Palo Alto, CA, USA).

Glucosinolates, flavonoids, and sinapic acid derivatives were quantified with an Acquity UPLC-MS/MS system (Waters, Millford, US) equipped with a diode array detector (DAD) and a triple quadrupole mass spectrometer (TQD) operated in negative electron-spray ionization mode (ESI−). An aliquot of 2 ml of the clean extract was evaporated to dryness with nitrogen and reconstituted with 1 ml of mobile phase A (5% ACN in 0.1% formic acid), filtered through a PTFE 0.2 µm filter, and 5 µl injected.

Chromatographic separation was using a BEH Shield C18 (1.7 µm particles, 1.0 mm id × 150 mm) column (Waters Corp., Manchester, UK) set at 35°C at a flow rate of

0.130 ml $min^{-1}$. The linear gradient of the mobile phase was from 100% A (0.1% formic acid: 95:5 *v*/*v* ACN) and 0% B (0.1% formic acid: 40:60 *v*/*v* ACN) to 50% A and 50% B at 23 min.

Glucosinolates and flavonoids were quantified by MRM, considering one MS/MS transition for each compound. Sinapic acid was quantified on the basis of the DAD signal ($\lambda = 330$ nm). Sinapic acid derivatives were quantified as sinapic acid equivalents (SAE), taking into account their molecular weight. Matrix-matched regression curves were calculated for each compound by plotting analyte/internal standard peak area ratio against the spiking concentration/internal standard concentration.

#### *(iv) Total phenols*

Total phenols were evaluated following the method of Singleton and Rossi (1965), with minor adjustments. An aliquot of 1.5 g of frozen sample was added to 7.5 ml of methanol and extracted as previously described. One millilitre of clean extract was added to 3 ml of water and 0.25 ml of Folin–Ciocalteau 2 N reagent. After 1 min the sample was mixed with 2.5 ml 20% $Na_2CO_3$ and 3.25 ml of water, and then thoroughly mixed. After 2 h, the color development was spectrophotometrically measured at 725 nm with a Shimadzu UV-240 Graphicord (Shimadzu Europe GmbH, Duisburg, Germany). Total phenols content was expressed as caffeic acid equivalents (CAE) with a caffeic acid calibration curve.

### 6.2.3. Statistics

Yield and fruit size data and bioactive compound data were analyzed with the Enterprise Guide software package (SAS institute Inc. 2006). Analysis of variance was conducted using the General Linear Models procedure, and the least significant difference test (LSD, $p < 0.05$) was used to establish differences between treatments.

### 6.2.4. Life Cycle Assessment methodology

LCA was used for evaluating the potential environmental impacts of cauliflower cultivation considering its entire life cycle. According to the ISO 14040 (ISO 2006a), an LCA is divided into four steps: the goal and scope definition (the current section), the inventory assessment (Section 6.3), the impact assessment (Section 6.4.3), and, finally, the interpretation of the results (Sections 6.4.3 and 6.5). A more detailed description of the system can be found in Martínez-Blanco et al. (2009b; 2011b).

#### *(i) Description of the system*

As mentioned above, the three cultivation options, characterized by the type of fertilization, were $C_H$, $C_L$, and M. The whole system, from obtaining raw materials required for the manufacture of the different elements to the management of generated waste, was considered for each stage.

The five stages used in the study were: compost production, mineral fertilizers production, compost transport, mineral fertilizers transport (both from the factory to the fields) and the cultivation stage including six sub-stages (Table 6.2). The cultivation options with compost, $C_H$ and $C_L$, considered the five stages while the M option did not consider the production and transport of compost.

**Table 6.2. Quality and origin of data for cauliflower cultivation in a Mediterranean open field. Experimental (E), local (L) and regional (R) data.**

| Stage | Processes / sub-stages included | Comments | Source | | |
|---|---|---|---|---|---|
| | | | E | L | R |
| **Mineral fertilizers production stage** | Production at plant including the production infrastructure, transport of raw materials, synthesis of the chemical components required and the deposition or treatment of waste generated [1,2]; dosages. | Mineral fertilizers considered ($HNO_3$ and $KNO_3$). | x | x | x |
| **Mineral fertilizers transport stage** | Mineral fertilizers transport from the plant to the crops [1,2]; distances. | Produced in Germany (1,950 km in a lorry of >16t MAL). | | x | x |
| **Compost production stage** | Collection and transport of municipal organic waste [3,4]; water, electricity and diesel consumed in the industrial composting process [3,4]; building and main machinery [3,4]; solid waste fraction landfilled [1,3,4]; biofilter characteristics and gaseous emissions [4,5]; and the physicochemical characteristics of compost. | Source separated collection of OFMSW and GW; decomposition in tunnels, with forced aeration and irrigation systems; biofiltration for the exhaust gases. | x | x | x |
| **Compost transport stage** | Compost transport from the plant to the crops [1,3]; distances. | Produced in the composting plant (66 km in a lorry of >3.5-16t MAL). | | x | x |
| | | ***Cultivation stage*** | | | |
| **Fertirrigation infrastructure** | System design; components production and transport [2,3]; and transport and management of waste [2,3]. | Including tanks, plumps, electrovalves, pipes, rods and micro-sprinklers. | x | x | |
| **Phytosanitary substances** | Type of substances needed; substance doses [6,7]; and phytosanitary substances production [1]. | Two applications. Minimal doses. | x | | x |
| **Machinery and tools** | Machinery and tools needed (type, hours of operation, characteristics and fuel consumption) [3]; machinery and tool production and maintenance [1]; diesel production and emissions; and transport and management of waste [1,3]. | Including tractor, agricultural machinery and harvesting elements. | x | | x |
| **Irrigation** | Water consumption; electricity consumption of pumps; and rainfall. | Water supplied were showed in Table 6.1. | x | x | |
| **Post-application emissions** | Emissions of $NH_3$, $N_2O$, $NO_x$ and $N_2$ to air and emissions of $NO_3$ to water [8,9]. | Emissions produced by nitrogenous mineral or organic fertilizers. | x | | x |
| **Carbon sequestration** | Carbon still bound to soil after 100 years. | Section 6.3.2. | | x | x |

[1] *SCLCI* (2012); [2] *Martínez-Blanco et al.* (2011b); [3] *Martínez-Blanco et al.* (2009b); [4] *Martínez-Blanco et al.* (2010); [5] *Colón et al.* (2009); [6] *MAPA* (2002); [7] *MAAMA* (2010); [8] *Brentrup and Küesters* (2000); [9] *Audsley* (2003).

### *(ii) Functional units*

For different bases of comparison between the cultivation options, five functional units were considered including mass (a ton of commercial cauliflower), which is the most usual functional unit for food production LCA (Antón et al. 2005a; Hayashi et al. 2006; Williams et al. 2006; Mourad et al. 2007; Reap et al. 2008a; Schau and Fet 2008; Blengini and Busto 2009; Martínez-Blanco et al. 2009b; Iribarren et al. 2010). The four additional bases dealt with the potential units of commercialization (including mass and fruit), the dry matter and the nutritional content of commercial cauliflower (considering the content of total sinapic acid derivatives and total phenols). The five functional units included in the study, which were the reference against which input and output flows were normalized, are summarized in Table 6.3.

**Table 6.3. Functional units considered in the environmental assessment.**

| Functional unit | Acronym | Related parameter | Data [a] |
|---|---|---|---|
| **1 t of commercial cauliflower** | CY | Commercial yield | Table 6.4 |
| **1 commercial fruit** | CF | Commercial fruit | Table 6.4 |
| **1 kg of commercial dry matter** | DM | Commercial dry matter harvest | Table 6.4 |
| **1 kg of SAE** | SA | Total sinapic acid derivatives content | Table 6.5 |
| **1 kg of CAE** | PH | Total phenols content | Table 6.5 |

*SAE, sinapic and equivalents; CAE, caffeic acid equivalents.*
[a] *Data used for the normalization of the inventory and the environmental results.*

### *(iii) Quality and origin of the data in the inventory*

The broad system of study required a detailed data-collection process. As shown in Table 6.2, most of this data were obtained experimentally by the authors in the plots and in the composting facility (e.g., industrial composting inputs and outputs, gaseous emissions from aerobic degradation, fertirrigation infrastructure design, irrigation, and rainfall). Furthermore, the parameters of soil characteristics, fertilizer dosages, harvests and fruit size were also obtained from the experimental plots. When local information was not available, bibliographical sources and the database ecoinvent data v2.1 (SCLCI 2012) were used to complete the life cycle inventory. Type of data and references used are specified in Table 6.2.

The tunnel composting plant, described in detail in Martínez-Blanco et al. (2009b; 2011b), was at Castelldefels, in the Barcelona metropolitan area (41°17′18″N, 1°58′16″E, Spain). The experimental plots were located in Santa Susana (41°38′27″N, 2°43′00″E, Spain) and are described above.

### *(iv) Impact distribution procedure*

For the management of several waste flows generated within the boundaries of the system, the "cut-off" method (Ekvall and Tillman 1997) was used. According to this method, each system was assigned the burdens for which it was directly responsible.

Regarding conflicts with distribution of environmental burdens, they are very common in agri-food (Blengini and Busto 2009) and waste treatment systems as these systems usually have one or more further functions in addition to waste management, such as by-products or energy production, and are usually at the limit of the system boundaries defined for the LCA. When comparing the impacts of the three cultivation options, it must be remembered that compost production, as well as providing a fertilizer, is an option for organic fraction of municipal solid waste (OFMSW) and green waste (GW) treatment, which is not the case in the production of mineral fertilizers. To take this it into account and trying to avoid allocation, as advised in the ISO 14040 (ISO 2006a), the boundaries of the system should be expanded (Finnveden 1999) to make the two fertilizer systems comparable and the environmental burdens of another way of treatment of the organic waste should be subtracted. Although dumping of organic waste is theoretically very restricted in Europe (EC1999), it is still common practice in the municipal waste management. Furthermore, the alternatives are also multi-functional ways of treatment, such as incineration and anaerobic digestion that have a major capacity for energy recovery. Therefore, the environmental burdens of dumping organic waste were subtracted from those options that include composting, so that only its fertilizing function was compared.

#### *(v) Categories of impact and LCA methodology*

Six impact categories (abiotic depletion, acidification, eutrophication, global warming, ozone layer depletion and photochemical oxidation potentials) defined by the CML 2001 (Guinée 2001), and cumulative energy demand as an energy flow indicator, were considered. Owing to the lack of relevant data for the assessment of most of the phytosanitary substances, toxicity categories were not assessed. The SimaPro v7.2.2 program (PRé Consultants 2012) was used for impact analysis, with the obligatory classification and characterization phases defined by the ISO 14040 standard regulation (ISO 1997).

## 6.3. Life cycle inventory

The description of stages and sub-stages, and the processes included in each one, are shown in Table 6.2. Most of the considerations were according to the inventory descriptions of Martinez-Blanco et al. (2009b; 2011b) but considering all the agricultural data for cauliflower cultivation obtained from the experimental plots. In addition, the inclusion of the high nitrogen content of water, the calculation of carbon sequestration in the soil and the subtraction of the composting-avoided burdens are stated below.

### 6.3.1. Nitrogen content in the irrigation water

The nitrogen content of the irrigation water was considerably higher (192 g $m^{-3}$ of $NO_3^-$) than the limit established by the European Directive 91/676 (ECC1991) for ground water (50 g $m^{-3}$ of $NO_3^-$) as a result of the excessive use of mineral fertilizers in the region (Muñoz et al. 2008c). The major additional contribution of nitrogen from the irrigation water was taken into account as synthetic added mineral fertilizer for two reasons. Firstly, to reflect its influence on the results, particularly for the cultivation options with combined compost and mineral fertilizer options. We verified that compost without addition of other fertilizers was not enough from an agricultural point of view, at least during the first few years of compost application (Muñoz et al. 2008a; Martínez-Blanco et al. 2011b). Secondly, to present more general results and to allow the comparison and extrapolation of data to other areas (with or without contamination of groundwater by nitrates). In other circumstances, irrigating with nitrate-polluted water may be considered making good use of a waste flow. For the inventory, irrigation water was given a virtual concentration of 50 g $m^{-3}$ of $NO_3^-$ and the extra 142 g $m^{-3}$ of $NO_3^-$ were accounted as added mineral fertilizer ($HNO_3$), considering its production, transport and application. The irrigation water used during the cultivation period is shown in Table 6.1.

### 6.3.2. Carbon sequestration

After the compost is produced and applied to the land, it continues to degrade, releasing more carbon dioxide and forming humic compounds that then mineralize much more slowly than the organic matter originally applied in the compost. Compost applications may therefore increase the store of soil organic matter (Smith et al. 2001). The Intergovernmental Panel on Climate Change clearly identified this long-term carbon sequestration as one of the possible GHG mitigation measures for agriculture at an early stage. Apart from promoting the build-up of carbon in the soil, Favoino and Hogg (2008) stated that the application of high-quality composted products would contribute to restoring soil fertility and health, preventing desertification and erosion, and avoiding floods.

Previous studies have estimated that the carbon still bound to soil after 100 years is between 2% and 14% of the carbon introduced with compost (Smith et al. 2001; Hansen et al. 2006a; Favoino and Hogg 2008; Boldrin et al. 2009). An average value of 8% was used in this study, similar to the value reported by Smith et al. (2001) and Hansen et al. (2006a). For cauliflower cultivation, the total carbon introduced was 197 g per kg of compost, therefore carbon sinks of 31.5 and 15.7 g $m^{-2}$ were considered for $C_H$ and $C_L$, respectively. This sink was accounted for in the calculation of greenhouse gas emissions as a negative contribution to the total emissions.

### 6.3.3. Avoided burdens of dumping OFMSW and GW in landfill

As previously mentioned, in the allocation procedure for the multi-functional systems, dumping was the method selected as an alternative to composting for organic waste management. Environmental burdens for dumping the same amount of OFMSW and GW as used in the production of the compost used in $C_H$ and $C_L$ were subtracted from the total impacts of these cultivation options.

The process of dumping OFMSW or GW was calculated with the Calculation Tool for waste disposal in Municipal Sanitary Waste Landfill MSWLF from the database ecoinvent data v2.1 (SCLCI 2012). These organic wastes were assimilated in the waste flow “compostable material”, in the calculation tool and the impurities material content was also included. To this was added the collection and transport of waste to the local landfill in a municipal waste collection lorry of 21 t MAL (a distance of 17 km), including the production and cleaning of the collection containers. The construction of the landfill and road access, the machinery operation, the combustion of methane without energy recovery, and the land used, were all considered with a time limit of impact of 100 years (Doka 2007).

## 6.4. Results and discussion

We present the results for the yield, fruit sizes, and the bioactive compound contents, followed by the environmental results for the functional units considered, using these data.

### 6.4.1. Agricultural results: yield and size parameters

The yield and the fruit size parameters are shown in Table 6.4. Significant statistical differences were observed for commercial yield between the option with high dosage of compost and the two others. The commercial yield obtained for $C_L$ and M were 27%

and 39% higher than that for $C_H$, respectively, while no significant differences were found between the $C_L$ and M cultivation options. These values were within the average levels expected for this method of cauliflower cultivation in the area (Doltra and Muñoz 2010). Differences between non-commercial yields were lower than for commercial ones, and non-significant, while there was a significant difference between $C_H$ and M for total yield.

**Table 6.4. Parameters of yield and fruit size.**

| Parameter | Unit | Cultivation option | | | LSD [a] |
|---|---|---|---|---|---|
| | | $C_H$ | $C_L$ | M | |
| | | ***Cauliflower yield*** | | | |
| **Commercial yield** | t ha$^{-1}$ | 12.2b | 15.5a | 17.0a | – |
| **Non-commercial yield** | t ha$^{-1}$ | 36.8 | 43.6 | 45.8 | ns |
| **Total yield** | t ha$^{-1}$ | 49.0b | 59.1ab | 62.7a | – |
| | | ***Fruit size parameters*** [b] | | | |
| **Fruit average wet weight** | g | 705.3c | 894.0b | 994.4a | – |
| **Fruit average dry weight** | g | 74.2b | 89.1a | 94.8a | – |
| **Fruit average diameter** | cm | 19.2b | 20.3a | 20.2ab | – |

*[a] Least significance different taste. Different letters (a, b, c) indicate significant effect and "ns" non-significant effect at p = 0.05 (Section 6.1.3).*

*[b] These parameters were assessed for commercial fruits, considering commercial fruits those without dark patches, and tight and compact, with a diameter of more than 18 cm.*

The fruit average weights were significantly different between the three cultivation options (see Table 6.4), being 41% and 11% greater for M than for $C_H$ and $C_L$, respectively. There was a significant difference between the average dry weight of fruit in cultivation options M and $C_H$, being 28% higher in the M cultivation option. There was no significant difference in this parameter between M and $C_L$. The fruit average diameter was 6% greater in $C_L$ compared to $C_H$ but non-significant differences were found between these options and M.

## 6.4.2. Nutritional results: bioactive compounds content

The concentrations of total sinapic acid derivatives and total phenols (Table 6.5) were significantly higher in $C_H$ samples than in the other two cultivation options, $C_L$ and M, for which the values were very similar. This was also true for individual compounds considered within total sinapic acid derivatives. For total sinapic acid derivatives and for the individual compounds, except for 1-sinapoyl-2-feruloyldiglucoside, the concentration for $C_H$ was between 75–89% higher than for M. The total phenols content in cauliflower samples was 24% higher in $C_H$ than in M.

As shown in Table 6.5, significant differences were found in some of the individual, but not in total, glucosinolates and flavonoids. While the concentrations of total glucosinolates were equal in $C_H$ and M, significant differences were detected for some of the individual compounds without a specific trend. The concentrations were lowest in $C_L$ for all the individual compounds and therefore for the total glucosinolates. Although they are not significant differences, the content of total glucosinolates for the cultivation options $C_H$ and M were double that of the $C_L$ content. The total content of flavonoids in the $C_L$ and $C_H$ samples was 22% and 10%

higher than for M samples, respectively, with a significant difference between $C_H$ and M for one of the individual compounds of the total flavonoids (see Table 6.5).

**Table 6.5. Concentration of bioactive compounds in cauliflower samples.**

| Bioactive compound | Unit (per kg$^{-1}$) | Cultivation option | | | LSD [a] |
|---|---|---|---|---|---|
| | | $C_H$ | $C_L$ | M | |
| **Total sinapic acid derivatives** | mg SAE | 40.9a | 23.3b | 23.3b | - |
| **Sinapic acid** | mg | 1.2 | 0.6 | 0.7 | ns |
| **1,2-disinapoyl-diglucoside** | mg SAE | 10.1a | 5.9b | 5.8b | - |
| **1-sinapoyl-2-feruloyldiglucoside** | mg SAE | 3.5 | 3.2 | 2.7 | ns |
| **1,2,2 Œ-trisinapoyldiglucoside** | mg SAE | 20.2a | 10.1b | 10.7b | - |
| **1,2 Œ-disinapoyl-2-feruloyldiglucoside** | mg SAE | 6.0a | 3.5b | 3.4b | - |
| **Total phenols** | mg CAE | 275.5a | 213.3b | 222.0b | - |
| **Total Glucosinolates** | mg | 169.6 | 84.3 | 164.3 | ns |
| **Glucoiberin** | mg | 128.1 | 65.4 | 137 | ns |
| **Sinigrin** | mg | 1.5 | 1 | 1.6 | ns |
| **Glucoraphanin** | mg | 26 | 10.1 | 12.5 | ns |
| **Progoitrin** | mg | 0.8 | 0.4 | 0.4 | ns |
| **Glucoalyssin** | mg GTPE | 0.6b | 0.6b | 1.1a | - |
| **Glucoiberverin** | mg GTPE | 1.1 | 0.3 | 1.4 | ns |
| **Glucoerucin** | mg | 1.5a | 0.3b | 0.6b | - |
| **4-OH-Glucobrassicin** | mg GTPE | 1.1 | 0.3 | 0.7 | ns |
| **Glucobrassicin** | mg GTPE | 3.8 | 2.4 | 3.9 | ns |
| **Metoxiglucobrassicin** | mg GTPE | 2.8 | 2.3 | 3.8 | ns |
| **Neoglucobrassicin** | mg GTPE | 1.3 | 0.7 | 0.7 | ns |
| **Total Flavonoids** | μg Q3RE | 790 | 876 | 718 | ns |
| **Kaempferol-3-diglucoside-7-diglucoside** | μg Q3RE | 309a | 134ab | 127b | - |
| **Quercetin-3-diglucoside-7-glucoside** | μg Q3RE | 453 | 714 | 568 | ns |
| **Kaempferol-3-diglucoside-7-glucoside** | μg Q3RE | 28 | 28 | 24 | ns |

*SAE, sinapic acid equivalents; CAE, caffeic acid equivalents; GTPE, glucotropaeolin equivalents; Q3RE, Rutin equivalents.*

[a] *Different letters indicate significant effect. "ns" non-significant effect at $p = 0.05$ (Section 6.1.3).*

Diverging trends were found in the bioactive compounds content for $C_H$ and $C_L$, particularly for total glucosinolates and the individual compounds, despite the differences not being significant. It is not easy to explain the behavior of the bioactive compounds we observed in the different cultivation options. Glucosinolates and phenolic compounds are secondary metabolites and could be affected indirectly by different agronomic factors, such as the fertilization practices or the available nitrogen levels, as described earlier. Their distribution in specific areas of the fruit, as well as differences in the water content of the tissues, could also influence the final result. More research is needed in this area to identify potential connections between all these factors.

### 6.4.3. Environmental results

#### *(i) Total environmental impacts*

As mentioned previously, five different functional units were considered to compare the environmental impacts between the cultivation options: the yield, the fruit, the dry matter and the content of two nutritional compounds (see Table 6.3). The results are presented in Table 6.6 and Figure 6.1.

**Table 6.6. Total environmental impacts for the cultivation option $C_H$ (mineral fertilizers plus high-dose compost) considering the five functional units.**

| Impact category | Unit (per corresponding FU) [a] | Functional unit [a] | | | | |
|---|---|---|---|---|---|---|
| | | CY | CF | DM | SA | PH |
| | | per t | per fruit | per kg DM | per kg SAE | per kg CAE |
| **ADP** | kg Sb eq. | 3.01E+00 | 2.12E-03 | 2.86E-02 | 6.14E+01 | 1.09E+01 |
| **AP** | kg $SO_2$ eq. | 5.14E+00 | 3.63E-03 | 4.89E-02 | 1.05E+02 | 1.87E+01 |
| **EP** | kg $PO_4^{3-}$ eq. | -1.79E+01 | -1.26E-02 | -1.70E-01 | -3.65E+02 | -6.50E+01 |
| **GWP** | kg $CO_2$ eq. | -3.45E+03 | -2.43E+00 | -3.28E+01 | -7.03E+04 | -1.25E+04 |
| **OLDP** | kg CFC-11 eq. | 2.55E-05 | 1.80E-08 | 2.43E-07 | 5.21E-04 | 9.26E-05 |
| **POP** | kg $C_2H_4$ eq. | 3.02E+00 | 2.13E-03 | 2.87E-02 | 6.16E+01 | 1.10E+01 |
| **CED** | MJ eq. | 6.69E+03 | 4.72E+00 | 6.36E+01 | 1.37E+05 | 2.43E+04 |

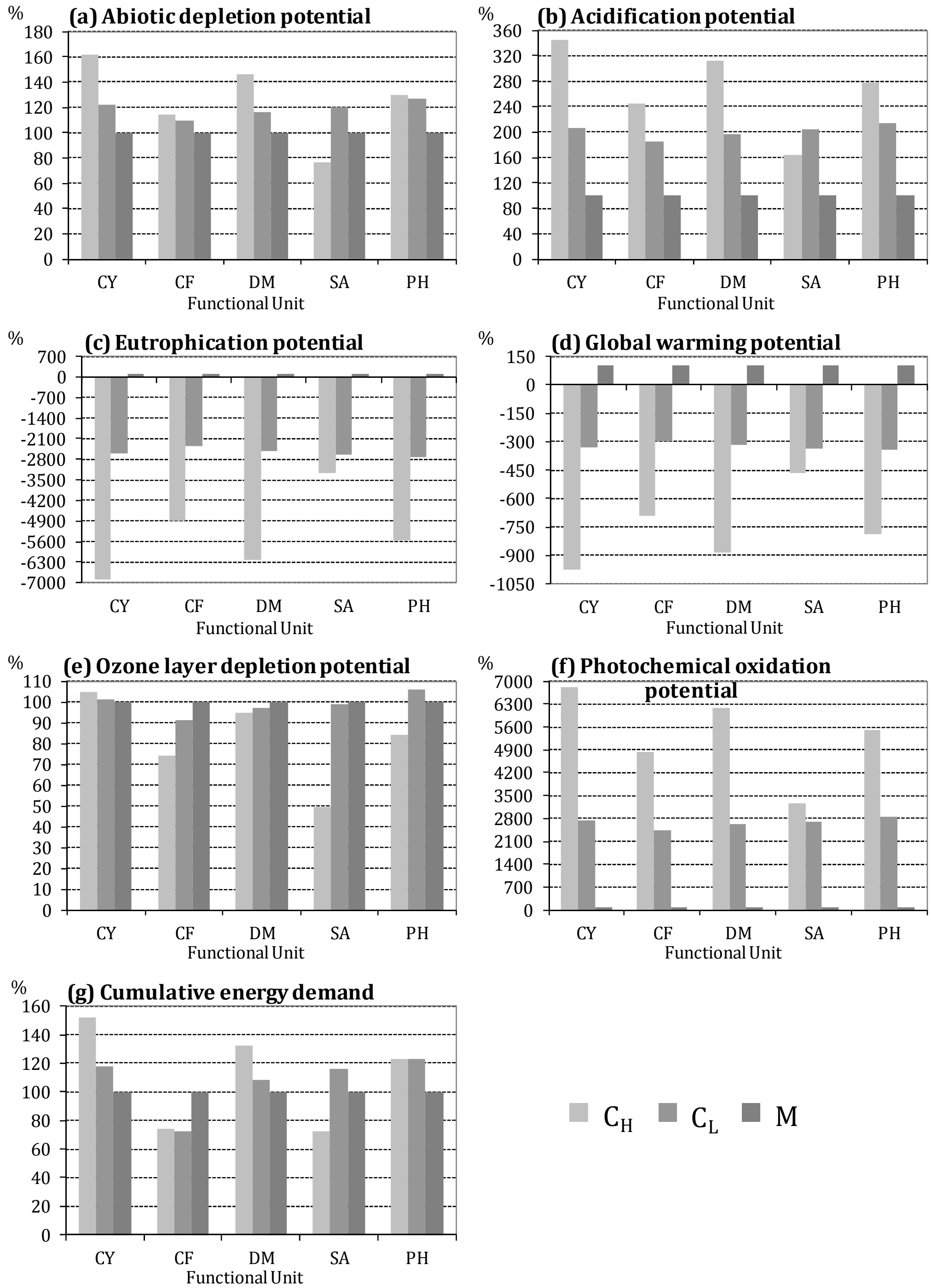
[a] *See Table 6.3 for the acronyms of functional units.*

The environmental impacts of cultivation option $C_H$, for each functional unit, are shown in Table 6.6. From these results, and using the percentages from Figure 6.1, the absolute values could be calculated for the other two cultivation options.

In Figure 6.1, for each impact category, the results are shown as a percentage of the environmental impact of M, which was considered as 100% for the five functional units. For example, the abiotic depletion potential (ADP) for $C_H$ and $C_L$, considering the functional unit of commercial yield, had almost 22% and 62% more impact than M option, respectively.

For the eutrophication potential (EP) and the global warming potential (GWP) categories (see Figure 6.1), $C_H$ had the minimum environmental impacts and M the maximum for all functional units. The environmental impacts of $C_H$ and $C_L$ for both impact categories were negative due to the avoided burdens by composting and not dumping municipal organic waste (see Section 6.3.3). Although leaching during cultivation was considered in the inventory, fertilization dosages were adjusted to the real needs of the crops and therefore few nutrients were leached. The environmental impacts of cultivation option $C_H$ were between 32 and 68 times less than for M for EP and between 4 and 9 times lower for GWP, depending on the functional unit.

For photochemical oxidation potential (POP), the impact order was reversed ($C_H > C_L > M$) in all the categories. The high impacts of options with compost were due to the emissions of volatile organic compounds during organic waste decomposition in the composting facility (Martínez-Blanco et al. 2011b). The impacts for cultivation option $C_H$ were between 32 and 67 times greater than for M, depending on the functional unit.




*See Table 6.3 for the acronyms of functional units.*

**Figure 6.1. Total environmental impacts (%) for the three cultivation options considering the five functional units.**

For these impact categories (EP, GWP, and POP), the impact order did not vary with the functional unit, but the magnitude of the differences between cultivation options changed considerably (see Figure 6.1). For example, the impact distances between $C_H$ and $C_L$, considering the total sinapic acid derivatives content as a functional unit, were lowest for EP, GWP, and POP as the lower production of $C_H$ was compensated for by its higher sinapic acid content.

Regarding ADP and acidification potential (AP), the order of impact was $C_H > C_L > M$ for all the functional units apart from sinapic acid derivatives content, for which $C_L$ had the higher impact. The cultivation option $C_H$ had between 14% and 61% more impact than M, depending on the functional unit, for ADP and between 62% and 244%, for AP.

For the impact categories ozone layer depletion potential (OLDP) and cumulative energy demand (CED), the impact order between the cultivation options changed considerably depending on the functional unit. For the former, the impact order for three out of the five functional units was M ≥ CL > CH, due to the impacts saved by not dumping municipal organic waste. The cultivation option CH had between 5% and 50% less impact than M, depending on the functional unit. Considering commercial yield, dry matter harvest and total phenols content as functional units, the CH cultivation option had the maximum impact and M the minimum for CED, with differences up to 52% between them.

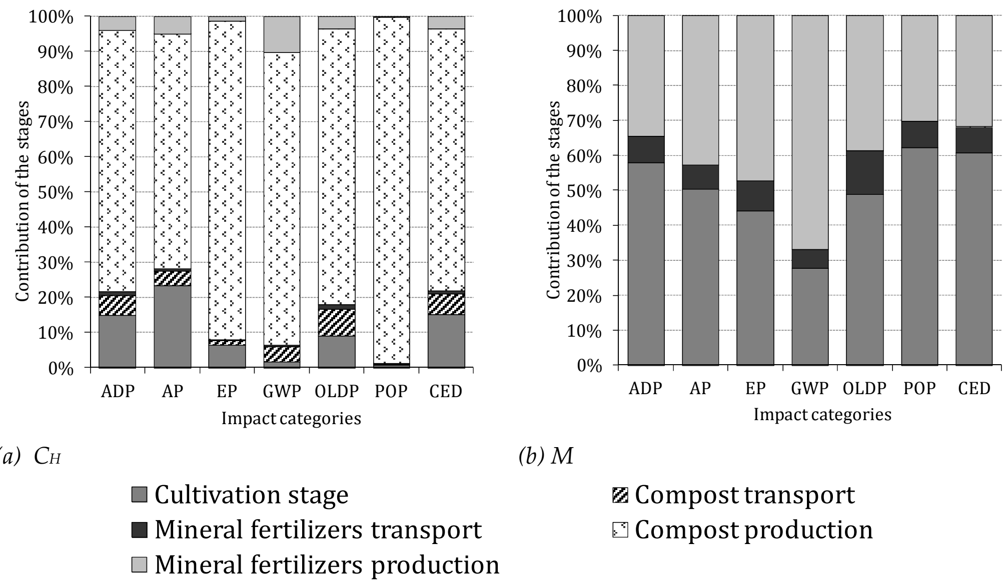
Other functional units were not used in the study as data were not available or non-significant differences between cultivation options were measured. Total glucosinolates content, total flavonoids content and other principal micro and macro-nutrients were assessed but were not considered as functional units as no significant differences were detected between the three cultivation options. Another potential functional unit related with the price of the product (i.e., one euro or one dollar paid for a quantity of commercial cauliflower) would be of interest if market prices differed depending on the type of fertilization or the nutritional content. Other functional units used in previous studies, for example the energy or protein content of fruit or the reaction of the consumer to the food product, were not considered due to the lack of data.

### *(ii) Contributions to environmental impacts*

An environmental contribution analysis was carried out for the three cultivation options, considering the five stages (described in Section 6.2.4.i) and the avoided burdens of dumping OFMSW and GW in landfill (Section 6.3.3). These results are independent of the functional unit considered. The results of the analysis are shown in Figure 6.2 for the cultivation options $C_H$ and M, the two extreme cases.

The compost production stage clearly had the greatest contributor to the total environmental impact of $C_H$, between 74% (ADP) and 99% (POP). As has been pointed out in previous publications (Martínez-Blanco et al. 2010; 2011b), the energy consumption of the facility, the organic waste collection, the VOC emissions and the solid waste fraction to landfill are mainly responsible for the high environmental impacts of this stage. The total impacts in Figure 6.1 were the summation of the contributions of the five stages, with the avoided burdens subtracted. These subtracted burdens were particularly important for EP, GWP, and OLDP, which are the impact categories most affected by the dumping of organic waste: the $CO_2$ and $CH_4$ emissions contribute to category GWP and the emissions and lixiviates of nitrogen and phosphorous contribute to EP and OLDP (Martínez-Blanco et al. 2009b).

For the M cultivation option, impacts were mainly distributed between the cultivation stage, contributing between 28% and 62% depending on the impact category, and the mineral fertilizers production, that contributed with 30–67% to the total impacts.




(a) $C_H$ (b) M

**Figure 6.2. Contribution to total environmental impacts by stages (plus avoided burdens of dumping OFMSW and GW in landfill).**

# 6.5. Conclusions and perspectives

Cauliflower cultivation in Mediterranean fields was chosen due to the high levels of natural bioactive compounds. Apart from functional units linked with the production, others connected with potential differences in the content of these compounds, depending on the type of fertilization, were used.

Higher commercial yield and higher wet average weight of fruit were found for the M cultivation option, with only mineral fertilizers, than for options with compost ($C_H$ and $C_L$). Where there were significant differences in content of bioactive compounds, the content in $C_H$ was higher than in M, (for total sinapic acid derivatives and total phenols).

For EP and GWP, cultivation options with compost had negative impacts for all functional units, and for POP, the impact order was $C_H > C_L > M$ regardless of the functional unit. The environmental results obtained for the other four impact categories (ADP, AP, OLDP, and CED) depended on the functional unit considered.

With production functional units (commercial yield, commercial fruit and commercial dry matter harvest), when commercial yield was considered, M had lower environmental impacts than $C_H$, apart from the EP and GWP categories, due to the higher yield of cauliflower. The number of cauliflowers cultivated per unit area was similar between cultivation options but the fruits for options with compost were smaller and had higher environmental impacts per commercial fruit, apart from EP, GWP, and OLDP. As the differences between cultivation options for moisture content were not significant, the impact distribution for commercial dry matter harvest was similar to that for commercial yield functional unit.

Comparing the environmental impacts of the three cultivation options depending on their bioactive compounds content, the results were reversed for sinapic acid derivatives content, apart from POP, due to the higher content of this compound in the $C_H$ cultivation option. Although $C_H$ presented the highest contents when there was a significant difference in these compounds, the highest levels were found in $C_H$, but $C_L$ did not follow the same trend. The content of total sinapic acid derivatives in $C_H$ is twice that of $C_L$ and M, so that, in all categories apart from POP and AP, $C_H$ had the lowest impacts. In contrast, smaller differences were measured for total phenols content, so that the order of the cultivation options with this functional unit was similar to that of commercial production but with lower differences.

Measuring the differences in bioactive compounds for cauliflower grown with only mineral fertilizers or adding compost is costly and difficult, with a huge number of potential bioactive compounds. As a result, it is difficult for a farmer using compost to demonstrate the higher value of their agricultural product and increase the market price. National statistics and further research in this field is necessary to establish clear trends that would affect the price of agricultural products and promote healthier consumption. More research is necessary on the effects of different cultivation techniques, with fertilizing alternatives, apart from the increase or decrease in yield. In particular, the quantification of the total production of secondary metabolites and their real benefits to human health could contribute to better knowledge of nutrition, aid institutional decision making and lead to more accurate labelling of products. Strategies are also needed to maintain or increase yield using organic fertilizers without reducing their beneficial effect on nutrition.

From the results and trends reported in this study, the importance of comparing environmental impacts with several functional units is clear, especially for food products assessment. Functional units related with economic, quality, safety, and consumer satisfaction parameters, among others, can give considerably different results for environmental impacts.

# Chapter 7. Assessing the environmental benefits of compost use-on-land through an LCA perspective: a review

**Chapter 7** is based on the following book chapter:

Martínez-Blanco, J., Lazcano, C., Boldrin, A., Muñoz, P., Rieradevall, J., Möller, J., Antón, A., & Christensen, T.H. Assessing the environmental benefits of compost use-on-land through an LCA perspective: a review. Accepted for publication (May 2012) in the Volume 11 of the Sustainable Agriculture Reviews, Eric Lichtfouse (editor).

Moreover, the following paper outlines what is said in **Chapter 7**:

Martínez-Blanco, J., Lazcano, C., Boldrin, A., Muñoz, P., Rieradevall, J., Möller, J., Antón, A., & Christensen, T.H. State of the art and future challenges for quantification of compost use-on-land through an LCA perspective. Accepted in May 2012 for publication in the international journal Agronomie for Sustainable Development.

# Abstract

Composting is a widespread alternative for managing organic waste that has been claimed as a way to recycle nutrients and improve soil properties. A large body of scientific evidence shows that the benefits of the use of compost on soil go well beyond the supply of nutrients. Most of these benefits have been so far excluded from LCA studies, mainly because of scarcity of data or lack of appropriate impact assessment methods. In this study, nine of the environmental benefits of compost application typically reflected in the scientific literature – nutrient supply, carbon sequestration, weed pest and disease suppression, increase in crop yield, decreased soil erosion, retention of soil moisture, increased soil workability, enhanced soil biological properties and biodiversity, and gain in crop nutritional quality – were intended to be quantified through literature revision and then their effects on the foreground and background system were discussed. Additionally, the suitability of currently available assessment methodologies was expounded. For "nutrient supply" and "carbon sequestration", both quantification and impact assessment of the effects could be performed and thus they should be regularly included in LCA studies. For four of the nine benefits, quantitative figures could not be provided, either because of complete lack of data or because the effects are both very variable and too depending on specific local conditions. For "soil erosion" and "soil moisture content" effects could be quantitatively addressed, but suitable impact assessment methodologies were not available. Finally, "weed pest and disease suppression" could not be generally proved. Additional impact categories – dealing with phosphorus resources, biodiversity, soil losses, and water depletion – may be needed for a comprehensive assessment of compost application.

## Keywords

Sustainable agriculture; Environmental impact; Life Cycle Assessment; Compost; Bio-waste; Organic matter; Soil.

# 7.1. Introduction

There is increasing concern about soil interrelated environmental problems such as soil degradation, desertification, erosion, and loss of fertility (European Commission 2006c). These problems are partially consequence of the decline in organic matter content in soils. A level of 2% of soil organic carbon (SOC) is commonly considered desirable for maintaining good soil structure for agricultural activities (Van-Camp et al. 2004). An estimated 45% of European soils have low (<2%) soil organic matter (SOM) content, principally in southern Europe but also in areas of France, the UK, and Germany (European Commission 2006c).

A parallel concern is the massive generation of organic waste by human activities – for instance, organic waste represents 30–40% of the municipal solid waste (MSW) generated in the European Union (European Parliament2010) –, which has led to the proposal of several alternatives to avoid landfilling and promote recycling. Among these alternatives, composting is one of the best-known processes. Composting allows the stabilization and sanitation of organic waste through accelerated aerobic decomposition under controlled conditions. In addition, depending on the quality of the initial feedstock, the final compost can be used for agricultural purposes in a local context.

Besides the supply of plant nutrients, several studies indicate that the use of compost on land may improve several plant and soil parameters, which would make compost an interesting fertilizing option not only for agriculture but also for soil restoration purposes. Compost addition increases SOM content, which enhances aggregation and stability, thereby ameliorating soil structure (Diacono and Montemurro 2010). Due to the stability of soil aggregates, the application of compost prevents surface sealing, improves water infiltration, and enhances water holding capacity thus reducing runoff generation and soil erosion (ROU 2007). Moreover, increasing SOM levels promote soil as a sink of carbon, also called carbon sequestration (Favoino and Hogg 2008; Marmo 2008). Other potential benefits of compost application are improved biological activity (Bastida et al. 2008; Hargreaves et al. 2008), enhanced nutrient availability (Boldrin et al. 2009), and the suppression of soil borne diseases (Bonanomi et al. 2007). Furthermore, several authors have reported higher yields with compost application and better quality of the harvest. Nevertheless, the application of compost may also produce environmental and agronomic drawbacks. The most relevant ones are gaseous and leachate emissions, immobilization of soil mineral N, increase in salt content, and accumulation of heavy metals in soil (Hargreaves et al. 2008). These issues are in general related to the quality of the feedstock waste used, the maturity of the compost, and the application of the compost (e.g. crop rotation, soil type, etc.). In fact, insufficiently stabilized compost may have an immediate negative impact on crop productivity thereby producing great economic losses. The choice of fertilizer to be used in agricultural operations is generally based in productivity and profit-based criteria. Nevertheless in recent times, the increasing societal concern about pollution, land degradation and resource consumption is introducing other type of criteria such as environmental impacts and benefits in the decision process. Decision tools are now forced by society and regulations to be focused on environmental aspects.

Several tools are available to quantify positive and negative environmental impacts of compost in a comprehensive manner. Among them, Life Cycle Assessment (LCA) was promoted in different European directives as a robust quantitative tool, and a keystone in decision making by producers and stakeholders. LCA has been widely used for the assessment of the waste and agricultural sectors. For example, several studies used LCA for comparing the environmental performance of compost production and application with conventional fertilization (Lundie and Peters 2005; Hansen et al. 2006a; ROU 2007; Tidaker et al. 2007; Blengini 2008; Martínez-Blanco et al. 2009b; Meisterling et al. 2009; Ruggieri et al. 2009a; Boldrin et al. 2010; Martínez-Blanco et al. 2010; 2011a; 2011b). While the environmental impacts associated to compost production have been successfully assessed in these studies, assessment of the impacts associated to the use of land have not yet been fully quantified as the abovementioned benefits of compost on plant and soil have not been taken into account. Up-to-date and due to methodological difficulties, carbon sequestration and nutrient supply were the unique environmental benefits taken into account when evaluating the environmental impacts of compost application. Regarding carbon sequestration, Favoino and Hogg (2008) proposed some bases for its quantification in LCA. Hansen et al. (2006a) developed a model for quantifying savings of mineral fertilizers, emissions to water and air, and carbon sequestration effects. Boldrin et al. (2009) reviewed the available data for greenhouse gas (GHG) accounting of composting systems, including the savings from carbon sequestration and mineral fertilizers. Martínez Blanco et al. (2011a) took into account changes in yield and nutritional content for cauliflower production.

Because of the modeling complexity, ROU (2007) is, to our knowledge, the only study where an attempt was made to include most of the abovementioned effects within LCA of two Australian case studies. The results were however only presented at the inventory stage and the obtained figures were not included within the impact categories. Some other studies (e.g. Hansen et al. (2006a); Boldrin et al. (2009); Martínez-Blanco et al. (2011a; 2011b)) addressed the issue in a qualitative manner, recommending that further research should be undertaken on the subject.

The main goal of this paper is to quantitatively address LCA modeling of those positive effects traditionally associated to land application of compost originating from organic MSW and garden waste. This is done by: (1) explaining some of the main dynamics determining the considered environmental and agronomic benefit (from now on benefit); (2) providing quantitative data (inventory) based on a critical revision of state-of-the-art literature; (3) describing existing impact methodologies and future challenges for assessing the benefits under a LCA perspective.

# 7.2. Methodology

A comprehensive revision of the literature dealing with the effects of compost on soil and plant growth was carried out between November 2010 and May 2011. The literature revision was divided into two parts, following the goals of the paper. First, a broad revision of those available studies assessing one or several of the potential benefits of compost application was made. In the second half of the paper, the current situation of the inclusion of each of these benefits in LCA studies was reviewed and discussed.

### 7.2.1. Literature review on potential compost benefits

The most relevant effects of compost on soil properties and plant growth were first identified through a literature search. Subsequently, 90 articles (including both reviews and case studies) were selected for further revision and collection of inventory data. The following criteria were used:

- Data from organic MSW or bio-waste (food and kitchen waste from households, restaurants, caterers and retail premises, and comparable waste from food processing plants) and green waste (coming from private gardens and public parks) were taken into account.
- Field studies were selected, when possible, instead of pot and laboratory incubation experiments.
- Compost used for agricultural purposes was in focus. However, land reclamation or restoration studies were also taken into consideration when necessary.
- Articles published earlier than 1990 were not taken into account, when possible, as they were replaced by better or newer studies.

Some of the studies included in the review did not fulfill one or more than one of the previous criteria. These studies were still included as they constituted the only scientific contribution in a specific area or because they were particularly relevant.

According to the literature revision, the effects were classified into short-term (1 year), mid-term (1–10 years), and long-term effects (10–100 years), depending on the length of the studies. Thus, quantification of specific effects is associated with the corresponding time frame.

Regarding the combination of compost, doses and experiment length, the majority of the reviewed papers on the short-term considered a unique application, while periodically applications are taken into account for longer studies. For the homogenization of the data, the annually proportional rates were considered in the Annex II, therefore in these cases the cumulative effects of compost application, rather than the residual effect were reported.

All the publications included for each of the assessed compost benefits were summarized and the main information is given as tables in Annex II. The more relevant parameters – such as the type of composted bio-waste, the dosages of compost and other fertilizers, the types of crop grown if any, the soil type, and the length of the experiment – were defined for each study, as well as the corresponding parameters used to measure the benefit. According to the specificity of each compost benefit, either average values or range values were stated when possible.

### 7.2.2. LCA of compost use on land

With regards to the LCA quantification and assessment methodologies, the ranges obtained in the literature review of the compost benefits were used as starting point and a broad revision of the current status of the corresponding assessment methodologies was prepared. The first step was to state and describe, for each benefit, the consequence or consequences through an LCA perspective of each one of its benefits on soil, plant, environment, farmer or harvest. After that, the quantification of the substituted or saved process based on up-to-date knowledge and data was discussed (Table 7.1 and 7.2). Thirdly, the impact categories in existing impact assessment methodologies which are most affected when considering compost application, were addressed, together with the current status of new assessment methodologies (Table 7.3). Finally, the nine benefits studied were classified according to the existing evidences for the positive effects on soil and plant, the possibility of quantification, and the current availability of tools for their consideration in LCA.

# 7.3. Review of compost benefits

A review of the literature dealing with the potential benefits resulting from compost application reflected in the scientific literature is presented in the next sections. The nine benefits included are nutrient supply, carbon sequestration, weed pest and disease suppression, increase in crop yield, decreased soil erosion, retention of soil moisture, increased soil workability, enhanced soil biological properties and biodiversity, and gain in crop nutritional quality. For each one, a discussion of the main factors affecting the performance of the benefit, the degree of proof and the range of the effects measured are included.

## 7.3.1. Nutrient supply

An adequate use of composted products, together with a mineral nitrogen supplementation, can provide the same fertilization level as mineral NPK fertilizers, while saving the economic and the environmental costs associated with the production and use of inorganic fertilizers. Moreover, compost can both provide essential trace minerals to the soil that are not supplied when mineral fertilizers are added (Favoino and Hogg 2008), and increase the cation exchange capacity (Diacono and Montemurro 2010). This favours a better retention of nutrients in the soil, thereby increasing fertilization efficiency.

The content of nutrients in composts is highly variable, depending on the raw material composition and the composting conditions. For example, Boldrin et al. (2009) reported that the typical contents of N, P and K in bio-waste compost are 7–28, 1–9 and 3–23 kg per ton of dry compost, respectively. However, these nutrients are not immediately available to the growing plant (Favoino and Hogg 2008). Because of low nutrient availability and slow mineralization rates, some authors claim that compost is not a good fertilization supplier (Fitzpatrick 2004; Moral and Muro 2008), although it is complementarily enhancing physical and biological fertility of soils (Gosling and Rayns 2008).

Besides, a period of net immobilization of N after compost application was reported for some case studies, and an extra application of mineral N was required to prevent N deficiency. Initial N-immobilisation is less likely with mature compost with a low C/N ratio, because N-immobilization already occurred during the composting process (Amlinger et al. 2003b; Bar-Tal et al. 2004; ROU 2007; Hargreaves et al. 2008).

The fate of N after compost application depends in general on the compost quality and composition (e.g. quality and nutrient content of raw material, management of composting process, etc.) together with site and management conditions (climatic conditions and period of the year, time of application, soil properties, mechanical aeration, crop rotation, and plant-soil interactions).

Based on several revisions (Amlinger et al. 2003a; 2003b; ROU 2007; Hargreaves et al. 2008; Diacono and Montemurro 2010), it could be stated that between 5–22% of the N contained in compost is available the first year, and 40–50% in 3 to 5 years. However, most of these studies have been done for temperate climates, while in warmest climates the mineralization rates can increase (Sikora and Szmidt 2004). Accordingly, Hadas and Portnoy (1997) and Elherradi (2005), both studies located in the Mediterranean region and including experimental measurement of the mineralization rates, are reporting rates of 15 to 24% in 4-5 months. For P, availability is 35–38% in the first year and 100% is available in 3 to 5 years, while 75–80% of the K is available the first year and 100% in 3 to 5 years (Table A6 in Annex II). Boldrin et al. (2009) reported that final – i.e. long-term – utilization efficiencies for N, P and K were in the order of 20–60% for N, 90–100% for P, and 100% for K, without taking into consideration the excess of P and K with respect to the N applied.

## 7.3.2. Carbon sequestration

The SOC pool is the largest terrestrial reservoir of organic C (Lal 2004a) More than twice as much carbon is held in soils as in vegetation or in the atmosphere (Whalen and Sampedro 2010). Pools of SOC are defined, as active (2 to 5% of SOC), slow (45–48% of SOC), and resistant (50% of SOC) according to their decomposition rate (Paul et al. 2001), where the latter is of major significance in long-term C sequestration. Increasing SOC stocks is one of the strategies with the largest potential for climate change mitigation (Rogner et al. 2007). The fate of SOC stocks is largely determined by the land management, as for example intensive agricultural practices can deplete the SOM and result in a net release of SOC to the atmosphere.

The use of compost could represent an effective alternative for increasing SOC stocks (Lal 2004a). Table A7 (in Annex II) shows some examples of compost effects on soil carbon sequestration. For example, Fortuna et al. (2003), reported that application of compost for 4 years to a crop rotation increased the slow pool C by 75% and the resistant SOC pool by 40%. Similarly, Lal (2002) estimated that the use of compost (10–20 t $ha^{-1}$ $yr^{-1}$) has the potential to increase SOC between 500 and 1,000 kg $ha^{1}$ $yr^{-1}$.

In general terms, sequestration rates are highly variable and depend on several compost, and site-dependent characteristics and management. Besides the application rate, the carbon sequestration potential of compost depends largely on its degree of humification, as the labile C fractions are rapidly metabolized and only the recalcitrant ones remain stable in the soil (Fabrizio et al. 2009; Sodhi et al. 2009b). As a result, C sequestration potentials are higher with mature composts (Eghball 2002). With regards to the soil characteristic, an important factor is the limited capacity of the soil to act as a C sink: SOC content increases rapidly when organic amendment is added to soils where SOM is depleted, while the increase rate is slower when the SOC content is close to saturation i.e. the maximum amount of stable C that one soil can maintain (Stewart et al. 2007). After a single compost application, the SOC sequestration rate increases remarkably in the short-term but decreases slowly over time due to the mineralization of the organic matter contained in compost, so that large differences can be seen depending on the time horizon chosen (Table 7.1). Audsley et al. (2003) stated that it can take over 50 years of continuous restorative practices to reach this new equilibrium in SOC level, although much of the change takes place in the first 10 years and slow down when soils are close to a saturation level. Finally, climatic conditions determine mineralization rates and the residence time of SOC in soil, as mineralization rates tend to be higher in warm climates (Sikora and Szmidt 2004).

As abovementioned, when estimating the sequestration potential, an important role is played by the time horizon chosen for the calculation. For example, in a short-term field study Fabrizio et al. (2009) observed that 150 days after application of 50 and 85 t $ha^{-1}$ of bio-waste compost, 40 and 53% of the applied C remained in the soil respectively. Similarly, Fagnano et al. (2011) reported that, six months after the incorporation of bio-waste compost to a sandy loam soil, the C retained in soil was between 63 and 70% of the C added. On a much longer time-horizon, more relevant for IPCC and LCA approach, other studies estimated that between 2% and 14% of the carbon introduced with a single compost application is still bound to soil after 100 years (Smith et al. 2001; Favoino and Hogg 2008; Boldrin et al. 2009).

To include all the above mentioned factors, quantification of the exact amount of C retained in soil after compost application may thus need to be assessed through the use of complex agroecosystem models that incorporate mineralization dynamics for the case-specific conditions (Bruun et al. 2006).

### 7.3.3. Weed, pest and disease suppression

Weeds reduce crop quality and yield by competing for light, water and nutrients. The use of compost amendments and mulches has been suggested as a potential alternative to the use of herbicides. According to ROU (2007) and De Cauwer et al. (2010), organic composted mulches could effectively control weeds in crops, thus avoiding herbicide applications totally or partially. However, authors also reported that compost does not have relevant effects on weed grow and density when used as a soil amendment.

Together with weeds, the incidence of pest and disease is one of the major factors limiting the productivity of agro-ecosystems. Modern agriculture heavily relies on the use of synthetic pesticides, although they are proved to damage the environment and human health (Margni et al. 2002; Streck 2003) and they often have unsatisfactory effectiveness (Litterick et al. 2004; Bonanomi et al. 2007). Finding efficient, low-cost, and environmentally acceptable alternatives against plant pests and diseases is one of the biggest challenges for modern agriculture. Litterick et al. (2004) pointed out several integrated crop protection strategies as potential alternatives to the use of synthetic pesticides, including crop rotation, local resistant varieties, biological control agents, and, as it is explained below, suppressive compost application.

Several factors concur for a successful biological control of plant pathogens with composts. During the composting process, the use of heterogeneous input material(s), a sufficient and controlled temperature, aeration and moisture, a correct maturation process, and non-anaerobic storage are the most important recommendations to obtain a suppressive compost (Litterick et al. 2004; Bonanomi et al. 2007; ROU 2007; Bonanomi et al. 2010; De Bertoldi 2010). Microbial diversity and the presence of specific suppressive micro-organisms or consortia in compost are also suggested to be essential for disease suppression (Amlinger et al. 2003a; Termorshuizen et al. 2006; De Bertoldi 2010). High compost application rates are related to higher suppression and a lag of time between compost application and planting is advisable (Amlinger et al. 2003a). The disease suppressive properties of composts are induced during the curing phase, because most biocontrol agents (mainly fungal and bacterial species) re-colonize the compost from the surrounding air and soil after peak heating (Litterick et al. 2004; ROU 2007; De Bertoldi 2010). However, ROU (2007) and Bonanomi et al. (2010) argued that excessively stabilized organic matter may not support adequate activity of biocontrol agents due to the scarcity of available nutrients.

A decreased incidence of diseases after compost application to soil could be attributed either to a direct suppression of the pathogens or to a systemic resistance induced in the crop (Noble and Coventry 2005). Direct suppression can be general or specific depending on whether there is only one suppressive agent or suppression is caused by the joint action of several factors. In any case, direct suppression is due to mechanisms such as competence, antibiosis and parasitism (De Bertoldi 2010). Nevertheless, recent publications indicated that there is still insufficient knowledge about the general principles of disease suppression by compost, because it is often pathogen-specific and related to the mechanism(s) of disease suppression (Termorshuizen et al. 2006; Bonanomi et al. 2010).

The application of composts can rarely – if ever – result in a disease control comparable to the use of chemicals (Litterick et al. 2004). Bonanomi et al. (2007) reviewed 250 studies about direct suppression of soil-borne fungal diseases in pot and field experiments, reporting relevant suppressive effects for several types of compost in 58–74% of the studied cases for *Phytium* spp, *Fusarium* spp, *Phytophthora* spp and *Verticillium dahlia* (Figure A1 in Annex II). For the other pathogens the suppression was lower or null. However, only 12% of the cases studied resulted in a disease reduction >80% when organic amendments were applied, which is the minimum pathogen population reduction that farmers are using as a criterion for replacing chemical pesticides (Bonanomi et al. 2007). Precise information about the duration of the suppressive effects was not found; however, most of the reviewed studies estimated the suppression was efficient for 3–6 months after the application of the amendment.

If on one hand compost can have a disease suppression effect, it should be mentioned that there is also a risk of pathogens and unwanted plant seeds being present in compost (Vinneras et al. 2010). This risk can be minimized with a proper combination of temperature, processing duration, and turning frequency during composting, thereby ensuring sanitization from human and plant pathogens, and weed seeds (Noble and Roberts 2004; Hargreaves et al. 2008; De Cauwer et al. 2010). Noble & Roberts (2004) reported that for most of plant pathogens and nematodes a peak temperature of 64–70°C and duration of 21 days is sufficient to reduce figures to below the detection limits.

In conclusion, the extent to which pest and diseases are suppressed when compost is used cannot be univocally determined, mainly because of lack of evidence in field conditions, as well as because the rate of suppression is closely dependant on the type of pathogen and the interrelation with the analysed plant. While there is a considerable amount of data covering pot experiments, field experiments lack and/or published studies only focused on soil-borne diseases. Results for laboratory experiments cannot be directly extrapolated to field crops, because field conditions are much more variable (Fuchs 2010) and because much smaller quantities of compost are typically used in the field compared to pot conditions.

### 7.3.4. Crop yield

Agricultural expansion and intensification are needed to meet the increasing global demand of food. Agricultural intensification involves increased fertilization; in most cases there is a large response to nitrogen fertilization measured as crop yield. As the cost of fertilizers is often small compared to the cost of lost yield, farmers prefer over-fertilization of crops with nitrogen rather than risking under-fertilization and consequent loss of revenue (Del Amor 2007). Moreover, agricultural producers often claim that crop yields are much lower with organic fertilization than with inorganic fertilizers (Mäder et al. 2002). However, excess nitrogen may result in lodging, greater weed competition and pest attacks, with substantial losses of production. In addition, the nitrogen not taken up by the crop is likely to be lost to the environment, which potentially contributes to groundwater and atmospheric pollution (Weinbaum et al. 1992; FAO 1998; Roy et al. 2006).

Sustainable agriculture would ideally produce good crop yields with minimal impact on the ecological system (Mäder et al. 2002). However, there is lack of knowledge about dosages and time applications of organic amendments for specific areas and crops, and often agricultural systems result in heavy pressure on the environment. Indeed, an optimized use of fertilizers should be targeted upon several factors, including the soil nutrient pool, soil management, compost composition and maturity, rates and methods of application, crop species and variety, fertiliser rates, and intervals between application and planting (Amlinger et al. 2003b; ROU 2007; Diacono and Montemurro 2010).

Application of MSW compost had positive effects on growth and yield of a wide variety of crops (Shiralipour et al. 1992). Experimental tests show that trials amended with bio-waste compost provide higher yields than controls not receiving any treatment (Table A8 in Annex II). However, when compost-based fertilization is compared to mineral fertilization, no relevant differences in crop yields are reported: non-significant differences were observed in 64% of the cases; while lower and higher yields for the composting option were seen in 32% and 4% of the cases respectively (Table A8, including 19 publications, 77 cases).

The best agronomic performance of compost is obtained with high dosages and high frequency of application, and by complementing compost with mineral fertilizers to achieve a balanced supply of nutrients (Amlinger et al. 2003a; ROU 2007; Diacono and Montemurro 2010). In general, complementing compost with mineral fertilizers leads to yields similar to the conventional production pathway (Table A8). When compost is used alone, long-term and repeated applications are needed to achieve a steady state that guarantees a crop yield close to the inorganic fertilizing scheme. According to Herencia et al. (2008), yield increases in fields transitioning from conventional to organic production systems usually require 3–5 years to be detected.

### 7.3.5. Soil erosion

Soil mass is the result of the balance between inputs (litter decay, water, etc.) and outputs (leaching, runoff, decomposition, etc.). The only factor likely to cause relevant changes in soil mass is soil erosion (Cowell and Clift 2000), which is a major issue particularly when soil is subject to intensive agricultural practices for many years. Globally, it is estimated that more than 75% of the arable soils of the world suffer from moderate to very high soil losses (Reich et al. 2001).

Soil erosion involves great economic losses and environmental damage due to the decline in soil productivity and soil functions and, in the worst scenarios, the irreversible loss of cultivable land. Indirect effects associated to soil erosion are the contamination of streams with soil particles and agrochemicals, the incidence of respiratory diseases in connection with particles transported by wind, and the associated loss of SOM and soil nutrients.

Organic matter has a key role in stabilizing soil structure, as it increases the inter-particle cohesion within aggregates and enhances the aggregate's hydrophobicity, thus decreasing their breakdown (Soane 1990; Ruehlmann and Körschens 2009; Diacono and Montemurro 2010). Aggregate stability determines soil sensitivity to crusting and erosion (Bissonnais 1996). The application of easily decomposable organic substances have an intense and transient effect on aggregate stability while more recalcitrant substances have a lower but longer term effect (Abiven et al. 2009).

Several studies showed that compost addition significantly increases soil aggregate stability therefore making it more resistant to erosion. Short-term incorporation of bio-waste compost to soil can increase aggregate stability as much as 41% and decrease aggregate instability between 6.9–30% depending on the dosage (Table A9 in Annex II). Short-term experiments under simulated rain showed that soil runoff can be reduced by 66% with compost incorporation. Data concerning mid- and long-term effects of bio-waste compost on soil erosion are scarce, and thus data regarding other types of compost were also included (Table A9). Large variability in compost effects was observed in mid-term experiments, with increases in aggregate stability ranging from 0% to 63% and decreases in aggregate instability of 21% (Table A9). As a consequence, soil loss was reduced between 5% and 36%, and soil runoff between 26% and 30% (Table A9). With regards to long-term compost application (10 year), Sodhi et al. (2009a) reported that rice straw compost application to soil may increase aggregate stability by 4%. No other studies covering long-term perspective could be found. Finally, it should be noted that field studies mainly regard the Mediterranean areas, while data from other regions are scarce.

### 7.3.6. Soil moisture content

Agriculture is by far the largest water-using sector, accounting for about 70% of the consumed freshwater coming from rivers and aquifers (FAO 2008). Several measures are proposed for reducing water demand in the agricultural sector, including efficient irrigation, reduction of distribution losses, maximisation of effluent reuse, and conservation of soil moisture.

One important function of soil is to store moisture and supply it to plants between rainfalls or irrigations. The water holding capacity (WHC) of soil is defined as the water retained in soil between field capacity and the permanent wilting point, and it represents the total available water storage of the soil. Water can be retained in soil if the moisture content is below the WHC, while run-off occurs when the moisture content is above WHC. Plant available water (PAW) is the portion of WHC that can be absorbed by a plant without stress problems.

The amount of soil water available to plants is determined by the depth of the roots and the moisture storage capacity of the soil. Because the latter is linked to porosity, the particle sizes (texture) and the arrangement of particles (structure) are the critical factors (Rawls et al. 2003), meaning that poor structure, low SOM, low carbonate content and presence of stones reduce the moisture storage capacity of a given soil texture class. According to Bot and Benites (2005), the addition of organic matter increases the number of micropores and macropores in the soil either by "gluing" soil particles together or by creating favorable living conditions for soil organisms. As a consequence, less irrigation water is needed to irrigate the same crop.

Compost has a high WHC because of its organic matter content, which in turn improves the WHC of the soil (Hargreaves et al. 2008). Due to improvements in soil physical and chemical properties (Dorado et al. 2003; ROU 2007), the application of composted organic matter reduces surface sealing, improves infiltration, lowers the water table while maintaining the same moisture, and reduces runoff generation (Pandey and Shukla 2006; ROU 2007). Large increases in WHC are reported for high-rate compost application on soils with initially low SOC content (Rawls et al. 2003; Olness and Archer 2005; ROU 2007; Glab et al. 2009).

Olness and Archer (2005) applied the General Energy Model for Limited Systems to U.S. National Soil Inventory Database and indicated that 1% increase in SOC causes a 2 to 5% increase in soil WHC depending on the soil texture. In other studies, the same authors reported increases in WHC ranging from about 0.8 to 8.4% for each percent increase in SOC. Accordingly, Hudson (1994) showed that for each 1% increase in SOM, the available WHC in the soil increased by 2.2–3.7%

With regards to application of bio-waste compost, the reviewed literature (Table A10 in Annex II) showed that when compost is applied at low rates (below 16 t $ha^{-1}$ $yr^{-1}$) no effects are observed in the moisture content in soils, even if compost application continues for several years (Hortenstine and Rothwell 1973; CIWMB 1997; Suzuki et al. 2007; Glab et al. 2009; Castillejo and Castello 2010). Sabrah et al. (1995) measured an increase on PAW from 16.5 to 43% when 16.5–66 t $ha^{-1}$ $yr^{-1}$ of compost were applied. Naeini and Cook (2000) measured less than 2% of increase in WHC after the application of 50 t $ha^{-1}$ of compost in a silt loam soil. Discordant results are reported for sandy soils; for example, Hortenstine and Rothwell (1973) found that applications of compost ranging 32–128 t $ha^{-1}$ $yr^{-1}$ led to WHC proportional increases of 5–50%, whereas Weber et al. (2007) did not detect any difference in WHC after one year of 120 t $ha^{-1}$ application. However, even in soils where compost does not increase PAW per unit of soil volume, compost amendments increased the availability of water to plants by facilitating denser and deeper root growth (Curtis and Claassen 2005).

### 7.3.7. Soil workability

Soil workability is a measure of how easy it is to till or plough a soil. Workability thus depends on several interrelated soil properties, including soil texture, SOM content, structure, bulk density, and presence of gravel and stones (Fischer et al. 2008). Intensive agricultural practices may lead to significant losses of the SOM and to increase its bulk density causing soil compaction and decreasing workability. According to Soane (1990), soil compaction is directly related to reduced yield and quality of crops, increased erosion, and increased power requirements for tillage.

Several studies have shown that organic matter addition to the soil decreases bulk density, and increases aggregate stability (Soane 1990; Ruehlmann and Körschens 2009) thereby reducing soil compaction. Addition of compost to soil supplies organic matter and can potentially decrease soil bulk density and soil compaction, thereby increasing soil workability. In the short-term (<1 year), the application of different types of bio-waste compost decreased soil bulk density between 2.5 and 21% as compared to soil without compost (Table A11 in Annex II), while in the mid-term (1–10 year) the decrease may be as much as 20.6 and 23.1%, with a minimum decrease of

0.7% (Table A11). Long-term (13 year) compost application can decrease soil bulk density by 20.6% (Table A11). Compost application seems to be more effective in the short-term (Weber et al. 2007) and with increasing dosages (Hemmat et al. 2010), as larger doses produce larger decreases in bulk density.

## 7.3.8. Soil biological properties and biodiversity

Farming systems have a strong impact on soil biodiversity. In particular, organic farming systems fostering the use of organic fertilizer showed to increase soil biodiversity significantly (Mäder et al. 2002; Bengtsson et al. 2005; Garratt et al. 2011).

In the same way, application of compost to soil has significant effects on above- and belowground soil biota, both in terms of amount, activity and diversity of organisms. Similarly, Griffiths et al. (2010) observed that addition of 2.6 t $yr^{-1}$ $ha^{-1}$ of bio-waste compost to a soil during six years increased earthworm biomass by 514%, enchytraeids by 1100%, collembola by 271%, oribatids by 92%, mesostigmatids by 84%, microbívorous nematodes by 35% and omnivorous nematodes by 45%.

Due to the central role of microorganisms in soil nutrient cycling, most of the studies dealing with the impacts of compost on soil biodiversity are focused on the impacts on the microbial community. Different studies showed that the biomass of microorganisms in the soil can increase between 5 and 116% after addition of compost (Tejada and Gonzalez 2006; Fuchs 2010). In mid-term applications, compost showed to increase soil microbial biomass between 10 and 225%, whereas smaller increases in this parameter were observed in the long-term (Table A12 in Annex II). Compost application also influences the activity of soil microorganisms (Diacono and Montemurro 2010), which are essential for the maintenance of soil functionality and fertility, as they are key players in several processes such as decomposition of organic matter and nutrient cycling. Short-term experiments showed increases in microbial activity of 0–57%, while mid-term trials showed that microbial activity can increase up to 263%. Microbial activity showed variations between 1–7% in long-term experiments (Table A12).

Besides changing the abundance and activity of soil microorganisms, several authors signalled that compost can also influence microbial diversity. Data determined through molecular techniques analyzing microbial DNA profiles such as denaturing gradient gel electrophoresis (PCR-DGGE) or terminal restriction fragment length polymorphism (TRFLP), showed that the effects on soil microbial diversity are highly variable, ranging from reductions (-2.2%) to increases (1.8%). Compost addition also alters soil microbial functional diversity as evidenced by community level physiological profiles (CLPP). Higher functional diversity indicates that the microbial community is able to carry out a broader range of metabolic reactions (Garland and Mills 1991). Short-term experiments showed that functional diversity increased between 5.21% and 9.38%, and between 4.09% and 7.02% in long-term experiments (Table A12). No data were found for mid-term experiments.

Literature covering aboveground species is limited, and results are in some cases inconclusive. For instance, Tejada et al. (2006) showed that compost can increase the biomass of natural vegetation by 68.7% in degraded soils, while Bastida et al. (2008) observed that composted sewage sludge increased natural plant biomass but decreased significantly species diversity as compared to a non-amended control. In contrast, Sutton-Grier et al. (2009) observed that compost addition to a degraded wetland had limited early effects on plant communities.

Finally, an issue that should also be considered is that heavy metals contained in bio-waste compost can have toxic effects on soil biota. The extent is influenced by metals concentration, the compost application rate, and the type of soil (De Araújo et al. 2010).

### 7.3.9. Crop nutritional quality

Increasing attention is being paid to the role of diet in human health, and in food safety and security (Podsedek 2007; Lairon 2010). Phytochemicals or secondary metabolites, many of which are essential vitamins and health-promoting antioxidants, have recently attracted the attention of the industry, scientists and consumers, because of their potential health benefits (Sun and Tanumihardjo 2007).

Phytochemicals are produced by plants in response to biotic or abiotic sources of stress. The type and amount of metabolites produced is influenced by several factors, with genetic and agronomical factors (i.e. species, variety, crop management strategies and circumstances, postharvest storage, plant stage, cooking, etc.) as main drivers (Gratacós-Cubarsí et al. 2010). In well fertilized agricultural fields, plants have typically a quick growth, thereby developing low concentrations of antioxidants and some vitamins as a consequence of this "dilution effect" (Benbrook 2005).

From a crop management perspective, studies suggested that organic farming has a tendency towards nutritional superiority but lower yields compared to conventional agricultural (Benbrook 2005; 2008; Lairon 2010). Based on 236 case studies, Benbrook et al. (2008) reported that the organic agricultural products were nutritionally superior in 61% of the cases, while conventional products contained more nutrients in 37% of the cases. De Pascale et al. (2006) demonstrated that both the farming system and N rate affect carotenoid content and antioxidant activity of tomato fruits.

In this context, few studies showed the beneficial effects of compost with respect to the level of phytochemicals in crops (Table A13 in Annex II). Wang and Lin (2003) reported higher levels of several antioxidants when compost was used in pot experiments with dosages above 50% in volume. Coria-Cayupán et al. (2009) assessed a higher content (between 20–35%) of chlorophylls and carotenoids in lettuce grown with urban or fruit-and-vegetable waste composts, but lower contents or non-significant differences for phenol compounds and antiradical activity. Martínez-Blanco et al. (2011a) also found up to 76% and 24% increase in the content of sinapic acids and phenols, respectively, in cauliflowers when compost was applied to soil.

Accordingly, increasing nutrient contents are often observed in organic production and when compost is employed. However, general figures cannot be drawn, as the response varies depending on the compounds being assessed, the crop type, and the environment conditions (mainly soil and climate), while compost affects more than one compound at the same time.

### 7.3.10. Summary of the benefits of compost application

According to all the literature summarized in the Annex III and explained in the previous sections, an overview is provided in Table 7.1 for each of the benefits.

**Table 7.1. Summary of the potential benefits of compost application in the short-, mid- and long-term retrieved from the literature review.**

| Related section | Effect | Unit | Short-term (<1 yr) | | Mid-term (<10yr) | | Long-term (<100) | | Number of case studies reviewed | Extra information (Annex II) |
|---|---|---|---|---|---|---|---|---|---|---|
| | | | Min. | Max. | Min. | Max. | Min. | Max. | | |
| **Nutrient supply** | **N mineralized** | % of N applied | 5 | 22 | 40 | 50 | 20 | 60 | 5 RW ,1 CS | Table A6 |
| | **P mineralized** | % of P applied | 35 | 38 | 90 | 100 | 90 | 100 | | |
| | **K mineralized** | % of K applied | 75 | 80 | 100 | 100 | 100 | 100 | | |
| **Carbon sequestration** | **C sequestered in soil** | % of C applied | 40 | 53 | | 30 | 2 | 16 | 14 CS | Table A7 |
| **Weed, pest and disease suppression** | **Weed suppression** | - | ns | ns | - | - | - | - | 1 RW | - |
| | **Pest & disease suppression** | - | nad | nad | - | - | - | - | 1 RW | Figure A1 |
| **Crop yield** | **Crop yield gain** | % from min. Fert. | -495 | 0 | -71 | 52 | - | - | 77 CS | Table A8 |
| **Soil erosion** | **Soil erosion decrease** | % soil loss | - | - | -5 | -36 | - | - | 4 CS | Table A9 |
| | | Δ structural or aggregate stability (%) | 29 | 41 | 0 | 63 | - | - | 8 CS | |
| **Soil moisture content** | **Soil moisture increase** | WHC increase (%) | 0 | 50 | - | - | - | - | 21 CS | Table A10 |
| | | PAW increase (%) | 0 | 34 | - | - | - | - | 10 CS | |
| **Soil workability** | **Improved workability** | Δ soil bulk density (%) | -2.5 | -21 | -0.7 | -23. | | -20 | 21 CS | Table A11 |
| **Soil biological properties and biodiversity** | **Biodiversity increase** | Δ Microbial diversity (%) | - | - | - | - | -2 | 4 | 2 CS | Table A12 |
| | | Δ Microbial biomass (%) | 22 | 116 | 10 | 242 | 3.2 | 100 | 8 CS | |
| | | Δ Microbial activity (%) | 0 | 344 | - | 264 | 0 | 43 | 5 CS | |
| **Crop nutritional quality** | **Nutritional content increase** | - | nad | nad | - | - | - | - | 42 CS | Table A13 |

*WHC, water holding capacity; PAW, plant available water; ns, no significant differences; nad, no average data because of complexity of available dataset; "-", no reported effects; RW, review; CS, case study. Negative values indicate a decrease in the indicator.*

Effects in the mid- and long-term were only reported for nutrient supply, carbon sequestration, crop yield, soil erosion, and soil workability; whereas for the other four potential benefits only short-term data could be reported.

For three out of the nine benefits assessed – pest and disease suppression, biodiversity increase and nutritional content increase –, although they were proved, it was not feasible to summarize the effect in a unique data range. They are affecting more than one indicator at the same time and the intensity of the effect is different for each one – e.g. compost application is inducing and increase of a large number of nutritional compounds while non-significant effects are being detected for other compounds.

**Table 7.2. Main factors reported in the literature affecting the potential benefits of compost application.**

| | | Potential benefits | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | **Nutrient supply** | **Carbon sequestration** | **Weed, pest and disease suppression** | **Crop yield** | **Soil erosion** | **Soil moisture content** | **Soil workability** | **Soil biological properties and biodiversity** | **Crop nutritional quality** |
| **Composting process** | **Raw material quality** | x | | | | | | | | |
| | **Composition/heterogeneity** | x | | x | | | | | | |
| | **Controlled conditions** | x | | x | | | | | | |
| **Compost characteristics** | **Maturity** | x | x | x | x | | | | | |
| | **C/N ratio** | x | | | | | | | | |
| | **Microbial diversity** | | | x | | | | | | |
| | **Heavy metals content** | | | | x | | | | x | |
| **Compost application** | **Dosages** | x | x | x | x | x | x | x | | |
| | **Gap application-cultivation** | x | | x | x | | | | | |
| | **Application method** | | | | x | | | | | |
| **Site conditions** | **Climatic conditions** | x | x | | | x | | | | |
| | **Soil nutrient pool** | x | | | x | | | | | |
| | **Type of soil** | x | | | | | x | | | |
| | **Saturation level of C** | | x | | | | | | | |
| **Crop management** | **Crop/rotation** | x | | x | x | | | | | x |
| | **Mechanical aeration** | x | | | | | | | | |
| | **Post-harvest management** | | | | | | | | | x |

Significant effects due to compost application were averaged for all the potential benefits, apart from weed suppression. Besides, for three of the assessed effects the share of studies with non-significant results was relevant: for crop yield, more than

60% of the case studies did not detect differences when compost was applied; non-significant effects were detected for soil moisture content for low rates of compost; and finally, crop nutritional quality was not relevantly different for a third of the case studies included. These were also the benefits with higher disparity in the measured effects among the results.

Besides the broadly used analytical parameters – such as nitrogen content, moisture, heavy metals, etc. –, other factors related to raw organic materials, composting process, site conditions, crop management, and compost maturity have been pointed out as relevant variables to explain the measured effects. Table 7.2 is detailing the factors reported by the literature to affect each one of the potential benefits. The maturity of the applied compost is described as a relevant factor to understand compost effects on soil and plants, although it is not a commonly analysed parameter. The dosages of compost applied as well as the existence of a lag of time between compost application and crop sowing, are also key to explain the strength of the observed effects.

# 7.4. Quantification and impact assessment

In the following sections, benefits from compost application are quantitatively described within an LCA perspective. Firstly, the consequential modeling for each benefit is briefly described, followed by quantification of the substituted or saved process based on up-to-date knowledge and data (reported in Table 7.1). Finally we address the impact categories in existing impact assessment methodologies which are most affected when considering compost application (summarized in Table 7.3). Other methodological issues are address in the Discussion section.

## 7.4.1. Nutrient supply

In an LCA context, supply of nutrients with compost substitutes for the use of mineral fertilizers, whose industrial production and transport is thus avoided. The amount of substituted fertilizers depends on the content of nutrients of the compost and their utilization rate. The substitution is often regulated by legislation (Hansen et al. 2006a). Boldrin et al. (2009) reported that final utilization efficiencies for N, P and K are in the order of 20–60% for N, 90–100% for P, and 100% for K. Using the nutrient contents presented in section 7.3.1, the potential amount of inorganic fertilizers replaced may be within the range of 1–13 kg of N, 1–5 kg of P, and 5–14 kg of K per ton of compost applied.

A Life Cycle Inventory (LCI) for fertilizer production include use of materials and energy, and emissions to different compartments, which would typically result in potential impacts on resource depletion, global warming, human- and eco-toxicity, and eutrophication categories. Datasets for N-P-K fertilizers are reported by different sources, as for example Wind and Wallender (1997), Hansen et al. (2006a) based on Patyk and Reinhardt (1997) and Audsley et al. (2003), and Ecoinvent based mainly on Davis and Haglund (1999).

**Table 7.3. Midpoint impact LCA categories involved in the evaluation of the potential benefits of compost use-on-land.**

| Related section | Effect | Impact categories | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | Currently accepted | | | | | | | | | | Under-development | | | |
| | | Acidification | Aquatic eutrophication | Global warming | Ecotoxicity | Energy consumption | Ozone depletion | Human toxicity | Ozone formation | Resource consumption | Terrest. eutrophication | P resource | Biodiversity | Land use & soil quality | Water use and quality |
| Nutrient supply | N mineralized | x | x | x | x | x | x | x | x | x | x | | | | |
| | P mineralized | x | x | x | x | x | x | x | x | x | x | | | | |
| | K mineralized | x | x | x | x | x | x | x | x | x | x | | | | |
| Carbon sequestration | C sequestered in soil | | | x | | | | | | | | | | | |
| Weed, pest and disease suppression | Weed suppression | | | x | x | x | | x | | x | | | | | |
| | Pest & disease suppression | | | x | x | x | | x | | x | | | | | |
| Crop yield | Yield increase/decrease [a] | | | | | | | | | | | | | | |
| Soil erosion | Prevention of soil erosion | | | | | | | | | (x) | | x | | x | |
| Soil moisture content | WHC increase | | | x | | x | | | | | | | | | x |
| | PAW increase | | | x | | x | | | | | | | | | x |
| Soil workability | Improved workability | | | x | | x | | | | x | | | | | |
| Soil biological properties and biodiversity | Biodiversity increase | | | x | x | | | x | | x | | | x | | |
| Crop nutritional quality | Nutritional content increase [a] | | | | | | | | | | | | | | |

[a] *It has to be included in the LCA using the adequate functional unit.*

Furthermore, compost is considered as an effective option for phosphorous recycling (Cordell et al. 2009), which is a growing issue as a consequence of the foreseen shortage of mineral P for agriculture fertilization (Syers et al. 2008). The use of a renewable P source rather than inorganic non-renewable supply is of great importance and this reduced raw resource consumption might be quantified during impact quantification.

### 7.4.2. Carbon sequestration

Sequestration of C into soil can be seen as removal of C from atmosphere. Thus, once the amount of C sequestered is estimated, the value can be translated into saved $CO_2$ emissions (i.e. using a conversion factor of 44/12, based on molar relation) which can be entered in the LCI. As previously mentioned, the time-horizon used in the assessment plays a crucial role when estimating the benefit from carbon sequestration. A time frame of 100 years is considered to be relevant for estimating contributions to global warming (Favoino and Hogg 2008). Boldrin et al. (2009) reported that the benefits from C retained in soil 100 years after the addition of bio-waste compost is between 2 and 79 kg $CO_2$-eq. $t^{-1}$. Higher values, 279 kg $CO_2$ eq. $t^{-1}$, were reported by ICF (2005). Most likely, this large variability is due to the synergetic effect of the different abovementioned environmental and site-specific factors, meaning that estimations should be done on a case-to-case basis.

### 7.4.3. Weed, pest, disease suppression

When the application of compost results in suppressive effect of weed, pest and diseases, the use of pesticides can be reduced or avoided. The avoided use can consequently be credited to the system as an environmental saving. The studies reviewed indicate that use of compost might increase, in most cases, the health of crops through resistance towards certain diseases, above all those related to fungi pathogens. However, these benefits are so case-specific that it is not possible to provide any general figures, regarding both the amount and the type of pesticides saved. For example, ROU (2007) estimated that 2–3 L per ha of 360 g $L^{-1}$ glyphosate could be saved when using garden waste compost as mulch, while no benefits were estimated if compost was used as soil conditioner.

When pesticides are not used, environmental benefits are related to both the avoided production/transportation and to the avoided use of the pesticides. Inventory data covering transport and production of pesticides can be found in different databases such as Ecoinvent, PestLCI and GEMIS. Assessment of the environmental effects induced from pesticide utilization requires the use of exposure-fate-toxicological models. In fact, on one hand pesticides are purposefully emitted to the environment to control soil and air-borne plant pests, and diseases, on the other hand they can also reach non-targeted life forms via wind drift, evaporation, leaching, and surface run-off, with consequent potential toxic effects on human or environment (Brentrup 2004; Juraske et al. 2009; De Bertoldi 2010).

The impact of chemicals on the environment depends on the amount of the active ingredient applied, the site of application, the crop stage development, the pesticides partitioning and degradation rates on the various environmental compartments, and the toxicity of the specific pesticide to the species present in these compartments (Antón et al. 2004). Unfortunately, dynamic and realistic models capable of predicting chemical fate taking into account the abovementioned factors are scarce (Birkved and Hauschild 2006) and require exhaustive information of the pesticide's characteristics and application conditions, which in many cases are unknown (Antón et al. 2004). An available model is PestLCI (Birkved and Hauschild 2006), developed for inventorying pesticide emissions to the different environmental compartments, and based on information which will normally be available to the model user. The model was firstly built for Danish conditions, but is adaptable to other regions of the world. In particular, PestLCI allows estimating pesticide emissions leaving the site under assessment and causing unwanted environmental effect, according to the ILCD Handbook (JRC and IES 2010b) recommendations.

Potential environment impacts from production/transportation and use of pesticides can be assessed using existing impact categories. The most relevant impact categories related to pesticides are the Toxicity categories, both Human- and Ecotoxicity.

### 7.4.4. Crop yield

Increased yield as a consequence of compost application results in avoided agricultural production and thus the involved burdens. The increase should be measured relative to the yield otherwise obtained by using the amount of fertilizer already substituted. Otherwise, the effect of the compost application is counted twice. From a consequential LCA point of view, this can have different consequences at a system level, depending on existing agricultural constraints. If arable land in a certain area is not constrained, the benefit is linked to avoided use of material and energy needed for the crop production. In the most likely regime of constrained arable land, the increased yield would have an effect on both intensification and expansion of agricultural production, and ultimately will prevent indirect land use changes (ILUCs), which are for instance a major source of GHG emissions throughout the life cycle of biofuels (Thamsiriroj and Murphy 2010).

The LCA modeling of increased or decreased agricultural productions is typically case-specific. Market driven mechanisms need to be taken into consideration when identifying both the specific crop directly affected by the compost-induced increased yield and the ILUC effects. For both effects, production inventories can then be used to credit the system for the avoided productions. Depending on the specific area and crop, most of the impact categories are influenced when agricultural production is involved.

### 7.4.5. Soil erosion

As mentioned earlier, the application of compost could prevent soil erosion and thereby avoid losses of arable land. Probably due to the absence of data, the reduction in soil erosion was not taken into account in previous LCA of use-on-land of compost. As shown in Table 7.1, losses of soil could decrease between 4 and 66% with the application of compost, depending on the time horizon considered. A more precise quantification is possible for specific conditions taking into account for instance climate, application rate, and type of soil.

Two approaches are possible during inventory modeling. One option is to model the avoided losses within traditional LCA impact categories. Here the consequential modeling should identify the agricultural production affected by the losses of arable land. Assuming a constrained agricultural production at a system level, the modeling is then done similarly to "crop yield" section, meaning that the consequences of intensification and/or land expansion are included in the assessment. The second option is to consider soil as a resource, thus either including loss of soil in the inventory as 'resource depletion' (Cowell and Clift 2000) or introducing 'soil erosion' as an independent endpoint impact category (Buratti and Fantozzi 2010). This issue is further elaborated in the discussion session.

### 7.4.6. Soil moisture content

Real water necessities of the crop are not affected by compost application; the potential benefit of compost is to raise the capacity of soil to retain rainfall and irrigation water (green water, i.e. rainwater stored in the soil as soil moisture) in order to reduce irrigation water consumption (blue water, i.e. water from surface and groundwater resources). This may result in two distinguished consequences: on one hand, blue water is saved; on the other hand, because more green water is available, crop yield could increase in those areas where irrigation water is not available. However, there is not a direct relation between the retention capacity and the amount of water saved, depending on site conditions, water demands of the plants, management practices, and previous moisture content of soils. The theoretical saving of irrigation water could be calculated for a particular case study, if all these data are available, as done for example in ROU (2007). Alternatively, results from experiment trials could be used. The only study on this issue was carried out by Ngoundo et al. (2007), who reported that, in both tropical humid and semi-arid regions, compost application was a reliable way of saving water. Authors concluded that compost application on water-intensive crops increased the net irrigation depth, thereby reducing the need for irrigation.

Environmental burdens from irrigation water supply are linked to water extraction, transport, and distribution in the field (electricity, pumps, pipes, etc.), and are found in several inventories, also at a regional level. Potential impacts from these processes are typically those related to energy supply and consumption, thus global warming, acidification, and eutrophication. There is indeed a growing consensus on the fact that water should be considered a resource, and thus its consumption included in LCA as Resource consumption (or depletion) (Heuvelmans et al. 2004; Berger and Finkbeiner 2010; Núñez et al. Accepted 2012). This issue is further elaborated in the discussion section.

### 7.4.7. Soil workability

Improved soil workability can potentially decrease energy requirements for agricultural operations (ROU 2007; Favoino and Hogg 2008). Soil ploughing typically involves large consumption of energy and fuel, being one of the farm operations with most environmental impact (Lal 2004b). Hillier et al. (2009) in a survey carried out in 57 farms in Scotland (UK), estimated that the carbon footprint of ploughing operations was on average 15.2 kg $CO_2$ eq. $ha^{-1}$ (corresponding to approx. 5.7 $L_{diesel}$ $ha^{-1}$). Lal (2004b) carried out an extensive literature review and reported that estimates of emissions for different tillage operations were between 2–20 kg $CO_2$ eq. $ha^{-1}$ (i.e. 0.75–7.5 $L_{diesel}$ $ha^{-1}$).

In an 8-year long study, McLaughlin et al. (2002) reported that, under a 100 t $ha^{-1}$ application of stockpiled and rotted manure on a corn field, the plough draft was reduced 27–38%, resulting in 13–18% reduction of fuel consumption. The fuel consumption accompanying the organic amendments was attributed to an improvement in soil tilt and quality related to the increase in SOM. No other studies were found linking compost application and fuel consumption for agricultural operations, meaning that more comprehensive data are needed to be able to relate, for example, fuel consumption with soil bulk density.

From an LCI point of view, reduced fuel consumptions can be credited to the system as avoided use of diesel. Inventory data for diesel consumption, including provision, are available in several databases and published report. Avoided diesel consumption mainly affect global warming impact category, while avoided emissions of nitrogen oxide could also have an influence on acidification and eutrophication.

### 7.4.8. Soil biological properties and biodiversity

Changes in soil biodiversity after compost addition might influence either positively or negatively the services "delivered" by the ecosystem, with consequences in terms of impacts associated to the substitution or compensation of those ecosystem services. Soil organisms are the basis of several ecosystem services and regulate soil processes such as, for instance, soil hydrological processes, nutrient cycling, and the incidence of pests, parasites and diseases (Brussaard et al. 2007).

In LCA terms, alterations in the system service in connection to biodiversity changes could be modelled within the traditional categories if those changes could be quantified in the inventory. If, for example, increased biodiversity could be related to better nutrient cycling and lower need for fertilization, then the benefits from increased biodiversity could be modelled in term of reduced production of fertilizers. However, data linking compost use, biodiversity and ecosystem services are non-existing – apart from a first attempt of establishing a preliminary relation by Nemecek et al. (2011) –, meaning that such quantification is currently not feasible. In addition, general figures could not be established in all case, as the effects of land management practices are highly variable depending on regional and scale dependent factors (Bengtsson et al. 2005). An alternative approach is to consider biodiversity and ecosystem services as independent endpoint categories when assessing the environmental impacts of land management alternatives (Zhang et al. 2010). This issue is further elaborated in the discussion section.

### 7.4.9. Crop nutritional quality

The differences in the nutrient content of crops do not directly affect resources consumption or emissions per ha or t yielded. Nevertheless, different nutrient contents in products can have a repercussion on the LCA modeling depending on how the functional unit is defined. If the functional unit is based on yield (i.e. mass, volume, surface) no consequences on Resource consumption or emissions need to be modelled in the inventory. However, when the functional unit includes qualitative aspects (e.g. nutritional and/or economic value), increased nutritional level of a food product may have as a consequence that lower amounts are needed. In general terms, including qualitative aspects in the functional unit (e.g. in Marshall et al. (2001) and Martínez-Blanco et al. (2011a)), would have an effect on the agricultural production, which could be modelled similarly to what is described under the "crop yield" section.

# 7.5. Discussion

Regarding the proof of the effects and the environmental assessment – including quantification and characterization – of the benefits of compost application to soils, four different scenarios were identified (Table 7.4):

- Scenario 1: The positive effects of compost application are proved, effects are quantifiable, and there are tools for their consideration with LCA. This includes nutrient supply and carbon sequestration which are (and should be) included in LCA studies.
- Scenario 2: The benefits are proved, but their magnitude is too variable as a consequence of the synergetic effect of many factors. Thus, inventory data cannot be unambiguously quantified. Impact categories and characterization factors exist for most of the benefits.
- Scenario 3: The benefits are proved and quantifiable. However, corresponding characterization factors and/or impact categories are non-existing.
- Scenario 4: Benefits are not fully proved and thus their inclusion in the modeling is not yet feasible.

Therefore, from all the nine benefits proposed, only two are proved and the quantification and assessment are possible, while different research efforts are required in the rest of the benefits for a full assessment. In the following sections, open issues related to quantification and characterization are discussed.

**Table 7.4. Overview of potential benefits derived from the use-on-land of compost in relation to their LCA modeling.**

| Scenario | Potential benefit | Proved | Quantification | Characterization |
|---|---|---|---|---|
| 1 | **Nutrient supply**<br>**Carbon sequestration** | ✓ | ✓ | ✓ |
| 2 | **Pest and disease suppression**<br>**Soil workability**<br>**Biodiversity**<br>**Crop nutritional quality**<br>**Crop yield** | ✓ | X | (✓) |
| 3 | **Soil erosion**<br>**Soil moisture content** | ✓ | ✓ | X |
| 4 | **Weed suppression** | X | X | ✓ |

*✓, done; (✓), done with some difficulties; X, non-sufficiently developed or non-existing.*

## 7.5.1. Quantification: improved modeling

Review of the existing literature, shows that in the future, modeling of effects from compost application should be improved in some areas in order to adequately assess its benefits.

First, LCA models are typically linear steady-state models of physical flows (Guinée et al. 2002). Fluxes of nutrients and pollutants after compost application to soil are not linear in most of the cases. This also applies, for instance, to repeated applications of compost: LCA studies typically look at the effects of a single application over 100 years, while the cumulative effects on several applications may not be linear with the amount of compost added.

Second, LCA models assume that impacts depend on the compost characteristics while they rarely include environmental parameters as determining factors. This links with the necessity of coupling LCA and agronomic models to gain a more precise picture.

Third, the amount of plant nutrients contained in compost is normally modelled for as a benefit. However, the use of compost could in some cases result in excessive application of P and K with respect to N, which is usually used for compost dosage calculation. The subsequent release of this unbalanced amount of nutrients could result in impacts to the environment and should be thus included in the LCA modeling through a more thorough mass balancing of the nutrients. To avoid the negative environmental impacts and the resources losses, compost doses should be better calculated based on crop P and K necessities and to add nitrogen in other forms.

Fourth, for LCA of the agricultural sector, the functional unit is typically defined per area used or product yield. The purpose of the functional unit is to provide a reference to which the inputs and outputs are related, and to ensure comparability of LCA results (ISO 2006a). As different functional units can lead to different results for the same product system (Marshall 2001; Martínez-Blanco et al. 2011a; Deytieux et al. 2012), there may be a need for a qualitatively more precise definition when dealing with compost application, especially in those cases where the product quality is affected. Better definitions could, for example, include the economic value (e.g. in Hayashi et al. (2006), Mourad et al. (2007), Reap et al. (2008a), and Schau and Fet (2008)) or the nutritional content of a product (e.g. in Hayashi et al. (2006) and Schau and Fet (2008)). A more accurate definition dealing with nutritional differences may include a combination of quality (nutritional quality) and quantity (yield). This was for example done by Charles et al. (1998) and Audsley et al. (2003), where the functional unit was defined as "1 equivalent ton grain with 12–13% protein". This involved the use of marginal productions to adjust the overall output of the system under assessment. However, identifying reference or recommended nutrient content for most crops may be a controversial issue.

Finally, the choice of the time horizon of the LCA should be harmonized. The studies reviewed showed in fact that such choice is in many cases very important, as both the foreground and background effects of compost application vary largely depending on the time frame.

### 7.5.2. Characterization: additional impact categories and proposed modifications

As highlighted in the text, when applying LCA for assessing use-on-land of compost, available impact assessment methodologies may not properly deal with some issues, and thus new impact categories or modifications of the current ones may be needed in the future to allow for a more holistic assessment. The proposals should deal with depletion of P resources, biodiversity, loss of arable soil, and consumption of water.

Depletion of P as a resource is currently modelled similarly to other natural resources. However, most metals, chemicals, even petroleum, could be replaced by other resources in much of their functions, while this is not the case of phosphate. A revision of the characterization factors is thus needed for the assessment of non-replaceable non-renewable resources such as P. In this respect, the ReCiPe model adds special value to resources. It is based on the geological distribution of mineral and fossil resources, and assesses how the use of these resources causes marginal changes in the efforts to extract future resources (Goedkoop and Spriensma 2000).

Assessment of the impacts of land management on biodiversity can be based on particularly endangered species (Nemecek et al. 2011). Other authors estimated the changes in biodiversity through the assessment of one relevant indicator group. For example, Weidema and Lindeijer (2001) developed an indicator that includes species richness of vascular plants, inherent ecosystem scarcity, and ecosystem vulnerability. Similarly, the Eco-Indicator 99 measures ecosystem quality based on the potential disappeared fraction of vascular plants (Goedkoop and Spriensma 2000). Other methods estimated the change in the overall number of species per year (Suer and Andersson-Sköld 2011) or the change in the number of species in the affected area as compared to the regional level (Köllner 2002). Besides being considered as independent endpoint categories, other authors have included biodiversity and ecosystem services as midpoint categories within the endpoint category Land use (Udo de Haes 2006; Milà i Canals et al. 2007). As different options exist, a harmonization may be needed in order to develop a consensus methodology.

Soil loss involves the loss of cultivable land but also the loss of SOC, plant nutrients, as well as the associated plant, animal and microbial biodiversity (Cowell and Clift 2000). Loss of soil can thus be included in some of the abovementioned impact categories. However, for a more comprehensive assessment the loss of soil mass could be considered as the loss of a resource and included in the inventory as 'resource depletion' (Cowell and Clift 2000; Núñez et al. submitted 2011). The amount of soil loss could, for example, be first normalized against the available soil reserves (Cowell and Clift 2000; Núñez et al. submitted 2011) or against the estimated soil eroded in the region of study within a specific time frame (Buratti and Fantozzi 2010). Further weighting can be based on the exergy use for the backup technology needed to maintain the soil productivity (Núñez et al. submitted 2011), or by relating the amount of soil loss to reference values of soil erosion where no significant decline in soil productivity is observed (Buratti and Fantozzi 2010). In alternative, soil erosion can be included within the impact category Land use, whose characterization factors are based on soil quality indicators such as SOM, structure, heavy metals, biodiversity, aesthetic value, etc. (Mattsson et al. 2000; Brentrup 2004).

Depletion of water resources is gradually gaining importance, particularly in certain geographical regions. There is currently only a preliminary scientific consensus about the parameters to consider and the methodology to follow (Núñez et al. Accepted 2012). Methodological issues concerning impact assessment methods include the types of water use accounted for, the inclusion of local water scarcity conditions, and the differentiation between watercourses and quality aspects (Berger and Finkbeiner 2010). While available inventories account for the total volume of water used within the life cycle of products (Hoekstra et al. 2009), at the impact assessment level, more emphasis is given to the blue water consumption (i.e., consumption from surface and groundwater resources). However, from an environmental point of view interest of green water consumption by crops is also important because of its influence in ecosystems (Berger and Finkbeiner 2010).

## 7.6. Conclusions

Use of bio-waste compost on land can have beneficial effects on the plant-soil system. Most of these benefits have been so far excluded from LCA studies, mainly because of scarcity of data or lack of appropriate impact assessment methods. In this study, a literature search was carried out in which nine benefits of compost application were identified and studied its influence and consequences on potential environmental problems. Availability and quality of the data for quantification differed largely among the assessed benefits, with no data or large variability in the observed benefits. Data concerning long-term effects of compost, which are relevant for LCA purposes, were particularly scarce. Therefore there is a need for more long-term studies or estimations.

When data were available, local conditions and ecosystem complexity were the main obstacles for a precise quantification. Furthermore, most of the studies consulted used different compost doses – annually or a unique application – and for different periods of time, thereby increasing the difficulty of quantifying the benefits.

A discussion on the suitability of currently available impact assessment methodologies indicated that additional impact categories may be needed for a comprehensive assessment of compost application. For two of the nine benefits – nutrient supply and carbon sequestration – the review showed that both quantification and impact assessment of the effects could be performed, meaning that these two benefits should be regularly included in LCA studies. For four of the nine benefits – increase in crop yield, soil workability, crop nutritional quality, and enhancement of soil biological properties and biodiversity –, quantitative figures could not be provided, either because of complete lack of data or because the effects are both very variable and too depending on specific local conditions. For "soil erosion" and "soil water content" effects could be quantitatively addressed, but available impact assessment methodologies were considered unsuitable to comprehensively evaluate the implication of compost application with regards to these two benefits. Finally, based on the available literature, "suppressive effects of compost on weed, pests, and diseases" could not be generally proved.

# Part IV

## Consumer, territorial and sustainability perspectives

# Chapter 8. Carbon Footprinting and Life Cycle Assessment for greenhouse gas impact quantification in horticulture

**Chapter 8** is based on the following paper:

Martínez-Blanco, J., Rieradevall, J., Muñoz, P., Pascual, P., Oliver-Solà, J., Gabarrell, X., & Gasol, C.M. (2012) Carbon Footprinting and Life Cycle Assessment for greenhouse gas impact quantification in horticulture. Under publication in the International Journal of Life Cycle Assessment.

# Abstract

***Aim, Scope and Background***. Evaluating the environmental behavior of products has become an essential issue, not only to fight against global warming and other threats but also because productive sectors (industry and services), consumers and public administrations demand this information. Life cycle thinking is one of the better approaches for assessing the environmental performance of products (services or processes). Although Life Cycle Assessment (LCA) is one of the most used methodologies, Carbon Footprinting (CF) is currently gaining strength. Our aim was to detect the repercussions on the quantification of greenhouse gas (GHG) emissions when LCA or CF is used. Several case studies for horticultural systems are compared in order to identify the critical issues in the choice of methodology.

***Method***. In this paper, we used LCA and to quantify GHG emissions from production of tomato and cauliflower in the Mediterranean, considering four types of cultivation. These options combine two fertilizing treatments and greenhouse covered or open fields. The ISO 14044 and the PAS 2050:2008 were used as the guide specifications for LCA and CF, respectively.

***Results and discussion***. The most important difference between the assessments is that CF excludes GHG emissions from the production of capital goods and from changes in the soil carbon sink. This may be an adequate decision in those systems with a low infrastructure for production, such as open field options in the case study, with differences on in the order of 5-10%. However, the excluded GHG emissions may be up to 26% of the total when a higher level of infrastructures is considered, as in our results for the greenhouse tomato options.

***Conclusions and perspectives***. The reported differences lead to underestimating the environmental effects communicated to consumers when CF is used. Due to the variability of production processes in the agri-food sector and the different use of capital goods, we strongly recommend including them when agri-food systems are studied.

## Keywords

PAS 2050:2008; Greenhouse gas emissions; Agri-food sector; Horticulture; Global warming; Capital goods.

# 8.1. Introduction

In the last century, production techniques in the agri-food sector, and particularly horticulture, have intensified. This trend is a response to the growing demand of an increasing population, technological changes, year-round production, human diet modifications and oil dependence, as well as due to economic interests (Smith et al. 2007; Heinberg and Bomford 2009). Consequently, food production has become a relevant contributor to the depletion of natural resources and climate change (Nonhebel 2004; Tukker et al. 2005). Smith et al. (2007) reported that agriculture currently accounts for 10-12% of the total global anthropogenic greenhouse gas (GHG) emissions generated worldwide.

This rise in GHG emissions has had a major impact on global warming, one of the major environmental issues today for all policy makers (Brown 2003; IPCC 2007; Weidema et al. 2008; Christensen et al. 2009; De Koning et al. 2010; Scipioni et al. 2010).

The global anthropogenic GHG emissions (measured in kilograms of carbon dioxide equivalent, kg $CO_2$ eq.) were 24% higher in 2004 compared to 1990 and 70% higher than in 1970 (IPCC, 2007). It is, therefore, of major interest to quantify the environmental behavior of products, processes and services, and particularly their carbon dioxide equivalent emissions. Of the many tools for monitoring and managing GHG emissions, the scientific community considers life cycle thinking the most appropriate for assessing environmental impacts.

Life Cycle Assessment (LCA) has been the more widely used in research for analyzing products, processes and services in general (ISO 2006b), including the agriculture sector (Audsley et al. 2003; Guinée et al. 2006; Gasol et al. 2009; Muñoz et al. 2010; Scipioni et al. 2010; Nemecek et al. 2011; Nemecek et al. 2011). Impact categories affecting several compartments – from eutrophication to global warming – may be considered in order to assess the potential damage to the environment and to human wellbeing (Pennington et al., 2004). LCA has also been promoted by different European directives for specific sectors and products as a suitable environmental quantitative tool (European Parliament2000; 2002; 2004; 2005). However, nowadays Carbon Footprinting (CF) methodology, which is also based on the life cycle approach, is gaining strength in the field of environmental assessment, mainly within the private sector. As pointed out by Scipioni et al. (2010), apart from the environmental benefits, these tools can lead to a reduction of production costs, competitive advantages and technological leadership, all highly valued by enterprises and companies.

In contrast with LCA, CF methodology is confined to the analysis of GHG emissions (2008a; BSI 2008b). This indicator is calculated in a similar way to the global warming potential (GWP) used in LCA (IPCC 2007; Weidema et al. 2008). It has to be calculated through one of the current, at least 16, specifications from private (e.g. PAS 2050:2008, from the British Standards Institution, or the Bilan Carbone method, from ADEME) or government initiatives, such as the New Zealand Ministry of Agriculture and Forestry, the Korean Product Based Reduction Scheme and the Japanese Ministry of Economy (Hospido et al. 2009; Brenton et al. 2010; De Koning et al. 2010). The UNEP/SETAC Life Cycle Initiative, the World Business Council for Sustainable Development and the World Resources Institute (WRI and WBCSD 2011) are examples of international associations that have also drawn up methods and guidelines to calculate $CO_2$ equivalent emissions or obtain the carbon footprint of products, processes and services (Finkbeiner 2009). Meanwhile, the International Standards Organisation (ISO) is developing the ISO 14067 for Carbon Footprinting, which it plans to release in the near future, for reliable specifications agreed at an international level. This standard will provide requirements for quantifying and communicating GHG emissions associated with products, processes and services.

As reported by Weidema et al. (2008), CF has been driven by retail chains, proactive companies and nongovernmental organizations. Johnson (2008) and Weidema et al. (2008) pointed out several reasons for its popularity. Firstly, there is the spread of online calculators to estimate 'personal footprints' for laypeople. Secondly, the analysis is limited to emissions that have effects on climate change, which makes the study easier, quicker and cheaper, whereas LCA is often complicated, difficult to communicate and it is frequently difficult to make clear-cut decisions based on the results. Finally, the CF results can easily be presented as an eco-label (ISO 2000a).

However, for experts working with detailed LCA, focusing on GWP alone is a crude approach that may often give a misleading picture compared to using multiple-indicators (Frischknecht et al. 2007; Weidema et al. 2008; Finkbeiner 2009).

As aforementioned, CF results can be presented to consumers as an eco-label, following the ISO 14020 series (ISO 2000a), as with the Carbon Reduction Label developed by the Carbon Trust. Companies have realized that consumers will buy products, or avoid their purchase, partially based on environmental considerations (Lampe and Gazda 1995; De Koning et al. 2010). Eco-labels do enable the consumer to influence how products are made (Rex and Baumann 2007) and authorities to legislate and oblige companies to internalize the environmental costs in the product prices (Weidema et al. 2008; De Koning et al. 2010).

For these reasons, identifying the critical points of a product by means of LCA or CF has developed into a major discussion. Accordingly, in this paper, we evaluate the main differences between LCA and CF results for the Mediterranean agri-food sector focusing on three main issues: crop factor (tomato and cauliflower), fertilizing factor (organic and inorganic types) and, finally, technological factor (open field and greenhouse cultivation). Tomato is the major crop in terms of production in the EU-27 (around 14 million tonnes produced per year), particularly in Mediterranean countries; Spain is the second main producer, contributing 27% to European tomato production. Tomato is mainly produced in Mediterranean countries, and cauliflower in most European countries, although less than tomato, being one of the most common winter crops consumed in Europe.

The environmental performance of tomato and cauliflower crops have previously been assessed from different points of view: the impacts of distribution systems (Roy et al. 2008); nitrogen leaching related to different fertilization options (Cavero et al. 1999; Muñoz et al. 2008c); greenhouse production considering several options for waste management (Antón et al. 2005a); the environmental performance of several cultivation options (Martínez-Blanco et al. 2011b); and the use of several functional units for cauliflower impact assessment (Martínez-Blanco et al. 2011a). We based our evaluation on the two latter studies because they provide data from six case studies combining several crops, fertilizing alternatives and technologies, key issues in horticultural production (Hillier et al. 2009).

## 8.2. Environmental tools

This section describes the methodologies used for LCA and CF quantifications, focusing on the main differences between them.

### 8.2.1. Life Cycle Assessment (ISO 14044)

LCA involves the evaluation of a system to determine its environmental impacts. It is based on the ISO 14044 series and is divided into well-known steps: goal and scope, inventory assessment, impact assessment and interpretation of results (ISO 2006b).

The impact assessment was made according to the CML 2001 baseline method (Guinée 2001) which elaborates the problem-oriented (midpoint) approach to give a list of impact categories that have to be assessed (ISO 2006b). Taking into account the main aim of this paper, only the GWP indicator was calculated for LCA and compared with that of CF.

### 8.2.2. Carbon Footprinting (PAS 2050:2008)

The Publicly Available Specification 2050:2008 for assessment of GHG emissions, developed by the British Standards Institution (BSI 2008a) and co-sponsored by the Carbon Trust and the Department for Environment, Food and Rural Affairs (DEFRA), was used. It has been widely implemented on the market, is readily available and is also considered the base guidelines for the ISO 14067.

However, because much effort has been internationally made for CF developing and agreement, during the publication process of the paper, relevant events occurred. The new PAS 2050:2011 (BSI 2011) was launched, superseding PAS 2050:2008. Additionally, the Greenhouse Gas Protocol (GHG Protocol) – a decade-long partnership between the World Resources Institute and the World Business Council for Sustainable Development – made available the GHG Protocol Product Life Cycle Standard (WRI and WBCSD 2011).

PAS 2050:2008 methodology has been used by Walkers Crisps (Walkers Snacks 2010), Silver Spoon sugar (Silver Spoon 2010) and Tesco supermarkets (TESCO 2011) to calculate the CF of at least 20 products including potatoes, washing detergents and orange juice (Carbon Trust 2008; Climatedec 2010).

According to the BSI (2008b), there are four steps for calculating the CF. The first is to build a process map to identify all materials, activities and processes which contribute to the life cycle of the product. The following step is to define the scope and perform a high-level footprint calculation to identify non-relevant contributions. Any single source resulting in less than 1% of total emissions can be excluded from the analysis, with a maximum of 5% of the full product CF omitted. The third step is to collect data on materials quantity, activities and emission factors throughout the life cycle for a more detailed CF assessment. Here we used the data of Martínez-Blanco et al. (2011a; 2011b) for this step. The last step is to calculate the CF, which requires a mass balance to ensure all input, output and waste streams are accounted for. There is a further optional step where technical uncertainty is checked in order to improve confidence in the results and any decisions made that are based on them.

The business-to-business (B2B) approach was chosen as this has the same boundaries as the LCA (Martínez-Blanco et al. 2011a; 2011b). For this approach, CF ends when the product is delivered to another business, consistent with the "cradle-to-gate" approach of the ISO 14044.

### 8.2.3. Methodological differences in the system boundaries

Reviewing the ISO 14044 and PAS 2050:2008 specifications for GHG emissions accounting, several differences and similarities were identified regarding system boundaries for agricultural systems.

Neither capital goods (such as machinery, equipment and buildings), carbon storage in vegetation and soils nor workers/consumers transport are taken into account when PAS 2050:2008 is used, while the ISO 14044 leaves the decision to the LCA practitioners. Nevertheless, other CF methodologies, such as the GHG Protocol initiative (WRI and WBCSD 2011), are not leaving capital goods out of the scope of study.

Regarding land use change emissions, those due to changes in soil use after 1990 have to be included with PAS 2050:2008, but ISO 14044 is not specific about this issue. Both methodology specifications recommend not considering human or animal energy and outside services.

## 8.3. Case study

Both LCA and CF have been applied to tomato and cauliflower production in Mediterranean fields. To compare the methodologies, we based our evaluation on the LCA data of Martínez-Blanco et al. (2011a; 2011b), where a wider explanation of the assumptions and the data can be found, and used the data from their case study for the inventory for the CF calculation.

The experimental field has Typic Xerothent soil. The cultivation followed the best available techniques for integrated crop management guidelines aiming to compare efficient systems in resources, energy and emissions. Crops were irrigated (Table 8.1) depending on the evapotranspiration. The doses of fertilizers were decided by taking into account the previous nutrient content of soil and the real agricultural necessities. The high nitrogen content of the irrigation water, which is the result of the excessive use of mineral fertilizers in the region, was included in the inventory as a mineral fertilizer addition, considering its production and transport. The experiment had a block design with three replicates of 35 $m^2$ per crop. Commercial and non-commercial yields were determined per block and per cultivation option during harvest time. The nitrogen applied and the uptake were considered for the emissions calculation.

Six case studies were assessed and compared, two for cauliflower and four for tomato (Table 8.1). The cultivation options considered were two fertilization scenarios (as described in Martínez-Blanco et al. (2011a; 2011b)), with or without greenhouse protection (for tomato only).

The importance of comparing two fertilizer management options is that fertilizer production is one of the processes with a major contribution to the total impact in horticultural production (Muñoz et al. 2008c; Martínez-Blanco et al. 2011a; 2011b). Moreover, the application of compost has been proven to increase carbon sequestration in soils, which is treated differently in CF and LCA.

The comparison of two levels of technology, open field technology using few infrastructures compared to a higher level of infrastructures for greenhouse production, may allow us to determine the importance of excluding capital goods when CF is used.

The main characteristics of the two crops are given in Table 8.1.

**Table 8.1. Summary of agronomic parameters for the six case studies.**

| | Parameters | T- OF_CL | T-OF_M | T-GH_CL | T-GH_M | W- OF_CL | W- OF_M |
|---|---|---|---|---|---|---|---|
| **Case study factors** | **Crop** | Tomato | | | | Cauliflower | |
| | **Location** | Open field | | Greenhouse | | Open field | |
| | **Fertilization** | Compost & mineral fertilizers | Mineral fertilizers | Compost & mineral fertilizers | Mineral fertilizers | Compost & mineral fertilizer | Mineral fertilizers |
| **Crop** | **Variety** | El Virado® | | Caramba® | | Trevi | |
| | **Scientific name** | Lycopersicon esculentum | | | | Brassica oleracea | |
| | **Plant density (plants m$^{-2}$)** | 2.30 | | 2.8 | | 2.08 | |
| **Period** | **Starting day** | 12/05/2007 | | 06/03/2006 | | 28/09/2007 | |
| | **Finishing day** | 25/09/2007 | | 01/08/2006 | | 18/01/2008 | |
| | **Length (days)** | 133 | | 145 | | 110 | |
| **Inputs** | **Mineral fertilizers** | $HNO_3$, $KNO_3$ | $HNO_3$, $KNO_3$ | $HNO_3$ | $HNO_3$, $KNO_3$, $KPO_4H_2$, $K_2SO_4$ | $HNO_3$, $KNO_3$ | $HNO_3$, $KNO_3$ |
| | **Irrigation water (l m$^{-2}$)** | 571 | 555 | 490 | 622 | 229 | 238 |
| **Harvest** | **Commercial harvest (t ha$^{-1}$)** | 100 | 103 | 153 | 159 | 15 | 17 |

*Source: data from Martínez-Blanco et al.* (2011a; 2011b).

## 8.3.1. Goal of the study

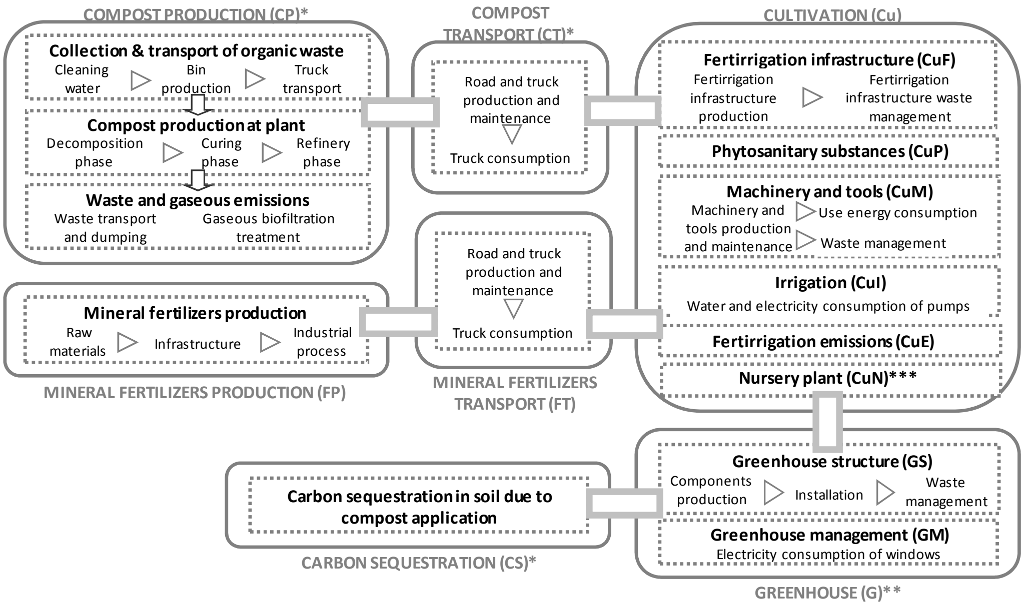
This paper assesses the value of CF and LCA as tools to calculate GHG emissions from horticultural production systems and to communicate environmental performance. The main repercussions on the quantification of greenhouse gas (GHG) emissions when either LCA or CF is used are stated. Several case studies for horticultural systems were compared in order to identify the critical issues in the choice of methodology that can give misleading information to the final consumer.

## 8.3.2. Data sources

Primary data used in this study was collected, in 2007 and 2008, from experimental fields of the research group located in the Maresme (Barcelona, Spain), where integrated cultivation management techniques for horticulture are used, and from a representative composting plant at Castelldefels (Barcelona, Spain), which uses in-vessel decomposition technology with a curing phase in turned windrows (Antón 2004; Colón et al. 2009; Martínez-Blanco et al. 2009b; 2011a; 2011b). When experimental and local information was not available, bibliographical sources and the database ecoinvent data v2.2 (SCLCI 2012) were used to complete the inventory as secondary data.

## 8.3.3. System description

This section gives a general description of all the system stages and sub-stages for the six case studies. Details of the differences in the stages and sub-stages taken into account in CF and LCA are given in section 8.3.4. As shown in Figure 8.1, the system analyzed is divided into five stages.




*See section 8.3.4 for LCA and CF system boundaries.*
** Only used in cultivation options GH_CL and OF_CL.*
***Only used in cultivation options GH_CL and GH_M.*
****Only used in the case studies with tomato.*

**Figure 8.1. Processes assessed in the stages and sub-stages of a Mediterranean tomato production system. All the stages and processes are represented.**

*Source: Modified from Martínez-Blanco et al.* (2011b).

### *(i) Fertilizer and compost production stage*

The mineral fertilizer production sub-stage includes the production of the raw and intermediate materials and their transport to the plant, synthesis of the chemical components and the waste treatment. It is important to highlight that the dose and type of mineral fertilizers added are different in each cultivation option due to the different field necessities.

The compost production sub-stage is only accounted for in two cultivation options (GH_CL and OF_CL). It includes organic waste collection, machinery and tools, water and energy consumption, gas emissions and wastes generated during industrial composting. Compost application was also considered in the total accounting of the nutrients applied.

### *(ii) Fertilizer and compost transport stage*

For the mineral fertilizer transport sub-stage, only the outward journey was included as it is considered that the means of transport return with another load. The mineral fertilizers are transported either from Israel, by road and freight ship, or from Germany by road, to the experimental fields.

For compost transport, the sub-stage considers from the plant at Castelldefels, to the experimental fields in the Maresme. Open fields are in the municipality of Santa Susana while the greenhouse experimental plots are in Cabrils (Maresme, Barcelona, Spain).

#### *(iii) Cultivation stage*

This includes six sub stages: (i) Fertirrigation infrastructure to carry water, fertilizers and phytosanitary substances to the plots, including pumps, tanks, pipes and other elements; (ii) Phytosanitary substances applied for plant and soil protection; (iii) Machinery and tools, including tractor and associated machinery and harvesting tools; (iv) Water and electricity consumption for irrigation; (v) Post-application emissions to water and air; (vi) Nursery plants (not included for cauliflower, according to Martínez-Blanco et al. (2011a)).

#### *(iv) Greenhouse stage*

This is divided into the greenhouse structure and greenhouse management sub-stages. The management sub-stage is the electricity consumption for window movement. The greenhouse stage is only included in the two cultivation options GH_CL and GH_M. The different considerations for CF and LCA are given in section 8.3.4.

#### *(v) Carbon sequestration stage*

The application of organic amendments from outside the system assessed leads to a genuine increase in soil carbon (Brenton et al. 2010). It has been recognized by the Intergovernmental Panel on Climate Change (IPCC) as one of the possible measures through which GHG emissions can be mitigated. It is considered that the carbon which remains in the soil 100 years after application is 8% (Smith et al. 2001; Hansen et al. 2006a), which was subtracted from the total GHG emissions. The different considerations for CF and LCA are given in section 8.3.4.

### 8.3.4. System boundaries. Main differences between CF and LCA in the case study

As shown in Table 8.2, the system boundaries differ depending on which methodology was applied. The methodological differences are described in section 8.2.3.

For the fertilizer and compost production stage, industrial processes were accounted in both methodologies, but production, maintenance and disposal of the production plants and machinery were not included for CF.

For the fertilizer and compost transport stage, the production, maintenance and disposal of capital goods as lorries, roads or ships are included in LCA and excluded in CF.

As Table 8.2 shows, the differences in the remaining stages are again infrastructures omission. In the cultivation stage for CF, the fertirrigation infrastructure the tractor and harvesting tools and the infrastructure for the production of nursery plants are excluded, as is the greenhouse structure.

According to the specifications in section 8.5.6 of the PAS 2050:2008, carbon sequestration in soils due to compost application is not included when CF methodology is applied. Only changes in the carbon content of soils, either emissions or sequestration, arising from direct land use changes before 1990 should be included in the assessment of GHG emissions under PAS 2050:2008. Carbon changes due to land use changes have not been considered in the study because the conversion of the experimental fields to agricultural land occurred before 1990. Carbon changes due to land use were also not accounted with LCA but carbon sequestration was.

All infrastructure for processes included in the inventory (materials, processing, transports, energy, waste treatments, etc.) were omitted for CF methodology and included for LCA. For example, the production plant was not included for phytosanitary substances production, nor the vehicle or road production, maintenance and disposal, for all transports, when CF was assessed.

Accounting of the processes included in the inventory for each methodology was from the production of raw materials required for the manufacturing processes to the final disposal of materials.

**Table 8.2. Processes included in the case studies using CF and LCA methodologies.**

| | CF [a] | LCA | | CF [a] | LCA |
|---|---|---|---|---|---|
| **FERTILIZER & COMPOST PRODUCTION** | | | **CULTIVATION** | | |
| C raw materials | Yes | Yes | ***Fertirrigation infrastructure*** | | |
| C electricity and diesel | Yes | Yes | P&M elements | No | Yes |
| P&M machinery | No | Yes | Transports | No | Yes |
| P&M production plants | No | Yes | WD elements | No | Yes |
| WD of machinery and plants | No | Yes | ***Phytosanitary substances*** | | |
| Transports | Yes | Yes | P substances | Yes | Yes |
| Emissions | Yes | Yes | Transports | Yes | Yes |
| **FERTILIZER & COMPOST TRANSPORT** | | | ***Machinery and tools*** | | |
| C diesel | Yes | Yes | C diesel and electricity | Yes | Yes |
| P&M lorries | No | Yes | P&M tractor | No | Yes |
| P&M road | No | Yes | P&M associated machinery | No | Yes |
| WD road and lorries | No | Yes | P harvesting tools | No | Yes |
| Emissions | Yes | Yes | P packaging | Yes | Yes |
| **GREENHOUSE (GH options)** | | | ***Irrigation*** | | |
| ***Greenhouse structure*** | | | C water | Yes | Yes |
| P&M structure | No | Yes | *Post-application emissions* | | |
| Construction | No | Yes | Water emissions | Yes | Yes |
| Transports | No | Yes | Air emissions | Yes | Yes |
| WD structure | No | Yes | ***Nursery plants (only for tomato)*** | | |
| ***Greenhouse management*** | | | Infrastructure & machinery | No | Yes |
| C electricity | Yes | Yes | C inputs | Yes | Yes |
| **CARBON SEQUESTRATION ($C_L$ options)** | | | Transports | Yes | Yes |
| Carbon sequestration | No | Yes | | | |

*C, consumption of; M, maintenance of; P, production of; WD, waste disposal of.*

[a] *For CF methodology, the infrastructures are excluded from all the processes accounted for in the inventory (materials, processing, energy, waste treatment, etc). They are included in the LCA.*

### 8.3.5. Functional unit

The functional unit used was the production of one ton of either commercial tomato or commercial cauliflower. All input and output flows were related to this reference value. The yields are shown in Table 8.1.

### 8.3.6. Waste management impact distribution

Recycled or recovered waste during the production cycle was not considered in the inventory, according to Martínez-Blanco et al. (2011a; 2011b), because it is attributed to the system which uses the waste as a raw material. Dumped waste was accounted for in all processes.

## 8.4. Results

The results for the six case studies using both methodologies are presented. The GHG emissions were measured in kilograms of carbon dioxide equivalent per functional unit (from now on kg $CO_2$ eq.), i.e. kilograms of carbon dioxide equivalent either per ton of tomato or per ton of cauliflower.

Tables 8.3, 8.4 and 8.5 show the results of the GHG emissions of the two methodologies and the differences between them in absolute value or percentage for each cultivation option, for each stage and sub-stage, and the totals for the six case studies. Figures 8.2 and 8.3 show the total GHG emissions for the two methodologies and the six case studies and illustrate the contributions of the stages and sub-stages to the differences in GHG emissions assessed with LCA and CF.

### 8.4.1. Main contributors to the total GHG emissions

According to Hillier et al. (2009), for all cultivation options, the maximum GHG emissions were in the fertilizer and compost production stage, for both methodologies (up to 233 kg $CO_2$ eq. for tomato and 714 kg $CO_2$ eq. for cauliflower), as reported by Martínez-Blanco et al. (2011a; 2011b) for LCA, equivalent to 46-87% of the total GHG emissions (Table 8.3-5). The highest levels were with the cultivation options that included compost application.

The cultivation stage also accounted for environmental impacts between 13-34% of the total GHG emissions, mainly due to the machinery and tools, irrigation sub-stages (and nursery plants for tomato).

The greenhouse stage, in the T-GH_$C_L$ and T-GH_M case studies, represented around 20% of the total GHG emissions using LCA methodology (Table 8.3 and 8.4).

### 8.4.2. Main contributors to the difference between methodologies

Figures 8.2 and 8.3 show the contributions of the stages and sub-stages to the differences in the amounts of GHG emissions assessed with the LCA and CF methodologies.

The fertilizer and compost production stage had the least percentage variation between LCA and CF, less than 3% (Tables 8.3-8.5). However, as this stage makes the highest contribution to the total impacts (section 8.4.1), thus the absolute value of the GHG emissions omitted for this stage with CF was relevant. The differences were particularly high for case studies in open field, being 28-56% of the total GHG emissions omitted for the fertilizer and compost production stage (Figure 8.2a-b and 8.3).

There were differences of more than 20% between the methodologies for the fertilizer and compost transport stage (Tables 8.3-5), as production, maintenance and disposal of the vehicles and roads are not considered with CF. However, this was not a major contribution to the total variation of GHG emissions between methodologies, which was between 12-17% around for open field and 3% for greenhouse cultivation (Figures 8.2 and 8.3).

**Table 8.3. Greenhouse gas emissions for open field tomato options applying LCA and CF methodologies.**

| | Mineral Fertilizers (T-OF_M) | | | | Compost & Mineral Fertilizers (T-OF_$C_L$) | | | |
|---|---|---|---|---|---|---|---|---|
| | LCA | CF | Difference | | LCA | CF | Difference | |
| | kg $CO_2$ eq. $FU^{-1}$ | | | % [a] | kg $CO_2$ eq. $FU^{-1}$ | | | % [a] |
| *Fertilizer and compost production stage* | | | | | | | | |
| **Mineral fertilizer production** | 96.0 | 91.9 | 4.1 | *4.3* | 77.7 | 74.8 | 2.8 | *3.6* |
| **Compost production** | na | na | na | *na* | 155.6 | 146.4 | 9.2 | *5.9* |
| **Stage total** | **96.0** | **91.9** | **4.1** | ***2.42*** | **233.3** | **221.2** | **12.0** | ***2.42*** |
| *Fertilizer and compost transport stage* | | | | | | | | |
| **Mineral fertilizer transport** | 8.0 | 6.4 | 1.6 | *20.5* | 6.2 | 4.9 | 1.3 | *20.5* |
| **Compost transport** | na | na | na | *na* | 6.8 | 5.3 | 1.5 | *21.7* |
| **Stage total** | **8.0** | **6.4** | **1.6** | ***20.5*** | **12.9** | **10.2** | **2.7** | ***21.2*** |
| *Cultivation stage* | | | | | | | | |
| **Fertirrigation infrastructure** | 5.9 | 0.0 | 5.9 | *100.0* | 6.1 | 0.0 | 6.1 | *100.0* |
| **Phytosanitary substances** | 1.9 | 1.8 | 0.1 | *7.2* | 2.0 | 1.8 | 0.1 | *7.2* |
| **Machinery and tools** | 18.1 | 17.1 | 1.0 | *5.8* | 18.7 | 17.5 | 1.2 | *6.6* |
| **Irrigation** | 11.5 | 11.3 | 0.3 | *2.4* | 12.2 | 11.9 | 0.3 | *2.4* |
| **Post-application emissions** | 0.0 | 0.0 | 0.0 | *0.0* | 0.1 | 0.1 | 0.0 | *0.0* |
| **Nursery plants** | 13.9 | 12.6 | 1.4 | *9.7* | 14.4 | 13.0 | 1.4 | *9.7* |
| **Stage total** | **51.5** | **42.7** | **8.7** | ***17.0*** | **53.5** | **44.4** | **9.2** | ***17.1*** |
| *Carbon sequestration stage* | | | | | | | | |
| **Carbon Sequestration** | na | na | na | *na* | -11.1 | 0.0 | -11.1 | *100.0* |
| **Stage total** | **na** | **na** | **na** | ***na*** | **-11.1** | **0.0** | **-11.1** | ***100.0*** |
| *Total GHG emissions* | | | | | | | | |
| **Subtotal [b]** | **155.5** | **141.0** | **14.5** | ***9.3*** | **299.7** | **275.8** | **23.9** | ***8.0*** |
| **Total** | **155.5** | **141.0** | **14.5** | ***9.3*** | **288.7** | **275.8** | **12.9** | ***4.5*** |

[a] *The percentages show the differences in the amount of GHG emissions assessed with both methodologies for each stage or sub-stage.*
[b] *Carbon sequestration not included.*

In the cultivation stage, there were differences of 11-22% between LCA and CF (Tables 8.3-5), depending on the case study, mainly due to excluding the fertirrigation infrastructure sub-stage (and the infrastructures in the nursery plants sub-stage) for CF. The contribution of these differences to the total variation between methodologies was around 12% for greenhouse options, but considerably higher for open field options, between 32-61% (Figures 8.2 and 8.3).

The highest percentage variation between the methodologies (93%) was in the greenhouse stage for tomato (Table 8.4), mainly due to the omission of the greenhouse structure sub-stage in CF. This stage also had the highest contribution (72-73%) to the GHG omitted by CF analysis compared to LCA (Figure 8.2c-d), for the cultivation options affected.

**Table 8.4. Greenhouse gas emissions for greenhouse tomato options applying LCA and CF methodologies.**

| | Mineral Fertilizers (T-GH_M) | | | | Compost & Mineral Fertilizers (T-GH_$C_L$) | | | |
|---|---|---|---|---|---|---|---|---|
| | **LCA** | **CF** | **Difference** | | **LCA** | **CF** | **Difference** | |
| | kg $CO_2$ eq. $FU^{-1}$ | | | % [a] | kg $CO_2$ eq. $FU^{-1}$ | | | % [a] |
| | | | | ***Fertilizer and compost production stage*** | | | | |
| **Mineral fertilizer production** | 71.1 | 66.3 | 4.8 | *6.7* | 27.5 | 26.8 | 0.7 | *2.5* |
| **Compost production** | na | na | na | *na* | 88.8 | 83.5 | 5.3 | *5.9* |
| **Stage total** | **71.1** | **66.3** | **4.8** | ***2.42*** | **116.2** | **110.3** | **5.9** | ***2.42*** |
| | | | | ***Fertilizer and compost transport stage*** | | | | |
| **Mineral fertilizer transport** | 6.7 | 5.3 | 1.4 | *20.5* | 1.9 | 1.5 | 0.4 | *20.5* |
| **Compost transport** | na | na | na | *na* | 3.9 | 3.0 | 0.8 | *21.7* |
| **Stage total** | **6.7** | **5.3** | **1.4** | ***20.5*** | **5.7** | **4.5** | **1.2** | ***21.3*** |
| | | | | ***Cultivation stage*** | | | | |
| **Fertirrigation infrastructure** | 3.0 | 0.0 | 3.0 | *100.0* | 3.1 | 0.0 | 3.1 | *100.0* |
| **Phytosanitary substances** | 0.0 | 0.0 | 0.0 | *7.2* | 0.0 | 0.0 | 0.0 | *7.2* |
| **Machinery and tools** | 16.7 | 16.3 | 0.5 | *2.8* | 16.9 | 16.3 | 0.5 | *3.1* |
| **Irrigation** | 6.6 | 6.5 | 0.2 | *2.4* | 5.1 | 5.0 | 0.1 | *2.4* |
| **Post-application emissions** | 6.2 | 6.2 | 0.0 | *0.0* | 1.8 | 1.8 | 0.0 | *0.0* |
| **Nursery plants** | 10.9 | 9.8 | 1.1 | *9.7* | 11.3 | 10.2 | 1.1 | *9.7* |
| **Stage total** | **43.5** | **38.8** | **4.7** | ***10.8*** | **38.1** | **33.3** | **4.9** | ***12.7*** |
| | | | | ***Greenhouse stage*** | | | | |
| **Greenhouse structure** | 29.1 | 0.0 | 29.1 | *100.0* | 30.3 | 0.0 | 30.3 | *100.0* |
| **Greenhouse management** | 2.2 | 2.2 | 0.1 | *2.4* | 2.3 | 2.3 | 0.1 | *2.4* |
| **Stage total** | **31.3** | **2.2** | **29.2** | ***93.1*** | **32.6** | **2.3** | **30.3** | ***93.1*** |
| | | | | ***Carbon sequestration stage*** | | | | |
| **Carbon Sequestration** | na | na | na | *na* | -10.4 | 0.0 | -10.4 | *100.0* |
| **Stage total** | **na** | **na** | **na** | ***na*** | **-10.4** | **0.0** | **-10.4** | ***100.0*** |
| | | | | ***Total GHG emissions*** | | | | |
| **Subtotal** [b] | **152.6** | **112.6** | **40.0** | ***26.2*** | **192.7** | **150.3** | **42.3** | ***22.0*** |
| **Total** | **152.6** | **112.6** | **40.0** | ***26.2*** | **182.2** | **150.3** | **31.9** | ***17.5*** |

[a] *The percentages show the differences in the amount of GHG emissions assessed with both methodologies for each stage or sub-stage.*
[b] *Carbon sequestration not included.*

In the two cultivation options where compost was applied, a certain amount of carbon was sequestered into the soil when LCA methodology was applied, but not with CF (Tables 8.3-5). The carbon sequestration stage, with a storage value of 10-11 kg $CO_2$ eq., for tomato case studies using compost and 47 kg $CO_2$ eq., for cauliflower, was fully omitted by CF (section 8.3.4.).

**Table 8.5. Greenhouse gas emissions for open field cauliflower options applying LCA and CF methodologies.**

| | Mineral Fertilizers (W-OF_M) | | | | Compost & Mineral Fertilizers (W-OF_$C_L$) | | | |
|---|---|---|---|---|---|---|---|---|
| | LCA | CF | Difference | | LCA | CF | Difference | |
| | kg $CO_2$ eq. $FU^{-1}$ | | | % [a] | kg $CO_2$ eq. $FU^{-1}$ | | | % [a] |
| | | | | *Fertilizer and compost production stage* | | | | |
| **Mineral fertilizer production** | 234.0 | 224.0 | 10.0 | *4.3* | 188.7 | 181.9 | 6.9 | *3.6* |
| **Compost production** | na | na | na | *na* | 524.8 | 490.0 | 34.9 | *6.6* |
| **Stage total** | **234.0** | **224.0** | **10.0** | ***2.42*** | **713.6** | **671.8** | **41.7** | ***2.42*** |
| | | | | *Fertilizer and compost transport stage* | | | | |
| **Mineral fertilizer transport** | 20.0 | 15.9 | 4.1 | *20.5* | 14.8 | 11.7 | 3.0 | *20.5* |
| **Compost transport** | na | na | na | *na* | 27.8 | 21.8 | 6.0 | *21.7* |
| **Stage total** | **20.0** | **15.9** | **4.1** | ***20.5*** | **42.6** | **33.5** | **9.1** | ***21.3*** |
| | | | | *Cultivation stage* | | | | |
| **Fertirrigation infrastructure** | 6.0 | 0.0 | 6.0 | *100.0* | 18.7 | 0.0 | 18.7 | *100.0* |
| **Phytosanitary substances** | 0.5 | 0.4 | 0.0 | *7.1* | 0.5 | 0.5 | 0.0 | *7.1* |
| **Machinery and tools** | 54.6 | 50.9 | 3.7 | *6.8* | 57.5 | 53.0 | 4.5 | *7.8* |
| **Irrigation** | 28.9 | 28.2 | 0.7 | *2.4* | 30.3 | 29.5 | 0.7 | *2.4* |
| **Post-application emissions** | 0.0 | 0.0 | 0.0 | *0.0* | 0.0 | 0.0 | 0.0 | *0.0* |
| **Stage total** | **89.9** | **79.5** | **10.4** | ***11.6*** | **107.0** | **83.0** | **24.0** | ***22.4*** |
| | | | | *Carbon sequestration stage* | | | | |
| **Carbon Sequestration** | na | na | na | *na* | -47.4 | 0.0 | -47.4 | *100.0* |
| **Stage total** | **na** | **na** | **na** | ***na*** | **-47.4** | **0.0** | **-47.4** | ***100.0*** |
| | | | | *Total GHG emissions* | | | | |
| **Subtotal [b]** | **343.9** | **319.4** | **24.5** | ***7.1*** | **863.1** | **788.3** | **74.8** | ***8.7*** |
| **Total** | **343.9** | **319.4** | **24.5** | ***7.1*** | **815.7** | **788.3** | **27.4** | ***3.4*** |

[a] *The percentages show the differences in the amount of GHG emissions assessed with both methodologies for each stage or sub-stage.*

[b] *Carbon sequestration not included.*

## 8.4.3. Total differences between methodologies

A total amount of 14-75 kg $CO_2$ eq. was omitted with CF methodology, depending on the case study (Table 8.3-5). For the OF_$C_L$ and GH_$C_L$, options, considering compost application, a fraction of the emissions calculated by LCA was offset by 10-47 kg $CO_2$ eq. carbon sequestration.

Regarding open field case studies, the absolute values of the variation in the GHG emissions assessed with CF and LCA were 14-24 kg $CO_2$ eq. for tomato and 25-27 kg $CO_2$ eq. for cauliflower (Tables 8.3 and 8.5). However, the relative variation was slightly higher with mineral fertilizers (7-9%) than with compost ones (3-4% with subtraction of carbon sequestration, and 8-9% without) as the total impacts were lower for T-OF_M and W-OF_M than for T-OF_$C_L$ and W-OF_$C_L$ (Tables 8.3 and 8.5). The differences were mainly due to the aforementioned changes in the cultivation, and the fertilizer and compost production stages for the CF methodology.

For greenhouse tomato case studies, the variations, in absolute values, were double those for open field; 42.3 and 40.0 kg $CO_2$ eq. for T-GH_$C_L$ and T-GH_M, respectively (Table 8.4). The variation in percentage was higher for T-GH_M (26%) than for T-GH_$C_L$ (22%, with subtraction of carbon sequestration and 18%, without) because the total impacts were lower for the former than for the latter (Table 8.4). The differences were mainly due to the omission of the greenhouse stage in CF methodology.

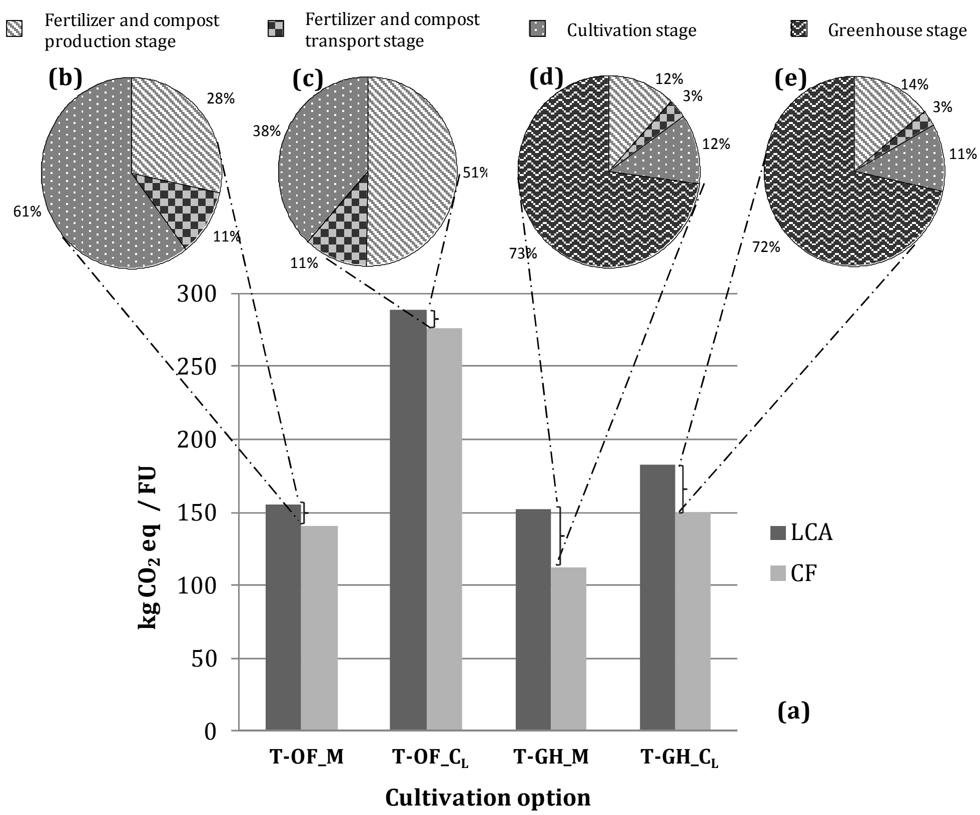



**Figure 8.2. Total greenhouse gas emissions and differences between methodologies for tomato cultivation options. (a) Total GHG emissions for the four cultivation options applying LCA and CF methodologies. Contribution of the stages and sub-stages to the amount of GHG emissions omitted by CF methodology compared to LCA results for T-OF_M (b), T-OF_$C_L$ (c), T-GH_M (d), and T-GH_$C_L$ (e).**

## 8.4.4. Case study comparison

The results show that, with both methodologies, using only mineral fertilizers had the least impact.

For tomato, the T-OF_M and T-GH_M case studies have the least impact, followed by T-GH_$C_L$ and finally T-OF_$C_L$, the latter emits 86-89% more kg $CO_2$ eq. than the mineral fertilizers options (Figure 8.2a). The difference in the GHG emissions between OF_M and GH_M was below 2% (Figure 8.2a) because, although more resources are used for the structure in the greenhouse option, tomato production was higher, offsetting GHG emissions per ton. The order of impact between the cultivation options when applying CF was as with LCA. However, the difference between T-OF_M and T-GH_M increased from less than 2% to 25%, as the CF methodology did not consider greenhouse burdens and, therefore, the differences in tomato production were not offset as in the case of LCA. There was also a significant difference in the assessment between T-OF_M and T-GH_$C_L$, which was lower than 7% with CF, while it was 17% for LCA.

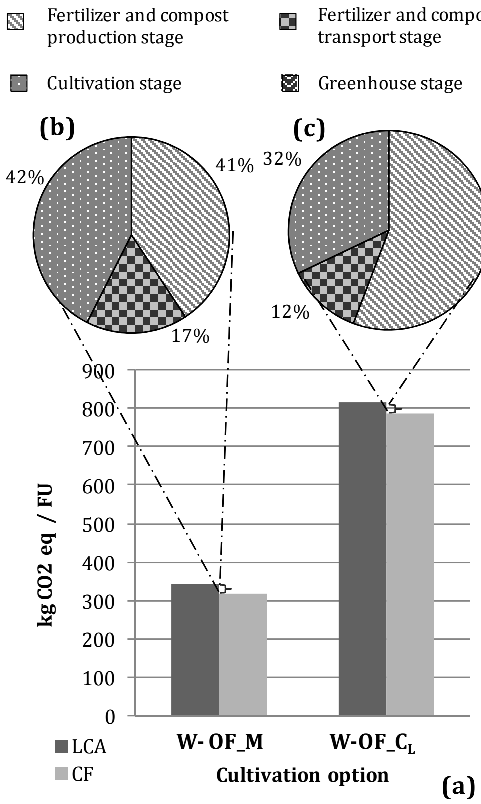



**Figure 8.3. Total greenhouse gas emissions and differences between methodologies for cauliflower cultivation options. (a) Total GHG emissions for the two cultivation options applying LCA and CF methodologies. Contribution of the stages and sub-stages to the amount of GHG emissions omitted by CF methodology compared to LCA results for W-OF_M (b) and W-OF_$C_L$ (c).**

According to the previous section (8.4.3), differences between methodologies for cauliflower were lower than 9%, therefore comparison between cultivation options were nearly the same: W-OF_$C_L$ was 137% and 147% more impacting than W-OF_M for LCA and CF, respectively (Figure 8.3a).

On the other hand, impact figures for cauliflower were much higher than for tomato (Figure 8.2 and 8.3), because of the lower yields of cauliflower (Table 8.1) which resulted in higher burdens per functional unit.

# 8.5. Conclusions and discussion

The differences in the assessment of GHG emissions using LCA and CF methodologies were evaluated for horticultural production, through several case studies of Mediterranean horticultural crops, including several crop, fertilizing and technology options.

There was significant disparity in the results due to the exclusion of capital goods in the CF calculation, as recommended by the PAS 2050:2008, with up to 26% of the GHG emissions measured with LCA for tomato not taken into account using CF. The emissions calculated for the greenhouse options using CF methodology were significantly lower than the emissions calculated using LCA. It was because the greenhouse structure, which is a major contributor to the total impacts in LCA results, was excluded from the CF analysis. Using CF, the tomato cultivation option under greenhouse and applying only mineral fertilizers (T-GH_M) was 25% less pollutant than the same option but in open field (T-OF_M), whereas the difference was below 2% when LCA was used, giving misleading information to the consumer when CF is applied. Regarding open field options, which consider fewer infrastructures; the differences in the GHG emissions assessed with the two methodologies were in the order of 5-10%.

The PAS 2050:2008, regarding the exclusion of emissions arising from capital goods, argues that capital goods, which last many years, are usually relatively smaller contributors to total burdens than operation inputs and outputs. Moreover, there is a lack of data to accurately identify sectors where capital goods emissions are material and the cost/complexity of specific analysis is too high. Consequently, the CF results for the GWP indicator could be similar to that using LCA only when there is a low level of infrastructure (or high energy consumption during operation or short life span). Frischknecht et al. (2007) reported that in renewable energy, wastewater treatment, landfill and transport services, the capital goods may be of major or substantial relevance. The same authors stated that, for agricultural products, the share of impacts caused by infrastructures on cumulative totals is of minor priority, and recommended not including them in case studies (Frischknecht et al. 2007). However, our results indicate that this exclusion for tomato and other Mediterranean horticultural products, with low energy demand, seems to increase the disparity between CF and LCA results.

In addition, the estimation of carbon and nitrogen dynamics in soils is a weak and complex methodological step, due to the high number of factors interacting. The lack of reliable data and the complexity of the calculation are the main reasons why PAS 2050:2008 avoids the inclusion of methods to calculate these emissions. According to Martínez-Blanco et al. (2011a; 2011b) and for the options considering compost application, when LCA methodology was used, it is assumed that a certain quantity of carbon is incorporated to soils, meaning a credit of about 5% of the total impacts. It is an important issue to take into account, as the soil as a carbon sink is a new perspective that could help to achieve the EU aims of reducing the $CO_2$ concentration in the atmosphere and could be a major measure for mitigation of climate change within the agricultural sector. Both methodologies fail on the inclusion of beneficial properties linked to the application of organic matter to the soil, for instance improving soil texture, reducing erosion risk and increasing water retention.

CF and LCA are both based on the life cycle approach and compiling the inventory data is very similar. The process stages or sub-stages excluded to simplify the methodology, which seriously affect the impact assessment, need to be reintroduced within the system boundaries, or specific recommendations should be set for affected sectors,

But even if CF is a tool limited to only one impact category, global warming, and has certain methodological limitations compared to LCA, it is valuable as an environmental assessment framework based on life cycle concepts for products in general and horticultural products in particular. CF is a simple and understandable environmental indicator and could be the first step in the environmental communication strategy to help consumers assimilate the environmental information present in products. In addition, while the ISO 14044 does not omit some of these processes, it does not clearly specify their quantification.

Due to the relevancy that CF could have in consumer choice, it needs standardization and a deeper methodological definition than that given by PAS 2050:2008. For example, not considering capital goods can lead to prioritization of products or processes that do not have a better environmental performance when they are assessed with LCA. There is a risk that carbon accounting and labelling instruments do not properly represent the complexity of production systems (Brenton et al. 2010) or may generate more confusion in consumer decision making.

There is a high degree of variability with horticultural, and agri-food, cultivation processes, such as type and intensity of fertilization, use of soil or growing media, application of chemicals and use of infrastructures for protection, lighting or heating, depending on factors such as climate, soil, pests and diseases, and cultivation intensity. With more intense use of capital goods, more miscalculation is likely by applying CF rather than LCA methodology. An increased use of technology in horticultural systems may lead to a rise in the energy consumption and so a decrease in the relevance of infrastructures on impacts. The omission of capital goods in the assessment could punish those growers using low technology and having lower yields (e.g. in developing countries or organic agriculture). Relevant differences could also be assessed between methodologies regarding nitrogen and carbon dynamics.

## 8.6. Recommendations

Due to the variability of production processes in the agri-food sector and the different use of capital goods they should be included when agri-food systems are studied, as pointed out by Antón et al. (2005a). This case study on Mediterranean tomato production with different cultivation options shows the need for specific methodological reports for each productive sector to obtain reliable results for the carbon footprint.

The current version of PAS 2050:2008 excludes GHG emissions arising from capital goods and the carbon credits from carbon sequestration in soils, generated by management and labour applied to crops. We consider that both should be included in future revisions of the PAS 2050:2008, in the ISO 14067 and in specific reports for the production of horticultural products, such as Product Category Rules (PCR).

Further case studies and examples of horticultural production systems considering different climate areas, technologies and crops management, are also necessary, as is further research to include carbon and nitrogen soil dynamics in both methodologies, for more representative CF and GWP assessments.

# Chapter 9. Life Cycle Sustainability Assessment of compost and mineral fertilizers production for agriculture - application challenges for social LCA

**Chapter 9** is based on the following paper:

Martínez-Blanco, J., Lehmann, A., Muñoz, P., Antón, A., Traverso, M., Rieradevall, R., & Finkbeiner, M. Life Cycle Sustainability Assessment of compost and mineral fertilizers production for agriculture - application challenges for social LCA. Accepted with major revisions in May 2012 to the Special Issue on Life Cycle Sustainability Assessment in the International Journal of Life Cycle Assessment.

# Abstract

***Purpose.*** The study aims to assess the three dimensions of sustainability (environment, economy and society) related to mineral fertilizers and compost, from a life cycle perspective considering a real case study – the tomato production in the Mediterranean region. The available nitrogen in the short-medium term from the applied fertilizers is the functional unit considered. The system boundaries of the Life Cycle Sustainability Assessment (LCSA) study include fertilizers production, transport as well some stages of the cultivation. Due to the novelty of the method, particular focus is laid on presenting the application challenges of Social life cycle assessment (SLCA) within LCSA.

***Methods.*** LCA is following the obligatory classification and characterization phases defined by the ISO 14044. For LCC we select three types of internal costs. Regarding SLCA, upstream and mainstream processes of the production chain are taken into account and a three level-approach (country, sector and company) for data collection is used. Social impacts related to upstream processes have so far not been considered comprehensively in the few existing SLCA case studies. The amount of working time which is spent on each unit process is used to score the relevance of each process. The Life Cycle Sustainability Dashboard presents the LCSA results.

***Results and discussion.*** For the environmental dimension, compost is the worst option, for all impact categories chosen, whereas results for nitric acid and potassium nitrate – the two mineral fertilizers included – are in the same order of magnitude. Regarding the LCC evaluation, even though the price of compost is lower than the price of the mineral fertilizers, transport and application costs are higher for compost. Similar upstream sectors are involved for compost and mineral fertilizers production, although working time is significantly higher for the former. Showing a high share of working time in the product´s life cycle, energy provision is detected as one of the main upstream sectors that should be assessed in social assessments. For mainstream sectors, nitric acid has the best social results, even though compost is promoting local employment. Regarding the aggregation of the results for LCSA, nitric acid seems to have a better performance for the three dimensions.

***Conclusions and recommendations.*** Since social impacts play a major role in sustainability assessment and as there is no a commonly agreed methodology so far beyond conceptual levels, every effort to go further in the application for real case studies is highly recommended.

## Keywords

Fertilizer; Compost; Company behavior; SLCA; LCA; LCC, Sustainability.

# 9.1. Introduction

Sustainability describes a development that meets the needs of the present without compromising the ability of future generations to meet their own needs (Bruntland 1987). It comprises three "pillars" – environment, economy and society – which are addressed by Life Cycle Sustainability Assessment (LCSA) when the product is assessed from a life cycle perspective (Finkbeiner et al. 2010). While the LCSA is generally accepted conceptually (UNEP 2011), the application experience through real world case studies is still very limited (Traverso and Finkbeiner 2009; Traverso et al. Accepted 2011).

The maturity of methods and tools, under a life cycle framework, is different for the three sustainability dimensions. While the environmental dimension can be covered quite well today with the environmental Life Cycle Assessment (LCA) (ISO 2006b; JRC and IES 2010b), the economic and social indicators and evaluation methods of products still need fundamental scientific progress (Klöepffer 2008; Finkbeiner et al. 2010).

The main concerns regarding Social life cycle assessment (SLCA) are the criteria for the selection of social indicators and the valuation method, as well as the difficulties to relate the social results for each indicator to the functional unit of the product-system. Besides, the existence of social databases is limited and on-site company data collection for processes along the life cycle of a product is highly time demanding and not always feasible (Jørgensen et al. 2008; Klöepffer 2008; Swarr 2009; Finkbeiner et al. 2010; Ciroth and Franze 2011).

While environmental impacts are generally aggregated over the entire life cycle of the product by relating impacts to a functional unit, the aggregation in SLCA is recommended to be carried out by use of working time in several references (Hunkeler 2006; UNEP 2009). However, no specific approach is presented to use this in the evaluation of social impacts.

Regarding Life cycle costing (LCC), the methodology for economical dimension evaluation, three main concerns are reported: different perspectives for the assessment can be used; data is very volatile; and no agreement is made for external costs quantification (Klöepffer 2008; Finkbeiner et al. 2010; Swarr et al. 2011).

Several studies dealing with methodological and practical questions for SLCA were published (Hunkeler 2006; Kruse et al. 2009; Ciroth and Franze 2011; Franze and Ciroth 2011; Lehmann et al. 2011). However, an implementation of SLCA to fertilizer has not been carried out yet, and more importantly no case study so far considers the upstream processes – this is crucial also because a LCSA ideally intends to have same system boundaries for the three dimensions. From the best of our knowledge, apart from the current paper, only two other SLCA studies are assessing agricultural systems (Andrews et al. 2009; Franze and Ciroth 2011); while Vinyes et al. (Accepted 2012) published the first social performance assessment for waste management.

Similarly, hardly any attempt is found for LCSA application. Lindner et al. (2010) developed a tool assessing few sustainability indicators throughout the forest-wood-chains. Vinyes et al. (Vinyes et al. Accepted 2012) proposed a Sustainability Index for three domestic used cooking oil collection systems. Traverso and Finkbeiner (2009) used the Life Cycle sustainability Dashboard for presenting environmental, economic and social performance of several hard floor coverings production (Traverso and Finkbeiner 2009; Capitano et al. 2011; Peri et al. 2011; Traverso et al. Accepted 2011).

This paper addresses the chosen case study from all three sustainability dimensions – according to the LCSA approach. Because the environmental assessment and cost-benefit for the full case study, including fertilizers production, transport and processes involved in crop cultivation (i.e., machinery, pesticides and water) were already reported by Martínez-Blanco et al. (2010; 2011a; 2011b; Submitted 2011), the paper is mainly focused on the presentation of social aspects, and is conducted with the SLCA methodology (UNEP 2009). Environmental and economic results are adapted from previous works. It focuses on fertilizing decisions from a real case study – tomato production in the Mediterranean region. From the best of our knowledge, no studies applying SLCA and the whole LCSA to fertilizers production or organic waste treatment have been performed.

# 9.2. Methodological approach

The current section is split in three main parts. First, the goal and scope common for all the study are presented. Following, the methods and assumptions for SLCA, LCA and LCC are defined. Finally, we include a briefly explanation of the LCSA dashboard used for presenting the LCSA results and we explain the assumptions taken while applying this tool.

## 9.2.1 Goal and scope

The objectives, functional unit, system boundaries, data sources and fertilizing alternatives are presented here. For some of the sub-sections additional comments are made for SLCA.

### *(i) Objectives*

The studies' main objective is the case study application of SLCA within Life Cycle Sustainability Assessment framework for organic and mineral fertilizers.

Three sub-objectives could be differentiated:

To develop a theoretical proposal for SLCA application in a real agricultural case study. This especially focuses on the three level-approach, the indicator selection process and the method for the social assessment of upstream company performance, using the Social Hotspot Database (SHDB) and working time along the product chain as a aggregation unit.

Practically, to identify social hotspots of the two fertilizing alternatives, revealing potential risks and improvements regarding its social performance. This can be of direct interest for the fertilizer user (the farmer).

Complementarily, environmental and economic performances are also assessed for the fertilizing alternatives. To be one of the few real case application of the LCSA approach and to apply dashboard for presenting the results of the three dimensions.

### *(ii) Functional unit*

For the current study, 1 kilogram of potential available nitrogen (i.e., mineralized) in the short-term from the applied fertilizers to tomato fields is the functional unit considered (henceforth, 1 kg of N available). The nitrogen available in the rotation period where the crops were cultivated is considered in the study, as sum of the mineral nitrogen and the easily hydrolysable nitrogen (Martínez-Blanco et al. 2011a; 2011b). The selected functional unit is reflecting that the focus of the study is put on fertilizers consumption by the farmer rather than tomato production.

### *(iii) System boundaries*

With focus on a comparative analysis, only those stages and sub-stages within compost and mineral fertilizers production and application system are considered that differ in the management between either using compost or do not using it. This includes compost transport and production, mineral fertilizers transport and production, farm operations strictly related to compost application and direct emissions occurred due to the application of fertilizers.

Although several authors reported the potential savings in water, pesticides or working efforts, among others, when compost is applied to soils (ROU 2007; Hargreaves et al. 2008), differences on these inputs were not evaluated in the experimental trials (Martínez-Blanco et al. 2011a; 2011b). Therefore, such differences between fertilizing alternatives were not included in the LCSA boundaries.

Figure 9.1 shows the elements included in the three LCSA dimensions. The black boxes are representing the main stages (mainstream companies for SLCA), while the white ones are referred to chain stages (upstream companies). Two different color shaded polygons were used to show the system boundaries used for SLCA, LCA and LCC.

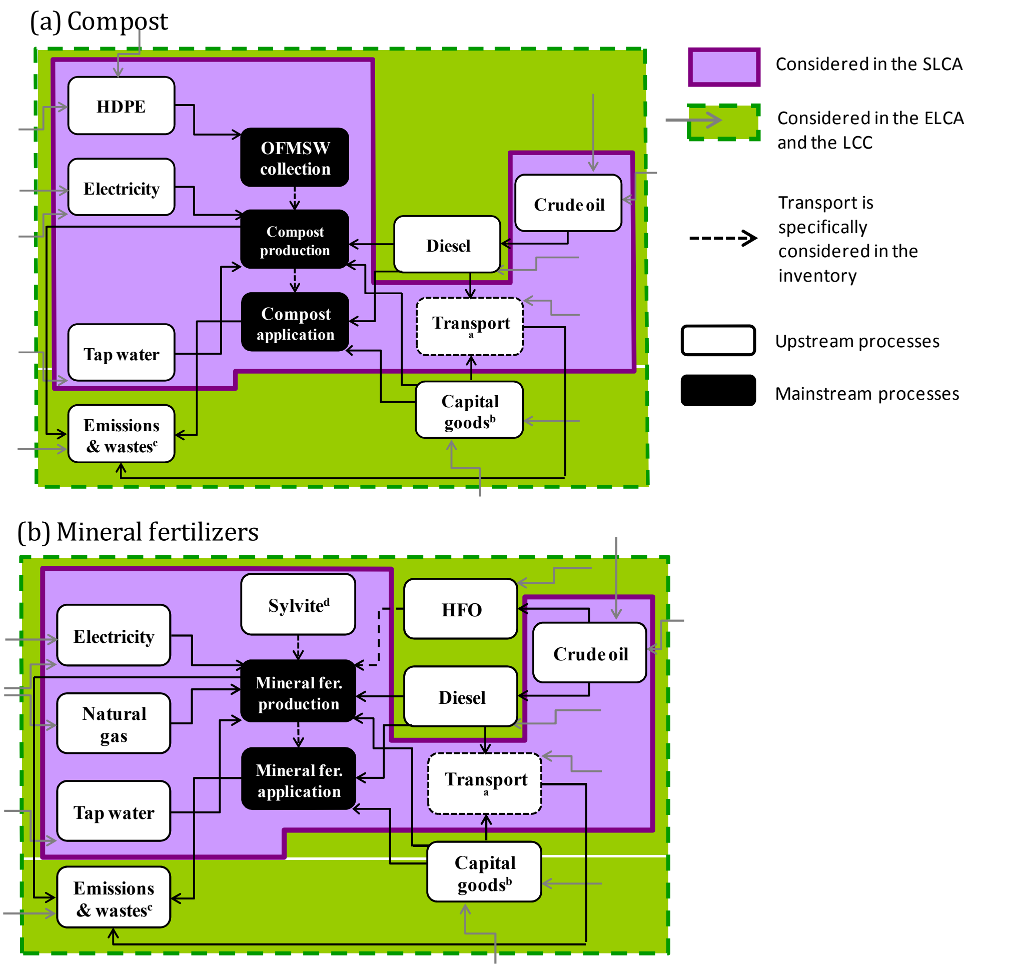
According to UNEP (2011), LCSA ideally considers equivalent system boundaries for the LCA, LCC and SLCA. Therefore, the system boundaries of the study have been defined as similar as possible for the three of them. However, there are some exceptions that are explained in the following.

In the study, LCA is including both mainstream and upstream processes (Figure 9.1). Additionally, the background life cycles of the included processes (e.g. energy consumption and infrastructure provision for tap water obtaining and distribution) are assessed in the used databases. Those inputs and outputs at the chain of these processes are included and represented as grey arrows in Figure 9.1.

LCC is sharing the system boundaries with LCA, but under a different perspective: the chain processes are indirectly reflected on the price of the elements. As a matter of fact, although only fertilizer production, transport and application prices (see section 9.2.3.) are taken into account, it is assumed that they are including all the costs for the processes along the life cycle of the product – e.g. mineral fertilizers price is reflecting the costs of energy, raw materials obtaining, water, etc.

**System boundaries for SLCA**

Facing the current state of SLCA methodology and data availability, difficulties are detected to analyze all the processes along the life cycle of the fertilizing alternatives in detail. Systematic databases are not currently available for SLCA, as they are for LCA (ecoinvent, ELCD, national databases, etc.). Therefore, the assessment of systems involves the data collection and the evaluation for all the downstream and upstream flows, being a highly time consuming process.




[a] *Doted arrows are involving transport processes/companies/costs.*
[b] *Capital goods include machinery, buildings and infrastructures.*
[c] *For fertilizers production, emissions are included in costs as technologies to reduce emissions, while no real money flows are occurring for emissions produced during fertilizers application.*
[d] *Sylvite is only included for $KNO_3$ production. HFO, Heavy fuel oil, HDPE, high density polyethylene.*

**Figure 9.1. Compost (a) and mineral fertilizers (b) system boundaries for the three LCA approaches.**

However, a simplified methodological proposal for a partial assessment of social impacts along the chain is used in the paper. The foremost upstream processes (energy, transport, etc.) are included.

Regarding the differences with LCA and LCC, diesel and heavy fuel oil are only indirectly within SLCA limits because social data from crude oil is used for fuels assessment (see section 9.2.4.iv). Waste, emissions and capital goods stages – as well as all the other background processes included in LCA and LCC – are not considered within this SLCA study, although they could be included in the future after the specific data obtaining.

### *(iv) Fertilization alternatives*

Within the framework of previous works of the authors assessing horticulture production in the Mediterranean region with several fertilizing alternatives (Martínez-Blanco et al. 2011a; 2011b; Submitted 2011), three fertilization alternatives are included and compared here: compost, nitric acid and potassium nitrate. The fertilizers are supposed to be applied by a farmer whose fields are located in Spain (Barcelona region).

Compost from source-separated organic fraction of the municipal solid waste (OFMSW), which is collected at nearby municipalities and the area main market, is produced in an in-vessel composting plant with biofiltration of decomposition gaseous emissions. The composting plant is located in Spain (Barcelona region) and the consumer (farmer) is directly buying the product at the plant and transporting it by truck.

Regarding mineral fertilizers, nitric acid ($HNO_3$) is produced in Spain (Tarragona region) and transported to the distributor by truck; later the consumer buys fertilizer and transports it to the fields by van. Nitric acid is widely produced as the production inputs are easily available.

Finally, potassium nitrate ($KNO_3$), it is produced in Israel (Haifa region), mainly using sylvite (from the Dead Sea) and natural gas, and it is transported to the Spanish distributor by freight ship and truck. The consumer buys fertilizer and transports it to the fields by van. Despite the fact that Catalonia is mining potash, potassium nitrate is being mainly imported and the main sources for the region are Italy and Israel.

### *(v) Data quality and sources*

A comprehensive inventory for environmental, economic and social data is required for the study. Data sources for the environmental assessment are referenced in previous works of the authors (Martínez-Blanco et al. 2010; 2011a; 2011b; 2012) and adapted for the current LCSA study definition – system boundaries, functional unit, etc. Specific prices considered for the cost-benefit analysis in Martínez-Blanco et al. (Submitted 2011) are considered here for LCC.

**Data sources and type of data for SLCA**

Regarding SLCA, not only quantitative (T) data is used, as is the case in LCA and LCC. Semi-quantitative (S) and qualitative (L) data also have to be integrated, according to UNEP Guidelines (UNEP 2009).

As social impacts are mainly due to companies and organizations behavior than due to the processes themselves, the necessity for site-specific data and assessment is higher than for LCA (Jørgensen et al. 2008; UNEP 2009). In the study, for mainstream processes, all three data types (country, sector, and company data) are ascertained, whereas for upstream processes, data on country and sector level are considered solely.

The type of social data used (T/S/L and geographical validity) and data sources are pointed out throughout the several tables and explanations about the SLCA methodological proposal application to the case study (section 9.2.4). However, because they are broadly used within the study, Social Hotspot Database (SHDB) and Life Cycle Working Environment (LCWE) are briefly presented in the following.

The SHDB, developed by Benoît and Norris among others (SHDB 2011), is a meta-analysis of the best country (and sector) international social data available. The database is including four social categories – labour rights and decent work, human rights, governance and access to community services – divided in social themes, which are assessed with one or more social issue. Tables are all characterized by level of risk or opportunity that the social impact exists at the country or sector-country-specific level. Characterization levels are determined based on distributions of the data for all countries or by consensus among experts. Working time is proposed by SHDB framework and by other authors as a weighting measure for social impacts aggregation (section 9.2.4.ii). The concept of integrating LCWE in LCA was presented in Poulsen and Jensen (2004). By now, LCWE data is also available in GaBi 5 database (PE International 2011) (Beck et al. 2009; Bos and Beck 2011) and is also used here. Although in the near future more information is expected to be included in the database, nowadays it is offering data on working time (several qualification levels) and lethal and non-lethal occupational accidents. Data is derived from statistics, mainly USA data (PE International 2010).

## 9.2.2. LCA methodology

Environmental assessment is following the same methodology as in previous studies of the authors (Colón et al. 2010; Martínez-Blanco et al. 2011a; 2011b; Submitted 2011). The impact categories included in the study are mid-point indicators. Ten impact categories – abiotic depletion, acidification, eutrophication, global warming, ozone layer depletion, photochemical oxidation, human toxicity and fresh water aquatic, marine aquatic and terrestrial ecotoxicity potentials – which had been defined by the CML 2001 (Guinée 2001), and cumulative energy demand (Frischknecht and Jungbluth 2003), as an energy flow indicator, are considered. The LCA software SimaPro v7.3 (PRé Consultants 2012) is used for the impact analysis, with the obligatory classification and characterization phases defined by the ISO 14044 standard (ISO 2006b). The score for each impact category is assigned by multiplying the amount of substance emitted by the corresponding characterization factor.

## 9.2.3. LCC methodology

The LCC method has been applied for the economic dimension assessment. Guidelines for LCC need further development and only few case studies are available relating LCC to LCSA (e.g. Schau et al. (2011)). According to Hunkeler et al. (2008), LCC assesses "(…) all costs associated with the life cycle of a product that are directly covered by one or more of the actors in the product life cycle (…)". In this study we determine the costs from the perspective of the *actor* farmer (consumer). As the systems under study are located in Spain, the currency selected is euro (€) and costs assumed are valid for period 2008-2011.

Three economic indicators referring to internal costs are selected. These include costs in the upstream processes of production, transport and application of fertilizers – i.e. for fertilizers production, the costs for raw material extraction, energy and transport of mineral fertilizer from factory to the distributor, among others.

Fertilizer market price: The price of the fertilizer that the farmer is paying for. It has to be noted that compost production costs are defrayed by the municipalities paying for the OFMSW treatment and by the consumers of the compost produced, as the composting plant is fulfillling a double function (waste management and fertilizer production). Though, from a farmer – (consumer) perspective the waste treatment function is not relevant and therefore only the price of the fertilizer is used.

Transport costs: The price of transporting the fertilizers from plant (compost) or distributor (mineral fertilizers) to the fields.

Application costs: For mineral fertilizer application the irrigation system is used. Thus, extra costs, specific for mineral fertilizer application, do not occur. If compost is used for fertilizing (and taking into account that no watering differences between fertilizing alternatives are considered), the irrigation costs are the same, but the farmer is incurring extra costs, e.g. for its application. Only those additional costs (specific for compost application) are considered in the LCC study.

Besides internal costs, LCC also includes "(…) externalities that are anticipated to be internalized in the decision relevant future (…)" (Hunkeler et al. 2008). This study, however focuses on internal costs, following the proposal of Swarr et al. (2011) to consider only the real money flows

### 9.2.4. SLCA methodology

A more comprehensive explanation of the SLCA methodology is presented here than for LCA and LCC, as a main goal of the paper is to use social Life Cycle Assessment in a real case study according to the reference documents for SLCA, UNEP Guidelines (UNEP 2009) and Methodological Sheets (UNEP 2010b). Data at three levels is included: country, sector and company level, which are explained in the following (sections 9.2.4.iii – iv). Before, two common issues for the three level-approaches are expounded, the stakeholders and social indicators evaluated and the use of working time as scoring variable.

#### *(i) Stakeholders and social indicators*

The UNEP (2009) methodology has listed 31 subcategories and more than 200 indicators according to stakeholder categories – Worker, Consumer, Local community, Society and Value chain actors –, throughout the life cycle of a product. Because there are not previous studies applying this methodology to fertilizers production and because social background and foreground data is not easily available, it is not feasible to assess all the subcategories and indicators stated in the Guidelines of UNEP (2009) for all the levels.

Therefore, we are reducing the number of subcategories based on our knowledge of the case study and the availability of data – specifications are provided in the following. Similarly, it was not feasible to use exactly the same stakeholders, subcategories and indicators for the several level-approaches. The specific assumptions for each level-approach are discusses in the corresponding section.

The stakeholder Value chain actors is not contained in any of the level-approaches, as all subcategories of this group focus on the behavior of single companies and cannot be applied on sector level in most cases and, regarding company level-approach, no specific data is found for two out of three firms.

#### *(ii) Working time as scoring variable*

As Guidelines (UNEP 2009) are not recommending a specific way to score the relevance of each process in the product chain (sector level), the proposal from several authors is used here.

The amount of working time (related to the functional unit) which is spent on each unit process within the product system is used for the sector level assessment. Those processes which are labour intensive in the life cycle will dominate the results of the SLCA, while those with less working time contribution will be nearly neglected.

Even though the use of working time as activity variable is quite coherent for the stakeholder group Worker, it is less impact-related for the other stakeholders. However, we are considering working time for the scoring as it is the simplest procedure and as it was pointed out as a potential and useful method by several authors (Hunkeler 2006; Norris 2006; Andrews et al. 2009; UNEP 2009; Dreyer et al. 2010; Ciroth and Franze 2011).

### *(iii) Country level*

We are applying and testing SHDB in a practical case, as well as providing other indicators with data from other sources (governmental and non-governmental organizations: ILO, OECD, Human Development Index, etc.), which can be used to complement the SHDB. This assessment is including the corresponding countries for the mainstream companies involved in the production, i.e. Israel and Spain (Table 9.2).

It is in order to assess the levels of risk or opportunity that the social impacts exist at the country level and how is this inducing sector and country data. If the countries involved have considerable differences in their social performance, the companies operating in these countries would reproduce, maybe partially, this variability.

**Stakeholders and social indicators**

The recommended stakeholders in UNEP (2009), apart from Value chain actors, are included in this level (i.e. Worker, Local community, Society and Consumer). SHDB social issues and the added indicators, some of them listed in the Methodological sheets (UNEP 2010b), are arranged to fulfill most of the subcategories suggested by the UNEP (2009) for Worker, while only few subcategories are considered for the other stakeholders.

**Assessment**

The results are presented using the four-level-scale of the SHDB (low, medium, high and very high risk); the assessment criteria are explained in Benoît et al. (2010a). Additional indicators specific for fertilizer assessment are evaluated considering the same scale. For the evaluation, a specific baseline has been defined for each one, which is specified at footnotes of Table 9.2.

### *(iv) Sector level*

Sector level-approach provides with information for indicators not always available for individual companies – without a high investment of time – and serves as a reference value for corporate data. Two types of sectors are included at the sector level-approach: mainstream sectors of the production chain (OFMSW generators and fertilizer producers) and sectors involved in the supply of upstream processes (energy producers, transport providers, water suppliers, plastic producers and sylvite producers). The assessment is done separately for mainstream (Table 9.3) and upstream sectors (Table 9.4 and 9.5), as the level of detail is different.

**Mainstream sectors**

An assessment of the sectors, which are referred to the specific countries where they are located, is provided here for the mainstream sectors involved in the production.

*Stakeholders and social indicators*

The recommended stakeholders in UNEP (2009) – Worker, Local community and Consumer – are included in this level. Value chain actors and Society are not included as they are considered to be even more company related and no relevant sector data is founded. The Consumer for the mainstream sector level-approach is the farmer buying and applying the fertilizers.

Additionally, a new stakeholder is proposed for this case study, the Citizens collecting the OFMSW, as their willingness to collect the waste is crucial for this type of compost production. In previous environmental assessments, no environmental load is given to the food waste, apart from its collection. For SLCA we add some social indicators about waste, such as citizen's efforts collecting it in houses and containers.

We are using some of the subcategories and indicators from the Guidelines; and we are defining some extra subcategories and indicators relevant for fertilizers production (such as "Neighborhood acceptance" within stakeholder Local community and "Working requirements for fertilizers application" within stakeholder Consumer).

An extensive list of subcategories and indicators is proposed that would be assessed for a complete social assessment of fertilizers production (other extra issues could be also assessed if data is available). Nevertheless, in this paper we present results for 20 first indicators (with data for all the sectors) and list the other proposed indicators (Table 9.3). The selection of proposed and filled out subcategories and indicators is according to data availability and to the most relevant issues for the studied systems.

*Assessment*

The social inventory data is compared for the three fertilizing alternatives and for each indicator. Therefore, four scores were defined, with regard of the fertilizer social performance: best, intermediate (results within the best and the worst option ones), worst option and similar values (color scale is applied). Although the inventory for background sectors involved in compost production – i.e. OFMSW collection and Compost production – have been separately collected (additional data in Annex III.I), the assessment, when compost production is compared with the other two alternatives, is made for the both together (Table 9.3).

**Upstream sectors**

This is the first attempt to develop a methodology to include the social performance of upstream organizations. It is mainly based on SHDB data, thus most figures are for social risk impact rather than real impact assessment.

The upstream organizations considered as being especially relevant for the case study are included in the paper: transport providers, including terrestrial and maritime; energy production, including crude oil, electricity and natural gas; and others, including plastic production, water supply and sylvite extraction. However, the same methodology could be applied to include other sectors (such as machinery production, coal, etc).

SHDB is not providing with specific data for diesel and heavy fuel oil production. Apart from that, crude fuel oil is detected in a first screening, as the more working time intensive input for both diesel and heavy fuel oil production and it is the most common form to fuel importation. Therefore, for the time being, only the crude fuel oil sector is assessed, obviating diesel and heavy fuel oil producers. For the inventory, we are considering the amount of crude oil required for diesel and heavy fuel oil production.

*Stakeholders and social indicators*

Although SHDB is designed to include sector data for those social issues considered to be sector related, rather than whole country issues (Benoît et al. 2010a), few sector data is currently available for most of sectors and countries. All the social issues of the SHDB defined to be sector depending are included in Table 9.4, which are mainly the ones included in the social theme Labour rights and decent work. Moreover, two issues related with gender equity and children schooling are added. Additionally to SHDB data, occupational lethal and non-lethal accidents occurrence was also evaluated for each sector, in the specific country when it is feasible. All the indicators assessed are referring to Worker stakeholder (UNEP 2009).

*Assessment*

The same levels of risk or opportunity as defined by the SHDB are used here, whereas for Occupational lethal and non-lethal accidents real numbers are provided instead of risk levels.

When the energy source is imported from other countries (i.e. for crude oil and natural gas) or more than one source of energy is used for the production (i.e. for electricity), the sector-country specific data for the several countries and energy sources is collected and aggregated, using country mixes for Spain and Israel.

For the impact assessment approach of energy sectors, first of all, the four ranges of risk or opportunity from SHDB are translated to numbers: low (1), medium (2), high (3) and very high (4), additionally no evidence (0). Then, a weighted average is calculated using the country mixes. For example, the risk or opportunity ranges for natural gas production in Algeria, Nigeria, Qatar, Egypt, Trinidad-Tobago and Norway, which were the main suppliers for Spain in the period 2007-2009 (INE 2012), are multiplied by the share of each country to the Spanish national mix and summed. Therefore, a score between 1 and 4 is obtained. Scores below 1.5 are labeled "low", scores between 1.5-2.5 are "medium", between 2.5-3.5 are "high" and higher than 3.5 are "very high". Moreover, when half or more of the countries have range 3, we rise one level the score of the average if it is lower than high; while when one third or more of the countries have range 4, we rise one level the averaged score (see Annex III.II).

Accordingly to the aggregation of the social performance for energy sectors, the social performance for all upstream sectors is aggregated in a unique figure for each indicator (Table 9.5) using the working time as weighting variable and the same scoring criteria as in previous paragraph.

### *(v) Company level*

Company level-approach is essential for the investigation of the specific situation of the corporation. For the collection of company data, the first step would be to focus on the hotspots (or main positive impacts) identified at national and sector level. Complementarily, a prioritization of the companies that are more relevant to be assessed (if the time resources are limited) could be done according to the highest shares of working time throughout the production chain (Benoît et al. 2010a).

However, nowadays social company data is not systematically collected and publicly available. Data collection on-site from the individual companies would be highly time consuming and to visit the several companies involved in the chain (usually including several countries) would not be feasible.

Therefore, in this section, a description of preliminary information, regarding the individual company social behavior for Worker, Local community, Society and Consumer stakeholders for mainstream companies, is included. The information, mainly descriptive, is arranged according to Guidelines' stakeholders and subcategories (Table 9.6). The specific data sources are stated at the table footnotes.

Following the assessment method used for mainstream sector level-approach, the companies are compared and, when it is possible, a score is given to each indicator (best, intermediate and worst option) and a color scale is applied.

### 9.2.5. Integration of environmental, social and economic assessments: Presentation of LCSA results

Interpreting the results from LCA, LCC and SLCA in a combined fashion can be difficult and presenting clear LCSA results to compare similar products to support decision-making processes is a key challenge (UNEP 2011).

Few methodologies have been proposed for presenting LCSA results and even fewer have been applied to real case studies. Traverso and Finkbeiner (2009) reported the Life Cycle Sustainability Dashboard (LCSD), a particular adaptation of the Dashboard of Sustainability to present product´s performance. Dashboard of Sustainability was developed by Joint Research Centre of Ispra (Italy) to assess sustainability performance of communities and is now scientifically supported by the International Institute for Sustainable Development (IISD 2012). Other examples for a combined presentation of results from LCA, LCC and SLCA are the Life Cycle Sustainability Triangle (Finkbeiner et al. 2010) and the adaptation of the Sustainability Index for LCSA by Vinyes et al. (Accepted 2012).

Life Cycle Sustainability Dashboard (LCSD) is the only method for presenting LCSA results that so far has been applied in a real case study (Traverso and Finkbeiner 2009; Traverso et al. Accepted 2011). Thus, it is here selected for the sustainability assessment as well.

In the Dashboard of Sustainability software a certain number of indicators and their values, related to different products, can be inserted in order to be able to interpret the results. For the application to LCSA simply the indicator sets used for LCA, LCC and SLCA can be used and implemented. Regarding LCA, the study is assessing 11 indicators (section 9.2.2.) and 3 indicators are considered for LCC (section 9.2.3.). For SLCA – due to the amount and different level-approaches of the indicators considered in the SLCA study – not all the assessed indicators could be handled.

Consequently, for the time being, only Worker stakeholder and sector data are taken into account for LCSA: the indicators for LCC and LCA are reflecting processes from all the chain whereas in our SLCA this is true only for some indicators, all of them addressing the stakeholder group worker. To reflect the whole life cycle (and thus be consistent with LCA and LCC) we give priority to this stakeholder group and those indicators common for upstream and mainstream sector results are chosen.

Therefore, an aggregated result of the most relevant sectors involved in the fertilizer production, including the scores for mainstream (Tables 9.3) and upstream sectors (Table 9.5) according to the shares of working time (Table 9.1), is calculated for the fertilizing alternatives and presented in the LCSD (see Annex III.III). Future assessments should include the whole life cycle along with all the stakeholders proposed in SLCA guidelines.

As it is explained in Finkbeiner et al. (2010) and Traverso and Finkbeiner (2009), the Dashboard ranks all values for each indicator and gives 1000 points to the product with the best performance and 0 points to the worst performance. All other values of the same indicator are linearly interpolated. Although weighting factors to the indicators can be handled by the software, no weighting differences are considered here. The evaluation results for each topic are given by a weighted average of all included indicators values; the overall evaluation is the arithmetical average of the topic evaluations. As for the indicator values the resulting evaluations are obtained by scores (between 0 and 1000) and according colors.

# 9.3. Results and discussion

## 9.3.1. Social LCA results

### *(i) Input-output and working time inventories*

According to the abovementioned system boundaries (Figure 9.1), an inventory of the inputs and outputs calculated per kg of N available, for each fertilizing alternatives, is showed in left part of Table 9.1, which is using data from Martínez-Blanco et al. (2010; 2011a; 2011b; Submitted 2011). In the central column of Table 9.1, the working time needed for each process is showed. The working time rate has being used to calculate the working time inventory, at the right part of the table.

Total working time for compost alternative is more than 20 times higher than for nitric acid one, mainly due to the high working time rate for OFMSW collection and composting. This is partially consequence of nitrogen available concentration in compost that is about 50 times lower than in both mineral fertilizers. Additionally, it is also a consequence of the larger amounts of energy consumed during waste transport and management inside the composting plant and during composting tunnels aeration and biofiltration of gaseous emissions.

In the fertilizing alternatives assessed, the upstream companies contribute with 78% (for nitric acid) and 28% (for compost) to the total working time of the life cycle (Table 9.1), which shows the importance to consider them in the assessment. Because no data is available for the working time rates of potassium nitrate production and sylvite extraction, total working time for potassium nitrate alternative cannot be calculated. However, it is detected that working time for upstream sectors of potassium nitrate is of the same order of magnitude than for nitric acid fertilizing alternative.

**Methodological concerns**

Working time rates are calculated from GaBi 5.0, using average of available data for developed countries, apart from OFMSW collection and composting time. The LCWE from GaBi 5.0 is providing with the working time needed for all the life cycle. For the calculation of the working time occurred during the production stage (e.g. $HNO_3$ production plant), the seconds of work per unit of product required throughout the production chain (e.g. transport, electricity) should be subtracted. However, for confidentiality reasons GaBi 5.0 is not showing the amounts of each process/material in the chain (i.e. electricity amount needed for producing 1kg of $HNO_3$) but the total emissions of the product life cycle. Thus, and despite the fact that considerable differences could exist between databases assumptions, the only current alternative is to use chain inventory data from another database. The inventories per unit of product from the most similar processes in database ecoinvent data v2.2 (SCLCI 2012) are used for the working time subtraction from the production chain.

**Table 9.1. Input-outputs flows, working time rates and working time inventory for the 3 fertilizing alternatives.**

| | **Flows (Unit per kg N available)** | | | | **Working time rate (s per Unit)** | **Working time (s per kg N available)** | | |
|---|---|---|---|---|---|---|---|---|
| **Fertilizing alternatives** | **Compost** | **$HNO_3$** | **$KNO_3$** | **Unit** | | **Compost** | **$HNO_3$** | **$KNO_3$** |
| **MAINSTREAM PROCESSES** | | | | | | **6197.6** | **80.3** | **-** |
| **Compost production** | 0.2 | | | t compost | 33451.4 [a] | 5084.6 | | |
| **OFMSW collection** | 1.1 | | | t OFMSW | 1060.0 [b] | 1113.0 | | |
| **$HNO_3$ production** | - | 7.5 | | kg | 10.7 [c] | | 80.3 | |
| **$KNO_3$ production** | | | 7.2 | kg | nd | | | - |
| **UPSTREAM PROCESSES** | | | | | | **2585.2** | **287.1** | **323.1** |
| | | | | ***Other inputs*** | | | | |
| **Plastic (HDPE)** | 0.1 | | | kg | 522.7 [d] | 47.2 | | |
| **Process water** | 372.0 | 0.02 | 0.1 | L | 0.2 [d] | 78.7 | 0.0 | 0.0 |
| **Sylvite extraction** | | | 6.3 | kg | nd | | | - |
| | | | | ***Energy*** | | | | |
| **Crude oil** | 16.2 | 0.9 | 0.53 | kg | 95.1 [d] | 1540.1 | 85.5 | 50.3 |
| **Electricity** | 36.8 | 0.4 | 0.3 | kWh | 27.9/25.0 [d] | 919.2 | 9.5 | 9.0 |
| **Natural gas** | | 1.1 | 1.5 | kg | 178.8 [d] | | 192.1 | 263.8 |
| | | | | ***Transport*** | | | | |
| **Transport (freight ship)** | | | 21.4 | tkm | 0.21 [d,e] | - | - | - |
| **Transport (rail)** | | 1.6 | 4.1 | tkm | 0.4 [d,e] | - | - | - |
| **Transport (truck)** | 50.4 | 1.7 | 1.2 | tkm | 1.1 [d,e] | - | - | - |
| | | | | | **Total** | **8782.8** | **367.4** | **-** |

*It is produced/consumed in Spain* *It is produced/consumed in Israel*

*nd, no data;"-", Calculations are not feasible (see comment e); HDPE, high density polyethylene.*

[a] *Average working time for Catalan composting plants (1997-2007), from Huerta et al.* (2010). *Regional data.*

[b] *Calculated according to Iriarte et al.* (2009) *and own measures. Regional data.*

[c] *Average life cycle working time in developed countries (GaBi 5.0) is used. The processes in the life cycle (energy and transport) are subtracted. Thus, this is the working time in the company producing the fertilizer. Developed countries data.*

[d] *Average life cycle working time in developed countries (GaBi 5.0). Developed countries data.*

[e] *No consistent results are obtained when the working time from the production chain processes (i.e. energy sources) are subtracted using database ecoinvent data v2.2* (SCLCI 2012). *Transport is not included in the working time inventory. See section 9.3.1.i.*

**Table 9.2. Country level - Social risks for countries involved in the mainstream processes for fertilizer production.**

| STAKEHOLDER > Subcategory (shaded) > Social indicator (white) | Data quality | Spain | Israel |
|---|---|---|---|
| **WORKER** | | | |
| ***Freedom of Association and Collective Bargaining*** | | | |
| Risk of not having freedom of association rights | S [a] | M | M |
| Risk of not having collective bargaining rights | S [a] | M | M |
| Risk of not having the right to strike | S [a] | M | M |
| Potential of country not passing labour laws (number of labour laws) | T [a] | L (1489) | H (135) |
| Potential of country not adopting labour conventions | S [a] | M | M |
| ***Child labour*** | | | |
| Risk of child labour | L [a] | L | L |
| Number of children out of school (%) | T [a] | L (0.33) | nd |
| ***Fair Salary*** | | | |
| Potential of average wage being < minimum wage USD | T [a] | L | L |
| Potential of average wage being < non-poverty guideline USD | T [a] | L | L |
| Potential of minimum wages not being updated (year of last update) | S [a] | L (2010) | M (2008) |
| ***Working Hours*** | | | |
| Risk of population working > 48hours $week^{-1}$ | T/L [a] | M | M |
| Average working hours per week | T | L (38.6 [b]) | M(40-43[c]) |
| Maximum working hours per week in labour laws | T [c] | L (40) | M (48) |
| ***Forced labour*** | | | |
| Risk of forced labour | L [a] | M | M |
| ***Equal opportunities/Discrimination*** | | | |
| Overall fragility of gender equity | T [a,d] | L | M |
| ***Health and Safety*** | | | |
| Occurrence of occupational lethal accidents per year (per 100,000 people) | T [e] | M (4.1) | L (2.9) |
| Occurrence of occupational non-lethal accidents per year (per 100,000 people) | T [e] | VH(5641) | H(2314) |
| **LOCAL COMMUNITY** | | | |
| ***Safe & healthy living conditions*** | | | |
| Deaths due to outdoor air pollution (deaths per million people) | T [f] | M (136) | H (216) |
| Population living on degraded land (%) | T [f] | L (1.4) | M (12.9) |
| ***Access to material resources*** | | | |
| % population having access to improved drinking water | T [a] | L (100) | L (100) |
| % population having access to improved sanitation | T [a] | L (100) | L (100) |
| **SOCIETY** | | | |
| ***Risk of corruption*** | | | |
| % population who faced a bribe situation | T [f] | L (6) | M (11) |
| **CONSUMER** | | | |
| ***Feedback mechanism*** | | | |
| Existence of national entities assuring consumer rights (existence) | L [g] | L (Yes) | L (Yes) |

*L Low* *M Medium* *H High* *VH Very high*

*Data quality: T, Quantitative; S, Semi-quantitative; L, Qualitative; na, not applicable; nd, no data.*

[a] SHDB (2011). *Risk impact ranges (L, M, H and VH) are according to assumptions in Benoît et al.* (2010a).

[b] OECD (2011). *The assessment is according to ILO* (2007), *figures equal or below 40-hour workweek are set low risk; figures up to 40-hour and below 48-hour are set medium risk; figures above 48-hour are set high risk.*

[c] USDS (2011). *For the assessment see footnote b.*

[d] *The indicator is aggregating five recognised index assessing gender equity and including many sub-indicators.*

[e] ILO (2011).. *The assessment for data coming from this source is using quartiles of data for all the countries in the world or with available data (i.e. first quartile corresponds to score low, second quartile to medium, third to high and fourth to very high). The increasing or decreasing order of scores depends on the indicator. Countries with a better reporting are being penalised.*

[f] UNDP (2010; 2011). *For the assessment see footnote e.*

[g] *When they exist, low risk is considered.*

Regarding transport, no consistent results are obtained when the working time from the upstream processes (e.g. working seconds needed for production of 1 MJ electricity or 1 kg crude oil, obtained from Gabi 5.0) are subtracted from the total working time for transport (e.g. working seconds needed for provide 1 tkm transport by train or by truck, respectively, obtained from Gabi 5.0.) using database ecoinvent data v2.2 inventory data (e.g. to determine the amount of electricity or crude oil, respectively, needed for transport). Therefore, transport is not included in the working time inventory. In any case, according to the results, the working time rates of transport are not particularly relevant for the whole life cycle (Table 9.1). Data for working time rates for potassium nitrate production and sylvite extraction are not found in Gabi 5 or other sources and thus they are not taken into account.

### *(ii) Social situation of the countries involved in the mainstream processes of fertilizers chain*

According to results in Table 9.2, more social risks were identified for Israel than for Spain for eight of the scored indicators, which are related to the number of labour laws, working hours, wages being updated, fragility of gender equity, safe and living conditions and risk of corruption. Differences are of one level, for most of the cases. With regard of occupational accidents, the occurrence rates are higher for Spain. For the rest of indicators – 10 out of 23 –, the potential for social risks is similar in both countries.

Specific hotspots, with high or very high risk level, are not detected for these countries, apart from Potential of country not passing labour laws and Deaths due to outdoor air pollution in Israel. Besides, following the assessment method proposed for occupational non-fatal accidents, Israel and Spain have very high and high risk levels. Nevertheless, countries having a better accident counting and reporting system are being penalized when they are compared with the rest of countries. According to abovementioned, if specific hotspots were found they should be mandatorily included in the sector and company assessment.

### *(iii) Social situation of the sectors involved in the processes of fertilizers chain*

Two main difficulties are identified at sector level. First, regarding the data obtaining, specific country-sector data is not available for many indicators as social issues are hardly reported by companies; moreover, the sector division used in country data statistics tend to be not enough detailed and to be different among countries.

**Mainstream sectors**

Among the social indicators proposed, which are according to the ones recommended in UNEP (2009) and to the specific social issues regarding the case study, about 20 indicators are chosen for a preliminary mainstream sector assessment. Table 9.3 is showing the results for the three fertilizing alternatives.

As has been aforementioned, the total mainstream working time needed per kg of N available with compost is much larger than time for nitric acid and potassium nitrate. Therefore, much local employment is promoted per kilo of nitrogen with compost. Nevertheless, according to our scoring approach, nitric acid can be considered as the best alternative from a social perspective, while compost and potassium nitrate are alternatively sharing worst scores.

As it is pointed out in the previous section, the two countries involved for mainstream sectors are quite similar regarding worker right indicators. It is also visible at sector level results. The main differences for Worker are found for *Health and safety* and for *Overall fragility of gender equity*.

Nearly no evidence of Local community health and safety inconveniences is found in the literature for mineral fertilizers production, while some nuisances are reported for compost production (ARC 2012). However, more studies have been found for waste collection and composting addressing Local community and Worker hazards than for mineral fertilizers, as the former is a type of plant that is usually rejected by the communities living close to it.

From a farmer (Consumer) perspective, with compost alternative the farmer is closer to the production point and the costs per kg of N available are lower, even though extra work is needed for its application.

**Upstream sectors**

Table 9.4 and 9.5 show an example of the proposed methodology (section 9.2.4.ii), based on working time and SHDB social issues, for the social risk assessment of upstream sectors.

According to the results in Table 9.4 for the chain processes included in each of the three fertilizing alternatives, energy sectors seem to be the most relevant sectors for the social dimension. They are contributing in a larger extent to working time necessities and are the sectors involving more countries, which are the importers of the energy sources. Moreover, energy sectors risk assessment is showing social hotspots for *Freedom of association & collective bargaining* sub-category and *Overall fragility of gender equity* and *Risk of forced labour* indicators, among others. The risks identified for the sectors electricity and other inputs (Table 9.4) using the SHDB are similar between countries.

As was aforesaid, no working time data is available for transport provision and for sylvite production, thus it is not included in the aggregated upstream social risk table (Table 9.5).

Regarding the aggregated social risks, for most indicators, the scores for the upstream sectors of the three fertilizers are similar (Table 9.5). Half of indicators are scored high or very high for the three alternatives, mainly due to the social performance of the countries involved in natural gas and crude oil importation.

SHDB developers had pointed out their intention to improve the database (Benoît et al. 2010a), thus with new versions of the SHDB more detailed results per sectors and for new sectors would be obtained.

### *(iv) Social situation of the companies involved in the processes of fertilizers chain*

Using the data from country and mainstream sectors for compost, nitric acid and potassium nitrate production we have theoretically estimated the social impacts risks of companies producing the fertilizers. The same indicators as in the sector and country level-approaches can be used to assess the companies´ behaviour.

**Table 9.3. Mainstream sector level - Comparison of social performance of three fertilizing alternatives involved in the production chain of fertilizers. Apart from the assessed social indicators by SHDB (2011), other ones are proposed.**

| STAKEHOLDER > Subcategory (shaded) | Fertilizing alternatives | Compost | | $HNO_3$ | $KNO_3$ | Data quality |
|---|---|---|---|---|---|---|
| | **Mainstream processes** | **OFMSW collection** | **Compost production** | **$HNO_3$ production** | **$KNO_3$ production** | |
| | **Working time (s per kg N available)** | 1113.0 | 5084.6 | 80.3 | nd | |
| | **Social indicator** | Spain | Spain | Spain | Israel | |
| **WORKER** | | | | | | |
| **Freedom of Association and Collective Bargaining** | Potential of sector not passing labour laws | B | | B | W | T |
| | Potential of sector not adopting labour conventions | S | | S | S | S |
| | Others: Risk of not having the right to strike; Risk of not having collective bargaining rights; Risk of not having freedom of association rights | | | | | |
| **Child labour** | Risk of child labour in the sector | S | | S | S | L |
| **Working Hours** | Average working hours per week | [a] | | [a] | [a] | T |
| | Others: Work-life balance situation | | | | | |
| **Forced labour** | Risk of forced labour | S | | S | S | L |
| **Equal opportunities/Discrimination** | Overall fragility of gender equity (% women total workers$^{-1}$) | W | | B | I | T |
| | Others: Ratio of basic salary of men to women by employee category; Ratio of immigrant employees (%); Ratio of basic salary of immigrants to the rest by employee category | | | | | |
| **Health and Safety** | Gaseous emissions exposure effects | [a] | | [a] | [a] | T/L |
| | Biological agents exposure effects | W | | B | B | L |
| | Occurrence of occupational lethal accidents | W | | B | nd | T |
| | Occurrence of occupational non-lethal accidents | W | | B | nd | T |
| | Others: Biological agents protection and prevention measures; Workers comfort level; Level of noise; Presence of a formal policy concerning health and safety in the sector | | | | | |
| **LOCAL COMMUNITY** | | | | | | |
| **Safe & healthy living conditions** | Odor and gaseous emissions effects | nd | W | B | B | T/L |

| | | | | | | |
|---|---|---|---|---|---|---|
| | Biological agents exposure effects | nd | W | B | B | T/L |
| | Other hazards and nuisances | [a] | | [a] | [a] | |
| | Others: Biological agents protection and prevention measures; Level of noise; Emissions and noise records are recommended or mandatory for the company. | | | | | |
| **Local employment** | Promotion of local employment in the consumption area | B | | B | W | L |
| | Others: Training courses for the employees; % Employees with Higher education; % Employees with Basic education; % spending on locally-based suppliers | | | | | |
| **Neighbours acceptance** | Others: rate of willingness to have the sector close to home; participation of neighbours in decisions and incomes. | | | | | |
| **CONSUMER (FARMER)** | | | | | | |
| **Supplier relationships** | Fertilizer production scale with regard of consumer | B | | I | W | S |
| **Health and Safety** | Product application dangers | [a] | | [a] | | L |
| | Others: Existence of health and safety measure labels for application in the product | | | | | |
| **Requirements for fertilizers application** | Extra working time for consumer to apply the product | W | | B | | S |
| | Others: Level of complexity for dosages calculation. | | | | | |
| **Consumer acceptance** | Average prices in Catalonia (€ per kg N available) | B | | I | W | T |
| | Others: Main consumer concerns about the product | | | | | |
| **CITIZENS COLLECTING OFMSW** | | | | | | |
| **Education and responsibility** | Others: Existence of obligation of waste collection for citizens; Existence of educational campaigns for citizen's engagement. | | | | | |
| **Comfort and collecting effort for the citizens** | Others: Frequency of organic bin emptying; % public space used; % private space used. | | | | | |
| **Acceptance and willingness of citizens to collect organic waste** | Amount of organic waste collected (%) | [a] | | na | na | T |
| | % of improper materials in the organic waste | [a] | | na | na | T |

*B Best option* · *I Intermediate option* · *W Worst option* · *S Similar values*

*T, Quantitative; S, Semi-quantitative; L, Qualitative; nd, no sector data; na, not applicable (see section 9.2.4.iv).*

*Social inventory data, data sources and assumptions for Table 9.3 are comprehensively presented in Annex III.I.*

*[a] The inventory results are not comparable because either not enough information is found, very different type of data, units, etc.*

**Table 9.4. Upstream sector level - Social risks and occupational accidents for sectors and countries involved in the upstream chain processes. The working time needed for each process is included. Data from SHDB (2011) is used.**

| | Transport | | | Energy sources | | | | | | Other inputs | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Upstream processes** | **Terrestrial** | | **Water** | **Crude oil[b]** | | **Natural gas[b]** | | **Electricity** | | **Plastic** | **Water** | | **Sylvite** |
| **Country using the upstream process** | **I** | **S** | **I** | **I** | **S** | **I** | **S** | **I** | **S** | **S** | **I** | **S** | **I** |
| **Compost (working s per kg N available)** | na | na | na | 0.0 | 1540.1 | 0.0 | 0.0 | 0.0 | 919.2 | 47.2 | 0.0 | 78.7 | na |
| **$HNO_3$ (working s per kg N available)** | na | na | na | 0.0 | 85.5 | 0.0 | 192.1 | 0.0 | 9.5 | 0.0 | 0.0 | 0.0 | na |
| **$KNO_3$ (working s per kg N available)** | na | na | na | 49.2 | 1.1 | 263.8 | 0.0 | 9.0 | 0.0 | 0.0 | 0.0 | 0.0 | na |
| **STAKEHOLDER > Subcategory (shaded) > Social indicator (white)** | | | | | | | | | | | | | |
| **WORKER** | | | | | | | | | | | | | |
| ***Freedom of Association & Collective Bargaining*** | | | | | | | | | | | | | |
| Potential of country not passing labour laws | H | L[a] | M[a] | L | M | M | L | H | H | L | H | H | H |
| Potential of country not adopting labour conventions | M | M | M[a] | L | M | M | M | L[a] | L[a] | L[a] | L[a] | L[a] | M[a] |
| Risk of not having freedom of association rights | M | M | M | H | H | H | H | M | M | M | M | M | M |
| Risk of not having collective bargaining rights | M | M | M | H | VH | H | H | M | M | M | M | M | M |
| Risk of not having the right to strike | M | M | M | VH | VH | VH | VH | M | M | M | M | M | M |
| ***Child Labour*** | | | | | | | | | | | | | |
| Risk of child labour | L | L | L | M | M | L | M | L | L | L | L | L | L |
| Number of children out of school | L | L | L | M | M | L | H | L | L | L | L | L | L |
| ***Fair Salary*** | | | | | | | | | | | | | |
| Potential of minimum wages not being updated | M | L | M | L | L | na | na | M | L | L | M | L | M |
| ***Working Hours*** | | | | | | | | | | | | | |
| Risk of population working > 48 hours week$^{-1}$ | M | M | M | L | M | M | L | M | M | M | M | M | M |

| | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| ***Forced Labour*** | | | | | | | | | | | | | |
| Risk of forced labour | H [a] | H [a] | H [a] | H | H | H | H | H [a] | H [a] | H [a] | H [a] | H [a] | H [a] |
| ***Equal opportunities/Discrimination*** | | | | | | | | | | | | | |
| Overall fragility of gender equity | L [a] | L [a] | L [a] | M | VH | VH | VH | M [a] | H [a] | H [a] | M [a] | H [a] | M [a] |
| ***Health and Safety (cases per h of work)*** | | | | | | | | | | | | | |
| Occurrence of occupational lethal accidents | 4.8E-08 [c] | 4.8E-08 [c] | 4.8E-08 [c] | 2.5E-08 [d] | 2.5E-08 [d] | 2.4E-08 [d] | 2.4E-08 [d] | 2.4E-10 [b] | 1.9E-10 [b] | 1.7E-08 [e] | 2.8E-08 [d] | 2.8E-08 [d] | na |
| Occurrence of occupational non-lethal accidents | 1.3E-05 [c] | 1.3E-05 [c] | 1.3E-05 [c] | 4.2E-05 [d] | 4.2E-05 [d] | 4.2E-05 [d] | 4.2E-05 [d] | 2.4E-10 [b] | 1.9E-10 [b] | 3.5E-05 [e] | 4.0E-05 [d] | 4.0E-05 [d] | na |

| | | | |
|---|---|---|---|
| *L Low* | *M Medium* | *H High* | *VH Very high* |

*I, Israel; S, Spain; na, data not available (see section 9.2.4.iv).*

[a] *Specific sector data for the country is used. When no data by sectors is available, the country average data is used, following the procedure of the SHDB* (Benoît et al. 2010a).

[b] *Weighted average data according to the country mixes for crude oil and natural gas imports (see section 9.2.4.iv). For the calculation, specific sector and country data is used.*

[c] *ILO* (2011); [d] *PE International;* [e] *MTI* (2011).

**Table 9.5. Upstream sector level - Aggregated social risks** (SHDB 2011) **and occupational accidents for the background chain processes.**

| **Fertilizing alternative** | **Compost** | **$HNO_3$** | **$KNO_3$** |
|---|---|---|---|
| **Total working time upstream (s per kg N available)** | 2585.2 | 287.1 | 323.1 |
| **STAKEHOLDER > Subcategory (shaded) > Indicator (white)** | | | |
| **WORKER** | | | |
| ***Freedom of Association & Collective Bargaining*** | | | |
| Potential of country not passing labour laws | M | M | M |
| Potential of country not adopting labour conventions | M | M | M |
| Risk of not having freedom of association rights | H | H | H |
| Risk of not having collective bargaining rights | H | H | H |
| Risk of not having the right to strike | VH | VH | VH |
| ***Child Labour*** | | | |
| Risk of child labour | M | M | L |
| Number of children out of school | M | H | L |
| ***Fair Salary*** | | | |
| Potential of minimum wages not being updated | L | L | L |
| ***Working Hours*** | | | |
| Risk of population working > 48 hours $week^{-1}$ | M | L | M |
| ***Forced Labour*** | | | |
| Risk of forced labour | H | H | H |
| ***Equal opportunities/Discrimination*** | | | |
| Overall fragility of gender equity | VH | VH | VH |
| ***Health and Safety (cases per h of work)*** | | | |
| Occurrence of occupational lethal accidents | 1.6E-08 | 2.4E-08 | 2.4E-08 |
| Occurrence of occupational non-lethal accidents | 2.6E-05 | 4.0E-05 | 4.0E-05 |

*L Low* *M Medium* *H High* *VH Very high*

*na data not available (see section 9.2.4.iv).*

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**Table 9.6. Company level - First indications about the individual companies social performance.**

| Fertilizing alternatives | Compost producer[a] | $HNO_3$ producer | $KNO_3$ producer |
|---|---|---|---|
| **Region (country)** | Barcelona (Spain) – More than one plant | Tarragona (Spain) – More than one plant | Haifa (Israel) - More than one plant |
| **STAKEHOLDER > Subcategories and social indicators (shaded)** | | | |
| **WORKER** | | | |
| **Freedom of association and collective bargaining** [1] | It exists | It exists | It exists |
| **Fair salary and working conditions** | No data | In 2009, due to economical reasons, a relevant amount of workers were fire permanently or temporarily [2]. | In 2011, a united struggle and a six months strike of workers asking for 'contract labour', better labour facilities conditions and decent wages are reported [3,4]. |
| **Health and Safety** | No data | Higher incidence of related illnesses was proved for workers in one of the plants, which in the past released high amounts of mercury among other toxic substances [1,2].<br>The incidence of occupational accidents in the company was more than 80% lower than national average for the sector (in 2010) [5]. | The cancer rates among workers are very high [6]. |
| **LOCAL COMMUNITY** | | | |
| **Access to material resources** | Measure for pollution prevention are taken: Several devices to reduce and control the emissions from the organic matter decomposition are provided. The emissions are controlled and above the limits [8]. | Several toxic substances releases to a river and to the atmosphere are reported for the several plants [2,8] | Insufficient measures for pollution prevention are taken. Several court cases for dumping toxic sludge in a close Bay (90s-00s) and in 2011 the government voided the company's permits [6,9]. |
| **Safe & healthy living conditions** | A lot of complains from neighborhood do to nuisances (as odor and noise) has recently caused the plant closure [8]. | Higher incidence of related illnesses was not proved for local residents [1]. | The cancer rates among locals are higher than usual (mainly fisherman and diving commandos) [6]. |
| **Local employment** | Local employment is promoted in the consumption area | Local employment is promoted in the consumption area. | Local employment is not promoted in the consumption area. |
| **SOCIETY** | | | |

| | | | |
|---|---|---|---|
| **Quality, safety and environmental standards** [5] | ISO 9001 | ISO 14001; ISO 9001; OHSAS 1800; Responsible Care; EMAS | ISO 14001; ISO 9001; TA LUFT 2002; OHSAS 1800 |
| **Global Compact Commitment** [10] | No | Yes (since 2002) – Several related social responsibility programmes. | No |
| **CONSUMER** | | | |
| **Farmer adjacency to point of primary production** [7] | The producer is in the adjacency area of the consumer (66 km) | The producer is in the adjacency area of the consumer (3020 km). | The producer is not in the adjacency area of the consumer (152 km). |
| **Additional effort needed in application** [7] | Extra working time needed: 94.3 s per kg N available | No extra time needed | No extra time needed |
| **Application information at the company website** [5] | No data for product application is available. | No data for product application is available. | Detailed data for product application is available. |
| **Transparency** [5] | Hardly any information regarding social and environmental performance is available at company website | Plenty information regarding social and environmental performance is available at company | Some information for environmental issues; no information for social performance is available at company website. |

*Best option* *Intermediate option* *Worst option*

*(see section 9.2.4.v)*

[a] *The OFMSW collectors involve more than one company. In Catalonia it is usual that each municipality contracts each own collecting waste company.*

[1] *According to the several sources consulted;* [2] *www.elpais.com,* [3] *www.haaretz.com;* [4] *www.globes.co.il;* [5] *company websites;* [6] *Greenpeace* (2002); [7] *Martínez-Blanco et al.* (Submitted 2011); [8] *www.elperiodico.com;* [9] *MEP* (2012); [10] *GCC* (2012).

We could make an evaluation of those companies, in order to set whether they have a better or worse performance than specific sector average and a better or worse performance than country average. Additionally, different companies could be compared between them (company A better than company B). Positive impacts can be communicated to promote this company, negative ones to support improvement (Polonia Giese 2012).

As the goal of the study is also to compare different fertilizing alternatives under a life cycle perspective the paper concentrates on upstream processes characterization rather than specific company data collection. Nonetheless, company specific assessment would be mandatory for a social assessment when the goal is to compare, for example, two competing waste collecting companies.

No data on-site is collected here for the company level-approach, which is limited to mainstream companies. All the same, the several sources consulted could give a preliminary idea of the social performance of the real companies involved in the production of the fertilizers (Table 9.6). As most of the data used is qualitative and no available data is found for all the companies, the comparison with sector and country data is not feasible. Table 9.6 presents the social inventory for the three fertilizing companies and for each indicator (see section 9.2.4.v). A clear comparison is not feasible as for some social issues (e.g. Health and Safety) defined criteria are needed.

Although not enough data is provided in Table 9.6 to compare the social behavior of the companies regarding Worker, the company producing potassium nitrate seems to be a worse alternative from Worker and Local Community perspective than the other two fertilizing alternatives. Both mineral fertilizers are keeping more social and environmental standards and potassium nitrate seems to be better from a Consumer perspective than the other two alternatives.

### *(iv) Interpretation of SLCA results according to the three level-approach*

SLCA has been used to describe social performance of countries (section 9.3.1.ii), sectors (section 9.3.1.iii) and companies (section 9.3.1.iv) involved in the production chain of fertilizers.

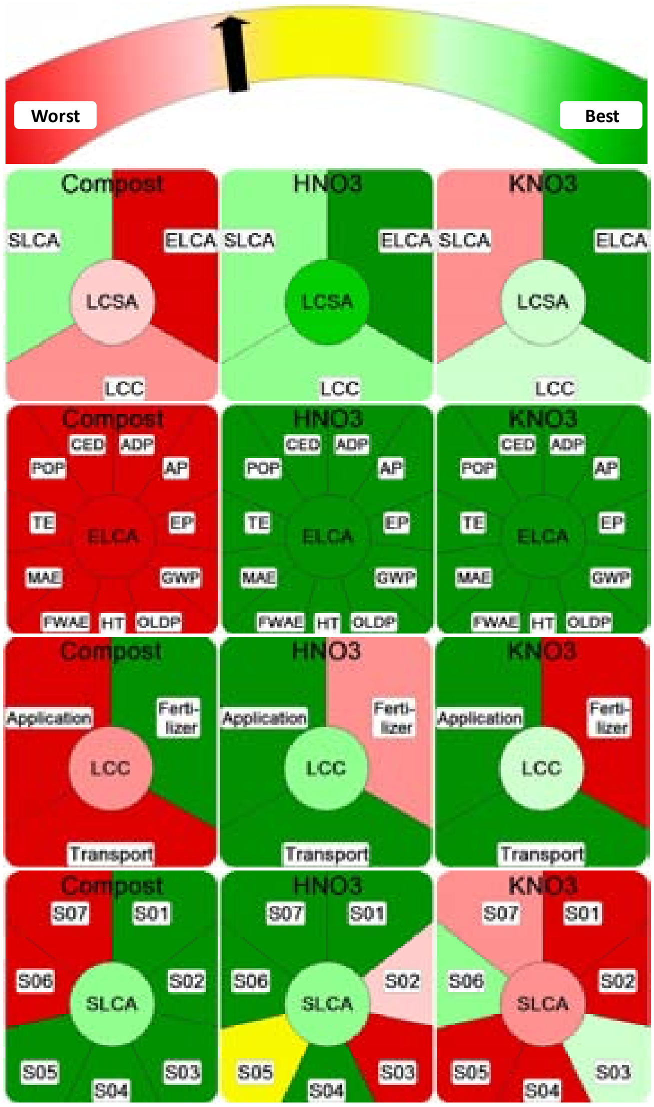
According to the results obtained in the study, from a Spanish decision maker perspective, country level-approach would indicate Spain products (compost and nitric acid) as the best options, as worse social impacts are identified for Israel (potassium nitrate). Nevertheless, country data should be used as baseline for sector and company data results rather than for a direct comparison, because the development level of countries would of course affect their social performance and that is not a reason to stop producing in developing countries. The focus could be on either choosing those companies behaving better than sector averages or on promoting concrete measures to improve social performance of companies involved in current fertilizer production.

Regarding the sector level-approach, similar upstream sectors are involved, and similar risks occur but working time is different for sectors involved in the different fertilizing alternatives (Table 9.5). Thus, less social impacts are expected for fertilizers production, as fewer working time is needed (maybe decision relevant). For mainstream sectors, nitric acid seems to be the best option than from a social perspective, according to the assessed indicators and the data available (Table 9.3).

However, compost is promoting local employment and is creating more work necessities than mineral fertilizers.

At the company level-approach, preliminary results are showing worse social performances for mineral fertilizers companies for Worker and Local Community stakeholders. Whereas they seem to be the best option for Society indicators and potassium nitrate obtain better results than the other two fertilizing alternatives.

### 9.3.2. Life Cycle Sustainability Assessment




*S01 Potential of country not passing labour laws, S02 Potential of country not adopting labour conventions, SO2 Risk of child labour, SO3 Risk of forced labour, SO4 Overall fragility of gender equity, SO5 Occurrence of occupational lethal accidents, SO6 Occurrence of occupational non-lethal accidents, P Fert Fertilizer market price, P Tran Transport costs, P Appl Application costs.*

**Figure 9.2. Presentation of the three dimensions and LCSA results for fertilizing alternatives: compost, potassium nitrate ($KNO_3$) and nitric acid ($HNO_3$).**

Although working time information is not fully provided for potassium nitrate, as working time for mainstream sectors has not been found, we give priority to the representation of the three fertilizing alternatives in the Life Cycle Sustainability Dashboard. Besides, full data is available for the LCA and LCC of the three fertilizing alternatives. Therefore, the next assumption is considered for the use of SLCA results at the LCSA: the mainstream working time stated for nitric acid – i.e. 80.3 s per kg N available – is used also for potassium nitrate aggregation of its own social results.

As it was stated by Martínez-Blanco et al. (2010; 2011a; 2011b; Submitted 2011), for the environmental dimension, compost is the worst option, regardless the impact category, whereas results for nitric acid and potassium nitrate are in the same order of magnitude. Nitric acid has lower impacts for most of the categories. Quantitative values are given in Annex III.IV.

Regarding the LCC evaluation (Martínez-Blanco et al. Submitted 2011), even though the price of compost is lower than the price of nitric acid, the other two cost indicators, related with transport and application, are higher for compost. Potassium nitrate has higher price than nitric acid for fertilizer price while transport and application costs are nearly equal for both mineral alternatives.

Finally, higher variability among fertilizing alternatives and the assessed social indicators – i.e. those related with Worker stakeholder and assessed thorough the life cycle of their production – is detected for SLCA. Compost has a better performance for the indicators related with labour rights, whereas higher occupational accidents are located for compost than for the two mineral alternatives. Nitric acid seems to be a worse alternative regarding *Potential of country not adopting labour conventions* and *Risk of child labour* and the best option for most of the other social indicators. Regarding the other mineral fertilizer, potassium nitrate is the worst alternative for four of the assessed indicators, mainly related with worker rights.

Finally, regarding the aggregation of the results in the LCSA representation (Figure 9.2), compost is truly the worst option for LCA and also for LCC, while it and nitric acid have the best performance for the social dimension. As a matter of fact, nitric acid is considered as a better fertilizing alternative for the three dimensions.

## 9.4. Recommendations and further research

Since social impacts play a major role in sustainability assessment and as there is no commonly agreed methodology so far, every effort to go further in the application for real case studies, even if—due to methodological and practical restrictions—only some aspects of (social) sustainability are addressed, is highly recommended. The set of indicators considered here and the methodology applied throughout data collection and assessment could be helpful in future agricultural and waste management LCSA studies

### *Application of SLCA Guidelines and SLCA 3 level-approach*

Regarding the level of approach, social assessment of country and sector data ideally lead to the identification of the main hotspots and complementarily the study at company stage could focus on them and be less time consuming. Besides, country data use should be restricted to hotspot identification rather than real social behavior evaluation, provided that data is available. Considering average data as a good indicator of companies' performance would lead to the punishment of those companies behaving better than the average and to the hiding of companies with a worst behavior than the national averages.

Unfortunately, if the whole life cycle of a product is to be included in the social assessment, it is currently not feasible to collect all the data at the company level, as too many companies would be involved. Thus, a balance between specific primary data and life cycle perspective should be considered while data collection method is designed.

### *Evaluation/scoring assessment methodology*

For a lot of social indicators no definition of what is a good or a bad performance is agreed in the international community. To evaluate the social performance often additional criteria and reference points are needed. Those can vary significantly between sectors (e.g. a low share of female workers in a certain sector does not necessarily indicate a risk for discrimination but can result simply from the type of work, for instance mining)) and regions. To develop framework conditions as well as regional, national targets in the respective sectors and regions can be used to define appropriate criteria to enable interpretation of the social assessment.

Two evaluation approaches are used here: the scoring of the results according to international conventions and current average situation and the scoring related to the comparison among alternatives. Although it is not used in the paper, another way to deal with the assessment would be to relate sector data to country data, e.g. if sector data (for compost) are better/equal/worse than country data, then compost has a better/neutral/worse effect regarding that social indicator. A similar idea could be used to assess company data, with regard of country and sector data. Future assessments should try to use similar or related scoring systems for the several level-approaches assessed.

### *Assessment of upstream processes – the use of SHDB and working time scoring*

The current study could be claimed as one of the first examples for applying the SHDB to a real case study. Most of previous SLCA are not including most processes of the life cycle but the final processes, therefore they can be claimed to be more an input-output balance rather than a real LCA approach.

According to the results for the fertilizing alternatives, the social performance of upstream companies, such as transport, water and mainly energy sources, could represent a relevant share of the total working time invested in a product. Therefore, if a Life Cycle Assessment is being performed, these companies should be included in the system boundaries. A relatively simple method is proposed here using data from SHDB among others and working time as weighting variable.

SHDB is good starting point. However, several indicators are not filled with real data but extrapolations (mainly from US and developed countries) and hardly any sector-country specific data is provided yet. Besides, as the SHDB is still under development, obtaining sector data from the SHDB website is not always possible due to technical problems.

Besides, for some indicators at the SHDB (such as *Potential of country not adopting labour conventions* or *Risk of forced labour*) the country risk is higher than the risk considered for all the sectors. For example, for Iran, the Potential of Country not adopting Labour Conventions of the country is very high whereas all its sectors have the level risk medium. Despite being explained in Benoît et al. (2010a), who argue that the reason is that no evidence of risk was found for the specific sectors, it not make sense if then the country level (that is supposed to be an average of all the sectors) has a major risk score.

PE International database (from GABI5) is providing with working time data (i.e., seconds of work throughout the life cycle of the production of one unit of the product). Nevertheless, because the contribution of the several processes in the database to the production chain is not available for the user, it is difficult to adapt this data when the social performance of a specific step (company) in the chain is assessed. Although data for extra countries was added in the last launch of the database, no specific data for all the regions or countries is available (e.g. although specific processes are provided for Spain, no data is currently available for Israel); which is a common concern with LCA.

### *LCSA on fertilizer production – Life Cycle Sustainability Dashboard*

The assessment of systems taking into account the three dimensions of sustainability is still a hard work to perform as two of the tools used, LCC and SLCA, are not enough developed to provide indicators sufficiently representative and accurate. Besides, social data is difficult and highly time demanding to collect.

Once data availability and assessment methods were enough agreed for all the three dimensions, the Life Cycle Sustainability Dashboard (LCSD) would be a good tool to present and interpret the results, as it is shown for the current case study.

Regarding the inclusion of social indicators, two approaches are proposed by the authors. The methodology applied here is the inclusion of social indicators that could be aggregated throughout the life cycle of the system; however, it could not be done at company level, as abovementioned, data at this stage is difficult to obtain for all the life cycle. Additionally, if higher importance is given to the specific behavior of companies rather than to the product performance along the chain, another option could be to include only those indicators that have differences between the compared companies, decisions are then only based on which is better or worse.

### *Further work*

No attention is paid in the study to the other nutrients, apart from nitrogen, that are applied with compost. Although nitrogen is usually the nutrient used for dosage calculation, an extra contribution of potassium and phosphorus in soils occurs with compost and it could induce to a reduction in the potassic and phosphoric mineral fertilizers necessities. Besides, compost has been considered as one of the sources for phosphor recycling (Cordell et al. 2009), a relevant issue for agriculture running, as there is an international shortage of this mineral crucial for agriculture fertilization.

The composting of OFMSW is a way to divert organic waste from being landfilled, apart from providing fertilizer, which is not the case in the production of mineral fertilizers. To take it into account, in previous environmental studies the boundaries of the system were expanded and the environmental burdens of dumping organic waste, an alternative treatment to this waste flow, were subtracted from those fertilizing treatment that included composting (Martínez-Blanco et al. 2011a; 2011b). The current paper is not including this assumption aiming to simplify the SLCA study and because from a farmer perspective (our consumer), composting having an extra function is not relevant for their decision. However, if we took it into account, it would certainly lead to a better sustainable performance of compost.

We are not including effects on tomato consumers as they are not currently affecting farmer decisions with regard of fertilization. For the time being, the use of compost from OFMSW in agriculture is restricted to different types of agriculture than organic one – although Regulation CCE 889/2008 (EC2008a) allows the application of this type of compost provided the quality of the product is within the limits, compost from OFMSW is not a common fertilizer in Catalan organic agriculture exploitations (Arco and Romanyà 2010). Therefore, the consumer is not aware about the choices regarding the fertilizers. Moreover, further work on the effects of compost application should be performed; exemplarily Martínez-Blanco et al. (2011a) reported higher nutritional value in cauliflowers when compost was applied. If special label is addressed and major value is given to this type of production, the use of compost in the surroundings of cities (where a continuous provision of this fertilizer is guaranteed and high transport burdens are not necessary) could be a way to obtain high valuable products and to reverse the currently trend to agriculture endangered in urban areas. Moreover, it would improve the organic matter content of soils, which is a common concern for Mediterranean soils among others.

# Chapter 10. Regional assessment of waste flow eco-synergy in food production: using compost and polluted ground water in Mediterranean horticulture crops

**Chapter 10** is based on the following book chapter:

Martínez-Blanco, J., Muñoz, P., Rieradevall, J., Montero, J.I., & Antón, A. (2011) Regional assessment of waste flow eco-synergy in food production: using compost and polluted ground water in Mediterranean horticulture crops. In: Matthias Finkbeiner (ed.) Towards life cycle sustainable management. Springer, New York, 318-330. ISBN-978-94-007-1898-2.


# Abstract

The potential eco-synergetic effects of using two waste flows for the substitution of mineral fertilizers is assessed from nutrient and environmental points of view. The two wastes are: composted organic municipal waste (slow release of nutrients) and nitrate polluted water (rapid nitrogen release). Catalonia is selected as a representative Mediterranean area of study. Macro-data at county level was used for the calculations, Geographic Information System, for the illustrations, and IPCC impact factors, for the environmental quantification. Compost and polluted water are able to supply 35-50% of the nutrient demand of Catalan horticulture production (330,000 tons of horticulture products per year), leading to reduction of 46% of the global warming potential of mineral fertilizers production. More mineral fertilizers are saved in urban and agriculture intensive areas.

# 10.1. Introduction

Intensive horticulture has produced increasing economic and social benefits and a more efficient use of resources; nevertheless, the increase of inputs has had bad consequences for the environment. Two of the main problems derived from the high use of mineral fertilizers are the loss of nutrients and the resulting pollution of aquifers. Most intensive horticulture areas in Europe have been declared regions vulnerable to nitrate pollution (EC 2006). Mild winter climates concentrate a major vegetable crops production, because weather is generally more favorable.

On the other side, the European Directive 2008/98/EC (EC2008b) on waste settled that the Member States should take measures for the treatment of their waste in line with the waste hierarchy, which considers recycling as one of the priority options. Therefore it is necessary to reduce the amount of the organic fraction of the municipal solid waste (OFMSW) being dumped, in order to minimise environmental impacts, and also the loss of organic resources.

Composting is one of the most broadly used OFMSW treatments in Europe and in the world (Boldrin et al. 2009). However, composting plant managers usually came across with the rejection of farmers to apply the organic product in their fields. One reason is that nutrients from compost are not immediately available for the plant after compost application in soils, but they are slowly released in the long-term, besides, there is a lack of training in its application. It could lead to shortage of nutrients during plant growth and, therefore, lower yields.

Nevertheless, an adequate use of composted products, together with a readily available nitrogen supplementation, have proved to reach similar yields as conventional fertilization (Favoino and Hogg 2008; Hargreaves et al. 2008; Diacono and Montemurro 2010; Martínez-Blanco et al. 2011b), saving a part of the economic and environmental costs of produce and use inorganic fertilizers. Highly nitrate polluted water may be a potential source of rapid nitrogen, apart from mineral fertilizers, and its use could mean a decrease in the concentration of contaminants.

Fertilization decisions are extremely relevant, as fertilization production has been reported as one of the determining factors for the environmental performance of horticulture products, particularly for those with low energy consumption (Martínez-Blanco et al. 2011a; 2011b).

Taking into account the principles of the industrial ecology, the aim of this paper is to study the potential eco-synergetic effects of using two waste flows, composted organic municipal waste and nitrate polluted water, which are slow and rapid sources of nutrients, respectively. We assess the maximum compost production in the area of study and state its potential use for supplying, together with the nitrogen in ground water, the nutrient demand of the horticulture sector. The results include the balance of nutrients and the global warming potential avoided due to mineral fertilizers savings. The area of study, Catalonia, was selected as representative of other Mediterranean regions due to the high population density, and therefore, large production of wastes; because it has firmly committed for composting as the major treatment for OFMSW (ARC 2007); because it has an annual production of almost 330,000 tons of horticulture products (DARPAMN 2012); and because it presents relevant levels of nitrates in ground water.

This paper provides also a good example of the potentialities of the joint use of Geographic Information System and Life Cycle Assessment.

# 10.2. Methods and area of study

Area of study, data sources and considerations for the calculations and tools used for the assessment are briefly explained below. The information was processed at county level.

## 10.2.1. Area of study

Catalonia is an autonomous region located at the north-east of Spain and with a total area of about 32,000 km$^2$. It borders on France to the north and on Mediterranean Sea to the south. At the beginnings of 2011, it had more than 7.5 million of inhabitants. The region of study is divided into 41 counties of 145-1,785 km$^2$.

### *(i) Horticultural production and nutrient demand*

The productions and areas cultivated per horticulture crop (for instance, tomato, onion, lettuce, etc.) and per county are obtained for 2008 (DARPAMN 2012).

Regarding the calculation of nutrient demand (nitrogen, phosphorus and potassium), fertilization recommendations for integrated cultivation management are used (DARPAMN 2010; MAMRM 2010).

### *(ii) Potential nutrients from OFMSW compost*

The total amount of organic waste from households, restaurants, caterers and retail premises generated (source-separated or collected with the bulky waste) in Catalonia is calculated at county level, using the following data: total municipal solid waste generation per county in 2009 (ARC 2012) and average content of organic waste in the municipal solid waste for Catalonia, which is 36% (ARC 2007). We consider a 10% of OFMSW non-composted due to likely technical, geographical and social difficulties for the whole source-separated collection.

Catalan average mass reduction during composting is 78% (Huerta et al. 2010). The average nutrient supply of compost the first year was calculated using analysis from Huerta et al. (2010) and considering the mineralization rates of 14% (N), 37% (P) and 78% (K), from several compost reviews.

### *(iii) Potential nutrients from ground water*

The available data from the Catalan quality controlling net is used (ACA 2012). Average nitrate content per county and for the last 5 years is calculated. In order to do calculations we have assumed the total irrigation water (6,500 m$^3$ ha$^{-1}$ year$^{-1}$) coming from ground water, which is supplemented with rainfall water. Total nitrogen applied with irrigation water is calculated multiplying nitrate content by the total amount of water used in each county. Supply of phosphorus or potassium is not taken into account as the concentrations in water are negligible.

## 10.2.2. Comparison scenarios

The nutrient savings and environmental improvement of the assessed eco-synergy scenario, which considers the use of nutrients coming from compost and polluted ground water, are calculated from the initial or current scenario. The maximum amount of compost per county depends on the potential compost production and is restricted to do not apply more than 170 kg N $ha^{-1}$ $year^{-1}$ from organic sources (Catalan Parlament2009). The rest of nutrients are supplied by mineral fertilizers.

The initial scenario, considers that the entire nutrient demand from horticulture is supplied by mineral fertilizers and the slender compost applied to soils is not substituting inorganic fertilization. Nowadays, less than 15% of the OFMSW generated in Catalonia is being applied to soils as compost (Giménez et al. 2005; ARC 2007).

### 10.2.3. Geographic Information System

The spatial information generated was presented with a Geographic Information System (GIS) software, MiraMon® v7, allowing data processing and visualization.

### 10.2.4. Global warming potential

For the global warming potential (GWP) calculation, impact factors from IPCC 2007 were used and only classification and characterisation steps, applied.

For the impact saving calculation only the avoided manufacture of mineral fertilizers is taken into account. Neither pumping station nor OFMSW management options nor transport of fertilizers are being included. The environmental data related to mineral fertilizers production was obtained from Boldrin et al. (2009).

## 10.3. Results

### 10.3.1. Nutrient demand and potential supply

The nutrient demand from horticulture sector and potential supply from compost and ground water for Catalonia region, at county level, are presented in this section. Nitrogen demand and supply are showed in Figures 10.1-3, respectively.

The total amount of required nutrients from horticulture crops, which is calculated following section 9.2.1.i, is about 1,900 tons of N, 300 tons of P and 2,150 tons of K per year. Nitrogen demand per county is represented in Figure 10.1. Coastal counties and 4 and 5 are the ones with higher demand of N and, therefore, higher horticulture production.

Figure 10.2 shows the amounts of nitrogen available (in the short-term) from the Catalan OFMSW generation if 90% of it is composted (see section 10.2.1.ii). Counties with higher populations obviously have upper amounts of OFMSW to manage. Catalan population is mainly concentrated in coastal counties, being Barcelona city (county 31) and its surroundings the higher populated and the major generators of waste. County 31 has a generation of nearly 600 thousand tons of OFMSW per year, which represents almost half of the total Catalan generation (1,450 thousand tons). The potential nutrient supply from compost is about 950 tons of N, 450 tons of P and 2,050 tons of K per year.

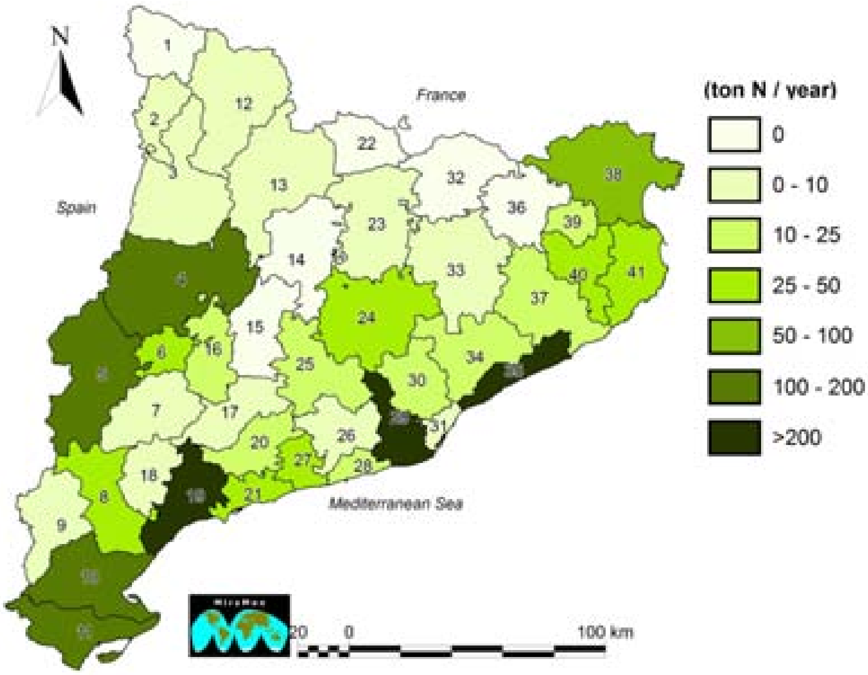



**Figure 10.1. Nitrogen demand from the Catalan horticulture sector.**

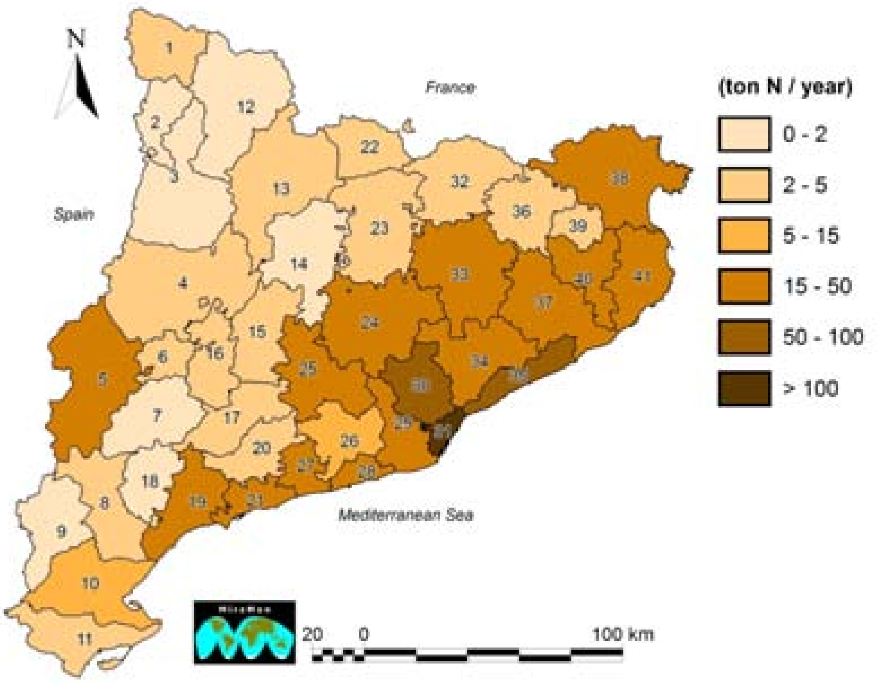



**Figure 10.2. Potential nitrogen available the first year from OFMSW composted.**

Upper nitrogen concentrations in ground water are measured in those counties with a major dedication to agriculture or stockbreeding: 4, 5, 6, 15, 19 have concentration above 50 mg $NO_3\,L^{-1}$ and 33 and 35 have levels above 75 mg (ACA 2012). These areas are under regional regulation (Catalan Parlament2009), in accordance with the Directive 2006/118/EC (EC 2006), for the protection of water against nitrate pollution from agricultural sources. Figure 10.3 shows the amount of nitrogen supplied by ground water according to the irrigation demand of each county (section 10.2.1.iii). The potential amount of nitrogen coming from irrigation water for Catalonia is almost 500 tons per year.

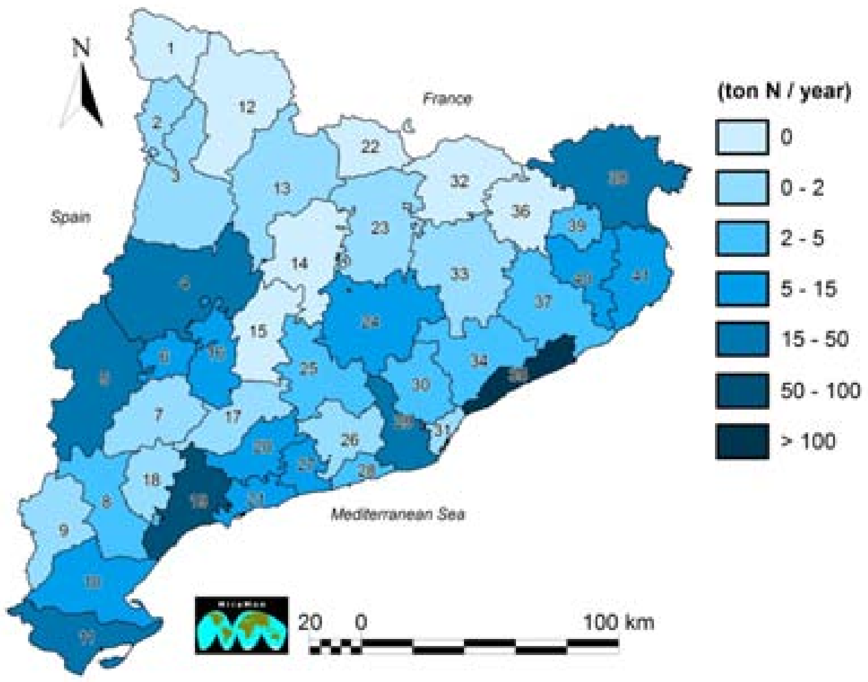



**Figure 10.3. Potential nitrogen available from ground water irrigation.**

## 10.3.2. Nutrients balance for compost scenario

According to the abovementioned amounts of nitrogen required by Catalan horticulture sector and supplied by compost and ground water (Figures 10.1-3), Figure 10.4 shows the balance of nitrogen at county level. Most coastal and west counties have shortage of nitrogen, whereas the rest have too much compost generation for their consumption.

Figure 10.5 illustrates the potential contributions from compost and ground water in the eco-synergy substitution of fertilizer. Almost 50% of mineral fertilizers are saved for N and P, half using compost and half using irrigation water, for the former, and using only compost, for the latter. Regarding potassium, compost supplies 35% of the demand.

However, the total amounts of truly saved mineral fertilizers by compost differ considerably from the potential nutrient supply for the entire Catalonia. According to section 10.3.1 and Figure 10.5, 41% of the potential nitrogen from compost is used, 30% of the P and 37% of the K. It is as a consequence of conflicts with places where compost is produced and places where nutrients are required (Figure 10.4), and because maximum amount of compost applied per hectare is limited (Catalan Parlament2009).

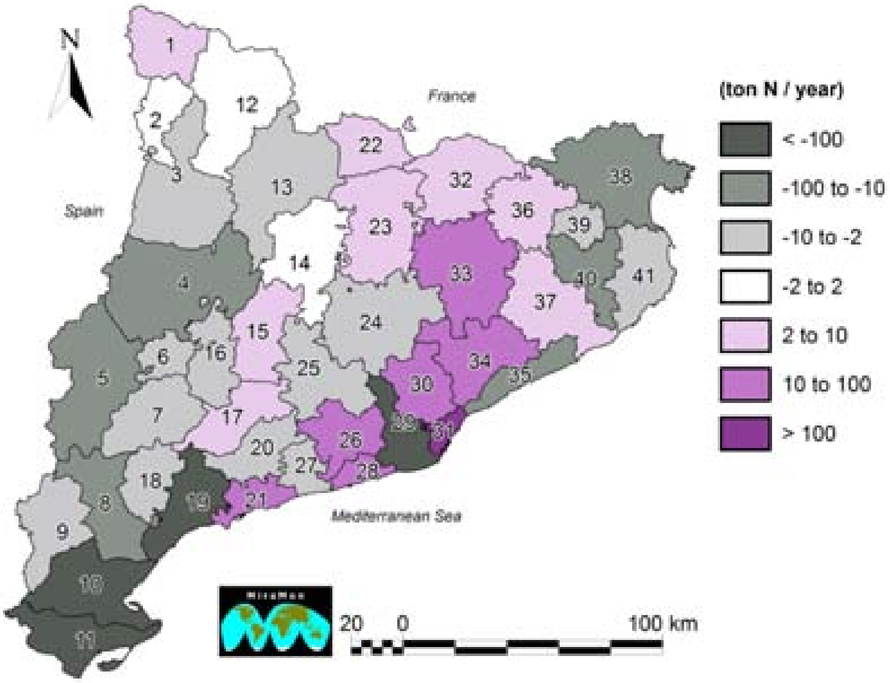



**Figure 10.4. Balance of nitrogen for the eco-synergy scenario (negative values show shortage, positive values show surplus).**

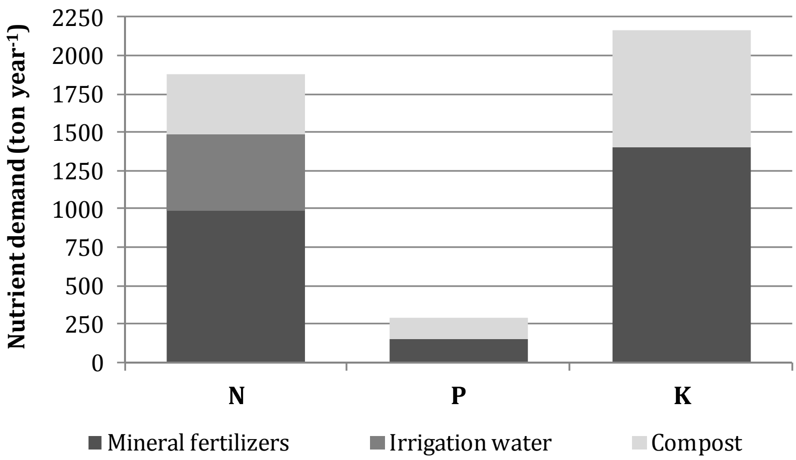



**Figure 10.5. Contribution of the three sources of nutrients to the total demand of nutrients of Catalan horticulture sector in the eco-synergy scenario.**

## 10.3.3. Environmental impact savings

The GWP of the mineral fertilizers saved, thanks to nutrient contribution of the eco-synergy compost and ground water, is presented in Figure 10.6. More than 9,000 tons of $CO_2$ equivalents, 46% of the current emissions from mineral fertilizers manufacture, are saved for the whole Catalonia.

Major GWP is avoided where there is high generation of OFMSW (i.e. population), high nitrate content in ground water and, specially, high demand of nutrients (i.e. horticulture production), for instance, county 35, 5 and 19.

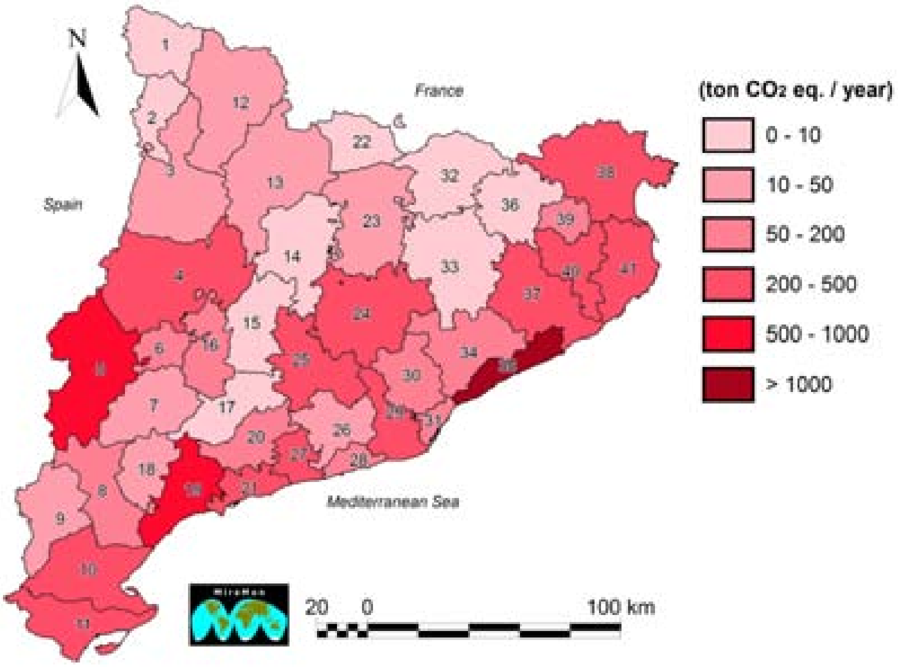



**Figure 10.6. Global warming potential savings for the eco-synergy scenario.**

# 10.4. Discussion and conclusions

The total amount of required nutrients from horticulture crops in Catalonia is about 1,900 tons of N, 300 tons of P and 2,150 tons of K per year. From them, the compost and polluted water eco-synergy combination is able to supply between 35-50% of the demand of nitrogen, phosphorus and potassium. Polluted water supplies rapid available nitrogen to the crops, while compost nitrogen and other nutrients are slowly released, providing a source of nutrients in the long-term.

Two waste flows, that before were problematic, become a way to reduce inorganic fertilizers consumption. The potential reduction would lead to avoid 46% of the current emissions from mineral fertilizers manufacture. Furthermore, other environmental savings would take place due to reduction in the amount of OFMSW going to landfill, decrease in the nitrate content of nitrates in ground water, thanks to bio-filtration effects, and effects of organic matter in the health of soils and plants.

Higher substitution rates, and therefore larger impact savings, with the eco-synergy of the two wastes are achieved in urban and agriculture intensive areas. It is consequence of major pollution of aquifers, major generation of organic waste and major demand of nutrients.

Taking into account the possibility of transporting compost from one county to another, it could increase even more the savings. Moreover, less than 41% of the nutrients of the potential production of compost would be used by horticulture sector; therefore other agriculture sectors could take profit of this surplus.

From the results of this paper, the joint use of the tools LCA and GIS inside the framework of industrial ecology appears a valuable strategy for territorial planning and decision taking.

## 10.5. Further research and points of concern

Aiming to build a more realistic model and to obtain more comprehensive results it is necessary to stress in several of the suppositions and data used throughout the paper. Further research would be focus on:

- Regarding OFMSW generation and management, it is necessary to:
  - → Consider real values of OFMSW generation per county or municipality.
  - → Quantify if there is enough bulky agent for the management of all the OFMSW generated.
- It is needed to use a better approach for the calculation of the average content of nitrates in ground water for each county.
- Improvement in agriculture productions and nutrient demand figures has to lead with:
  - → Adding other irrigated crops apart from horticulture ones.
  - → Taking into account the soil and climate particular conditions in each county or area.
  - → Using a better adaptation of nutrient requirements for Catalonia.
  - → Introducing the residual effects of nutrients release during the second and the following years after compost application.
- In reference to nutrients balance, it is required to:
  - → Assess the potential higher savings if transport of compost between counties is considered.
  - → Assess the excesses of P and K applied with compost.
- Finally, improvements in the impact assessment would come from:
  - → Adding other impact categories apart from GWP, as eutrophication potential savings from the filtration of polluted water could be relevant.
  - → Comparing the impacts of the current and the proposed waste management scenarios, as each option of treatments or disposal have different environmental performances.
  - → Including fertilizer application emissions, as the performance during application is different between mineral and organic fertilizers.
  - → Potential improvement in the quality of ground water.
  - → Assessing the introduction of the potential savings linked with carbon sequestration on soils and landfills.

# Part V

## General conclusions and future research

# Chapter 11. Conclusions

This dissertation presents a complete assessment of the production and use in horticulture of municipal compost, using life cycle approach and including environmental, socio-economic, territorial and methodological aspects. **Chapter 11** addresses the main research findings and formulates a list of general final comments, based on the objectives and extended conclusions presented in Chapters 3 to 10.

Conclusions are structured as follows:

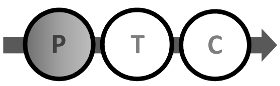
- Environmental assessment of composting technologies.
- Agronomic performance of compost use in horticulture.
- Environmental assessment of compost life cycle. Production, transport and application.
- Consumer, territorial and sustainability perspectives on compost production and application.
- Methodological contributions to life-cycle-thinking tools.

The acronyms used thorough this chapter are presented in Table 11.1:

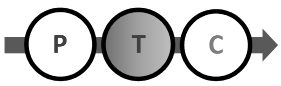
**Table 11.1. Acronyms for the Chapter 11 of the dissertation.**

| | |
|---|---|
| ***Impact categories*** | |
| ADP Abiotic depletion potential | OLDP Ozone layer depletion potential |
| AP Acidification potential | POP Photochemical oxidation potential |
| EP Eutrophication potential | CED Cumulative energy demand |
| GWP Global warming potential | |
| ***Cultivation options (see section 2.2.2)*** | |
| $C_H$ Option using a high-dose of compost (and the rest of nitrogen necessities supplied by mineral fertilizers). | |
| $C_L$ Option using a low-dose of compost and the rest of nitrogen necessities supplied by mineral fertilizers. | |
| M Option using only mineral fertilizers. | |
| ***Methodologies*** | |
| CF Carbon Footprinting | SLCA Social life cycle assessment |
| LCA (Environmental) Life Cycle Assessment | LCSA Life Cycle Sustainability Assessment |
| ***Others*** | |
| OFMSW Organic fraction of the municipal solid waste | VOC Volatile organic compounds |
| GW Green waste | GHG Greenhouse gases |

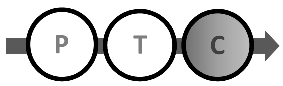
The following pictograms are used to show the stages involved in each section:



*Fertilizers and compost (P)roduction stages*



*Fertilizers and compost (T)ransport stages*



*(C)ultivation stages*

## 11.1. Environmental assessment of composting technologies

P T C

Inventory data, with particular focus on decomposition emissions, and the environmental assessment of two types of composting technologies, for OFMSW and GW, were performed in the dissertation. A high-technology, although quite widespread, type of composting plant – in-vessel decomposition plus maturation in turned windrows – was faced with home composting. Up to the date, few experimental studies have assessed the real gaseous emissions of composting processes under a broad sampling plan, especially for home composting.

### Industrial in-vessel composting

- The gaseous emissions **released after biofiltration** were 34 mg of $CH_4$, 1.21 kg of VOC, 92 mg of $N_2O$ and 110 mg of $NH_3$ per ton of OFMSW.
- **Waste collection** stage and the **electricity consumption** during the composting process were two of the major contributors to almost all the impact categories. Furthermore, **VOC emissions** were responsible of nearly the entire impact for POP and the **dumping of the solid waste**, which was produced at the plant, generated high contributions for the categories EP and GWP.

### Home composting

- The gaseous emissions **directly released** were 158 mg of $CH_4$, 559 mg of VOC, 676 mg of $N_2O$ and 842 mg of $NH_3$ per ton of OFMSW.
- **Ammonia**, **nitrous oxide** and **VOC** had impact contributions above 90% for the categories AP and EP, GWP, and POP, respectively. The **composter**, and particularly its manufacturing process, presented greater impacts for the remaining categories (ADP, OLDP and CED). Finally, electricity consumption contributed a third of the total impact for ADP and CED.

### Composting technologies comparison

- For the cases studied, **home composting had not particularly better quality than industrial compost**. Both composts were stable and according to the European standards. Nevertheless, home composting was more stable whereas industrial one presented higher fertilizing potential, i.e. higher content of organic matter and nitrogen.
- **Home composting emitted between 5 and 8 times more $NH_3$, $N_2O$ and $CH_4$** than industrial process, while **VOC emissions in the industrial process** doubled that of the home composting. The former was due to the lack of gas biofiltration for the domestic option. The higher VOC emissions were probably due to the forced aeration on the full scale plant.
- On the contrary of what is generally believed, **home system was more impacting than industrial one for 3 out of 7 categories** (AP, EP and GWP), although differences were below 40%. It is largely due to major gaseous emissions, in spite of the lower resources consumption, during the domestic composting process.
- **Industrial option was more harmful for 4 out of 7 categories** (ADP, OLDP, POP and CED), with differences of 100-500%, principally as a result of electricity consumption and waste collection.

## 11.2. Agronomic performance of compost use in horticulture

Apart from the environmental assessment of compost cycle, the agronomic viability of its application on Mediterranean horticultural crops, tomato and cauliflower, was assessed in the dissertation. Both productivity and quality were tested for the cultivation options considered, which were combining mineral and organic fertilization as well as open field and greenhouse production. Harvests were of major importance also for the environmental assessment as the ton of tomato or cauliflower was selected as functional unit for most of the chapters.

### Tomato crops

- Regarding the **tomato crops, similar commercial yields and product quality** – both including weight and size parameters – were obtained for all the fertilizing options in open field ($C_H$, $C_L$ and M).
- For $C_L$ and M, **greenhouse productions,** and the quality of the tomatoes, were **similar and 50% larger than open field productions**, although they were below expected values for greenhouse in the area. Regarding $C_H$ in greenhouse, it presented lower yield than the other fertilization options, because it was not complemented with mineral fertilizers and thus the high nitrogen demand of tomato plants in early cultivation stages could not be covered by compost alone.

### Cauliflower crop

- Regarding **cauliflower,** cultivation option **M had significantly better commercial yield, almost 40% higher, and product quality** than cultivation option $C_H$, probably as a result of the slower release of compost nutrients in autumn as well as the previous cultivation of tomato that diminished soil reservoirs. There were **similar agricultural performances between $C_L$ and M** option, as the former was relying less on compost nutrients contribution.
- When compost in a high dosage was applied ($C_H$), significant higher content of total sinapic acid derivatives and total phenols were detected. Their content in **$C_H$ was higher than in M** (about 80% higher for sinapic acid and 25% for the total phenols)**, while $C_L$ had similar content** to M. Other bioactive compounds did not present significant differences.

### Agronomic performance of compost

- Therefore, compost application, which is supplying nutrients on a slow rate and in the long-term, **needs to be complemented with a rapid source of nutrients**, especially nitrogen, to guarantee similar yields to options exclusively relying on mineral fertilization.
- Compost use **in greenhouse crops seems difficult as the demand of nutrients** on this type of cultivation **is very high**, especially at the beginning of the cultivation, while nutrients release has its own rate.

## 11.3. Environmental assessment of compost life cycle. Production, transport and application.

The environmental impacts of the whole life cycle of compost, including production, transport to the crops and cultivation, were assessed from experimental data of the studied sites. Firstly, the main stages contributing to the total impacts for each cultivation option were stated (Table 11.1) and subsequently the options were compared.

### <u>Critical environmental stages of compost life cycle</u>

- **Industrial compost production** had turned into **the most impacting stage** in those cultivation options using this organic fertilizer, with contributions above 30% of total impacts for $C_L$ and above 50% for $C_H$. For the impact category POP, compost was responsible of nearly 100% of the total impact (see section 11.1).
- For **cultivation option M** impact contributions are not as concentrated in only one stage. There are three main stages contributing to total impacts:
  → The most impacting one is **mineral fertilizers production**, contributing with 60% to GWP and between 30-50% for the remaining categories. It is principally due to energy consumption and gaseous emissions during the production process.
  → Machinery and tools, and particularly **harvesting crates**, involve 10-40% of the total impacts, with the higher contributions for ADP, GWP and CED.
  → When considered (i.e. for tomato), **nursery plant production** also had relevant contributions – up to 20% of the total impacts. It was consequence of the high consumption of energy for the temperature regulation with heating.

**Table 11.2. Main contributors to the total impacts of the cultivation options. Stages and sub-stages are placed in order of contribution.**

| Fertilization options | | Location options | |
|---|---|---|---|
| | | Open field | Greenhouse |
| $C_H$ | +<br>− | Compost production | Compost production<br>Greenhouse |
| $C_L$ | +<br>− | Compost production<br>Mineral fertilizers production<br>(Nursery plant production)<br>Machinery and tools<br>Post-application emissions | Compost production<br>Greenhouse<br>Nursery plant production<br>Machinery and tools |
| M | +<br>− | Mineral fertilizers production<br>(Nursery plant production)<br>Machinery and tools<br>Irrigation<br>Post-application emissions | Mineral fertilizers production<br>Greenhouse<br>Nursery plant production<br>Machinery and tools<br>Post-application emissions |

- Similarly, regarding **$C_L$**, the remaining impacts, apart from compost contribution, were largely due to **nursery plant production**, when considered, **harvesting crates production, and mineral fertilizers production**.
- For all the options and crops, the **post-application emissions** stage involved relevant impacts for AP and EP due to ammonia emissions to air and nitrate emissions to water, and had negative contribution to GWP due to **carbon sequestration** in soils throughout the compost application.
- Finally, for tomato options under **greenhouse,** it contributes with 15-30% of the total impacts. The production of several components of the greenhouse structure – plastic fixing bar, covering film, door sheets and galvanizing of steel posts –, entailed the major part of the total greenhouse stage.
- Transports, irrigation, fertirrigation infrastructure and phytosanitary substances had minor contribution for all or most of the impact categories.

**<u>Total environmental performance</u>**

***Avoided burdens were not accounted:***

- For tomato and cauliflower crops, the order of impact between the cultivation options was the following one:

| |
|---|
| ***$C_H$ > $C_L$ > M, per ton of commercial vegetable*** |

- It was mainly due to compost production burdens, and, for cauliflower, also due to the lower yields in $C_H$.
- Regarding the higher impact of compost options, it is important to bear in mind:
  - → Because **agricultural stage is a clean process** from an energetic point of view (photosynthesis), any industrial process involved becomes an important contribution.
  - → The **manufacture of mineral fertilizer is a highly efficient process** resulting in a product with a high concentration of nutrients, so only small amounts are required to be transported, even though this involves long distances.
  - → **Composting is not only a fertilizer production process**, but, more important, it is a way to divert organic waste from landfills.

***Avoided burdens are subtracted from total impacts:***

- Therefore, **composting avoids the environmental loads related to the disposal of organic wastes in landfills**. When the system boundaries were expanded and these avoided burdens included, the **differences between cultivation options were tempered or even reversed**, depending on the impact category.
- For the categories **EP, GWP, and OLDP** the order of impact was the following one, in most of the cases:

| |
|---|
| ***M > $C_L$ > $C_H$, per ton of commercial vegetable*** |

**For EP and GWP, the impacts for compost cultivation options were negative**, i.e. they were avoiding burdens:

- It is because the subtracted burdens were particularly important for these categories, due to the saved $CO_2$ and $CH_4$ emissions and the saved lixiviates of nitrogen and phosphorous, all occurring due to organic matter decomposition in landfills.

- For the categories **CED and ADP the impacts were quite similar** between the three cultivation options ($C_H$, $C_L$ and M), whereas **$C_H$ was still more impacting** option for **AP and POP**.

*Comparison between open field and greenhouse:*

- **Compost application has shown its major effectiveness in open field cultivation** than in greenhouse, due to the high requirements of nutrients, mainly in early stages of the crop; nevertheless, a correct combination between organic and mineral fertilizers could be a good option in greenhouse.
- The choice of greenhouse cultivation represents an environmentally attractive alternative, provided that **the increased production could justify a higher use of natural resources**. However, in this dissertation sufficiently larger yields under greenhouse are not obtained.
- **Bigger differences** in environmental impacts between cultivation options were found **due to fertilization** factor rather than the location factor (greenhouse or open field), for most of the impact categories.

*Comparison using other functional units:*

- When **functional units involving nutritional content** were used for cauliflower assessment, differences between $C_H$, $C_L$ and M were tempered or even reversed.
- When sinapic acid content was used as functional unit and including landfilling avoided burdens, $C_H$ had the lowest impacts in all categories apart from POP and AP.

## <u>State-of-the-art of compost benefits inclusion in LCA assessments</u>

Use of bio-waste **compost on land can have beneficial effects on the plant-soil system**. Most of these benefits have been so far excluded from LCA studies, mainly because of **scarcity of data or lack of appropriate impact assessment methods**.

- For **nutrient supply and carbon sequestration**, the review showed that both quantification and impact assessment of the effects could be performed, meaning that these two benefits should be regularly included in LCA studies.
- For **increase in crop yield, soil workability, crop nutritional quality, and enhancement of soil biological properties and biodiversity**, quantitative figures could not be provided, either because of complete lack of data or because the effects are both very variable and too depending on specific local conditions.
- For **soil erosion and soil water content**, effects could be quantitatively addressed, but available impact assessment methodologies were considered unsuitable to comprehensively evaluate the implication of compost application with regards to these two benefits.
- For **suppressive effects of compost on weed, pests, and diseases**, the effect could not be generally proved.

## 11.4. Consumer, territorial and sustainability perspectives on compost production and application

Based on the experience and the systems studied throughout the previous chapters of the disertation, the Chapters 8, 9 and 10 were dealing with other perspectives of compost production and application in horticulture further than environmental assessment.

### Consumer – Carbon Footprinting (CF)

CF could be the first step in the environmental communication strategy to help consumers assimilate the environmental information present in products because it results more **simple and understandable environmental indicator** than LCA.

- The results of CF methodology (PAS 2050) for **greenhouse fields were omitting 25% of the GHG emitted,** with regard to results using LCA. It is essentially because CF is not including capital goods, i.e. the greenhouse structure. For open field options, the differences in the GHG accounted by the two methodologies were 5-10%, as not many infrastructures were used.
- The omission of capital goods in the assessment could **punish those growers using low technology and having lower yields**. An increased use of technology in horticultural systems may lead to a rise in the energy consumption and so a decrease in the relevance of infrastructures on impacts, if they are not compensated by a high enough production.
- For the options considering compost application, when LCA methodology was used, it was assumed that a certain quantity of carbon was incorporated to soils, meaning a credit of about 5% of the total impacts, while **CF was not including that credit** because considers that there are not enough scientific knowledge to be sure about this figure.

### Life Cycle Sustainability Assessment of mineral fertilizers and compost

The dissertation includes the sustainability assessment of nitrogenated mineral fertilizers (nitric acid and potassium nitrate) and compost produced with OFMSW, along the entire production life cycle. It is one of the few attempts to apply SLCA and LCSA, two methodologies that still need further development. Therefore, some of the most controversial issues, particularly for SLCA, were faced according to the best of authors' knowledge, aiming to contribute to the methodologies improvement.

***The entire Social life cycle assessment results:***

- The **upstream sectors involved in the production** of the fertilizing options were analogous, and similar risks occurred. However, **working time was significantly higher for those upstream sectors of compost production** and, thus more social impacts were expected for it.
- High or very high levels of risk in upstream sector total result were owing to **diesel and natural gas imports from countries with worst social behaviors** than fertilizer producer countries.

- For **mainstream sectors, nitric acid seemed to be the best option from a social perspective**. However, compost was promoting local employment and was creating more work necessities than mineral fertilizers.
- At the **company level-approach**, preliminary results were showing worse social performances for mineral fertilizers companies for Worker and Local Community stakeholders. Whereas they seemed to be the best option for Society indicators.

***Life Cycle Sustainability Assessment results including the 3 dimensions:***

- For the **social dimension**: **potassium nitrate was the worst alternative, mainly for indicators related with worker rights**, whereas compost had better performance for 5 out of 7 indicators and nitric acid for 4 out of 7. Only indicators involving worker stakeholder were included for this dimension of LCSA, with the aim to assure that they were assessed thorough the entire life cycle of fertilizers production.
- For the **environmental dimension, compost was the worst option** (landfilling avoided burdens were not included), regardless the impact category, whereas results for nitric acid and potassium nitrate were in the same order of magnitude.
- For the **economic dimension**, even though the price of compost was lower than the price of nitric acid, the other indicators related with transport and application were higher for compost.
- Regarding the **LCSA aggregation** of the results, **nitric acid was the best fertilizing alternative for the three dimensions**, compost was the worst for economic and environmental approaches, and potassium nitrate was the worst for social dimension (Figure 11.1).

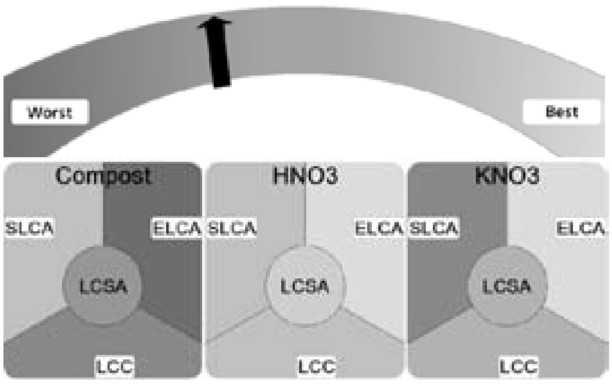



**Figure 11.1. Life Cycle Sustainability Assessment results for the three nitrate fertilizers.**

## <u>Regional assessment of compost use along with nitrate polluted water</u>

In those areas with nitrate polluted aquifer and nutrient demand from either horticulture or general agriculture, the combined use of compost from municipal waste and nitrate polluted water may be a potential fate for both flows, as well as a way to reduce resources consumption. Compost is a slow release source of nitrogen and other nutrients and, according to the results from section 11.2, it should be complemented by a rapid source of nitrogen, for instance, polluted water.

- Preliminary calculations showed that the **compost and polluted water eco-synergy combination** would be able to supply between **35-50% of the total**

**demand of nitrogen, phosphorus and potassium** from horticulture crops in Catalonia.

- The **potential substitution would lead to avoid 46% of the current emissions** from mineral fertilizers manufacture from horticulture. Higher substitution rates, and therefore larger impact savings, with the eco-synergy of the two wastes are achieved in urban and agriculture intensive areas.

## 11.5. Methodological contributions to life-cycle-thinking tools

In this dissertation several life-cycle-thinking tools were applied to real case studies. Moreover, some proposals were made for the currently available methodology and data and, some of them were moreover applied and tested.

### Life Cycle Assessment

- For the environmental assessment, the **entire life cycle of compost** – including OFMSW collection, composting, transport and all the cultivation steps – was inventoried and assessed, facing some of the **methodological concerns for LCA** and particularly for agricultural and waste systems.
- It was detected the need in **agriculture systems of using other complementarily functional units,** apart from area or yield, that include quality aspects, as different cultivation managements could lead to different qualities. **Functional units including the nutritional value of the products** were considered and used for the assessment.
- **System boundaries definition and allocation** were also **deciding factors for the obtained results**, for instance: the inclusion or exclusion of the environmental burdens saved thorough diverting organic waste from the landfills; and the decision to leave out of the boundaries of the system those waste being used by another system, especially organic crop waste.

### Environmental benefits of compost use-on-land assessed with LCA

- In the dissertation, **nine of the environmental benefits of compost application typically reflected in the scientific literature** were quantified through literature revision, when possible, and then their effects on the foreground and background system were discussed.
- **Key LCA methodological issues for the quantification of the benefits** were identified, which are sometimes common concerns for general and agricultural LCAs, such as time frame, availability of data, local data and ecosystems complexity, compost doses and periodicity of applications, among others.

### Carbon footprint evaluation

- Due to the relevancy that CF could have in consumer choice, it **needs standardization and a deeper methodological definition** than that given by PAS 2050.
- For example, **not considering capital goods can lead to prioritization of products or processes that do not have a truly better environmental**

**performance** when they are assessed with LCA. Dissertation results indicated that the exclusion of capital goods for tomato and other Mediterranean horticultural products, with low energy demand, increases the disparity between CF and LCA results.

- The **estimation of carbon and nitrogen dynamics in soils is a weak and complex methodological step**, due to the high number of factors interacting. The lack of reliable data and the complexity of the calculation are the main reasons why PAS 2050 avoids the inclusion of methods to calculate these emissions, whereas the wider and complex LCA allows the incorporation of scientific assumptions justified. The progressive review (every 2 years) of the PAS 2050 standard allows the incorporation of those cases duly justified.

## Social life cycle assessment

P T C

- A **balance between specific primary data and fulfillling life cycle perspective** was considered. Although the whole life cycle of a product is to be included in the social assessment, it is currently not feasible to collect all the data at the company level, as too many companies would be involved.
- The assessment was made at **three different levels: country, sector, and company**. Although knowing company behavior was the ultimate goal, apart from data obtaining difficulties, country and sector social performance are the baseline for companies.
  - → **Country level assessment is the most mature step**, as many data sources are available and as Social Hotspot Database would help both to data obtaining and to social risk quantification.
  - → **For the sector level, mainstream as well as upstream sectors** were included, coping with broader system boundaries along the life cycle than previous studies. However, collecting sectorial data for all the countries that could be involved on the life cycle of a product is still a huge task, which is difficult to be undertaken out of science community.
  - → To assess **life cycle of a product at the company level** is almost impossible nowadays, as too many companies would be involved. Nevertheless, to use primary data at least for mainstream company is recommended.
- The **viability of using working time as aggregating variable** of the social results throughout the life cycle was proved in the dissertation, although further development is needed.
- **Two assessment approaches were proved** - scoring according to international conventions and comparison among alternatives – however, the preferable use of one of the other, or other alternatives, is not clear yet and depends on data availability, level of assessment, etc.
- The dissertation could also be claimed as one of the first examples for **applying the Social Hotspot Database (SHDB) to a real case study**.

## Life Cycle Sustainable Assessment

P T C

- The assessment of systems taking into account the three dimensions of sustainability is still a hard work to perform as two of the tools used, **LCC and SLCA, are not enough developed** to provide indicators sufficiently representative and accurate. Besides, **social data is difficult and highly time demanding to collect**.

- Once data availability and assessment methods were enough agreed for all the three dimensions, the **Life Cycle Sustainability Dashboard would be a good tool to present and interpret the results**, as it is shown in the dissertation.

# Chapter 12. Future research and strategies

**Chapter 12** points out future lines of research that may follow this thesis are mentioned here. Some of them relate to one specific chapter of the dissertation, and others present general lines of action or research.

This chapter is structured as follows:

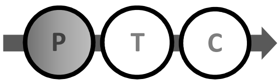
- Strategies for the improvement of the assessed systems and their operation.
- Furture research studies.
- Further research on life-cycle-thinking tools and new developments.

The few acronyms used thorough this chapter were presented in Table 12.1.

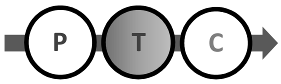
**Table 12.1. Acronyms for the Chapter 12 of the dissertation.**

| | |
|---|---|
| ***Methodologies*** | |
| CF Carbon Footprinting | LCSA Life Cycle Sustainability Assessment |
| LCA (Environmental) Life Cycle Assessment | |
| ***Others*** | |
| OFMSW Organic fraction of the municipal solid waste | GHG Greenhouse gases |

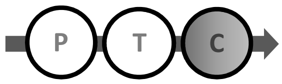
The following pictograms are used to show the stages involved in each section:




*Fertilizers and compost (P)roduction stages*




*Fertilizers and compost (T)ransport stages*




*(C)ultivation stages*

## 12.1. Strategies for the improvement of the assessed systems and their operation

### Composting technologies

- The environmental viability of cultivation with compost depends on **reducing the high impacts of the industrial composting plants**. This could be achieved by several actions:
  - → To **reduce the energy use in the facility**, for instance reducing the distances that the processed material has to be transported by tractors and other machinery, and using more efficient air collecting systems. Apart from the efficiency increase, the use of renewable sources of energy.
  - → To make **more efficient the organic waste collection transport**, either using more efficient trucks (Euro V) or alternative fuels. Moreover, to reduce as much as possible the **distance between OFMSW generators and the plant**.
  - → To guarantee a **good aeration** during organic waste decomposition, at the same time that **more efficient gas emission treatment** are used, especially for volatile organic compounds, to reduce finally released emissions. To install gas sensors along the composting plant would help to control the emissions.
  - → To **minimize the presence of impurities** in the source-separated OFMSW that reduces the facility treatment capacity and increases the amount of dumped solid waste. Door-to-door collection systems (in spite of the major transport requirements) and environmental education would lead to better selection.
- The **spread of home composting**, where possible, as an alternative to industrial scale should be **constrained by emissions reduction and** major efforts on the **eco-design of the composter**:
  - → To examine the **incorporation of gas treatment systems to the composter**, for example biofilters, is of vital importance.
  - → To use **low impact materials** in the composter construction, for instance recycled plastics or other reused materials.
  - → To set **community composters** owing to share composter and tools, thus reducing the burdens per composter.

### Compost use in horticulture

- Regarding the most impacting cultivation sub-stages, several impact reducing proposals are stated:
  - → With regard of the harvesting, further research is necessary on the materials and the reuse rates of the crates or in new systems of harvesting.
  - → With the aim to reduce post-application emissions, an accurate fertilizing plan should be designed including previous nutrients content in soils, weather, mineralization rates, soil characteristics, plant uptake, etc.
  - → Nursery plant production could be optimized by using renewable energy sources, improving the isolation of the greenhouses, and/or reducing the rate between nursery plant and volume of air to heat.

→ Regarding the **greenhouses,** the first measure should be to assure **sufficiently high yield** throughout a better cultivation operation (climatic conditions, irrigation and fertilizers). Furthermore, although Mediterranean greenhouses are of low technology, their **eco-design** could lead to lowering impacts: to minimize the consumption of materials as much as possible; to use local materials and those with lower environmental impact; and to increase their durability or the materials recyclability potential.

- To assure a **sufficient nutrient supply from compost**, two measures are proposed. **Delaying time of planting after compost incorporation**, about one month, and taking into account disparity in mineralization rates of organic fertilizers depending on the temperature. Furthermore, the promotion of **long-termed fertilizing plans**, including periodical compost applications, would assure a stable organic matter pool in soils.
- If **special label is addressed and major value is given** to this type of production, the **use of compost in the surroundings of cities** (where a continuous provision of this fertilizer is guaranteed and high transport burdens are not necessary) could be a way to obtain high valuable products and to reverse the currently trend to agriculture endangered in urban areas. Moreover, it would improve the organic matter content of soils, which is a common concern for Mediterranean soils among others.
- Furthermore, the **input of nitrogen from irrigation in those regions with high nitrate polluted** water should be taken into account in the fertilizing plan as a valuable source of rapid nitrogen source. This possibility could achieve higher performance if it is **planned at the regional level** and, for instance, composting plants are distributed according to potential consumers.

## 12.2. Future research studies

### Experimental research on compost use

P T C

- Further case studies and **examples of horticultural production systems** considering different climate areas, regional particularities, levels of technology and crops management, are also necessary.
- **Long-term fertilization experiments** with compost aiming to obtain stable high levels of organic matter in soils are necessary.
- Considerable disparity was observed on the compost mineralization rates in soils reported by several authors, it seems to be highly dependent on climatic conditions, while most of the experimental studies were located on cold regions. Therefore, further research on the **factors affecting mineralization rates and the building of models** is necessary, specially for Mediterranean regions, where rates seems to be higher than in cold areas. It is very relevant for compost dosages calculation and also for the allocation of compost burdens among the crops of a rotation.
- It is also necessary an **accurate monitoring of water resources** fate. Real water necessities and leachates should be continuously measured during the experiments, in order to detect differences between cultivation options. Furthermore, because they were quite relevant for the impact category

eutrophication potential, the real **nitrate leachates** should be measured and evaluated.

- Further research on the quantification and a better understanding of the relation between **compost application and the increase in crop yield**, **soil workability, crop nutritional quality, and enhancement of soil biological properties and biodiversity**. The improvements of soil characteristics are likely to appear mostly in the long-term, but **studies for the long-term** are scarce. Besides, data concerning effects further after application are also relevant for LCA purposes. Thus, this type of studies, assessing several parameters of soil, is necessary.
- Further research in the nutritional quality of the vegetable products grown with compost, which could be **reflected on the price of agricultural products** and promote healthier consumption.

## <u>Future environmental assessment studies</u>

### *Composting technologies:*

- The environmental **assessment of home composting for different** seasons of the year and, if possible, under very different climate conditions, as composter are operating exposed to the elements. Additionally, the assessment of the **potential impact savings if a biofilter is incorporated** in new composters and gaseous emissions are reduced.
- The **environmental performance of the horticultural production considering other industrial plants** and other compost qualities. Results could differ due to either type of technology, organic waste or operation. Colón et al. (2012) presented inventories and environmental assessments for five composting technologies and ZeroWaste project (Jojart et al. 2011) assessed cauliflower cultivation for other home and industrial composts.

### *Fertilizers dosages:*

- It would be interesting to analyze the **impacts from a complete cropping cycle perspective**, because compost is usually applied for a certain period (1-2 years) and it is a slow-releasing source of nutrients. This temporal boundary expansion would enable the advantages of well-planned crop rotations to be taken into account.
- Related to this, it is of main importance to assess the effect on the results of the **allocation method chosen for the distribution of compost burdens among the crops cultivated** after the application. Although compost is applied in a certain moment, it supplies fertilization to the crops intended to be fertilized and also to the subsequent ones. In this dissertation compost was allocated among the crops of a rotation as a function of nitrogen uptaken by each crop. However, several approaches could be taken to make the allocation, for instance the duration of each crop, the evolution of the nutrients released, etc.
- Future studies should try to **take into account also the mineral fertilizers saved in the following crops**. Although it was not considered in the dissertation, apart from the nutrients theoretically released during the cropping period, the remaining nutrients from compost would be mineralized in the long-term, and therefore used by other crops. For example, non-easily hydrolysable nitrogen of the compost would be available in the following 10-20 years after application.

*Other benefits due to compost application:*

- **Nutrient supply and carbon sequestration should be regularly included in LCA studies** where compost application is considered, as both quantification and impact assessment of the effects could be performed for them.
- It is necessary to also carry out a review for **drawbacks of compost application in soils** – such as salinity, increase on heavy metals content, phosphorous leaching in acid soils, etc. It must include quantification of the effects and their potential inclusion on environmental LCA.

*Toxicity impact categories:*

- Fresh water, marine, terrestrial and human ecotoxicity were not considered in this dissertation in order not to introduce relevant uncertainty, mainly in reference to heavy metal release and accumulation and pesticide use that would be relevant contributors to these impact categories. Nevertheless, ecotoxicity models and indicators for chemicals have been developed during last years and thus the assessment of the system including toxicity categories could be revised.

## Future sustainability assessment studies

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- Future LCSA should expand the system boundaries to include **the entire fertilizers life cycle.** They should **include the cultivation stage**, apart from fertilizers production and transport. Additionally, studies including compost should include the **avoided burdens through diverting organic waste** from landfills with composting.
- In the future, **the potential reduction of potassic and phosphoric mineral fertilizers** necessities in soils when compost is applied should be also considered for the SLCA and LCSA, as only nitrate fertilizers were included in this dissertation.
- **Compost technologies**, including home composting, one or more industrial composting, and additionally, community composting, should be assessed from the three dimensions of the sustainability.

## Integrated urban and agricultural planning

P T C

- Future regional studies on the application of compost in a certain region would consider **all the environmental savings related with compost application**, apart from mineral fertilizers saved, as the reduction in the amount of OFMSW going to landfill, decrease in the nitrate content of ground water thanks to bio-filtration effects, effects of organic matter in the health of soils and plants, etc.
- It is also necessary to **improve some suppositions and data sources**, with the aim to build a more realistic model and to obtain more comprehensive results.

## 12.3. Further research on life-cycle-thinking methodologies and new developments

### Goal and scope

- The **choice of the time horizon of the LCA agricultural studies should be harmonized**. This decision is in many cases (carbon sequestration, landfilling decomposition emissions, nutrients release in soils, etc.) very important, as results vary largely depending on the time frame.
- **Homogeneity among standards** should be stated for agricultural studies, as well as for other specific sectors, if results aim to be comparable. For instance: functional units including both productivity and quality wherever differences are detected; system boundaries and allocation of organic fertilizers and wastes; organic matter effects on soil parameters; etc.
- **GHG emissions arising from capital goods and the carbon credits from carbon sequestration in soils**, generated by management and labour applied to crops, should be included in future revisions of PAS 2050, and its specification for horticulture PAS 2050-1, as well in other standard specifications for CF, such as the forthcoming ISO 14067.

### Environmental impact assessment

- A revision of the **characterization factors is needed for the assessment of non-replaceable non-renewable resources** such as phosphorus, as their depletion is even worse than those resources having a potential replacement. With composting, phosphorus is recycled from organic wastes and supplied to soils.
- **Consensus among the different existing options for the assessment of changes in biodiversity** due to a certain activity or practice should be reached.
- Similarly, it is necessary an international agreement on the way **to include the loss of arable soil** on either the existing impact categories or a new one. Núñez (2011) could provide a useful starting point. It should include the loss of cultivable land, but also the loss of soil organic carbon, plant nutrients, as well as the associated plant, animal and microbial biodiversity.
- The inclusion of **impact categories related with the depletion of water resources** should be considered, because there is evidence of the major water retention on soils after compost application. Currently, only a preliminary scientific consensus exists about the parameters to consider and the methodology to follow for the assessment of the depletion of water resources.
- Impact assessment models should assume that **impacts are not only in function of compost characteristic, but also according to environmental parameters**. It is because fertilizers performance depends largely on location factors (such as climate, type of soil, management, etc.), this is especially true for organic fertilizers.

### Other dimensions of sustainability

- Because SLCA is a life-cycle-perspective tool, future assessments **should include at least the upstream companies/sectors transport, water and mainly energy sources**. According to the results for the fertilizing alternatives, these upstream sectors could represent a relevant share of the total working time invested in a product.

- **Social Hotspot Database should be filled with real data** but extrapolations and with sector-country specific data for all the countries and indicators. Additionally, scoring criteria for the few cases assessing sector data should be revised, as in some cases the country risk is higher than the risk considered for all the sectors.
- PE International database (from GABI5) or other databases should be developed to provide **specific working time data for all the regions or countries.** Additionally, data inventories must **be transparent and open** in order to re-adapt them.
- Future assessments should try to use **similar or related scoring systems for the several level-approaches** assessed. Furthermore, it is needed to develop framework conditions as well as regional, national targets in the respective sectors and regions to be used as reference points in the scoring.
- Regarding **Life Cycle Sustainability Dashboard**, it is necessary to achieve an international, or by regions, **agreement about the factors that must be used for the aggregation** of the several indicators included within each dimension as well as for the aggregation of the three dimensions.

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# Annexes

# Annex I. Supplementary information for Chapter 5

The Annex I corresponds to the Supplementary information of the Chapter 5 that is based on the following paper:

Martínez-Blanco, J., Muñoz, P., Antón, A., & Rieradevall, R. (2011) Assessment of tomato Mediterranean production in open-field and in standard multi-tunnel greenhouse with compost or mineral fertilizers from an agricultural and environmental standpoint. Journal of Cleaner Production, 19, 985-997.

This first annex is divided in the following section

- Harvesting and fruit size parameters for cultivation options with compost as only fertilizer.
- Environmental inventories of cultivation options with compost.
- Total environmental impacts of the four cultivation options.
- Environmental assessment considering normal tomato productions in the region.

References are at the end of Annexes part.

# Annex I.I. Environmental inventories of cultivation options with compost

Table A1 shows the main values obtained from our experiments including all the stages considered: total consumption of water, energy, mineral fertilizers, pesticides and infrastructure materials, transport and the total release of gaseous emissions for the composting plant and waste. Results are given per functional unit (ton of commercial tomato).

The environmental inventories presented are for options with compost plus mineral fertilizers, in open-field or greenhouse (OF_CL and GH_CL). The data for the inventories of these cultivation options were of high quality on account of being mainly obtained experimentally. With regard to mineral fertilizers options (OF_M and GH_M), all data used in production and transport of fertilizers was from available databases and not experimental, therefore neither of the two options were included in Table A1.

### *(i) Water consumption*

Regarding the importance of water in Mediterranean regions, an especial inventory assessment was carried on for water consumption throughout the system. Irrigation water (in open-field) was the main contribution to the total water consumption. In the greenhouse option (Table A1) there was a more conservative use of water, with 53% lower water consumption per functional unit than in the open-field option. Montero et al. (2009) and Stanghellini et al. (2003) found that solar radiation within the greenhouse was between 25-40% lower than outside and that air circulation was considerably reduced in a closed environment, resulting in less evaporation and evapotranspiration, which are of primary importance in regions with a semi-arid climate, such as the Mediterranean. The other reason for the lower water consumption is a result of the higher production per square meter (Table 5.3, Chapter 5).

With reference to irrigation water consumption per square meter, which depended on the tensiometer reading, the difference between OF_CL and OF_M was less than 3%, while for GH_M the consumption was 27% higher than for GH_CL (Table 5.2, Chapter 5). The latter would appear to be a result of the greater biomass production in option GH_M, with the consequently higher water requirements (Muñoz et al., 2008c) and of the improvement in the water holding capacity of the soil due to compost application in option GH_CL, particularly in sandy soils (Hargreaves et al., 2008). In spite of the differences in the amount of biomass, the tomato production was statistically equal for GH_CL and GH_M options, as a larger amount of biomass (leafs, stalk and roots) is not always related to an increase in commercial fruit production.

### *(ii) Energy, mineral fertilizers, pesticides and transport*

As can be seen from Table A1, the energy and mineral fertilizers consumptions and the transport per functional unit for OF_CL were between 40-100% higher than for GH_CL, mainly due to differences in the tomato harvests and to the large dosage of mineral fertilizers and pesticides in OF_CL. The energy and transport flows were almost all a consequence of compost production or transport. The main contributions to energy were the diesel and electricity consumption of the composting plant and for transport, OFMSW collection and compost transport to the fields.

**Table A1. Inputs and outputs inventory for the compost cultivation options including all the stages and sub-stages. Units per functional unit (ton of commercial tomato).**

| Flow | Units (Per $FU^{-1}$) | $OF\_C_L$ | $GH\_C_L$ | Comments |
|---|---|---|---|---|
| | | | | ***Inputs*** |
| **Water** | $m^3$ | 68.0 | 32.2 | The water used for cleaning the OFMSW containers and the rain and irrigation water. |
| **Energy** | MJ | 564.2 | 338.5 | The electricity and diesel consumption of the composting plant, the electricity of the irrigation pumps, the diesel of the tractor, the consumption of the installation machinery of the greenhouse and the electricity consumption of the ventilators (for options in greenhouse). |
| **Mineral fertilizers** | kg | 20.6 | 7.9 | In open-field, high nitrogen water content was accounted as an addition of synthetic acid nitric. |
| **Pesticides** | kg | 0.2 | 0.0 | The pesticides used in the fields. |
| **Infrastruc-ture materials** | kg | 22.5 | 46.8 | The OFMSW collection container and the building and main machinery of the composting plant, the fertirrigation infrastructure, the tractor and other agricultural machinery, the crates and steel handcarts for harvesting, the greenhouse structure and the machinery for its construction (for options in greenhouse). |
| **Transport** | tkm | 119.2 | 70.6 | The transport of OFMSW, the several infrastructure materials considered, the compost, the mineral fertilizers and the several wastes considered. |
| | | | | ***Outputs*** |
| **Waste** | kg | 283.0 | 174.9 | The waste of building and machinery and solid waste fraction of the composting plant, the fertirrigation infrastructure, the crates and steel handcarts and the greenhouse structure (for options in greenhouse). |
| **Emissions** | kg | 1.6 | 1.3 | Ammonia, volatile organic compounds, methane and nitrous oxide emissions generated in the composting plant. Post-application emissions were not included. |

*Cultivation options: $OF\_C_L$, Option in open-field with compost in a low dosage plus mineral fertilizers. $GH\_C_L$, Option in greenhouse with compost in a low dosage plus mineral fertilizers. FU, functional unit.*

### *(iii) Infrastructure materials*

The element with most material requirements of the system (in weight) was greenhouse structure, with the option under greenhouse needing 24 kg more of materials than the open-field option per functional unit (Table A1).

### *(iv) Waste and emissions*

Most of the waste and emissions released to the environment were produced during compost production. Such flows were, respectively, 38 and 20% higher for $OF\_C_L$ than for $GH\_C_L$, mainly due to differences in tomato production.

# Annex I.II. Harvesting and fruit size parameters for cultivation options with compost as only fertilizer

Although compost alone was applied as fertilizer in the experimental fields, it was not considered in the study for the reasons below. Under greenhouse, with compost fertilization alone the minimum tomato production for the zone (Muñoz et al. 2008b; Muñoz et al. 2008c) was not reached, with the harvest reduced by around 40% compared to the options GH_C$_L$ and GH_M. The high nitrogen demand of tomato plants in early cultivation stages cannot be covered by compost alone (Table A2), due to its slow release rates of nutrients (Hargreaves et al. 2008). In open-field cultivation, fertilization without added nitrogen was not an option, due to the particular characteristics of the irrigation water, with a high concentration of nitrates (Martínez-Blanco et al. 2009), but we have previously demonstrated considerably lower production in fields with compost as the only fertilizer (Muñoz et al. 2008a).

**Table A.2. Parameters of harvest and fruit size for cultivation options with compost alone.**

| Parameter | Units | Compost in open-field (OF_C$_H$) | Compost in greenhouse (GH_C$_H$) |
|---|---|---|---|
| **Commercial production** | t ha$^{-1}$ | 104 | 96 |
| **Total production** [a] | t ha$^{-1}$ | 118 | 101 |
| **Tomato average diameter** | mm | 79.1 | 82.7 |
| **Tomato average weight** | g | 209.1 | 236.64 |

*Cultivation options: OF_C$_H$, Option in open-field with compost in a high dosage plus mineral fertilizers. GH_C$_L$, Option in greenhouse with compost in a high dosage plus mineral fertilizers.*
*[a] Total production is the addition of commercial plus non-commercial tomato production.*

# Annex I.III. Total environmental impacts of the four cultivation options

The absolute values of the environmental impacts of the four cultivation options considering (Table A3) and without considering (Table A4) the avoided burdens through composting by not dumping OFMSW and green waste are showed in the following tables.

**Table A3. Total environmental impacts of the four cultivation options without considering the avoided burdens of composting by not dumping OFMSW and GW.**

| Impact category | Unit (Per FU$^{-1}$) | Open-field | | Greenhouse | |
|---|---|---|---|---|---|
| | | OF_C$_L$ | OF_M | GH_C$_L$ | GH_M |
| **ADP** | kg Sb eq. | 1.645E+00 | 9.459E-01 | 1.289E+00 | 1.057E+00 |
| **GWP** | kg $SO_2$ eq. | 1.685E+00 | 8.880E-01 | 1.036E+00 | 9.381E-01 |
| **OLDP** | kg $PO_4^{3-}$ eq. | 1.019E+00 | 2.343E-01 | 6.520E-01 | 3.488E-01 |
| **POP** | kg $CO_2$ eq. | 2.887E+02 | 1.555E+02 | 1.823E+02 | 1.526E+02 |
| **AP** | kg CFC-11 eq. | 3.136E-05 | 1.858E-05 | 1.846E-05 | 1.388E-05 |
| **EP** | kg $C_2H_4$ eq. | 5.389E-01 | 2.284E-02 | 3.153E-01 | 3.194E-02 |
| **CED** | MJ eq. | 3.989E+03 | 2.260E+03 | 3.137E+03 | 2.550E+03 |

*See acronyms from Table A4.*

**Table A4. Total environmental impacts of the four cultivation options considering the avoided burdens through composting by not dumping OFMSW and GW.**

| Impact category | Unit (Per $FU^{-1}$) | Open-field | | Greenhouse | |
|---|---|---|---|---|---|
| | | OF_$C_L$ | OF_M | GH_$C_L$ | GH_M |
| **ADP** | kg Sb eq. | 9.611E-01 | 9.459E-01 | 8.989E-01 | 1.057E+00 |
| **GWP** | kg $SO_2$ eq. | 1.151E+00 | 8.880E-01 | 7.308E-01 | 9.381E-01 |
| **OLDP** | kg $PO_4^{3-}$ eq. | -1.997E+00 | 2.343E-01 | -1.068E+00 | 3.488E-01 |
| **POP** | kg $CO_2$ eq. | -2.665E+02 | 1.555E+02 | -1.344E+02 | 1.526E+02 |
| **AP** | kg CFC-11 eq. | 1.642E-05 | 1.858E-05 | 9.994E-06 | 1.388E-05 |
| **EP** | kg $C_2H_4$ eq. | 3.815E-01 | 2.284E-02 | 2.256E-01 | 3.194E-02 |
| **CED** | MJ eq. | 2.244E+03 | 2.260E+03 | 2.142E+03 | 2.550E+03 |

*Cultivation options: OF_$C_L$, cultivation in open-field with compost in a low dosage plus mineral fertilizers: OF_M, cultivation in open-field with mineral fertilizers; GH_$C_L$, cultivation with compost in a low dosage in greenhouse plus mineral fertilizers; GH_M, cultivation in greenhouse with mineral fertilizers.*
*FU, functional unit.*

# Annex I.IV. Environmental assessment considering normal tomato productions in the region

According to previous studies in this area (Muñoz et al. 2005; Muñoz et al. 2008a; Muñoz et al. 2008b), normal tomato production in open-field is around 85 t $ha^{1}$ and that in greenhouses between 150 and 200 t $ha^{-1}$. In Table A5 the total environmental impacts of the four cultivation options considering the average tomato productions in the region are showed.

**Table A5. Total environmental impacts of the four cultivation options considering the average tomato productions in the region (open-field, 85 t $ha^{-1}$; greenhouse, 170 t $ha^{-1}$) and the avoided burdens through composting by not dumping OFMSW and green waste.**

| Impact category | Unit (Per FU-1) | Open-field | | Greenhouse | |
|---|---|---|---|---|---|
| | | OF_CL | OF_M | GH_CL | GH_M |
| **ADP** | kg Sb eq. | 1.176E+00 | 1.146E+00 | 5.076E-01 | 9.883E+00 |
| **GWP** | kg $SO_2$ eq. | 1.408E+00 | 1.076E+00 | 4.217E-01 | 8.774E-01 |
| **OLDP** | kg $PO_4^{3-}$ eq. | -2.444E+00 | 2.840E-01 | -6.034E-01 | 3.262E-01 |
| **POP** | kg $CO_2$ eq. | -3.261E+02 | 1.884E+02 | -7.590E+01 | 1.427E+02 |
| **AP** | kg CFC-11 eq. | 2.009E-05 | 2.252E-05 | 5.615E-06 | 1.299E-05 |
| **EP** | kg $C_2H_4$ eq. | 4.668E-01 | 2.767E-02 | 1.274E-01 | 2.987E-02 |
| **CED** | MJ eq. | 2.746E+03 | 2.738E+03 | 1.209E+03 | 2.385E+03 |

*See acronyms from Table A4.*

# Annex II. Supplementary information for Chapter 7

The **Annex II** corresponds to some supplementary tables and figures for **Chapter 7**.

**Chapter 7** is based on the following book chapter:

Martínez-Blanco, J., Lazcano, C., Boldrin, A., Muñoz, P., Rieradevall, J., Möller, J., Antón, A., & Christensen, T.H. Assessing the benefits of compost use-on-land through an LCA perspective: a review. Accepted for publication (May 2012) in the Volume 11 of the Sustainable Agriculture Reviews, Eric Lichtfouse (editor).

Moreover, the following paper outlines what is said in **Chapter 7**:

Martínez-Blanco, J., Lazcano, C., Boldrin, A., Muñoz, P., Rieradevall, J., Möller, J., Antón, A., & Christensen, T.H. State of the art and future challenges for quantification of compost use-on-land through an LCA perspective. Accepted on May 2012 for publication in the international journal Agronomie for Sustainable Development.

**Annex II** is including 8 tables and one figure referring to the following issues:

- Nutrient supply
- Carbon sequestration
- Weed, pest and disease suppression
- Crop yield
- Soil erosion
- Soil moisture content
- Soil workability
- Soil biological properties and biodiversity
- Crop nutritional quality

References are at the end of Annexes.

**Table A6. NUTRIENT SUPPLY: Literature review of nutrient released from compost to soils during 1-5 years after its application.**

| Study | Compost feedstock | N released from total (%) | | P released from total (%) | | K released from total (%) | |
|---|---|---|---|---|---|---|---|
| | | short-term | mid-term | short-term | mid-term | short-term | mid-term |
| | | | ***Reviews*** | | | | |
| **Amlinger et al.** (2003b) | bio-waste and garden waste | 5-15% (1 yr) | 2-8% (each yr) 50% (5 yr) | | | | |
| **Amlinger et al.** (2003a) | bio-waste | 10-15% (1 yr) | | 35% (1 yr) | | 75% (1 yr) | |
| **ROU** (2007) | food and garden waste | 15% N (1 yr) | 40% (3-5 yr) | 38% (1 yr) | 100% (3-5 yr) | 80% (1 yr) | 100% (3-5 yr) |
| **Hargreaves et al.**(2008) | MSW | 16-21% (6 months) | | | | | |
| | | 10% (1 yr) | | | | | |
| | | 10-22% (1 yr) | | | | | |
| **Diacono & Montemurro** (2010) | several feedstocks | 15-20 (1 yr) | 3-8% (each yr) | | | | |
| | | | ***Individual papers*** | | | | |
| **Terman et al.** (1973)[a] | MSW | 16% | | | | | |
| **Murillo et al.** (1995)[a] | MSW | 22% | | | | | |
| **Hadas & Portnoy** (1997)[a] | MSW | 22% (5 months) | | | | | |
| **Biala & Wynen** (1998)[a] | garden waste | 15% (1 yr) | 25-35% (2 yr) | 50% (1 yr) | 50% (2 yr) | 80% (1 yr) | 20% (2 yr) |
| **Sikora** (1997)[a] | MSW | 10% | | | | | |
| **Sullivan et al.** (1998)[a] | food waste | 7.6 % – 8.1% | | | | | |
| **Mamo et al.** (1999)[a] | MSW | 0-12 % (1 yr) | | | | | |
| **Frossard et al.** (2002)[a] | organic solid waste | | | 2-16% | 40-77% | | |
| **Houot et al.** (2002)[a] | MSW | 3% | | | | | |
| **Sullivan et al.** (2002)[a] | food waste | 0% | | | | | |
| **Guerini et al.** (2006) | MSW | 20% (1 yr) | 10% (2 yr)<br>5% (3 yr) | | | | |

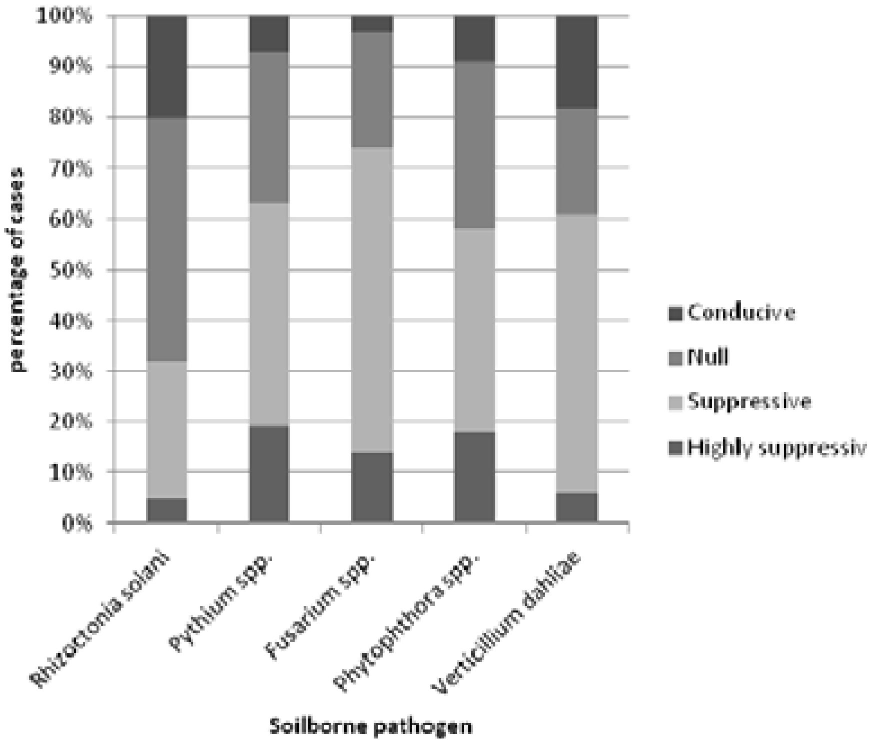
| | | | |
|---|---|---|---|
| **Hansen et al.** (2006) | MSW | 30% (1 yr) | 30%<br>30% (2 yr) |
| **Elherradi et al.** (2005) | household | 15-24% (4 months) | |

*MSW, Municipal Solid Waste; yr, year.*
*[a] Reviewed in ROU (2007)*

**Table A7. CARBON SEQUESTRATION: Literature review of compost effects on soil carbon sequestration.**

| Study | Compost feedstock | Dosage ww (t $ha^{-1}$ $yr^{-1}$) | C addition | Crop | Type of soil | Length (yr) | C retained (%) |
|---|---|---|---|---|---|---|---|
| **Smith et al.** (2001)[a] | | | | | | 100 | 8.2 |
| **Eghball** (2002) | feedlot manure | 21.5[b] | 1.7 t $ha^{-1}$ $yr^{-1}$ | maize | silty clay loam, mesic typic argiudolls | 4 | 36 |
| **Fortuna et al.** (2003) | oak leaves-manure (1:1) | 4.48[b] | | maize-wheat | kalamazoo loam | 4 | 63 |
| **Sánchez-Monedero et al.** (2008)[c] | two phase olive mill waste+ sheep litter | 48 | 25.44 t $ha^{-1}$ $yr^{-1}$ | no crop | na | 0.77 | 63.1 |
| | two phase olive mill waste+ sheep litter+ grape stalks | 48 | 25.48 t $ha^{-1}$ $yr^{-1}$ | no crop | na | 0.77 | 66.1 |
| | two phase olive mill waste+ sheep litter | 48 | 14.4 t $ha^{-1}$ $yr^{-1}$ | no crop | na | 0.77 | 91.6 |
| | two phase olive mill waste+ sheep litter+ grape stalks | 48 | 11.04 t $ha^{-1}$ $yr^{-1}$ | no crop | na | 0.77 | 91.93 |
| **Sodhi et al.** (2009) | rice straw compost | 4 | 1.24 t $ha^{-1}$ | rice–wheat | sandy-loam typic ustipsamment | 10 | 29 |
| | rice straw compost | 16 | 4.96 t $ha^{-1}$ | rice–wheat | sandy-loam typic ustipsamment | 10 | 29 |
| **Fabrizio et al.** (2009) | food residues and ligno-cellulosic wastes | 50 | 2.72 g $kg^{-1}$ soil | maize | silt-clay fluvic-eutric cambisol | 0.41 | 40 |
| | food residues and ligno-cellulosic wastes | 85 | 4.64 g $kg^{-1}$ soil | maize | silt-clay fluvic-eutric cambisol | 0.41 | 53 |
| **Fagnano et al.** (2011) | MSW | 10 | 1282 kg $ha^{-1}$ | lettuce | sandy loam eutric regosol | 0.5 | 63.1 |
| | MSW | 30 | 3847 kg $ha^{-1}$ | lettuce | sandy loam eutric regosol | 0.5 | 70.3 |
| | MSW | 60 | 7694 kg $ha^{-1}$ | lettuce | sandy loam eutric regosol | 0.5 | 69.9 |

*MSW, Municipal solid waste; yr, year; na, not available.*
[a] *modeling study.*
[b] *dry weight.*
[c] *Incubation study.*




*Data are the percentage of cases with highly suppressive (>80% disease reduction), suppressive (significant disease reduction), null (no significant effect) or conducive effects (significant disease increase). The total suppressive cases are the sum of HS and S. Only combinations with at least 10 studies are shown.*

**Figure A1. <u>WEED, PEST AND DISEASE SUPPRESSION</u>: Effect of compost amendments on disease incidence and severity caused by soilborne pathogens, compared to the non-amended control.**

*Source: Bonanomi et al.* (2007).

**Table A8. CROP YIELD: Literature review of crop yield effects after compost application in soils.**

| Study | Compost feedstock | Dosages ww (t $ha^{-1}$ $yr^{-1}$) | Supplementary fertilizing treatment | Crop | Type of soil | Length (yr) | Effect on yield (%) | Nutrient balance | Minerali-zation rate |
|---|---|---|---|---|---|---|---|---|---|
| | | | | *Comparison with mineral fertilizer treatment* | | | | | |
| **Iglesias-Jimenez & Alvarez** (1993)[a] | MSW | 10 | no | ryegrass | andeptic paleudult | 0.5 | M>C (+136%) | yes | nm |
| | MSW | 50 | no | ryegrass | andeptic paleudult | 0.5 | M>C (+495%) | yes | nm |
| **CIWMB** (1997) | grass clippings | 13[b] | NPK | cotton+winter forage crop | sandy loam | 1 | ns | no | no |
| | grass clippings | 15[b] | NPK | irrigated corn/wheat | sandy loam | 1 | M>C (+8%) | no | no |
| **Diez et al.** (1997) | MOW | 28[c] | 150 kg N $ha^{-1}$ | maize | sandy-loam | 3 | ns | nm | yes |
| | MOW | 28[c] | no | wheat | sandy-loam | 3 | ns | nm | yes |
| | MOW | 28[c] | no | maize | sandy-loam | 3 | ns | nm | yes |
| **Bazzoffi et al.** (1998) | MSW | 96[c] | no | maize | clay loam typic udorthent | 1 | ns | no | no |
| | MSW | 96[c] | no | maize | clay loam typic udorthent | 2 | ns | no | no |
| | MSW | 96[c] | no | maize | clay loam typic udorthent | 3 | M>C (+71%) | no | no |
| **Eriksen et al.** (1999) | MSW | 63[c] | no | maize | sandy, siliceous, mesic psammentic hapludults | 2 | ns | nm | no |
| | MSW | 126[c] | no | maize | sandy, siliceous, mesic psammentic hapludults | 2 | ns | nm | no |
| | MSW | 189[c] | no | maize | sandy, siliceous, mesic psammentic hapludults | 2 | ns | nm | no |
| **Mamo et al.** (1999) | MSW | 90 | 125 kg N $ha^{-1}$ | corn | loam sandy | 3 | ns | no | no |
| | MSW | 90 | 250 kg N $ha^{-1}$ | corn | loam sandy | 3 | ns | no | no |
| **Wolwoski** (2003) | MSW | 35[b] | no | grain | saybrook silt loam | 1 | M>C (+37%) | nm | nm |
| | MSW | 138[b] | no | grain | saybrook silt loam | 1 | M>C (+28%) | nm | nm |

| | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | MSW | 35[b] | no | grain | boyer loamy sand | 1 | M<C (-22%) | nm | nm |
| | MSW | 138[b] | no | grain | boyer loamy sand | 1 | M<C (-52%) | nm | nm |
| | MSW | 35[b] | no | grain | saybrook silt loam | 2 | M>C (+11%) | nm | nm |
| | MSW | 138[b] | no | grain | saybrook silt loam | 2 | ns | nm | nm |
| | MSW | 35[b] | no | grain | boyer loamy sand | 2 | ns | nm | nm |
| | MSW | 138[b] | no | grain | boyer loamy sand | 2 | ns | nm | nm |
| **Elherradi et al.** (2005)[a] | household | 10 | no | lettuce | sandy | 0.3 | M>C (+98%) | no | no |
| | household | 10 | no | lettuce | loamy-clay | 0.3 | M>C (+39%) | no | no |
| | household | 20 | no | lettuce | sandy | 0.3 | M>C (+81%) | no | no |
| | household | 20 | no | lettuce | loamy-clay | 0.3 | ns | no | no |
| | household | 30 | no | lettuce | sandy | 0.3 | M>C (+45%) | no | no |
| | household | 30 | no | lettuce | loamy-clay | 0.3 | ns | no | no |
| | household | 10 | 50% NPK | lettuce | sandy | 0.3 | M>C (+37%) | no | no |
| | household | 10 | 50% NPK | lettuce | loamy-clay | 0.3 | ns | no | no |
| | household | 20 | 50% NPK | lettuce | sandy | 0.3 | M>C (+27%) | no | no |
| | household | 20 | 50% NPK | lettuce | loamy-clay | 0.3 | ns | no | no |
| | household | 30 | 50% NPK | lettuce | sandy | 0.3 | ns | no | no |
| | household | 30 | 50% NPK | lettuce | loamy-clay | 0.3 | ns | no | no |
| | household | 10 | 100% NPK | lettuce | sandy | 0.3 | ns | no | no |
| | household | 10 | 100% NPK | lettuce | loamy-clay | 0.3 | ns | no | no |
| | household | 20 | 100% NPK | lettuce | sandy | 0.3 | ns | no | no |
| | household | 20 | 100% NPK | lettuce | loamy-clay | 0.3 | ns | no | no |
| | household | 30 | 100% NPK | lettuce | sandy | 0.3 | ns | no | no |
| | household | 30 | 100% NPK | lettuce | loamy-clay | 0.3 | ns | no | no |
| **Mkhabela and** | MSW | 17 | no | potato | pugwash sandy loam | 1 | M>C (+44%) | no | no |

| | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| **Warman** (2005)[d] | MSW | 33 | no | potato | pugwash sandy loam | 1 | M>C (+50%) | no | no |
| | MSW | 50 | no | potato | pugwash sandy loam | 1 | ns | no | no |
| | MSW | 17 | no | potato | pugwash sandy loam | 2 | ns | no | no |
| | MSW | 33 | no | potato | pugwash sandy loam | 2 | ns | no | no |
| | MSW | 50 | no | potato | pugwash sandy loam | 2 | ns | no | no |
| | MSW | 6 | no | sweet corn | pugwash sandy loam | 1 | ns | no | no |
| | MSW | 12 | no | sweet corn | pugwash sandy loam | 1 | M>C (+18%) | no | no |
| | MSW | 17 | no | sweet corn | pugwash sandy loam | 1 | M>C (+24%) | no | no |
| | MSW | 6 | no | sweet corn | pugwash sandy loam | 2 | ns | no | no |
| | MSW | 12 | no | sweet corn | pugwash sandy loam | 2 | ns | no | no |
| | MSW | 17 | no | sweet corn | pugwash sandy loam | 2 | ns | no | no |
| **Montemurro et al.** (2005) | MSW | 5 | 50% N | sunflower | silty-clay | 2 | ns | yes | no |
| | MSW | 9 | no | sunflower | silty-clay | 2 | ns | yes | no |
| **Guerini et al.** (2006) | MSW | 60 | NPK | maize | sandy loam | 5 | ns | yes | yes |
| | MSW | 60 | NPK | maize | clay loam | 6 | ns | yes | yes |
| **Montemurro et al.** (2006) | MSW | na | | alfalfa | | 3 | ns | yes | nm |
| | MSW | na | | cocksfoot | | 3 | M>C (+21%) | yes | nm |
| **Ros et al.** (2006a) | urban organic waste | 14.6 | no | maize | loamy silt | 12 | M>C (+23%) | nm | yes |
| | urban organic waste | 14.6 | 80 kg N $ha^{-1}$ | maize | loamy silt | 12 | ns | nm | yes |
| | green waste | 10.9 | no | maize | loamy silt | 12 | M>C (+16%) | nm | yes |
| | green waste | 10.9 | 80 kg N $ha^{-1}$ | maize | loamy silt | 12 | ns | nm | yes |
| **Ghorbani et al.** (2008) | household | 20[c] | no | tomato | sandy clay loam | 1 | ns | nm | no |
| | household | 20[c] | no | tomato | sandy clay loam | 2 | M>C (+33%) | nm | no |
| **Odlare et al.** (2008) | separated household waste + garden litter | 4[b] | 50 kg N $ha^{-1}$ | oat+spring barley | sandy-clay loam Eutric cambisol | 4 | ns | no | yes |

| | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | separated household waste + garden litter | 8[b] | no | oat+spring barley | sandy-clay loam Eutric cambisol | 4 | M>C (+44%) | no | yes |
| **Coria-Cayupan et al.** (2009) | fruit and vegetables | 10 | no | lettuce | nm | crop period | ns | yes | nm |
| | MSW | 10 | no | lettuce | nm | crop period | M>C (+138%) | yes | nm |
| **Montemurro** (2009) | MSW | 5 | 50% N | winter wheat | silty-clay | 4 | M<C (-7%) | yes | no |
| | MSW | 9 | no | winter wheat | silty-clay | 4 | M>C (+11%) | yes | no |
| **Celik et al.** (2010) | grass, wheat stubbles and plant leaves | 38[b] | no | winter wheat | typic xerofluvent clay–loam | 13 | M>C (+56%) | no | no |
| **Morra et al.** (2010) | MSW | 15 | no | horticulture rotation | sandy loam calcaric cambisol | 3 | ns | no | no |
| | MSW | 30 | no | horticulture rotation | sandy loam calcaric cambisol | 3 | ns | no | no |
| | MSW | 45 | no | horticulture rotation | sandy loam calcaric cambisol | 3 | ns | no | no |
| | MSW | 15 | 50% NPK | horticulture rotation | sandy loam calcaric cambisol | 3 | ns | no | no |
| | MSW | 15 | 25% NPK | horticulture rotation | sandy loam calcaric cambisol | 3 | ns | no | no |
| | | | | ***Comparison with control treatment*** | | | | | |
| **Iglesias-Jimenez & Alvarez** (1993)[a] | MSW | 10.0 | no | ryegrass | andeptic paleudult | 0.5 | N<C (+30%) | | |
| | MSW | 50.0 | no | ryegrass | andeptic paleudult | 0.5 | N<C (+106%) | | |
| **Sabrah** (1995) | urban waste | 16.5 | NPK | irrigated wheat | sandy | 1 | ns | | |
| | urban waste | 16.5 | NPK | irrigated wheat | sandy | 2 | ns | | |
| | urban waste | 33 | NPK | irrigated wheat | sandy | 1 | N<C (+11%) | | |
| | urban waste | 33 | NPK | irrigated wheat | sandy | 2 | N<C (+34%) | | |
| | urban waste | 49.5 | NPK | irrigated wheat | sandy | 1 | N<C (+36%) | | |

| | | | | | | |
|---|---|---|---|---|---|---|
| | urban waste | 49.5 | NPK | irrigated wheat | sandy | 2 | N<C (+43%) |
| | urban waste | 66 | NPK | irrigated wheat | sandy | 1 | N<C (+45%) |
| | urban waste | 66 | NPK | irrigated wheat | sandy | 2 | N<C (+48%) |
| **Mamo et al.** (1999) | MSW | 90.0 | no | corn | loam sandy | 3 | N<C (+71%) |
| **Movahedi Naeini & Cook** (2000) | urban compost | 50.0 | 125 kg N $ha^{-1}$ | rainfed maize | silt loam | 1 | N<C (+10%) |
| | urban compost | 50.0 | 125 kg N $ha^{-1}$ | rainfed maize | silt loam | 1 | ns |
| **Hartl et al.** (2003) | bio-waste | 22[d] | no | rye | calcaric fluvisol | 5 | ns |
| | bio-waste | 12[d] | no | rye | calcaric fluvisol | 5 | ns |
| **Tejada et al.** (2006a) | crushed cotton gyn | 16 | no | rice | aquic xerofluvent | 3 | N<C (+5%) |
| | crushed cotton gyn | 24 | no | rice | aquic xerofluvent | 3 | N<C (+7%) |
| | crushed cotton gyn | 32 | no | rice | aquic xerofluvent | 3 | N<C (+7%) |
| | crushed cotton gyn | 16 | 250 kg N | rice | aquic xerofluvent | 3 | N<C (+7%) |
| | crushed cotton gyn | 24 | 250 kg N | rice | aquic xerofluvent | 3 | N<C (+9%) |
| | crushed cotton gyn | 32 | 250 kg N | rice | aquic xerofluvent | 3 | N<C (+11%) |

*MSW, Municipal solid waste; MOW, Municipal organic waste; M, mineral fertilizers option; C, compost option; N, option of no fertilization.*
*na, not available; nm, not mentioned; ns, no significant differences.*
[a] *Case study of pot experiment.*
[b] *Only dry weight is available in the study. We considered 35% moisture content for the ww calculation.*
[c] *Compost was applied once at the beginning of the experiment.*
[d] *The dosages were different each year. An average value is provided.*

**Table A9. SOIL EROSION: Literature review of the effect of compost application on the soil parameters related to soil erosion.**

| Study | Compost feedstock | Dosage ww (t $ha^{-1}$ $yr^{-1}$) | Crop | Type of soil | Length (yr) | Change (%) after compost application |
|---|---|---|---|---|---|---|
| | | *% Water stable aggregates* | | | | |
| **Sodhi et al.** (2009) | rice straw | 8 | rice-wheat | typic ustipsamment | 10 | 4 |
| | | *Soil structural stability or aggregate stability* | | | | |
| **Tejada & González** (2006b)[a] | cotton gin crushed | 30 | wheat | typic xerofluvent | 5 | 0 |
| **Annabi et al.** (2006)[b] | MSW | | | | 5 | 29.3 |
| **Annabi et al.** (2007)[c] | MSW | 34.59 g $kg^{-1}$ | no crop | typic hapludalf | <1 | 41 |
| | bio-waste | 74.79 g $kg^{-1}$ | no crop | typic hapludalf | <1 | 29 |
| | green waste+ sludge | 21.3 g $kg^{-1}$ | no crop | typic hapludalf | <1 | 29 |
| **Bipfubusa et al.** (2008)[d] | paper sludge | 40 | corn | silt loam | 4 | 45 |
| **Leroy et al.** (2008) | vegetable, fruit and garden waste | 22.5 | corn | sandy loam | 9 | 63 |
| **Tejada et al.** (2008) | green manure + beet vinase | 10 | no crop | xelloric calciorthid | 4 | 10.5 |
| **Tejada et al.** (2009b) | plant residues (rapeseed) | 17.7 | no crop | xelloric calciorthid | 4 | 28.3 |
| **Arthur et al.** (2011) | garden waste | 30 $m^3$ $ha^{-1}$ | cauliflower | sandy loam, haplic podzol | 9 | 45 |
| | vegetable, fruit and garden waste | 30 $m^3$ $ha^{-1}$ | cauliflower | sandy loam, haplic podzol | 9 | 27.5 |
| | spent mushroom | 30 $m^3$ $ha^{-1}$ | cauliflower | sandy loam, haplic podzol | 9 | 25 |
| | | *Aggregate instability* | | | | |
| **Aggelides & Londra** (2000) | MSW+ sewage sludge+saw dust | 39 | grass fallow | Clay soil, humic fluvaquent | 1 | -6.89 |
| | MSW+ sewage sludge+saw dust | 78 | grass fallow | Clay soil, humic fluvaquent | 1 | -6.89 |
| | MSW+ sewage sludge+saw dust | 156 | grass fallow | Clay soil, humic fluvaquent | 1 | -17 |
| | MSW+ sewage sludge+saw dust | 39 | grass fallow | Loam soil, typic xerochrept | 1 | -13 |
| | MSW+ sewage sludge+saw dust | 78 | grass fallow | Loam soil, typic xerochrept | 1 | -19 |

| | | | | | | |
|---|---|---|---|---|---|---|
| | MSW+ sewage sludge+saw dust | 156 | grass fallow | Loam soil, typic xerochrept | 1 | -30 |
| **Tejada & González** (2008)[a] | Cotton gin crushed | 20 | no crop | xerollic calciorthid | 4 | -21 |
| | | ***Soil loss*** | | | | |
| **Bazzoffi et al.** (1998)[e] | | 64 | corn | clay loam, typic udorthent, calcaric regosol | 3 | -5 |
| **Tejada & González** (2006b)[a] | cotton gin crushed | 30 | wheat | typic xerofluvent | 5 | -36 |
| **Tejada & González** (2008)[b] | cotton gin crushed | 20 | no crop | xerollic calciorthid | 4 | -29.2 |
| | cotton gin crushed | 20 | no crop | xerollic calciorthid | 4 | -29.2 |
| **Tejada et al.** (2009a) [a] | beet vinasse+ green waste vermicompost | 15.82 | no crop | xerollic calciorthid | 3 | -28.9 |
| | | ***Runnoff*** | | | | |
| **Bresson et al.** (2001) [ac] | MSW | 50 | no crop | silt loam, typic hapludalf | 60 min | -66.5 |
| **Arthur et al.** (2011) | garden waste | 30 m$^3$ ha$^{-1}$ | cauliflower | sandy loam, haplic podzol | 9 | -26 |
| | vegetable, fruit and garden waste | 30 m$^3$ ha$^{-1}$ | cauliflower | sandy loam, haplic podzol | 9 | -30 |
| | spent mushroom | 30 m$^3$ ha$^{-1}$ | cauliflower | sandy loam, haplic podzol | 9 | -30 |
| | | ***Soil erodibility*** | | | | |
| **Arthur et al.** (2011) | garden waste | 30 m$^3$ ha$^{-1}$ | cauliflower | sandy loam, haplic podzol | 9 | -18 |
| | vegetable, fruit and garden waste | 30 m$^3$ ha$^{-1}$ | cauliflower | sandy loam, haplic podzol | 9 | -27 |
| | spent mushroom | 30 m$^3$ ha$^{-1}$ | cauliflower | sandy loam, haplic podzol | 9 | -18 |

[a] *Data obtained with simulated rain.*
[b] *Three applications over 5 years.*
[c] *Incubation study.*
[d] *Three applications per year during 2 years.*
[e] *Only one compost application at the beginning of the experiment.*

**Table A10. SOIL MOISTURE CONTENT: Literature review of soil moisture content after compost application in soils.**

| Study | Compost feedstock | Dosages ww (t ha$^{-1}$ yr$^{-1}$) | Crop | Type of soil | Length (yr) | Change on the water capacity indicator (%) |
|---|---|---|---|---|---|---|
| | | | ***Water moisture savings*** | | | |
| **CIWMB** (1997) | grass clippings | 13.3[a] | irrigated corn/wheat | sandy loam | 1 | 0 |
| | grass clippings | 14.9[a] | irrigated corn/wheat | sandy loam | 2 | 0 |
| **Glab et al.** (2009) | bio-waste | 7 | rotation cereal/potato | mollic-gleyic fluvisol | 13 | 0 |
| | bio-waste | 13 | rotation cereal/potato | mollic-gleyic fluvisol | 13 | 0 |
| | bio-waste | 18 | rotation cereal/potato | mollic-gleyic fluvisol | 13 | 0 |
| | | | ***Water holding capacity (WHC)*** | | | |
| **Hortenstine & Rothwell** (1973) | municipal waste | 16 | irrigated sorghum | sandy | na | 0 |
| | municipal waste | 32 | irrigated sorghum | sandy | na | 5 |
| | municipal waste | 64 | irrigated sorghum | sandy | na | 26 |
| | municipal waste | 128 | irrigated sorghum | sandy | na | 50 |
| **Mamo et al.** (1999) | urban waste | 138.5[a] | irrigated corn | loamy sand | na | 4.4 |
| **Movahedi Naeini & Cook** (2000) | urban waste | 50 | rainfed corn | silt loam | 1 | 2 |
| | urban waste | 50 | rainfed corn | silt loam | 1 | 0 |
| **Bazzoffi et al.** (2006)[b] | MSW | 60[c] | no crop | silty clay soil typic eutrochrept | 1 | 7 |
| | MSW | 60[c] | no crop | silty clay soil typic eutrochrept | 3 | 8 |
| | MSW | 180[c] | no crop | silty clay soil typic eutrochrept | 1 | 0 |
| | MSW | 180[c] | no crop | silty clay soil typic eutrochrept | 3 | 3 |
| | MSW | 60[c] | no crop | sandy loam texture typic udifluvent | 1 | 0 |
| | MSW | 60[c] | no crop | sandy loam texture typic udifluvent | 3 | 0 |
| | MSW | 180[c] | no crop | sandy loam texture typic udifluvent | 1 | 0 |

| | | | | | | |
|---|---|---|---|---|---|---|
| | MSW | 180[c] | no crop | sandy loam texture typic udifluvent | 3 | 0 |
| **Suzuki et al.** (2007) | leaf litter | 10 | forage sorghum | sandy soil | 1 | 0 |
| **Albaladejo et al.** (2009) | MSW | 205.5 | no crop | typic calcisol | 1 | 18 |
| | MSW | 342.5 | no crop | typic calcisol | 1 | 26 |
| **Castillejo & Castelló** (Castillejo and Castello 2010) [d] | MSW | 10 | fodder shrub (Atriplex halimus) | gypsum spoil (soil forming material) | 1 | 0 |
| | MSW | 30 | fodder shrub (Atriplex halimus) | gypsum spoil (soil forming material) | 1 | 19 |
| | MSW | 50 | fodder shrub (Atriplex halimus) | gypsum spoil (soil forming material) | 1 | 25 |
| | | | ***Plant available water (PAW)*** | | | |
| **Sabrah et al.** (1995) | urban waste | 16.5 | irrigated wheat | sandy | 2 | 16.5 |
| | urban waste | 33 | irrigated wheat | sandy | 2 | 30 |
| | urban waste | 49.5 | irrigated wheat | sandy | 2 | 35 |
| | urban waste | 66.0 | irrigated wheat | sandy | 2 | 43 |
| **Curtis and Claassen** (2005)[d] | garden waste | 415.4[a] | grass (Elymus elymoides) | serpentinitic, thermic lithic argixerolls | 1.5 | 0 |
| | garden waste | 830.8[a] | grass (Elymus elymoides) | serpentinitic, thermic lithic argixerolls | 1.5 | 126 |
| **Curtis and Claassen** (2009)[dc] | yard waste | 830.8[a] | big squirreltail | sandy loam Lahar | 0.5 | -19 |
| | yard waste | 830.8[a] | needle grass | sandy loam serpentinistic | 0.5 | -21 |
| | yard waste | 830.8[a] | needle grass | sandy loam sandstone | 0.5 | 32 |
| **Weber et al.** (2007) | municipal solid waste | 120,0 | no crop | sandy | 1 | 0 |

*MSW, Municipal solid waste; na, not available.*
[a] *Only dry weight is available in the study. We considered 35% moisture content for the ww calculation.*
[b] *Pot experiment.*
[c] *Non-tillage is compared with tillage and compost option. Therefore we are not only comparing compost effects.*
[d] *Case study of land reclamation.*

**Table A11. SOIL WORKABILITY: Literature review of the effect of compost application on the soil parameters related to soil workability.**

| Study | Compost feedstock | Dosage ww (t ha$^{-1}$ yr-1) | Type of soil | Crop | Length (yr) | Change in the workability indicator (%) |
|---|---|---|---|---|---|---|
| *Soil bulk density* | | | | | | |
| **Martens et al.** (1992) | | 25 | | | 2 | -10.8 |
| **Turner et al.** (1994) | MSW | 134[a] | | | 2 | -15 |
| **Illera et al.** (1999) | MSW | 80 | | | 1 | -13.1 |
| **Stamatiadis et al.** (1999) | green waste+ cow manure | 44 | silt clay loam | broccoli | 1 | -6.1 |
| **Zebarth et al.** (1999) | | 45 | | | 4 | -15.5 |
| **Aggelides & Londra** (2000) | MSW+ sewage sludge+saw dust | 75 m$^3$ ha$^{-1}$ | humic fluvaquent, clay soil | grass fallow | 1 | -6.3 |
| | MSW+ sewage sludge+saw dust | 150 m$^3$ ha$^{-1}$ | humic fluvaquent, clay soil | grass fallow | 1 | -12.5 |
| | MSW+ sewage sludge+saw dust | 300 m$^3$ ha$^{-1}$ | humic fluvaquent, clay soil | grass fallow | 1 | -16.7 |
| | MSW+ sewage sludge+saw dust | 75 m$^3$ ha$^{-1}$ | humic fluvaquent, clay soil | grass fallow | 1 | -12 |
| | MSW+ sewage sludge+saw dust | 150 m$^3$ ha$^{-1}$ | humic fluvaquent, clay soil | grass fallow | 1 | -17 |
| | MSW+ sewage sludge+saw dust | 300 m$^3$ ha$^{-1}$ | humic fluvaquent, clay soil | grass fallow | 1 | -19.7 |
| **Tejada et al.** (2008) | green manure + beet vinase | | xelloric calciorthid | no crop | 4 | -13.5 |
| **Tejada & González** (2008) | cotton gin crushed compost | 7.12 | xerollic calciorthid | | 4 | -19.6 |
| **Courtney & Mullen** (2008) | municipal sludge | 100 | nm | barley | <1 | -8.9 |
| | spent mushroom compost | 100 | nm | barley | <1 | -8.9 |
| **Diana et al.** (2008) | wine-producing residues | 1 | alluvial soil | lettuce | <1 | -21 |
| **Mylavarapu & Zinati** (2009) | 75% MSW + 25% biosolids | 20 | loamy, siliceous, hyperthermic, grossarenic, paleudult | parsley | 1 | -2.5 |
| **Hemmat et al.** (2010) | MSW | 25 | typic haplargids/ calcaric cambisols, fine-loamy, mixed | wheat-corn | 7 | -0.7 |
| | MSW | 50 | typic haplargids/ calcaric cambisols, fine-loamy, mixed | wheat-corn | 7 | -4.5 |
| | MSW | 100 | typic haplargids/ Calcaric cambisols, fine-loamy, mixed | wheat-corn | 7 | -23.1 |

| | | | | | | |
|---|---|---|---|---|---|---|
| **Celik et al.** (2010) | green waste | 25 | typic xerofluvents, clay loam | wheat-corn | 13 | -20.6 |
| | | | *Soil porosity (%)* | | | |
| **Aggelides & Londra** (2000) | MSW+ sewage sludge+saw dust | 75 $m^3$ $ha^{-1}$ | humic fluvaquent, clay soil | grass fallow | 1 | 5.4 |
| | MSW+ sewage sludge+saw dust | 150 $m^3$ $ha^{-1}$ | humic fluvaquent, clay soil | grass fallow | 1 | 8.5 |
| | MSW+ sewage sludge+saw dust | 300 $m^3$ $ha^{-1}$ | humic fluvaquent, clay soil | grass fallow | 1 | 9.9 |
| | MSW+ sewage sludge+saw dust | 75 $m^3$ $ha^{-1}$ | humic fluvaquent, clay soil | grass fallow | 1 | 11 |
| | MSW+ sewage sludge+saw dust | 150 $m^3$ $ha^{-1}$ | humic fluvaquent, clay soil | grass fallow | 1 | 27 |
| | MSW+ sewage sludge+saw dust | 300 $m^3$ $ha^{-1}$ | humic fluvaquent, clay soil | grass fallow | 1 | 32.8 |
| **Weber et al.** (2007) | MSW (1) | 18 | dystric cambisol, sandy | triticale | <1 | 0.39 |
| | MSW (1) | 36 | dystric cambisol, sandy | | <1 | 0.24 |
| | MSW (1) | 72 | dystric cambisol, sandy | | <1 | 0.23 |
| | MSW (2) | 18 | dystric cambisol, sandy | triticale | <1 | 0.46 |
| | MSW (2) | 36 | dystric cambisol, sandy | | <1 | 0.35 |
| | MSW (2) | 72 | dystric cambisol, sandy | | <1 | 0.29 |

*MSW, Municipal solid waste; nm, not mentioned.*
[a] *Only one compost application.*

**Table A12. SOIL BIOLOGICAL PROPERTIES AND BIODIVERSITY: Literature review of the effects of compost on soil microbial community.**

| Study | Compost Feedstock | Dosage ww (t ha$^{-1}$ y$^{-1}$) | Crop | Type of soil | Length (yr) | Change in the diversity indicator (%) |
|---|---|---|---|---|---|---|
| | | | *Microbial biomass Carbon* | | | |
| **Albiach et al.** (2000) | MSW | 24 | no crop | xerorthent sandy silty loam | 5 | 187.5 |
| **García-Gil et al.** (2000) | MSW | 20 | barley | typic haploxeralf | 9 | 10.2 |
| | MSW | 80 | barley | typic haploxeralf | 9 | 46.8 |
| **Ros et al.** (2003) | MSW | 300 | no crop | xeric torriorthents | 2 | 225 |
| **Ros et al.** (2006a; 2006b) | MSW | 22.5 | corn-wheat-barley | loamy silt | 12 | 8.2 |
| | green waste | 16.9 | corn-wheat-barley | loamy silt | 12 | 10 |
| | cattle manure | 20.8 | corn-wheat-barley | loamy silt | 12 | 3.2 |
| | sewage sludge | 12.2 | corn-wheat-barley | loamy silt | 12 | 4.7 |
| **Tejada & González** (2006a) | crushed cotton gin | 10 | rice | aquic xerofluvent | 1 | 21.7 |
| | crushed cotton gin | 15 | rice | aquic xerofluvent | 1 | 60.9 |
| | crushed cotton gin | 20 | rice | aquic xerofluvent | 1 | 116.3 |
| **Fließbach et al.** (2007) | cattle manure | 11.2 | | typic hapludalf | 21 | 38.3 |
| **Bastida et al.** (2008) | sewage sludge | 120 | no crop | haplic calcisol | 1 | 106.4 |
| **Tejada et al.** (2009a) | beet vinasse+ greenwaste vermicompost | 7.91 | no crop | xerollic calciorthid | 3 | 183.3 |
| | beet vinasse+ greenwaste vermicompost | 15.82 | no crop | xerollic calciorthid | 3 | 241.7 |
| | | | *Microbial Diversity H′ (PCR-DGGE)* | | | |
| **Ros et al.** (2006a; 2006b) | MSW | 22.5 | corn-wheat-barley | loamy silt | 12 | -2.24 |
| | green waste | 16.9 | corn-wheat-barley | loamy silt | 12 | 3.73 |
| | cattle manure | 20.8 | corn-wheat-barley | loamy silt | 12 | 2.24 |
| | sewage sludge | 12.2 | corn-wheat-barley | loamy silt | 12 | 2.24 |

| | | | | | | |
|---|---|---|---|---|---|---|
| **Ge et al.** (2010) | MSW | 25 | | hapludalf silty clay loam | nm | 1.8 |
| | MSW | 75 | | hapludalf silty clay loam | nm | 0 |
| | | ***Bacterial Functional Diversity H′ (CLPP)*** | | | | |
| **Gómez et al.** (2006) | household solid waste | 20.74[a] | no crop | vertic argiudoll | 0.5 | 5.21 |
| | household solid waste | 41.49[a] | no crop | vertic argiudoll | 0.5 | 9.38 |
| **Ros et al.** (2006a; 2006b) | MSW | 22.5 | corn-wheat-barley | loamy silt | 12 | 7.02 |
| | green waste | 16.9 | corn-wheat-barley | loamy silt | 12 | 5.26 |
| | cattle manure | 20.8 | corn-wheat-barley | loamy silt | 12 | 4.09 |
| | sewage sludge | 12.2 | corn-wheat-barley | loamy silt | 12 | 6.43 |
| | | ***Microbial Activity (basal respiration)*** | | | | |
| **Ros et al.** (2003) | MSW | 300 | no crop | xeric torriorthents | 2 | 263.6 |
| **Pérez-Piqueres et al.** (2006)[b] | green waste compost | 20% (vol) | | clayey soil | 10 days | 0 |
| | spent mushroom (UK) | 20% (vol) | | clayey soil | 10 days | 344.4 |
| | spent mushroom (Fr) | 20% (vol) | | clayey soil | 10 days | 555.6 |
| | green waste compost | 20% (vol) | | sandy silty clay soil | 10 days | 57.1 |
| | spent mushroom (UK) | 20% (vol) | | sandy silty clay soil | 10 days | 123.8 |
| | spent mushroom (Fr) | 20% (vol) | | sandy silty clay soil | 10 days | 120.6 |
| **Ros et al.** (2006a; 2006b) | MSW | 22.5 | corn-wheat-barley | loamy silt | 12 | 1 |
| | green waste | 16.9 | corn-wheat-barley | loamy silt | 12 | 7 |
| | cattle manure | 20.8 | corn-wheat-barley | loamy silt | 12 | 0 |
| | sewage sludge | 12.2 | corn-wheat-barley | loamy silt | 12 | 43 |
| **Fließbach et al.** (2007) | cattle manure | 11.2 | | typic hapludalf | 21 | 23.8 |
| **Bastida et al.** (2008) | sewage sludge | 120 | no crop | haplic calcisol | 1 | 121.7 |

*MSW, Municipal solid waste; vol, volume; nm, not mentioned.*

[a] *Only dry weight is available in the study. We considered 35% moisture content for the ww calculation.*

[b] *Laboratory incubation.*

**Table A13. CROP NUTRITIONAL QUALITY: Literature review of crop nutritional quality after compost application in soils.**

| Study | Compost feedstock | Dosage ww | Type of soil | Crop | Length (yr) | Phytochemical | Increase in the nutritional content from options without compost (%) |
|---|---|---|---|---|---|---|---|
| **Wang & Lin** (2003)[a] | dairy and poultry manure + livestock bedding+ landscape trimmings + greenhouse discards + wood chips + leaves | 50% soil + 50% compost | - | allstar and honeoye strawberry | 1 | ascorbic acid | 8.9-15.03 |
| | | | | | | dehydroascorbic acid | ns |
| | | | | | | reduced glutathione | 4.7-12.7 |
| | | | | | | oxidized glutathione | ns |
| | | | | | | ellagic acid | 16.0-140.0 |
| | | | | | | p-coumaroylglucose | 23.9-65.8 |
| | | | | | | dihydroflavonol | 44.6-116.0 |
| | | | | | | kaempferol 3-glucoside | 12.5-32.6 |
| | | | | | | kaempferol 3-glucuronide | 22.1-51.0 |
| | | | | | | cyanidin 3-glucoside | 39.5-76.6 |
| | | | | | | pelargonidin 3-glucoside | 17.2-25.9 |
| | | | | | | cyanidin 3-glucoside-succinate | 31.0-71.8 |
| | | | | | | pelargonidin 3-glucoside-succinate | 14.3-28.5 |
| **Coria-Cayupán et al.** (2009) | bio-waste + MSW | 10 t $ha^{-1}$ | - | lettuce | 0.15 | Chlorophylla | 15.3-21.7 |
| | | | | | | Chlorophyllb | 15.6-16.4 |
| | | | | | | β-carotene | 6.7-20.0 |
| | | | | | | lactucaxanthin | ns |
| | | | | | | lutein | 7.7-23.1 |
| | | | | | | violaxanthin | 11.5-26.9 |
| | | | | | | neoxanthin | 40.0-170.0 |
| | | | | | | vitA | 7.3-23.6 |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| | | | | | | caffeic acid | ns |
| | | | | | | coumaric acid | ns |
| **Martínez-Blanco et al.** (2011) | bio-waste | 200-100 t ha$^{-1}$ [b] | typic Xerothent | cauliflower | 1[c] | sinapic acid | ns / ns |
| | | | | | | 1,2-disinapoyl-diglucoside | 74.1 / ns |
| | | | | | | 1-sinapoyl-2-feruloyldiglucoside | ns / ns |
| | | | | | | 1,2,2′-trisinapoyldiglucoside | 88.8 / ns |
| | | | | | | 1,2′-disinapoyl-2-feruloyldiglucoside | 76.5 / ns |
| | | | | | | total phenols | 24.1 / ns |
| | | | | | | glucoiberin | ns / ns |
| | | | | | | sinigrin | ns / ns |
| | | | | | | glucoraphanin | ns / ns |
| | | | | | | progoitrin | ns / ns |
| | | | | | | glucoalyssin | -45.5 / -45.5 |
| | | | | | | glucoiberverin | ns / ns |
| | | | | | | glucoerucin | 150.0 / ns |
| | | | | | | 4-OH-Glucobrassicin | ns / ns |
| | | | | | | glucobrassicin | ns / ns |
| | | | | | | metoxiglucobrassicin | ns / ns |
| | | | | | | neoglucobrassicin | ns / ns |
| | | | | | | kaempferol-3-diglucoside-7-diglucoside | 143.3 / ns |
| | | | | | | quercetin-3-diglucoside-7-glucoside | ns / ns |
| | | | | | | kaempferol-3-diglucoside-7-glucoside | ns / ns |

*ns, no significant differences.*
[a] *Pot experiment.*
[b] *Two options were considered. The results in the last column are separately presented for the two dosages.*
[c] *The crop lasted 112 days. The crop is the fourth crop in a cultivation rotation. The crop was cultivated 313 days after compost application.*

# Annex III. Supplementary information for Chapter 9

The **Annex III** corresponds to the Supplementary information of the **Chapter 9** that is based on the following paper:

Martínez-Blanco, J., Lehmann, A., Muñoz, P., Antón, A., Traverso, M., Rieradevall, R., & Finkbeiner, M. Life Cycle Sustainability Assessment of compost and mineral fertilizers production for agriculture. Focus on application challenges for social LCA. Submitted on February 2012 to the Special Issue on Life Cycle Sustainability Assessment in the International Journal of Life Cycle Assessment.

**Annex III** is divided in the following sections:

- The whole inventory data for mainstream sector
- Example of aggregated social risks for natural gas importations of Spain
- Calculation of the social indicators used in the Life Cycle Sustainability Dashboard
- Data Table for Life Cycle Sustainability Dashboard

References are at the end of Annexes part.

# Annex III.I. The whole inventory data for mainstream sector

**Table A14. Mainstream sector level - Comparison of social performance of three fertilizing alternatives involved in the production chain of fertilizers. Source data for the Table 9.3 of the manuscript.**

| STAKEHOLDER > Sub-category (shaded) | Fertilizing alternative | | Compost | | $HNO_3$ | $KNO_3$ |
|---|---|---|---|---|---|---|
| | **Mainstream process** | | **OFMSW collection** | **Compost production** | **$HNO_3$ production** | **$KNO_3$ production** |
| | **Working time (s per kg N available)** | **Data qual.** | 1113.0 | 5084.6 | 80.3 | nd |
| | **Social indicator** | | Spain | Spain | Spain | Israel |
| **WORKER** | | | | | | |
| **Freedom of Association and Collective Bargaining** | Potential of sector not passing labour laws[a] | T | L | L | L | H |
| | Potential of sector not adopting labour conventions [a] | S | L | L | L | L |
| | Others: Risk of not having the right to strike; Risk of not having collective bargaining rights; Risk of not having freedom of association rights | | | | | |
| **Child labour** | Risk of child labour in the sector [a] | L | L | L | L | L |
| **Working Hours** | Average working hours per week | T | 39.1[b] | 39.1[b] | 40.6[b] | 43.6[c] |
| | Others: Work-life balance situation | | | | | |
| **Forced labour** | Risk of forced labour [a] | L | H | H | H | H |
| **Equal opportunities/Discrimination** | Overall fragility of gender equity (% women total workers$^{-1}$) [a] | T | H (12.4) [b] | H (12.4) [b] | M (27.1) [b] | L (38.0)[d] |
| | Others: Ratio of basic salary of men to women by employee category; Ratio of immigrant employees (%); Ratio of basic salary of immigrants to the rest by employee category | | | | | |
| **Health and Safety** | Gaseous emissions exposure effects | T/L | [e] | [f] | [g] | [g] |
| | Biological agents exposure effects | L | [h] | [i] | ne | ne |
| | Occurrence of lethal accidents per year (per 100,000 people) | T | 12.3 [b] | 12.3 [b] | 7.1 [b] | nd |
| | Occurrence of non-lethal accidents per year (per 100,000 people) | T | 10876.8 [b] | 10876.8 [b] | 2952.3 [b] | nd |
| | Others: Biological agents protection and prevention measures; Workers comfort level; Level of noise; Presence of a formal policy concerning health and safety in the sector | | | | | |
| **LOCAL COMMUNITY** | | | | | | |
| **Safe & healthy living conditions** | Odour and gaseous emissions effects | T/L | nd | [j] | ne[g] | ne[g] |
| | Biological agents exposure effects | T/L | nd | [k] | ne | ne |
| | Other hazards and nuisances | L | Noise and traffic | | [m] | [m] |
| | Others: Biological agents protection and prevention measures; Level of noise; Emissions and noise records are recommended or mandatory for the company. | | | | | |

| | | | | | | |
|---|---|---|---|---|---|---|
| **Local employment** | Promotion of local employment in the consumption area | L | yes | yes | yes | no |
| | Others: Training courses for the employees; % Employees with Higher education; % Employees with Basic education; % spending on locally-based suppliers | | | | | |
| **Neighbours acceptance** | Others: rate of willingness to have the sector close to home; participation of neighbours in decisions and incomes. | | | | | |
| **CONSUMER (FARMER)** | | | | | | |
| **Supplier relationships** | Fertilizer production scale with regard of consumer | S | Local | | Regional | Internation. |
| **Health and Safety** | Product application dangers | L | [n] | | [o] | |
| | Others: Existence of health and safety measure labels for application in the product | | | | | |
| **Requirements for fertilizers application** | Extra working time for consumer to apply the product | S | Necessary | | Not necessary | |
| | Others: Level of complexity for dosages calculation. | | | | | |
| **Consumer acceptance** | Average prices in Catalonia (€ per kg N available)[p] | T | 1-3.5 | | 5 | 6 |
| | Others: Main consumer concerns about the product | | | | | |
| **CITIZENS COLLECTING OFMSW** | | | | | | |
| **Education and responsibility** | Others: Existence of obligation of waste collection for citizens; Existence of educational campaigns for citizens engagement. | | | | | |
| **Comfort and collecting effort for the citizens** | Others: Frequency of organic bin emptying; % public space used; % private space used. | | | | | |
| **Acceptance and willingness of citizens to collect organic waste** | Amount of organic waste collected (%)[q] | T | Cat. 22.5% | | na | na |
| | % of improper materials in the organic waste[q] | T | Cat. 7% | | na | na |

*Type of data: T, Quantitative. S, Semi-quantitative. L, Qualitative*
*nd no sector data; ne no evidence; na not applicable*
[a] *Sector level risk according to SHDB. L Low, M Medium, H High, VH Very high*
[b] *MTI (2011);* [c] *ILO (2011);* [d] *CBE (2011);* [e] *It is possible that waste collectors may be exposed to doses of VOC exceeding the occupational emissions limits for very short periods (Poulsen et al. 1995). Workers exposure to VOC during waste collection was only modest for Kiviranta et al. (1999), and the pattern of VOC suggested that most of the VOC originated from the exhaust fumes of the vehicle rather than from the wastes.*
[f] *VOC concentrations, even 'worst case', were well below Directive 2000/39/CE (EC 2000) permissible workplace levels for toxicity effects but they are detectable and can be annoying. Ammonia concentrations in working environments may be close or above regulation limits but health effects are unlikely to be severe. The other gaseous species and dosages are not dangerous for health (Eitzer 1995; Lavoie and Alie 1997; Smet et al. 1999; EC 2000; Staley et al. 2006; Tsai 2008; Persoons et al. 2010; Colón et al. 2012; Health Protection Agency 2012).*
[g] *Few data was found about gaseous emission exposure for workers. Cholinesterase activity is a measure of an occupational exposure of neighbors and workers exposed directly or indirectly to the different gaseous emissions and waste products of fertilizer plants. Significant differences were found in workers of a fertilizer plant but not for neighbors (Shad et al. 2009).*
*h Gastrointestinal and respiratory symptoms were founded to be higher for waste collection workers than for the control group in some studies but they were not in others (Swan et al. 2003; Porta et al. 2009).*
[i] *Higher incidence of eyes, airways and skin symptoms were found for workers exposed to organic dust than for the control groups (Swan et al. 2003; Domingo and Nadal 2009; Porta et al. 2009; Persoons et al. 2010).*

*[j] Higher odor and VOC levels up to an area between 500-1200 m from the composting plant are demonstrated (lower areas for enclosed composting plants than for open plants), although health effects were not related (Fischer et al. 2008; Domingo and Nadal 2009).*
*[k] In most cases, until distances above 600–800 m from the plant the concentration of microorganisms did not reach the reference levels (Fischer et al. 2008). However, airway irritation problems were detected within the community residents up to 150-200 m from the bio-aerosol source (Porta et al. 2009).*
*[m] The toxic hazard of a potential large release of liquid ammonia (i.e. from a storage tank), although with a low incidence, may be the most serious one for the local population (EFMA 2000).*
*[n] Only if during composting areas of anaerobic decomposition appear and, thus, lower temperatures are achieved, sanitation problems are likely to appear (Solans et al. 2008). Relevant ammonia emissions during applications but health effects are unlikely to be severe (Cocco et al. 1996; Health Protection Agency 2012).*
*[o] Ammonia health effects are unlikely to be severe (Cocco et al. 1996; Health Protection Agency 2012).*
*[p] Average prices for Catalonia (Spain) are used. Several sources.*
*[q] ARC (2012).*

# Annex III.II. Example of aggregated social risks for natural gas importations of Spain

**Table A15. Social risks for countries involved in natural gas importations to Spain according to data from SHDB (2011). The contribution to the Spanish total natural gas consumption is specified.**

| Country exporting to Spain | Norway | Algeria | Qatar | Egypt | Nigeria | Trinidad-Tobago |
|---|---|---|---|---|---|---|
| **Contribution total natural gas consumption (%)[1]** | 7.7 | 34.4 | 12.6 | 11.4 | 18.8 | 10.3 |
| **STAKEHOLDER > Subcategory (shaded) > Social indicator (white)** | | | | | | |
| **WORKER** | | | | | | |
| *Freedom of Association & Collective Bargaining* | | | | | | |
| **Potential of country not passing labour laws** | low | low | high | low | low | nd |
| **Potential of country not adopting labour conventions** | low | medium | medium | medium | medium | nd |
| **Risk of not having freedom of association rights** | low | high | very high | high | high | nd |
| **Risk of not having collective bargaining rights** | medium | medium | very high | high | high | nd |
| **Risk of not having the right to strike** | medium | high | very high | very high | very high | nd |
| *Child Labour* | | | | | | |
| **Risk of child labour** | ne | medium | medium | medium | very high | nd |
| **Number of children out of school** | medium | medium | high | medium | very high | nd |
| *Fair Salary* | | | | | | |
| **Potential of minimum wages not being updated** | na | low | na | na | medium | nd |
| *Working Hours* | | | | | | |
| **Risk of population working > 48 hours week$^{-1}$** | low | low | medium | low | medium | nd |
| *Forced Labour* | | | | | | |
| **Risk of forced labour** | high | high | high | high | high | nd |
| *Equal opportunities/Discrimination* | | | | | | |
| **Overall fragility of gender equity** | medium | very high | very high | very high | very high | nd |

*Sector data; country data; na not applicable; nd no data; ne no evidence*

*[1] Average importation for the period 2007-2009. Data from INE (2012).*

*[2] Trinidad-Tobago is not included in the SHDB (2011). It is not taken into account in the aggregation.*

**Table A16. Social risks for countries involved in natural gas importations to Spain according to data from SHDB** (SHDB 2011)**. The contribution to the Spanish total natural gas consumption is specified.**

| | Ranges of risk translated to scores[1] | | | | | | Aggregated social risks | |
|---|---|---|---|---|---|---|---|---|
| **Country exporting to Spain** | **Norway** | **Algeria** | **Qatar** | **Egypt** | **Nigeria** | **T-T** | **Score[2]** | **Risk level[3]** |
| **Contribution total natural gas consumption (%)[1]** | 7.7 | 34.4 | 12.6 | 11.4 | 18.8 | 10.3 | | |
| **WORKER** | | | | | | | | |
| *Freedom of Association & Collective Bargaining* | | | | | | | | |
| **Potential of country not passing labour laws** | 1 | 1 | 3 | 1 | 1 | nd | 1.30 | low |
| **Potential of country not adopting labour conventions** | 1 | 2 | 2 | 2 | 2 | nd | 1.91 | medium |
| **Risk of not having freedom of association rights** | 1 | 3 | 4 | 3 | 3 | nd | 2.97 | high |
| **Risk of not having collective bargaining rights** | 2 | 2 | 4 | 3 | 3 | nd | 2.65 | high |
| **Risk of not having the right to strike** | 2 | 3 | 4 | 4 | 4 | nd | 3.41 | very high |
| *Child Labour* | | | | | | | | |
| **Risk of child labour** | 0 | 2 | 2 | 2 | 4 | nd | 2.26 | medium |
| **Number of children out of school** | 2 | 2 | 3 | 2 | 4 | nd | 2.59 | high |
| *Fair Salary* | | | | | | | | |
| **Potential of minimum wages not being updated** | na | 1 | na | na | 2 | nd | na | na |
| *Working Hours* | | | | | | | | |
| **Risk of population working > 48 hours week$^{-1}$** | 1 | 1 | 2 | 1 | 2 | nd | 1.37 | low |
| *Forced Labour* | | | | | | | | |
| **Risk of forced labour** | 3 | 3 | 3 | 3 | 3 | nd | 3.00 | high |
| *Equal opportunities/Discrimination* | | | | | | | | |
| **Overall fragility of gender equity** | 2 | 4 | 4 | 4 | 4 | nd | 3.82 | very high |

*na not applicable; nd no data*

*[1] The four ranges of risk or opportunity from SHDB are translated to numbers: low (1), medium (2), high (3) and very high (4), additionally no evidence (0)*

*[2] The weighted average is calculated using the country mixes. The ranges of risk of each country are multiplied by the share of each country to the Spanish national mix and summed. A score between 1 and 4 is obtained.*

*[3] Scores below 1.5 are labeled “low”, scores between 1.5-2.5 are “medium”, between 2.5-3.5 are “high” and higher than 3.5 are “very high”. Moreover, when half or more of the countries have range 3, we rise one level the score of the average if it is lower than high; while when one third or more of the countries have range 4, we rise one level the averaged score.*

# Annex III.III. Calculation of the social indicators used in the Life Cycle Sustainability Dashboard

The seven social indicators correspond to Worker stakeholder. They are the aggregation of the results for mainstream (Table 9.3 from Chapter 9) and upstream (Table 9.5 from Chapter 9) sectors involved in fertilizers production according to the shares of working time (Table 9.1 from Chapter 9).

**Table A17. Social risk level and translated scores for upstream and mainstream processes of the three fertilizing alternatives.**

| | | Upstream processes (Table 9.5 from Chapter 9) | | | | | | Mainstream processes (Table 9.3 of from Chapter 9)) | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | Risk level | | | Scores level[1] | | | Risk level | | | Scores level[1] | | |
| | Fertilizing alternative | Compost | $HNO_3$ | $KNO_3$ | Compost | $HNO_3$ | $KNO_3$ | Compost | $HNO_3$ | $KNO_3$ | Compost | $HNO_3$ | $KNO_3$ |
| | Compost (working hours per kg N) | 2585.1 | 0.0 | 0.0 | 2585.1 | 0.0 | 0.0 | 6197.6 | | | 6197.6 | | |
| | $HNO_3$ (working hours per kg N) | 0.0 | 287.0 | 0.0 | 0.0 | 287.0 | 0.0 | | 80.3 | | | 80.3 | |
| | $KNO_3$ (working hours per kg N) | 0.0 | 0.0 | 323.1 | 0.0 | 0.0 | 323.1 | | | 80.3[2] | | | 80.3[2] |
| | **SOCIAL INDICATORS** | | | | | | | | | | | | |
| S01 | Potential of country not passing labor laws | medium | medium | medium | 2.4 | 1.4 | 1.9 | low | low | high | 1.0 | 1.0 | 3.0 |
| S02 | Potential of country not adopting labor conventions | medium | medium | medium | 1.6 | 2.0 | 1.8 | low | low | low | 1.0 | 1.0 | 1.0 |
| S03 | Risk of child labor | medium | medium | low | 1.6 | 2.0 | 1.2 | low | low | low | 1.0 | 1.0 | 1.0 |
| S04 | Risk of forced labor | high | high | high | 3.0 | 3.0 | 3.0 | high | high | high | 3.0 | 3.0 | 3.0 |
| S05 | Overall fragility of gender equity | very high | very high | very high | 3.6 | 4.0 | 3.6 | high | medium | low | 3.0 | 2.0 | 1.0 |
| S06 | Occurrence of occupational lethal accidents (cases per h of work) | 1.62E-08 | 2.37E-08 | 2.37E-08 | 1.62E-08 | 2.37E-08 | 2.37E-08 | 6.55E-08 | 3.64E-08 | 3.64E-08 | 6.55E-08 | 3.64E-08 | 3.64E-08 |
| S07 | Occurrence of occupational non-lethal accidents (cases per h of work) | 2.66E-05 | 4.04E-05 | 4.06E-05 | 2.66E-05 | 4.04E-05 | 4.06E-05 | 5.80E-05 | 1.51E-05 | 1.51E-05 | 5.80E-05 | 1.51E-05 | 1.51E-05 |

*[1] The four ranges of risk or opportunity from SHDB are translated to numbers: low (1), medium (2), high (3) and very high (4)*

*[2] The mainstream working time stated for nitric acid –i.e. 80.3 s per kg N available– is used also for potassium nitrate aggregation of its own social results (see section 9.3.2. of Chapter 9). Occurrence of occupational lethal and non-lethal accidents for nitric acid is also used for potassium nitrate.*

**Table A18. Aggregated social risk level and translated scores the three fertilizing alternatives.**

| | | Aggregated social risks for the whole life cycle | | | | | |
|---|---|---|---|---|---|---|---|
| | | Scores level[1] | | | Risk level[2] | | |
| | Fertilizing alternative | Compost | $HNO_3$ | $KNO_3$ | Compost | $HNO_3$ | $KNO_3$ |
| | Compost (working hours per kg N) | 8782.8 | | | 8782.8 | | |
| | HNO3 (working hours per kg N) | | 367.4 | | | 367.4 | |
| | KNO3 (working hours per kg N) | | | 323.1 | | | 403.4 |
| | SOCIAL INDICATORS | | | | | | |
| S01 | Potential of country not passing labor laws | 1.4 | 1.3 | 2.6 | low | medium | medium |
| S02 | Potential of country not adopting labor conventions | 1.2 | 1.8 | 2.1 | low | medium | medium |
| S03 | Risk of child labor | 1.2 | 1.8 | 1.4 | low | medium | low |
| S04 | Risk of forced labor | 3.0 | 3.0 | 3.7 | high | high | high |
| S05 | Overall fragility of gender equity | 3.2 | 3.5 | 3.9 | high | very high | very high |
| S06 | Occurrence of occupational lethal accidents (cases per h of work) | 5.10E-08 | 2.65E-08 | 3.28E-08 | 5.10E-08 | 2.65E-08 | 3.28E-08 |
| S07 | Occurrence of occupational non-lethal accidents (cases per h of work) | 4.87E-05 | 3.48E-05 | 4.44E-05 | 4.87E-05 | 3.48E-05 | 4.44E-05 |

[1] *The weighted average is calculated using the working time. The scores of upstream and mainstream columns are multiplied by their contribution to total working time and summed for each fertilizing alternative. A score between 1 and 4 is obtained.*

[2] *Scores below 1.5 are labeled "low", scores between 1.5-2.5 are "medium", between 2.5-3.5 are "high" and higher than 3.5 are "very high".*

# Annex III.IV. Data for Life Cycle Sustainability Dashboard

**Table A19. Life Cycle Sustainability assessment results for the three fertilizing alternatives (per kg N available).**

| | | Indicator | Unit | Compost | $HNO_3$ | $KNO_3$ |
|---|---|---|---|---|---|---|
| **ELCA** | **ADP** | **Abiotic depletion** | kg Sb eq. | 6.46E-01 | 5.38E-02 | 8.41E-02 |
| | **AP** | **Acidification** | kg $SO_2$ eq. | 8.75E-01 | 1.37E-01 | 1.24E-01 |
| | **EP** | **Eutrophication** | kg $PO_4^{3-}$ eq. | 6.61E-01 | 3.19E-02 | 4.01E-02 |
| | **GWP** | **Global warming (GWP100)** | kg $CO_2$ eq. | 1.39E+02 | 2.86E+01 | 2.17E+01 |
| | **OLDP** | **Ozone layer depletion (ODP)** | kg CFC-11 eq. | 1.45E-05 | 1.00E-06 | 1.58E-06 |
| | **HT** | **Human toxicity** | kg 1,4-DB eq. | 7.08E+01 | 5.74E+00 | 7.72E+00 |
| | **FWAE** | **Fresh water aquatic ecotox.** | kg 1,4-DB eq. | 8.37E+02 | 1.30E+00 | 2.08E+00 |
| | **MAE** | **Marine aquatic ecotoxicity** | kg 1,4-DB eq. | 3.02E+05 | 3.63E+03 | 4.86E+03 |
| | **TE** | **Terrestrial ecotoxicity** | kg 1,4-DB eq. | 1.35E+00 | 7.17E-02 | 5.42E-02 |
| | **POP** | **Photochemical oxidation** | kg $C_2H_4$ eq. | 2.52E-01 | 1.08E-03 | 1.62E-03 |
| | **CED** | **Cumulative Energy Demand** | MJ eq. | 1.58E+03 | 1.18E+02 | 1.87E+02 |
| **LCC** | **Fertilizer** | **Price of fertilizer** | € | 2.95 | 4.95 | 5.76 |
| | **Transport** | **Price of transport** | € | 0.40 | 0.0291 | 0.0279 |
| | **Application** | **Extra application costs** | € | 4.40 | 0.00 | 0.00 |
| **SLCA** | **S01** | **Potential of country not passing labour laws** | Score | 1.40 | 1.28 | 2.62 |
| | **S02** | **Potential of country not adopting labour conventions** | Score | 1.18 | 1.76 | 2.07 |
| | **S03** | **Risk of child labour** | Score | 1.18 | 1.76 | 1.40 |
| | **S04** | **Risk of forced labour** | Score | 3.00 | 3.00 | 3.75 |
| | **S05** | **Overall fragility of gender equity** | Score | 3.18 | 3.54 | 3.89 |
| | **S06** | **Occurrence of occupational lethal accidents** | cases per h of work | 5.10E-08 | 2.65E-08 | 3.28E |
| | **S07** | **Occurrence of occupational non-lethal accidents** | cases per h of work | 4.87E-05 | 3.48E-05 | 4.44E |

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