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Environmental Issues in Logistics and Manufacturing

Subramanian Senthilkannan Muthu
Editor

Assessment of Carbon Footprint in Different Industrial Sectors, Volume 1

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Preface

The carbon footprint is an important environmental impact and a frequently heard term these days. This terminology has received great attention from the public, government, and media. The dreadful consequences of global warming (and the importance of addressing them) have been widely discussed by newspapers, media, government, and various nongovernmental organizations. Development of green products (such as low-carbon products) and the demand for such products are increasing each day. The assessment of carbon footprint and reduction of greenhouse gas emissions are measures that should be followed by any manufacturer or producer. Although the assessment and declaration of the carbon footprint of products are currently voluntary, soon they will become mandatory. Today's market is gradually receiving products that display their carbon emissions; likely, this will eventually become mandatory for every product produced on earth.

The mitigation of carbon emissions is an important topic for any government's agenda, and nations are trying their best to reduce its carbon footprint to the maximum possible extent. Many companies would like to reduce the carbon footprint of their products, and consumers are looking for products that emit lower carbon emissions in their entire life cycle. Assessment of the carbon footprint for different products, processes, and services, as well as the carbon labelling of products, have become familiar topics recently in various industrial sectors. Every industry has unique assessment and modelling techniques, allocation procedures, mitigation methods, and labelling strategies for its carbon emissions. Therefore, this book has been framed with dedicated chapters on carbon footprint assessment in various industrial sectors.

Each chapter provides details pertaining to the assessment methodologies of carbon footprint for a particular industry, challenges in calculating the carbon footprint, case studies of various products in that particular industry, mitigation measures to reduce the carbon footprint, and recommendations for further research. This first volume includes the carbon footprint assessment methodology for the agricultural, telecommunication, food, ceramic, packaging, building and construction, and solid waste sectors.

The concepts of eco-design and lifecycle assessment are the crux of carbon footprint assessment. Hence, the first chapter introduces eco-design methodology and the basic concept of a product's carbon footprint. For the benefit of the readers, every chapter in the book briefly touches upon the concept of carbon footprint, assessment methods, and standards.

Chapter 2 provides detailed discussions pertaining to carbon footprint estimation and mitigation of greenhouse gas emissions in the agricultural sector. Chapter 3 focuses on the carbon footprint estimation of building and construction products with the aid of a case study of a residential building. Chapter 4 discusses the details pertaining to the carbon footprint estimation of food products using various case studies, challenges in calculating the carbon footprint of food products, methodological limitations, uncertainties, recommendations for further research, and mitigation measures to be followed in the food sector. Chapter 5 deals with the carbon footprint estimation of ceramic products.

Introducing the process flow followed in the ceramic industry with the implications for carbon footprint calculation, this chapter examines case studies on various products of the ceramic industry in terms of their carbon footprint assessment. Assessment of carbon footprint in the telecommunication sector is discussed with a case study of mobile devices in Chapter 6. This chapter addresses the open questions for carbon footprint assessment of emerging mobile ICT technologies, such as how to obtain the reliable inventories for various components and subassemblies, as well as the ultimate effect of consumer behavior on recycling. Chapter 7 focuses on the carbon footprint estimation of pigment in Flanders. Chapter 8 is dedicated to discuss the carbon footprint estimation of different industrial spaces in mainland China, along with discussions pertaining to policy recommendations to achieve a low-carbon society. Packaging is an indispensable part of any industry today, so the carbon footprint of packaging products deserves considerable attention. Chapter 9 is a dedicated chapter dealing with the carbon footprint assessment of packaging used in different sectors. Chapter 10 focuses mainly on the agricultural sector in China. Finally, Chap. 11 discusses the implications of carbon footprint estimation in the solid waste sector. For this, one of the important cities in south India, Bangalore, has been chosen as the case study. This chapter deals with aspects such as the management of solid waste, methods of estimating the carbon footprint of solid waste, and implications on carbon footprint due to the mismanagement of waste.

I take this opportunity to thank all the contributors to this book. I am sure that the readers will certainly benefit from this book, which brings the minute details of carbon footprint assessment for various industrial sectors together in one resource. This first volume about product carbon footprint will certainly become an important reference for researchers and students.

Contents

Introduction to the Eco-Design Methodology and the Role of Product Carbon Footprint

Esther Sanyé-Mengual, Raul García Lozano, Ramon Farreny, Jordi Oliver-Solà, Carles M. Gasol and Joan Rieradevall

Abstract Eco-design is used as a tool in the manufacturing and services sectors for improving the sustainability of products by integrating environmental aspects into the design stage, where most of the product impacts are determined. Laws (e.g., EU eco-design directive) and international schemes (e.g., ISO 14006) have encouraged the use of eco-design by companies; in addition, the literature has reported advances in methodology and widespread case studies in different economic sectors. This chapter aims to show a combined design for environment (DfE) and life cycle assessment (LCA) methodology for the implementation of eco-design by companies. The steps and tools of the methodology, as well as the most common strategies, are described. Product carbon footprint (PCF) plays an important role in the methodology in two main ways. First, PCF is one of the indicators that can be calculated with LCA, which has become a common environmental indicator used by companies, not only as quantitative data of the current environmental performance but also as a benchmark for further improvements. Second, PCF is used as a strategy for environmental communication to consumers through eco-labeling. The main strength of the carbon footprint is that stakeholders (business and consumers) are aware of and understand its meaning due to the presence of carbon emissions and global warming in mass media and public science studies.

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1 Introduction

Design for environment (DfE) or eco-design has been increasingly used in sustainable manufacturing during recent decades. The design step of a product has been identified as an important life cycle stage in which 80 % of the environmental burdens are determined (Tischner et al. 2000).

The first EU Directive on eco-design was directive 2005/32/EC on Eco-design of energy-used products (European Council 2005a). This directive replaced former energy efficiency directives for hot-water boilers (Directive 92/42/EEC) (European Council 1992), household appliances (Directive 96/57/EC) (European Council 1996), and fluorescent lighting (Directive 2000/55/EC) (European Council 2000a). In this context, the focus of the directive was on reducing energy consumption and enhancing product efficiency. Moreover, at the same time the European Climate Change Program encouraged energy saving and energy efficiency as key points to achieve the objectives of greenhouse gas (GHG) emissions. These environmental aspects became the main goals of the 2005 eco-design directive (Fig. 1).

The current Directive 2009/125/EC on eco-design requirements for energy-related products (European Council 2009a) emphasizes the assessment of the entire life cycle of a product. This is mainly based on a growing policy-making concern about the environmental impact of products and services and the development of life cycle thinking, particularly since the Integrated Product Policy was implemented (European Commission 2003). Furthermore, other environmental policies also positively influenced the development of eco-design, such as Directive 94/32CE on packaging and packaging waste and the later amending documents (European Council 1994, 2004, 2005b, 2009b) and Directive 2000/53/CE on end-of-life vehicles (European Council 2000b) (Fig. 1).

Finally, international schemes were designed in order to address environmental management and eco-design in companies. The ISO14006 (2011) standards (Environmental management systems—guidelines for incorporating eco-design) provide guidance for working on eco-design as part of an environmental management system. Finally, ISO/TR 14062 (2002) (Environmental management—integrating environmental aspects into product design and development) describes eco-design concepts and practices related to the integration of environmental aspects into product design and development (European Council 2009c, 2010) (Fig. 1).

This chapter introduces the eco-design methodology that is used to integrate the environment into the design stage in order to improve the environmental performance of a product. First, the benefits and opportunities of implementing eco-design in companies are reviewed (Sect. 2), as well as the scope and the implementation of

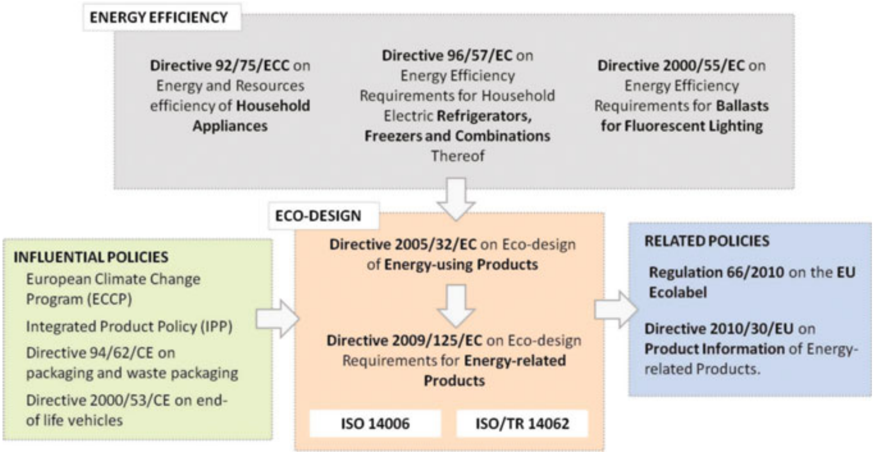


Fig. 1 Legal framework of eco-design in EU countries

eco-design in different sectors (Sects. 2.1 and 2.2). Second, an overview of the methodology is presented (Sect. 3) while showing the different steps and tools employed. In the following sections, attention is paid to qualitative tools (Sect. 4), quantitative tools (Sect. 5), the determination of the design requirements through the eco-briefing (Sect. 6), the definition of eco-design strategies (Sect. 7), and the final prototype design (Sect. 8). The chapter also focuses on the product carbon footprint (PCF) as a quantitative tool (Sect. 5.2) and as a communication-to-user strategy (Sect. 7.3) within the eco-design methodology.

2 Benefits and Opportunities of Eco-Design Implementation

Eco-design offers different benefits and opportunities to companies, not only environmental but also economic and social (Boks 2006; Borchardt et al. 2011; Brezet and van Hemel 1997; Clarimón et al. 2009; CPRAC 2012; Esty and Winston 2006; Knight and Jenkins 2008; Plouffe et al. 2011; Rieradevall et al. 2005; Rupérez et al. 2008; van Hemel and Cramer 2002) (Table 1).

The environmental performance of a product improves (i.e., there is a footprint reduction) by optimizing inputs and outputs of the production process, which reduces resource consumption (i.e., energy, materials, water), and, consequently, the environmental impact (e.g., emissions, waste) and increases the efficiency of the system. Moreover, the implementation of eco-design methodologies might promote the application of environmental management systems (EMS). As external drivers, environmental data can be used for communication-to-user and marketing purposes while expanding the presence of the environment as a

Table 1 Internal and external drivers for eco-design in companies for environmental, economic and social aspects

	Internal drivers	External drivers
Environmental	Decrease of resource consumption	Use of environmental communication
	Decrease of environmental impact	Compliance with environmental legislation
	Increase of efficiency	Contribution to global sustainability
	Enhanced environmental management systems	
	Continuous improvement	
Economic	Variable cost savings	Market differentiation
	Fixed cost reduction	Green purchasing
	Introduction into new markets	Supply for new green market demands
	Development of new products	Enhanced supply chain information
	Improved product quality	
Social	Improved company image	Environmental awareness
	Enhance of innovation and entrepreneurship	Environmental responsibility
	Increased staff motivation	

Adapted from Boks 2006; Borchardt et al. 2011; Brezet and van Hemel 1997; Clarimón et al. 2009; CPRAC 2012; Esty and Winston 2006; Knight and Jenkins 2008; Plouffe et al. 2011; Rieradevall et al. 2005; Rupérez et al. 2008; van Hemel and Cramer 2002

decision-making criterion during purchase. Moreover, improved environmental profiles comply with current regulations but also anticipate more restrictive normative conditions. Finally, eco-design contributes to the global sustainability along with current legislation to establish a framework for promoting continuous environmental improvement (i.e., ISO 14006).

Thanks to the application of efficient production systems, both variable and fixed costs can be reduced (e.g., less demand for the cleaning treatment of outflows by internal recycling and lower demand for resources by energy efficiency). Companies have the opportunity to differentiate themselves from competitors, enter into new markets, and develop new products. Finally, the image of the product is usually improved with the incorporation of environment criteria into its design. Apart from differentiation, companies can benefit from green procurement and from new green market demands. Further, the application of eco-design enhances environmental information along the supply chains.

The social image of the company is also upgraded by the inclusion of environmental criteria and reporting environmental responsibility. Companies generally turn out to be more innovative and entrepreneurial than their competitors when promoting eco-design; the staff motivation also increases. Lastly, environmental communication and marketing can promote environmental knowledge and awareness among customers and consumers.

2.1 *The Scope of Eco-Design*

DfE or eco-design is defined as the integration of environmental aspects in the product design process during its life cycle (Directive 2009/125/EC). Eco-design can be applied pursuing different objectives depending on the product life cycle stage that must be improved. In this sense, different “Design for X” tools were developed.

- Design for remanufacture (Borchardt et al. 2011; Okumura et al. 2001; Pigosso et al. 2010) focuses on the redesign of an existing product.
- Design for manufacture and assembly (Boothroyd et al. 1994) addresses the improvement of the production process.
- Design for disassembly (Cser and István 1996) aims to optimize the lifespan of the product (e.g., substitution of pieces and reparation) and enhance product recyclability.
- Design for reuse (Hoffmann et al. 2001) aims to optimize the lifespan of the product.
- Design for recycling (Seliger et al. 1999; Oyasato et al. 2001) enhances the product recyclability by avoiding end-of-life treatments with higher impacts.

2.2 *Eco-Design Implementation*

Eco-design has been applied to different types of products. Several guides have been developed, not only about methodology (IHOBE 2000) but also for specific sectors: urban furniture (e.g., streetlight, bin, bench) (Fundació La Caixa 2007), household products (e.g., appliances) (Rieradevall et al. 2003), electric appliances and electronic devices (Rodrigo and Castells 2002), and packaging (Rieradevall et al. 2000).

Furthermore, the implementation of eco-design in different product sectors has been also analyzed in the literature through case studies, such as wooden products (González-García et al. 2011a, 2012a, b, c), electronics (Unger et al., 2008; Mathieux et al. 2001; Aoe 2007), lighting (Gottberg et al. 2006; Casamayor and Su 2013), automobiles (Alves et al. 2010, Muñoz et al. 2006), packaging (Almeida et al. 2010), and printing (Tischner and Nickel 2003).

Finally, some entities and private companies has developed guidelines and procedures focused on eco-design. For example, Philips published the eco-design manual for electronic products (Cramer 1997; Stevels 1997). Volvo wrote an environmental guidance for car designers (Westerlund 1999). The British Marine Industries Federation made an environmental code of practice for boats (BMIF 2000). The Institute for Product Development (DTU Denmark) published a general eco-design guide (DTU 2005). In Sweden, different handbooks were published by the public administration, such as the guide for electronic products (Bergendahl et al. 1994), or by private companies, such as the construction handbook (AutolivSverige 1999).

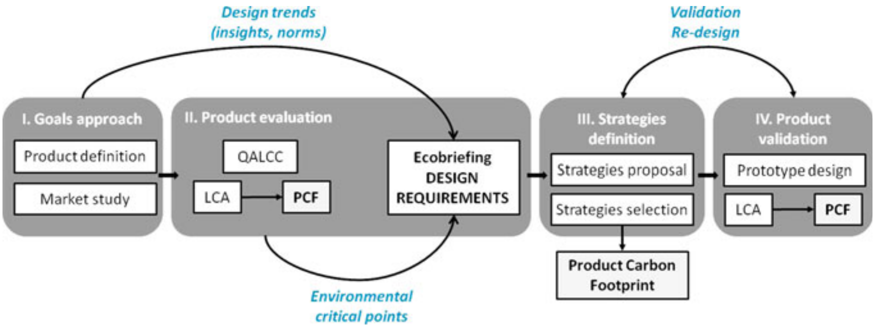


Fig. 2 Steps and tools of the eco-design methodology and role of the product carbon footprint

3 Overview of the Eco-Design Methodology

The presented methodology is based on the combined eco-design and life cycle assessment (LCA) procedure described by González-García et al. (2011b). However, qualitative tools were incorporated for the product evaluation. Although several tools were developed to implement eco-design (Bovea and Perez-Belis 2012; Le Pochat et al. 2007; Pigosso et al. 2010), this methodology was used and improved during the development of real pilot projects and, therefore, was optimized for the participation of companies.

The methodology is divided into four main steps (Fig. 2). First, the product is defined in order to approach the goals of the eco-design process (step I). During this first step, a market study is also completed to detect the design trends (insights and norms) that can contribute to the design requirements. Then, a product evaluation (step II) is performed through the application of the qualitative assessment of life cycle criteria (QALCC) (CPRAC 2012) and a quantitative analysis by means of LCA (ISO 2006a) and carbon footprint (ISO14067, PAS2050). The outputs of both tools are compiled in an eco-briefing (Smith and Wyatt 2006) of the critical points in the life cycle. As a result, the proposal of eco-design strategies (step III) can be defined and selected by the company after a technological, social, and economic assessment. Finally, the prototype is determined by the company with the integration of the chosen strategies (step IV) and the product is validated (step V) through LCA and PCF. These last stages usually interact with each other. The new design is validated and redesigned until it is optimized.

PCF (PAS 2050, ISO 14067) can be used during the eco-design process in different steps and for pursuing different purposes. First, PCF can be used as environmental indicator in the quantitative assessment (steps II and IV). Second, PCF can be used as strategy for environmental communication to the consumer (step III).

4 Qualitative Tools for the Environmental Assessment

4.1 Qualitative Assessment of Life Cycle Criteria

QALCC (CPRAC 2012) is a qualitative methodology that aims to obtain a first environmental assessment of the product at the life cycle stage scale (i.e., the different stages are analyzed separately). Through the interpretation of the QALCC results, the stages that have the largest potential to be environmentally improved are detected. The QALCC is a basis to create an eco-briefing or checklist of the environmental requirements for the eco-design process.

The QALCC methodology consists of three main steps (Fig. 3). First, the life cycle stages of the product are identified to define the system. Second, environmentally relevant criteria are determined for each life cycle stage. Finally, the assessment is performed by the creation of an expert team that evaluates the criteria in order to obtain a spider diagram, in which the valuation of each life cycle stage is represented.

In the system definition, the different life cycle stages of the product are specified. Usually, the life cycle stages are divided into concept, materials, production, packaging, distribution, use, and end-of-life. The life cycle stages considered depend on the LCA perspective: cradle-to-gate, cradle-to-consumer, or cradle-to-cradle.

As second step, various environmentally relevant criteria are defined for each life cycle stage. Criteria for the concept of the product can be the optimization of the function and timeless design. When considering the materials and packaging life cycle stages, the amount and variety of materials should be considered. In the case of the processing of products, the number of production steps and the amount of production wastes are common criteria for the QALCC analysis. Regarding the distribution stage, the distance requirements or logistics optimization can be evaluated as criteria that show the potential environmental contribution of this stage. Communication-to-user about maintenance or resource consumption during maintenance can be criteria for the use stage. Finally, for the end-of-life stage, criteria can be the presence of separable components or separable materials.

For the assessment, a multidisciplinary team must be created. The team may involve the largest number of departments of the company. Therefore, it is recommended to engage employees ranging from directors to workers and from the design department to the sales department. The team can also be complemented by eco-design experts, such as research entities involved in pilot projects. The role of

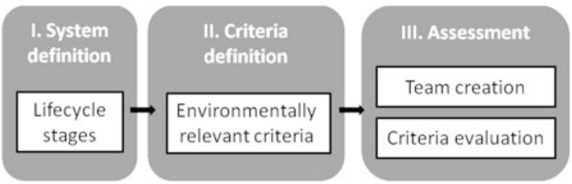


Fig. 3 Steps of the Qualitative Assessment of Life Cycle Criteria (based on CPRAC 2012)

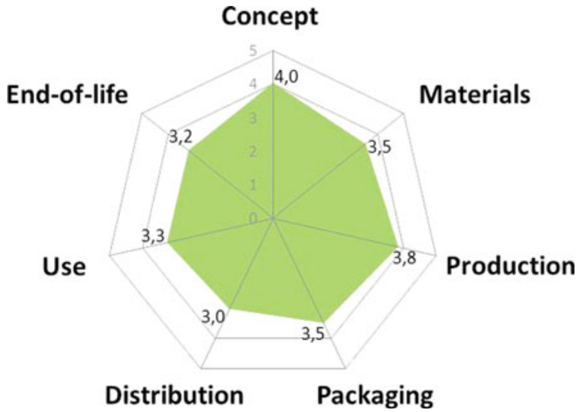


Fig. 4 Example of a QALCC spider diagram. In this case, the perception is that distribution and end-of-life stages are the least environmentally friendly ones, whereas concept and production are the most valued stages

the team is evaluating the criteria according to their possibilities in order to make them more environmentally friendly, grading them on a scale from "enormous room for improvement" (1) to "no room for improvement" (5).

After this evaluation, punctuation is averaged per criteria and per life cycle stage. Results may be represented in a spider diagram that enables the identification of the worst and best life cycle stages and, therefore, those stages where more efforts and strategies should be focused. The area represents the environmental impact: the lower the potential impact, the larger the area (closer to 5: no room for improvement) (Fig. 4).

As an example, the QALCC method was applied to an indoor chair for its entire life cycle (Fig. 4). The highest rated aspects of the design were the multipurpose design (4.5, concept), the use of local materials (4.2, materials), the low number of processing steps (4.1, production), the use of renewable materials (3.7, packaging), the reduced distance for raw materials extraction (4.7, transport), the low consumption of resources during maintenance (4.1, use), and the easy disassembly of materials (4.3, end-of-life). On the other hand, the lowest rated issues were low eco-innovation of the product (2.8, concept), non-use of recycled materials (2.1, materials), large amount of leftovers (3.4, production), non-use of reusable packaging systems (2.1, packaging), no hiring of low impacting transportation (2.6, transport), and no communication about either the maintenance (2.2, use) or the end-of-life management (1.9, end-of-life). The global values highlighted that eco-design strategies should focus on the distribution, use, and end-of-life stages (Fig. 4).

The QALCC tool has some advantages compared to other assessment methods. First, it is comprehensive and accessible to professionals who are not familiar with environmental tools. This enables the involvement of the different departments of a company. Second, the representation of the results facilitates interpretation by professionals who are not familiar with the tool. Third, the application of the tool and the results extraction steps are quick. Fourth, the life cycle concept is introduced to the company and the professionals involved in the project. Finally, this tool facilitates the communication of the environmental profile of the product as well as potential improvements.

However, QALCC should be complemented with a quantitative method due to its disadvantages. First, only qualitative data can be obtained. Second, the contribution of each life cycle stage to the product impact is not measurable, as each stage is assessed separately. Finally, results are linked to the expertise of the professionals who perform the assessment. For this reason, the presented methodology combines QALCC as a qualitative tool and LCA and PCF as quantitative assessment methods for hotspot identification in the eco-briefing.

5 Quantitative Tools for the Environmental Assessment

The quantitative assessment may complement the result of the qualitative assessment to complete the final eco-briefing. As because the entire life cycle of a product is considered, the LCA methodology is used. The PCF can be used as environmental indicator to assess a product's impact on global warming.

5.1 Life Cycle Assessment

LCA (ISO 2006a) is a methodology that allows systematizing the compilation and generation of information to establish objective criteria in the decision-making process for a sustainable development. The LCA method is defined by the ISO 14040 series [Environmental Management—Life Cycle Assessment—Principles and framework] (ISO 2006a). Moreover, this tool efficiently detects the improvement opportunities of an entire system.

The consideration of the environmental impacts of a product, process, or service along its life cycle has been performed since the 1960s and has been expanded in recent years. According to the ISO 14040, the LCA methodology is used for:

- identifying opportunities to improve the environmental performance of products at various points in their life cycle
- informing decision-makers in industry, government, or non-government organizations (e.g., for the purpose of strategic planning, priority setting, product or process design, or redesign)
- selecting relevant indicators of environmental performance, including measurement techniques
- marketing (e.g., implementing an eco-labeling scheme, making an environmental claim, or producing an environmental product declaration)

The depth and range of an LCA study can considerably vary depending on the specific goal of an study (Baumann and Tillman 2004). However, the current standard practice of LCA (ISO 2006a) includes four steps (Fig. 5):

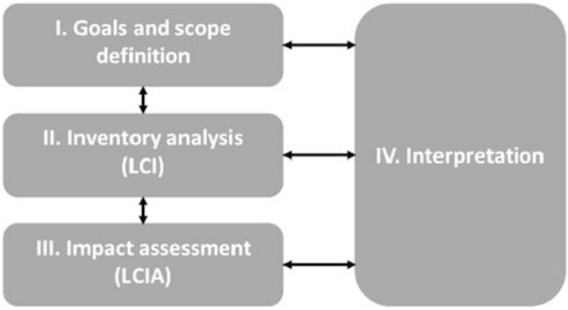


Fig. 5 Steps of the Life Cycle Assessment Methodology (ISO 2006). LCI, life cycle inventory; LCIA, life cycle impact assessment

- *Definition of the goal and scope of a project*. The objectives of the study, the intended audience, and the scope are defined. The last one includes the system that must be analyzed, the functional unit, the system boundaries, and the assumptions as the main items. The functional unit is a measure of the function of the system under study that provides a reference to which the inputs and outputs are related.
- *Inventory analysis* (or life cycle inventory) involves the data collection stage of the method. The objective is to quantify the relevant inputs and outputs of the product system (i.e., energy and raw material requirements, atmospheric emissions, waterborne emissions, solid wastes, and other releases for the entire life cycle of a system).
- *Impact assessment* The life cycle impact assessment is aimed at evaluating the significance of potential environmental impacts using the life cycle inventory results. The process relates the inventory data to specific environmental impact categories (e.g., climate change, ozone depletion).
- *Interpretation* of the significance of impacts. Interpretation is the phase of LCA in which the findings from the inventory analysis and the impact assessment are considered together.

5.2 Product Carbon Footprint as an Assessment Tool

The carbon footprint (CF) is a globally accepted tool for quantifying the environmental burdens of products. This indicator can be obtained through the implementation of an LCA analysis, like other environmental impact categories. The CF is an estimation of the GHG emissions from business activities. The goal of the method is to quantify the global GHG emissions related to the entire life cycle of a product, process, or service. This quantification is expressed in CO_2 equivalent (a unit for expressing the irradiative forcing of a GHG to carbon dioxide) and has become a common indicator for environmental assessment.

This calculation has increasingly more relevance for organizations as customers and consumers perceive and increasingly support environmentally responsible firms. Carbon footprint can be performed at different levels (BSI 2011):

- Organizational carbon footprint, at the company level
- PCF, which includes:

(a) an activity performed on a consumer-supplied tangible product (e.g., automobile to be repaired)
(b) an activity performed on a consumer-supplied intangible product (e.g., the income statement needed to prepare a tax return)
(c) the delivery of an intangible product (e.g., the delivery of information in the context of knowledge transmission)
(d) the creation of ambience for the consumer (e.g., in hotels and restaurants)
(e) software, which consists of information and is generally intangible and can be in the form of approaches, transactions, or procedures

At the organizational level, the international standard family ISO 14064 describes the specifications and guidance for the certification: ISO 14064-1 (ISO 2006b) Greenhouse gases—Part 1: Specification, establish the basic specifications and guidance for certification, ISO 14064-1 (ISO 2006c) Greenhouse gases—Part 2: Specification with guidance at the project level for quantification, monitoring, and reporting of greenhouse gas emission reductions or removal enhancements, and ISO 14064-3 (ISO 2006d) Greenhouse gases—Part 3: Specification with guidance for the validation and verification of greenhouse gas assertions.

The calculation of the PCF is standardized by the specification PAS 2050 (BSI 2011), where a method is provided for accounting for the GHG emissions in the life cycle of goods and services (products). The international standard ISO 14067 (ISO 2013) has recently been published and provides the standard for the application of this method (Carbon footprint of products—Requirements and guidelines for quantification and communication).

PCF quantification follows the LCA stages (Fig. 5). The approach of the method can be cradle-to-grave or cradle-to-gate, based on the system boundaries of the analysis (i.e., the emissions and removals considered arise from the full life cycle of the product or up to the point at which the product leaves the organization) (BSI 2011). PCF aims to measure the overall GHG emissions of a product by considering both emissions to the atmosphere and removals from the atmosphere and by assessing both carbon and biogenic carbon sources (BSI 2011) (Fig. 6).

PCF might characterize the energy use, combustion processes, chemical reactions, loss to atmosphere of refrigerants and other fugitive GHGs, process operations, service provision and delivery, land use and land use change, livestock production, and other agricultural processes and waste management. The assessment of GHG emissions and removals should be performed in a 100-year assessment period. The global warming potential (GWP) of the emissions is calculated according to the latest Intergovernmental Panel on Climate Change coefficients. Multiplier factors for aircraft emissions should not be applied and the

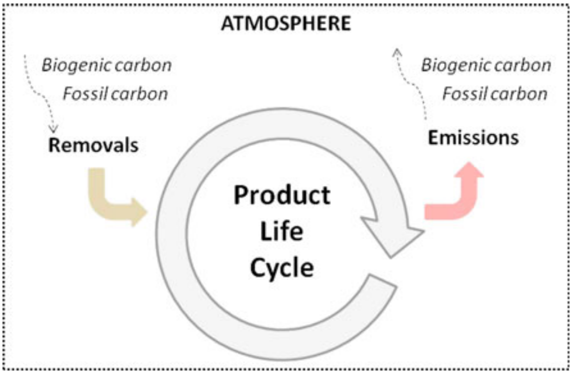


Fig. 6 Flows considered in the product carbon footprint (BSI 2011)

carbon that will not be emitted within this time period (100 years) should be treated as carbon storage (BSI 2011).

In the case of eco-design, the PCF can play two main roles as environmental indicator or assessment tool. First, PCF is used for the quantitative assessment of the initial product because it shows the contribution to global warming of a product in order to be considered in its design (Jeong and Lee 2009). Therefore, the source of GHG emissions and the related environmental impact can be detected within the life cycle of the product. Second, as other environmental indicators, PCF can also be used to define eco-design goals (e.g., reduce 20 % the PCF of the product) and to establish thresholds.

Moreover, PCF is a useful indicator in the realization of projects in companies, such as:

- PCF is an indicator of an increasing importance for reporting the environmental performance of products and organizations (CPRAC 2012)
- PCF has an important role as an eco-label, which has already presence in the market (e.g., Carbon Trust)
- The consumer has knowledge about the topic and can understand the meaning of the PCF, because:

– There is environmental awareness regarding global warming and climate change (e.g., development of laws and standards)
– These topics are covered by mass media (e.g., documentaries, press)
– Environmental laws have already been developed for domestic consumables and marketing has been done regarding carbon emissions (e.g., cars)

Therefore, PCF can be used as a quantitative assessment tool and as indicator to show the environmental burdens of the initial product (step II), the potential improvement of the strategies (step III), and to validate the environmental performance of the prototype design (step IV).

6 Eco-Briefing: Design Requirements Definition

The eco-briefing (Smith and Wyatt 2006) shows, in a concise and clear way, the environmental critical points where attention must be paid to minimize their contribution to the environmental burdens through the eco-design methodology, as well as to observe the life cycle stages where they are concentrated.

The eco-briefing may compile data from the environmental assessment (both qualitative and qualitative). In this case, eco-briefing collects the results from the QALCC and LCA tools. The main requirements for a conventional briefing are contextualization (general description, market trends), product (objective, range of product), design aims and conditioning (available technology, cost, time), project definition, and expected results. An eco-briefing complements this information with the identified environmental critical points in the different life cycle stages (Table 2).

Following the same example, the eco-briefing of an indoor chair included impacts from the QALCC and LCA analysis (Table 2). First, the multidisciplinary team of the QALCC pointed out that the chair design was not innovative and, therefore, it might be a requirement for the prototype definition. Second, both methods noted that some materials were high-impacting and that minimal recycled materials were used. However, the quantitative assessment also highlighted the significant contribution of the energy consumption and paint to the environmental impact. The low efficiency of the volume for distribution was identified in the qualitative assessment and confirmed with the high impact of the distribution in the LCA. Finally, packaging and communication requirements were mainly determined in the qualitative assessment.

Table 2 Example of eco-briefing of an indoor chair

	Life cycle stages						
Critical points	C	M	P	D	P	U	EoL
Lack of innovation	●						
High impact of the wood board		●			●		
Low use of recycled materials		●			●		
High impact of the energy consumption during processing			●				
High impact of the paint			●				
Low efficiency of the volume for distribution	●			●	●		
High impact distribution				●			
Multi-material packaging					●		
Insufficient environmental communication of the maintenance	●					●	
Insufficient environmental communication of the end-of-life	●						●

C Concept, *M* Materials, *P* Production, *D* Distribution, *P* Packaging, *U* Use, *EoL* End-of-life

7 Definition and Selection of Strategies

Once the design requirements for each life cycle stage are identified in the product evaluation, eco-design strategies are proposed in order to improve the product profile by solving the main issues. As a second step, the strategies are selected by the company for designing the prototype.

7.1 Strategies Definition

The first life cycle stage of a product is the conception, where several environmental aspects can be approached and different life cycle stages can be improved at the same time (Table 3). Strategies regarding this life cycle stage improve the design of the product by considering concepts such as functionality, temporality, and lifespan. However, other stakeholders can be involved in this stage, such as consumers (when providing environmental information) and suppliers (when demanding environmental information of production inputs).

The materials of a product (Table 4) can be improved through different ways. The amount of resources can be reduced or optimized (e.g., dematerialization, reused components). Materials can be switched to more environmentally friendly options: renewable materials, recyclable materials, or low-impact materials. Moreover, material selection can be done based on other life cycle stages. A single material design can enhance the recyclability of the product, materials with low maintenance requirements can reduce the contribution of the use stage to the environmental burdens, and materials associated with low-impact end-of-life options can improve this stage of the product. Finally, the selection of local suppliers for the material inputs reduces the transport requirements.

Table 3 Main strategies regarding concept (based on CPRAC 2012)

Life cycle stage	Strategy	Environmental aspect
Concept	Dematerialization	Reduction of resource consumption
	Product sharing	Maximization of product use
	Multifunctional product	Reduction of resource use per function
	Timeless design	Increased lifespan
	Design for updating	
	Environmental information (e.g., carbon footprint)	Market differentiation
	Demand for suppliers' environmental information	Environmental impact of products Environmental responsibility
	Design for assembly and disassembly	Reduction of environmental impact of other life cycle stages of the product

Table 4 Main strategies regarding materials (based on CPRAC 2012)

Life cycle stage	Strategy	Environmental aspect
Materials	Dematerialization	Reduction of resource consumption
	Single material design	Enhanced recycle options
	Recyclable materials	
	Renewable/natural resources	Decoupling of non-renewable resources
	Low-impact materials	Product impact reduction
	Use of abundant materials	
	Local resources	Local positive impacts and distribution impact reduction
	Materials with easy end-of-life management (biodegradable)	Reduced product end-of-life impact
	Materials with low maintenance	Reduced product use impact
	Reused components	Reduced resources extraction

Table 5 Main strategies regarding production (based on CPRAC 2012)

Life cycle stage	Strategy	Environmental aspect
Production	Internal recycling closed-loop production	Waste and emissions reduction
	Optimize production processes	Reduced production impact
	Choose cleaner production processes	Enhanced efficiency
	Use of low-impact energy sources	Reduced energy consumption impact
	Local production	Reduction of logistics requirements

Production strategies commonly address efficiency (Table 5). Material inputs and their environmental impact can be reduced by enhancing internal recycling processes (e.g., waste flows from the production can become a resource for the same or other products). Attention is also paid to the energy flow, where the consumption can be reduced and the energy source may be as clean as possible (e.g., renewable energy systems). Finally, the production process can be revised to optimize and invest in cleaner production systems while reducing the environmental impact of this life cycle stage.

Although packaging is a secondary life cycle of the analyzed product, it is an important stage to assess and it can result into transversal environmental actions (i.e., some products have the same packaging) (Table 6). Main strategies focus on the reduction of resource consumption and the associated impacts, and they also consider the volume of the design, which affects other life cycle stages. The lifespan of the product can be enlarged by designing a reusable or multifunctional packaging.

The transportation of the material inputs and the finalized product can be improved in different ways (Table 7). First, the design of the product can be

Table 6 Main strategies regarding packaging (based on CPRAC 2012)

Life cycle stage	Strategy	Environmental aspect
Packaging	Avoid superfluous packaging Dematerialization	Reduction of resource consumption
	Reusable packaging Multifunctional packaging	Increased lifespan
	Low-impact materials	Reduced of product packaging impact
	Recyclable materials	Enhanced recycle options
	Volume reduction	Optimization of product distribution

Table 7 Main strategies regarding distribution (based on CPRAC 2012)

Life cycle stage	Strategy	Environmental aspect
Distribution	Optimization of product weight	Reduction of transportation energy consumption
	Local distribution Biofuels transportation Efficient transportation	Reduction of transportation environmental impact

Table 8 Main strategies regarding use (based on CPRAC 2012)

Life cycle stage	Strategy	Environmental aspect
Use	Optimization of resources consumption during use	Reduction of product use impact
	Easy installation/assembly Easy maintenance	Reduction of resources consumption
	Easy reparation Modular design Availability of spares Design for customize Product reliability and durability	Increased lifespan
	Communication-to-user (e.g., best practices)	Reduction of product use impact

optimized to reduce the product weight and, thereby, to reduce the environmental burdens of transportation. Second, transportation requirements can be diminished by looking for local suppliers and retailers. Finally, more environmentally friendly means of transport (i.e., biofuels, efficient systems) reduce the environmental impact.

Regarding the use stage (Table 8), strategies may focus on an optimized use of resources (e.g., design for an easy installation and maintenance), an enlargement of the lifespan of the product (e.g., design for disassembly and easy reparation), and communication-to-user to boost best practices by reducing the environmental impact of this stage.

Table 9 Main strategies regarding end-of-life (based on CPRAC 2012)

Life cycle stage	Strategy	Environmental aspect
End-of-life	Simplification of disassembly	Enhanced recycle and reuse options
	Component disassembly	
	Recyclability	
	Material identification	
	Reusability	Increased lifespan
	Biodegradability	Reduction of product impact
	Communication-to-user (e.g., end-of-life options)	Reduction of product end-of-life impact

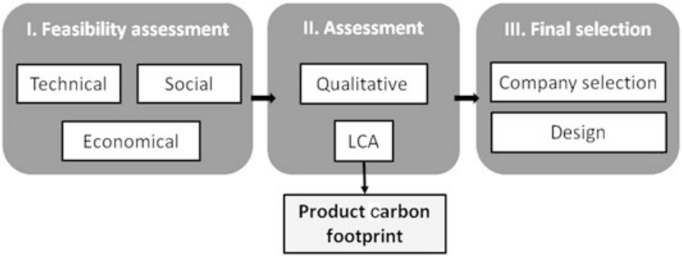


Fig. 7 Steps of the eco-design strategies selection process. LCA, life cycle assessment

Finally, different strategies can be used to improve the end-of-life stage (Table 9). Materials can be recyclable or can be biodegradable to reduce the environmental impact of the disposal process. The design can be performed for disassembly (facilitate end-of-life management) or for reusability (increase the lifespan). Finally, communication-to-user strategies can enhance best practices regarding end-of-life treatments to reduce their impact.

7.2 Strategy Selection

Once the strategies are defined, two selective steps are done in order to determine the eco-design strategies that should be integrated in the prototype (Fig. 7). First, a feasibility assessment is performed by the company to observe technical, economic, and social constrains. This step leads to a first selection of the most potential strategies. Second, the selected strategies are assessed (both quantitative and qualitative) to observe their potential improvements. Finally, the company picks some of the strategies to be incorporated in the prototype design.

The feasibility assessment (Table 10) is performed for each proposed strategy. The technical, economic, and social feasibility are determined as “Feasible” (F+), “Feasible at mid-term” (F-), “Unfeasible” (U) or “Not applicable” (NA) when the

Table 10 Example of the feasibility assessment of potential concept strategies

Id	Strategy	Criterion	Feasibility				Priority
			F+	F−	U	NA	
Concept	Increase of functionality	T		X			Low
		E	X				
		S	X				
	Design for customize	T	X				Medium
		E	X				
		S	X				
	Volume reduction	T			X		Low
		E			X		
		S				X	

T technical, *E* economic, and *S* social

criterion is not considered. Finally, a priority value is defined as "High," "Medium," or "Low" in order to establish a classification for the strategies selection.

For the feasible eco-design strategies, a quantitative assessment (LCA and/or PCF) is used to show the potential environmental improvement of the strategy. For those conceptual strategies (e.g., multifunctional product, communication) the assessment is carried out from a qualitative perspective (e.g., description of the potential benefits and involvement of stakeholders). After the assessment, the company selects the final strategies for defining the prototype.

7.3 Product Carbon Footprint as a Communication Tool

Environmental communication is the process of sharing environmental information, not only with suppliers but also with customers. The main goal is to generate confidence, credibility, and association as well as to increase the environmental awareness during decision-making processes.

Labeling for environmental communication is used for showing the environmental contribution of products to human health and sustainability. With this standard labeling system, the company shows their reliability and becomes competitive in their sector.

For environmental communication, the PCF is a useful and understandable indicator to show the environmental aspects of a product. This can be used within a set of indicators, such as a part of an Environmental Product Declaration (Ecolabel type III, norm ISO 14025) (ISO 2006e) or as a unique indicator. In this second case, the PCF can be done through the standard certification (following the PAS 2050) and by means of an official entity (e.g., Carbon Trust), or through a self-declared environmental claim (eco-label type II, norm ISO 14021 (ISO, 1999)) (Fig. 8).

Fig. 8 Examples of the use of product carbon footprint as a communication-to-user tool: **a** environmental product declaration (EPD) of a chair (©Arper), **b** Carbon Trust declaration of a wine (©Mobiu), and **c** Self-declared product carbon footprint of an eco-designed knife by ARCOS (©Sostenipra)

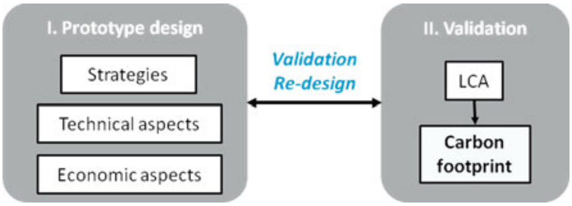
Moreover, environmental communication can be done through different pathways. As mentioned before, labels can be used in the product or packaging. However, this communication can also be integrated in the company's functions (e.g., marketing, advertisement, events, customers, suppliers, and website). For example, the ISO 14063 (ISO 2006f; on Environmental Management—Environmental communication of the company) establishes guidance on the communication of the environmental policy (general, strategy, and activities) of the company, based on the EMS ISO 14001 (ISO, 2000).

8 Design of the Prototype

The final step of the presented eco-design methodology is the design of the prototype. The selected strategies are integrated and combined to obtain a more environmentally friendly product with environmental aspects as added value.

The final prototype is obtained as a result of two interactive steps: prototype design and validation (Fig. 9). First, the company designs the prototype as a result of integrating eco-design strategies but also adapting them to the technical and economic constraints. Second, the environmental performance of the new product is validated through a quantitative method (LCA) in order to assess the environmental improvement of the entire eco-design process. Both steps interact in order to optimize the outputs of the eco-design efforts by considering the company context (e.g., technical availability, economic costs) and the aimed environmental

Fig. 9 Steps of the prototype design and validation. LCA, life cycle assessment



improvements. The PCF can be used in this step as an environmental indicator for the quantitative validation. Moreover, it can also be used as goal of the eco-design process (e.g., minimum reduction of the PCF value), as mentioned before.

9 Conclusions

The presented eco-design method is a comprehensive way to assess the potential environmental improvement of products, processes, and services. It combines qualitative (VEA) and quantitative (LCA) methods for the assessment of product as well as for analyzing the strategies. As a life cycle-based method, each life cycle stage is assessed in detail in order to optimize the impact contribution of the product. The eco-briefing method establishes the design requirements and results in a complete basis for defining eco-design strategies and for transferring complex environmental information to the design team. The strategies selection method combines two selection steps, where the company is involved, and optimizes the efforts for defining the new prototype design.

Finally, the PCF may have an important role in the eco-design methodology because it can be used in different steps and for achieving diverse targets. Carbon footprint is a well-known and understandable communication tool, not only from companies to consumer but also between businesses. Moreover, PCF can be a pioneer indicator in the implementation of quantitative environmental communication in products and services. For communicative purposes, the use of PCF can be also complemented with other life cycle environmental indicators in order to show different environmental aspects of the product (e.g., LCA indicators, water footprint).

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Carbon Footprint Estimation in the Agriculture Sector

Divya Pandey and Madhoolika Agrawal

Abstract The term "carbon footprint" has evolved as an important expression of greenhouse gas (GHG) intensity for diverse activities and products. Widespread public acceptance and the ease of conveying information about GHG intensity with this term has also attracted scientists and policy makers to review and refine its calculations. Standard methods for carbon footprinting have been prepared, and sector-specific standards are under development. These standards direct the procedures to carry out carbon footprinting through life cycle assessment in conjunction with GHG accounting, classifes activities into three tiers based on the order of emissions. Agriculture is the largest contributor to anthropogenic emissions of greenhouse gases, so the quantification of different agricultural practices is essential for identification of more sustainable practices. Carbon footprinting has potential as a tool for assessing and comparing GHG performances of different agricultural products along with identification of points to improve environmental efficiencies. Case studies on the application of carbon footprinting to cultivation practices are increasing in the scientific literature, but the majority of studies do not comply with the standard three-tier methodology. This leads to nonuniformity among different studies and their comparisons. Hence, a standard guideline addressing carbon footprinting specifically for agriculture is essential for the effective application of this tool in the quantification of GHG intensity, mitigation of global warming, and adaptation against future climate change scenarios.

Keywords Cultivation practices · Mitigation · Agricultural management · Three-tier methodology

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1 Introduction

Climate change has emerged as the biggest environmental and developmental challenge of the present time; it also influences the focal possibilities for sustainable development. The effects of climate change have already been felt all over the world, in diverse forms ranging from shifting weather patterns, receding ice caps, crop losses, altered distribution of precipitation, increased frequency and intensities of floods and droughts, and serious ecological imbalances. All of these effects also have resulted in significant economic losses (Stern 2006). To prevent projected and unforeseen disasters, global temperatures must not exceed 2 °C more than 1990 levels. For this, the atmospheric stock of greenhouse gases (GHGs) should be controlled to remain below 550 ppm in terms of CO_2 equivalents (CO_2-e). Among different GHGs, carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O), sulfur hexafluoride (SF_6), perfluorocarbons (PFCs), and hydrofluorocarbons (HFCs) are the six important anthropogenic GHGs. GHG inventories can identify, quantify, and manage all sources and sinks of GHGs. Among different quantitative indicators, the carbon footprint has gained popularity and widespread application. Moreover, because of its ease of conveying information about the GHG intensity of variety of products and activities among the general public, carbon footprint also offers a simple mode of communication about climate responsibility of different entities between people, scientists, and policy makers. Scientific analyses of carbon footprinting are being conducted, mainly for consumer products and industrial processes; its application to agricultural systems is less, despite the fact that agriculture alone is responsible for GHG emissions to the largest degree. Here, we review the available scientific literature on the concept and calculations of carbon footprint, and its application to the agriculture sector. We begin with an overview of agriculture's role in regulating GHG fluxes, followed by the concept and general principle of carbon footprinting. Applications and challenges in using carbon footprinting in agriculture are also discussed.

2 Agriculture as a Source of Greenhouse Gases

Covering about 35 % of the land area, agriculture accounts for nearly 13.5 % of the total global anthropogenic GHG emissions, contributing about 25, 50, and 70 % of CO_2, CH_4, and N_2O, respectively (Montzka et al. 2011). As it is recognized that cereal production must increase at a rate not less than 1.3 % annually (Cassman et al. 2003), related emissions are also expected to increase. GHG emissions from agriculture originate mainly in the form of CH_4 from rice cultivating systems and cattle rearing and N_2O from fertilizer management practices.

Rice fields alone emit 32 to 44 Tg CH_4 yr^{-1} (Le Mer and Roger 2001). Del Grosso et al. (2008) estimated that agricultural activities add into the atmosphere about 4.2 to 7 Tg N annually in the form of N_2O. Due to their high global warming potential of 298, emissions of N_2O, even in a small quantity, cause significant radiative forcing. Increased soil temperatures coupled with high moisture conditions during cooler months will increase N_2O production in soil. Elevation in CO_2 concentrations is also projected to increase N_2O emissions from upland agricultural soils (Van Groeningen et al. 2011). Regarding CO_2, soil respiration is an important source, but the majority of the farm operations and inputs, such as fertilizers, pesticides, and energy, also have embodied CO_2 content.

The majority of GHG estimates cover only soilborne emissions, generally of CH_4 and N_2O only, whereas numerous studies have been carried out targeting only CH_4 measurements (Le Mer and Roger 2001) and its mitigation from rice fields, mainly through water (Pathak et al. 2003), fertilizer, and manure managements (Linquist et al. 2012). Among different management techniques, mulching and organic manure applications are found to increase the emissions of CH_4 (Ma et al. 2007), whereas midseason drainage can cut CH_4 emissions significantly (Zou et al. 2005). Aerobic soils, on the contrary, may act as CH_4 sinks (Le Mer and Roger 2001; Smith et al. 2008) or sources (Ma et al. 2013), but they too are poorly quantified (Robertson 2000).

As a widely recognized effect, application of mineral nitrogen increased the emissions of N_2O. However, the effects of different management practices on emissions of all the GHGs are highly inconsistent, depending on the cultivation system and environmental conditions. Some inhibitors to methanogenesis and nitrification have also been tested in agricultural soils (Liu et al. 2010). It is found that frequency and timing of tillage also influence fluxes of soilborne GHGs. In the long term, the elimination of tillage reduced the emissions of CH_4 and N_2O, but increased CO_2 from rice cultivation (Pandey et al. 2012) as compared to conventional practice of regular tillage.

3 Agricultural Management as a Carbon Offsetting Option

Although, agriculture is an emissions source, there are opportunities for reducing the emissions and even using cultivated soils as a GHG offsetting tool if better management practices are identified and adopted (Hutchinson et al. 2007). Soils are the largest terrestrial carbon store; they hold carbon in the form of organic and inorganic molecules. Due to erosion and oxidation, a significant part of soil organic carbon has been lost. Scientific evidence suggests that 50–66 % of the cumulative historic carbon loss from soil can be recovered if managed intelligently (Lal 2004b). Increasing the organic carbon content in soil may lock the carbon out of the atmosphere for centuries a phenomenon is termed as carbon sequestration. The two fundamental keys to support carbon sequestration in soils is minimization of soil disturbance and increasing inputs of organic matter. Therefore, cover crops, mulching, no tillage, organic manure, and decreasing the fallow period are among the recommended management practices (Lal 2004a). Improvement in nutrient status, particularly of nitrogen and phosphorous, also strengthens carbon sequestration; hence, intercropping with legumes and certain permutations of crop rotations are found to be greatly effective (Nishimura et al. 2008). It is estimated that under recommended management, soils of the European Union and UK, respectively, can offset 0.09–0.12 and 0.010 Pg C annually (Smith et al. 2005); at the global level, soils offer an annual sequestration potential of 0.6–2.0 Pg C (Lal 2000). Long-term studies have shown that organic manure application increases the carbon sequestration capacity of soil in the range of 70–551 kg C ha^{-1} as compared to mineral fertilizer use (Mandal et al. 2007, 2008).

4 Understanding Product Carbon Footprints: Concept, Scope, and Calculation

Carbon footprints originated as a subset of the "ecological footprint" proposed by Wackernagel and Rees (1996). *Ecological footprint* referred to the biologically productive land and sea area required to sustain a given human population, expressed as global hectares. According to this concept, carbon footprint was the land area that will assimilate the CO_2 produced during the lifetime of a person or total global population. The calculation of carbon footprint as a part of the ecological footprint was very tedious and complex. But as the issue of global warming gradually gained prominence on the global environmental forefront, carbon footprinting emerged independently, in a modified form (East 2008). The present form of carbon footprints can be regarded as a hybrid that derives its name from "ecological footprint" but conceptually is a global warming potential indicator. However, few studies still report carbon footprints in terms of global hectares with regard to its origin (Browne et al. 2009).

In spite of its widespread popularity among the public as an indicator of contribution of an entity to the global warming, until few years back, there was confusion over what carbon footprints exactly meant (Wiedmann and Minx 2007; Pandey et al. 2011). This was particularly due to the lack of a standard methodology for carbon footprint calculation and its scientific analyses. Most studies have been carried out by private organizations and companies for business purposes rather than environmental responsibility (Kleiner 2007; East 2008). However, recognizing the public response to carbon footprinting studies and increasing financial transactions in the carbon market, standards are now under construction by the International Organization for Standardization (ISO); British Standards Institution (BSI) is also developing and upgrading their guidelines. Scientific literature is also growing, with more and more case studies of carbon footprinting, thus adding to the development of standard methods.

Based on a survey, Wiedmann and Minx (2007) recognized that definitions of carbon footprints were also different among different studies. They suggested that the term carbon footprint should reflect measure of the exclusive total amount of CO_2 emissions that is directly and indirectly caused by an activity or is accumulated over the life stages of a product. A similar indicator, "climate footprint," was proposed to be used, if all the GHGs were included in the calculation instead of only CO_2. But keeping in mind the motive of carbon footprinting, (i.e., assessing the impact of the activity on global climate), new studies and guidelines suggested inclusion of all the GHGs that are covered under the Kyoto Protocol (Kelly et al. 2009; Eshel and Martin 2006). However, still there are terms that are used interchangeably with carbon footprints, such as embodied carbon, carbon content, embedded carbon, carbon flows, virtual carbon, GHG footprint, and climate footprint (Courchene and Allan 2008; Peters 2010). Selection of direct and embodied emissions is also inconsistent among different studies. Direct emissions take place onsite. For example, in an industrial unit, CO_2 released during the combustion of gasoline fired in boiler is a direct emission. On the other hand, if the boiler was electrically powered, no direct emissions will be observed on the site. But during production of that electricity in a thermal power plant, a certain amount of CO_2 should have been released. Such an emission is referred as the embodied or indirect emission. In most cases, it becomes too complicated to include all possible indirect emissions; hence, many carbon footprinting case studies report only direct or first-order indirect emissions (Carbon Trust 2007; Wiedmann and Minx 2007; Matthews et al. 2008). But indirect emissions may constitute the major share of carbon footprints for many activities (Matthews et al. 2008). In spite of prevailing differences among the calculations, the CO_2 equivalent (CO_2-e) mass based on 100 years global warming potential of GHGs is used as the reporting unit of carbon footprints (WRI/WBCSD 2004; Carbon Trust 2007; BSI 2008), although there had been certain critical comments over it. Hammond (2007) and Global Footprint Network (2007) hold the opinion that "footprints are spatial indicators"; therefore, the carbon footprint should precisely be called a "carbon weight" or "carbon mass" (Jarvis 2007). However, convenient calculations and widespread acceptance makes CO_2-e mass the practical unit of carbon footprints.

The definition of carbon footprints is therefore proposed as follows: "The quantity of GHGs expressed in terms of CO_2-e, emitted into the atmosphere by an individual, organization, process, product or event from within a specified boundary" (Pandey et al. 2011).

4.1 Scope of Product Carbon Footprinting

The main drivers of carbon footprint calculations are legislative requirements, carbon trading, corporate social responsibility, and scientific analyses for devising effective policies to combat global warming (Carbon Trust 2007). The scope of carbon footprinting is wide and includes virtually all kinds of products, services, activities, and processes. Carbon footprinting of products and services has proven useful in not only managing the emissions more effectively across the supply chain, but also as a business tool (Kleiner 2007). It is proven by the continually increasing number of companies participating in the Carbon Disclosure Project (CDP 2009). This rush gained momentum from changing marketing strategies as more consumers began to prefer products with low carbon footprints (LEK Consulting 2007). Therefore, regulated carbon labeling of products has also been introduced. Suh (2006) calculated carbon footprints for different products in the USA and concluded that lime was the most GHG intensive product (22.1 kg CO_2-e $\$^{-1}$), followed by chemicals, fertilizers, and meat production. Services such as health care, water supply, computing and data processing, and amusement left smaller carbon footprints, ranging from 42.1 to 46.1 Tg CO_2-e for an average household. Hoefnagels et al. (2010) used product carbon footprinting for comparing the overall performances of different energy options. Under optimum conditions, biofuels production emitted between 17 and 140 g CO_2-e MJ^{-1}. Carbon footprints of different fuels are calculated to decide about the import of nonconventional vehicular fuels in California (Courchene and Allan 2008). Gemechu et al. (2012) also advocated the application of carbon tax based on product carbon footprinting for kraft pulp production, in which energy usage was the most polluting sector with nearly 0.32 kg CO_2-e kg^{-1} of pulp produced.

Among services, aviation has been identified among the highest GHG emitters; hence, the carbon footprinting of airlines is ongoing, covering different aspects such as aircraft types, load factors, and seat configurations (Miyoshi and Mason 2009). The European Union has taken the lead in formulating legal bindings for reduction in emissions embodied in aviation. Schools and universities are also participating in such calculations. GAP et al. (2006) in the UK Schools Carbon Footprint Scoping Study estimated that, in 2001, all schools in the United Kingdom left a carbon footprint of 9.2×10^9 kg CO_2-e. Elsewhere, the University of British Columbia and University of Pennsylvania left carbon footprints of 8.2750×10^7 and 3.0×10^8 kg CO_2-e, respectively (Ferris et al. 2007; TC Chan Centre for Building simulation and Energy Studies/Penn Praxis 2007). Carbon footprints have also been included in the management of cities and organizations to improve environmental policies (Courchene and Allan 2008; Good Company 2008). The UNDP (2007) and Edgar and Peters (2009) used per capita CF of different countries to compare the contributions of countries, cities, and sectors to global warming. These reports clearly indicated that high-income countries leave the biggest footprint, while it was substantially lower for developing countries. Carbon footprint is now used as an indicator in event management as well (London-2012 Sustainability Plan 2007).

In addition to the above, voluntary carbon footprinting by organizations as well as individuals is growing at a fast rate. Consultancies and online calculators have further promoted individual carbon footprinting, particularly in developed countries (Padgett et al. 2008; Kenny and Gray 2008). Such calculators also offer carbon offsetting options, mainly through supporting forestation and renewable energy resources (Murray and Day 2009). A dramatic growth in the voluntary carbon market has been reported since 1989 (Hamilton et al. 2007). Carbon footprinting is also extended to the natural and semi-natural systems, which may help compare natural versus anthropogenic impacts on the environment (Chambers et al. 2007). Hence, we see that there is hardly any entity that cannot be a candidate for carbon footprinting.

4.2 Calculation of Product Carbon Footprints

Being a quantitative expression of GHG emissions, carbon footprinting helps in emission management and evaluation of mitigation measures (Carbon Trust 2007). Through carbon footprint analyses, important sources of emissions can be identified and areas of emission reductions can be prioritized. For carbon footprint calculation, estimates of GHGs emitted/embodied at each identified step of the product's/activity's/individual's life cycle are conducted, which is technically known as GHG accounting. Standards and guidance are available for GHG accounting. Common resources are:

(a) GHG protocol of World Resource Institute (WRI)/World Business Council on Sustainable Development (WBCSD): Nearly all GHG accounting guidelines, including ISO 14064 and PAS 2050 of BSI (2008), are based on this protocol. The GHG protocol provides separate guidelines for GHG accounting and reporting during the life cycles of products and corporate organizations. ISO 14064 (parts 1 and 2): International Organization for Standardization has developed this standard for determination of boundaries, quantification of GHG emissions, and removal (ISO 2006a, b). Part 1 deals with carbon footprinting of organizations, addressing guidance for the quantification, monitoring, and reporting of GHG emissions. Part 2 deals specifically with well-defined activities and projects.
(b) Publicly Available Specifications-2050 (PAS 2050) of British Standard Institution (BSI): It specifies the requirements for assessing the life cycle GHG emissions of goods and services (BSI 2008). PAS 2050 is preparing a standard method to calculate carbon footprints of agricultural systems as well.
(c) Intergovernmental Panel on ClimateChange (IPCC) guidelines for National Greenhouse Gas inventories: IPCC categorizes all anthropogenic sources of GHG emissions into four sectors—energy, industrial process and product use, agriculture, forestry, and other land use and waste.

All of these guidelines and standards proceed through life cycle assessment (LCA) or 'cradle-to-grave analyses' for the activity for which the carbon footprint is to be calculated. ISO formulated standard methods for conducting LCA as a part of the ISO 14000 series. ISO 14040 provides the principles and framework for carrying out LCA (ISO 2006c), whereas ISO 14044 provides guidelines on detained methodology (ISO 2006d). It also directed the Life Cycle Impact Assessment (LCIA) as the last and compulsory stage of LCA. For effective application of ISO 14044, two technical revisions have been made: ISO 14047

(ISO 2012a) and 14049 (ISO 2012b). They focus on the key points of LCIA that are important for carbon footprinting, with specific examples and sample practices.

To provide guidelines and principles of product carbon footprinting, ISO 14067 is under development (ISO 2013). This technical specification is based on the ISO's standards of LCA and environmental labeling of products. Although there are provisions of different modes of communicating product carbon footprints and performance tracking, it is under critical review and evaluation so that an international standard can be developed.

According to the available standards, following structured framework is suggested for carbon footprinting (WRI/WBCSD 2004; Carbon Trust 2007; BSI 2008):

a. Selection of GHGs
b. Setting boundaries
c. Collection of GHG emission data
d. Footprint calculation

4.2.1 Selection of GHGs

Selection of the set of GHGs covered in the calculation depends on the guideline followed, the need for carbon footprinting, and the type of activity. For example, in a thermal power plant, where CO_2 is a predominant emission and other gases are almost negligibly emitted, only CO_2 emission measurement will be feasible, whereas for a cattle farm, CH_4, CO_2, and N_2O emissions may be significant. Although some studies include only CO_2 emissions in carbon footprinting (Patel 2006; Wiedmann and Minx 2007; Craeynest and Streatfeild 2008), the guidelines recommend all six Kyoto gases (Bokowski et al. 2007; Garg and Dornfeld 2008; Good company 2008; Matthews et al. 2008). All guidance and standards also direct to include all Kyoto gases.

4.2.2 Setting Boundaries

A boundary refers to an imaginary line drawn around the activities that will be used for calculating carbon footprints. It depends on the objective of footprinting and characteristics of the entity for which footprinting will be done. Defining the boundary is crucial as it determines the activities, which will be included in the study. To facilitate convenient accounting, the following tiers have been suggested (WRI/WBCSD 2004; Carbon Trust 2007; BSI 2008):

Tier$_1$: direct, i.e., onsite emissions

Tier$_2$: emissions embodied in purchased energy

Tier$_3$: all indirect emissions not covered under tier$_2$, such as those associated with the transport of purchased and sold goods, business travels, waste disposal, etc. Carbon footprint has also been divided into two parts: basic/primary and full carbon footprint. Primary carbon footprint is calculated from tier$_1$ and tier$_2$ only, whereas full carbon footprint covers emissions up to tier$_3$ (Carbon Trust 2007; Lynas 2007).

Most carbon footprinting studies limit up to tier$_2$ because going beyond tier$_2$ increases the complexity and uncertainty in estimates (Matthews et al., 2008). Even during trading of carbon offsets, only tier$_1$ and tier$_2$ emissions are important. It is also advocated that embodied emissions are beyond the control of the organization of process for which carbon footprints are to be calculated and hence tier$_3$ should be left out during carbon footprinting (Lenzen 2001). For this reason, PAS-2050, GHG protocol, and other registries and consultancies based on these have kept tier$_3$ optional. Critical analyses of carbon footprinting case studies, however, reveal that indirect emissions compose a significant part of total carbon footprint (Matthews et al. 2008). Hence, attempts must be taken to count tier$_3$ emissions. To make the definition of tier$_3$ more clear, Mathews et al. (2008) proposed that emissions exclusively related to delivery, use, and disposal of products also should be kept out of tier$_3$ An additional tier$_4$ can be used for the same.

Advancement in the tracking and management of emissions in the supply chain is expected to promote tier$_3$ accounting (Matthews et al. 2008; CDP 2009). In the Carbon Disclosure Project (CDP), 72 % of respondents among 500 companies reported their basic carbon footprints, but the number of companies reporting up to tier$_3$ is increasing (CDP 2009). As more and more organizations carry out their complete LCA, a database can be developed through which average sector-specific emission factors can be calculated (Matthews et al. 2008; Weidema et al. 2008). International trade of raw materials and finished products poses further challenges in tier$_3$ estimation (Courchene and Allan 2008). Appropriate assumptions over sharing of responsibilities of countries and organization related with emissions associated with international trade of goods and services need to be developed (Peters 2010).

Regarding natural systems and land uses, almost all the carbon footprinting studies focus on emissions; the amount of GHG removal and sequestration appears to be neglected (Peters 2010).

4.2.3 Collection of GHG Data

Estimation of GHG emissions and removals associated with all the activities identified within the boundary can be carried out by direct measurements or estimated using emission factors or models. Direct measurements yield near accurate estimates and are clearly prescribed in globally accepted protocols, but their cost and application may be prohibitive in certain cases (WRI/WBCSD 2004). Under such conditions, indirect estimations through models and emission factors are applicable. If developed or modified specifically for a particular region or sector, they yield fairly accurate results. Usually, customized tools relying on combinations of direct measurements, emission factors, and models are popular and practicable (USCCTP 2009). For large-scale GHG estimation, observation networks such as FLUXNET have been initiated (Sundareshwar et al. 2007), but due to high costs and nonuniform global distribution of sites, they are still far away from global representativeness. To overcome the patchy coverage of ground-based monitoring networks, satellites have been launched to monitor sources and sinks of CO_2 and other GHGs (Haag 2007). A Japanese satellite, (the "greenhouse gas observing satellite") and Vulcan, a joint project of NASA and the US Department of Energy, are two such examples (Gurney et al. 2009; Kelly et al. 2009). Remote sensing and geographic information systems are also used extensively for large and relatively less accessible areas. Such an example is the case of carbon footprinting of Hurricane Katrina on the US coast, carried out by Chambers et al. (2007) using LANDSAT imageries.

4.2.4 Footprint Calculation

The collected GHG data is translated into CO_2-e using global warming potentials of different GHGs as provided by IPCC (2007). The final unit of the carbon footprint depends on the nature of the entity. For individuals and dynamic processes, carbon footprints need to be calculated periodically, but events such as conferences, sports events, etc. have one-time emissions. Some entities have a combination of both; for example, for building, a one-time emission take place during construction phase, while periodic calculations are needed during the operation phase. For such activities, there is a provision of sharing one-time emissions over the operation phase. Natural processes are highly complex; hence, they have a temporally as well as spatially dynamic carbon footprint.

5 Carbon Footprinting as a Tool to Estimate Agriculture's Contribution to Atmospheric Stock of Greenhouse Gases

As discussed, the emissions as well as sink capacity of the agriculture sector are still highly uncertain, and available estimates must be refined through an extensive monitoring network covering different geographical regions, environmental conditions, and management practices (Seip 2011). In addition to soilborne GHGs and carbon sequestration, keeping in mind the increasing energy and chemical inputs in farming, the boundaries of agriculture must be expanded to include all relevant emissions and/or removals of GHGs. Carbon footprinting therefore can be utilized for cultivation systems by producing a detailed map of different sources and sinks of GHGs. This will identify the points where environmental efficiencies can be improved. This also facilitates a comparison of different management options and their environmental cost-benefit analyses. Although scientific literature is still sparse in carbon footprinting studies targeting cultivation practices, such estimates are essential for upgrading national GHG budgets and to improve environmental efficiency of the agriculture sector.

6 Calculating Carbon Footprints for Agricultural Products

The GHG protocol acts as a common resource for carbon footprinting, but it is important to keep in mind the role of the agriculture sector in anthropogenic GHG emissions and sensitivity of this sector to a number of environmental and social factors. Therefore, the development of an agriculture-specific carbon footprinting method is proposed. BSI is developing PAS 2050 specifically for agriculture.

6.1 Selection of Boundary and Tiers

For carbon footprinting of agricultural practices, all activities associated with farming must be identified. A generalized illustration of different activities involved in cultivation practices that are relevant to carbon footprinting is presented in Fig. 1. Because there is no agriculture-specific standard for carbon footprint calculation, the generalized standard three-tier approach of the GHG protocol must be followed in order to maintain uniformity among different studies. The selection of the boundary will depend on the level up to which carbon footprints are to be calculated, as presented in Table 1. For carbon footprinting in the production of cereals, vegetables, fruits, etc., the activities related to cultivation of the concerned crop up to the final harvest and readiness for use as raw material will be covered. To cover the activities up to the shelf of the store, activities related to processing, packaging, and transportation of farm produce must also be included. To calculate carbon footprints of food, the boundary is set to cover home preparations also. Among the three proposed boundaries, carbon footprints of cultivation (i.e., up to the farm gate) is more helpful for comparing different agricultural practices and efficacies of different management systems on GHG performances. Extending the boundary beyond this introduces activities such as the transportation of products to the market, their distribution, and food preparation techniques and preferences, which are more sensitive to local and personal conditions.

Direct emissions to be covered under $tier_1$ for agricultural systems include CH_4, N_2O, and CO_2 emissions from soil and onsite CO_2 emissions from fossil fuel-powered farm machines such as tractors, harvesters, threshers, grain cleaning

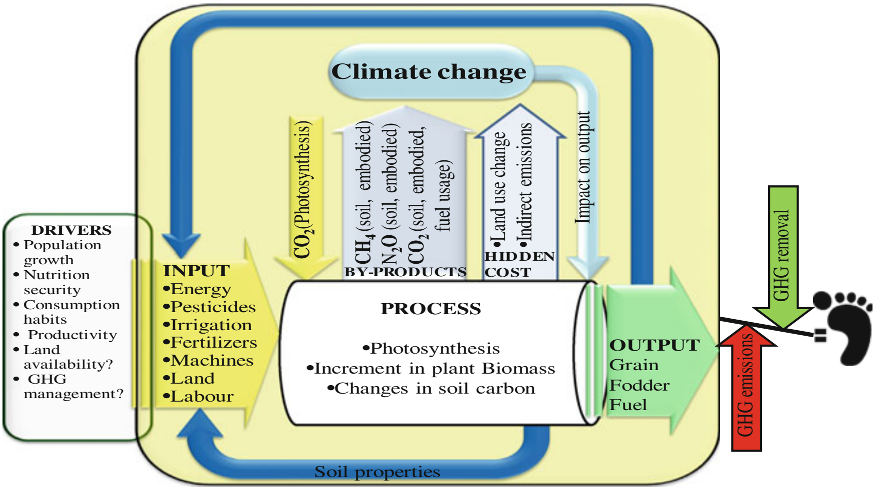

Fig. 1 A generalized illustration of activities and inputs associated with cultivation of a crop to be considered in the boundary (sowing to farm gate) for carbon footprinting

Table 1 Choice of boundaries for carbon footprinting of agricultural products

Objective	Boundary
Carbon footprinting of cultivation	Up to the farm gate
Carbon footprinting of finished farm products	Up to the shelf
Carbon footprinting of food	Up to the table

systems, etc. Electricity use in activities such as irrigation constitutes tier$_2$. Table 2 lists common farm activities and their classification into different tiers. Most of these activities are also performed manually, but human labor is not considered under carbon footprint calculations (WRI/WBCSD 2004). Because agricultural soils can sequester atmospheric CO_2 (Lal 2004b), it is proposed to be a part of tier$_1$.

In addition to these, agricultural inputs such as fertilizers, pesticides, herbicides, and soil conditioners carry embodied emissions, and hence they constitute tier$_3$.

6.2 *Estimation of GHG Emissions/Removals*

Because agricultural practices depend significantly on region, traditional practices, economic conditions of farmers, and the crop under cultivation, emission factors and models developed to express emissions of GHGs need to be validated and refined before their application in a particular agricultural system. Particularly for soilborne GHG emissions, which are also sensitive to environmental conditions, direct measurements are the most reliable (Pandey et al. 2012). For this purpose,

Table 2 Farm activities and their classification into tiers

Activity	Cultivation practices	Energy source	Tier
Land preparation	Plow	Diesel	$Tier_1$
	Harrow	Diesel	$Tier_1$
	Spader	Diesel	$Tier_1$
	Subsoiler	Diesel	$Tier_1$
	Spreader	Diesel	$Tier_1$
Sowing	Seed drill	Diesel	$Tier_1$
	Broadcast	Diesel	$Tier_1$
	Seeders/spreaders	Diesel	$Tier_1$
	Transplanter	Diesel	$Tier_1$
Irrigation	Channel	Electricity	$Tier_2$
	Sprinkler drip	Electricity	$Tier_2$
		Electricity	$Tier_2$
Fertilizer application	Spreader	Diesel	$Tier_1$
	Self-propelled sprayers	Diesel	$Tier_1$
	Agricultural aircrafts	Petroleum spirit	$Tier_1$
Pesticide application	Self-propelled sprayers	Diesel	$Tier_1$
	Agricultural aircrafts	Diesel	
Irrigation	Channel	Diesel/electricity	$Tier_1$/$Tier_2$
	Drip	Electricity	$Tier_2$
	Sprinkler/central pivot	Electricity	$Tier_2$
Harvesting	Harvester (reaper, thresher)	Diesel/electricity	$Tier_2$
Threshing	Thresher	Diesel/electricity	$Tier_2$
Seed processing	Seed processing systems	Diesel/electricity	$Tier_2$

closed chamber systems are a simple, low-cost, and most applied technique (Parkin et al. 2012). There have been different shapes, sizes, and sampling procedures for estimating GHG fluxes using chamber systems depending upon the crop, rates of GHG fluxes, and analyses procedures. The periodically sampled gases are usually analyzed through gas chromatography and infrared gas analyzers; however, with advancements in chamber technique, GHG flux rates can now be measured in situ (Arnold et al. 2001). Automated chambers are capable of carrying out continuous monitoring of GHGs covering diurnal and daily variations under different environmental conditions. However, their cost, maintenance, and electricity requirement prohibits their application in the farmer's field and remote rural areas. The sensitivity of chambers also poses a challenge; in many cases, emission rates are very low and remain below the detection limits of chambers (Parkin et al. 2012). This is particularly important when soil acts as a net sink of GHGs. Also, due to poor chamber sensitivities, low or negative fluxes are often discarded as experimental errors (Chapuis-Lardy et al. 2007). Such low positive or negative fluxes become significant for large cultivated areas. With more refined chamber designs and flux calculation methods, sensitivity has been improved significantly. Flux towers are meant for large farming areas under similar cultivation practices and cropping systems because they provide the cumulative flux of the entire coverage area (Sundareshwar et al. 2007).

Regarding changes in soil carbon, which also constitutes $tier_1$, actual measurement of stock difference in soil over a long period of time is needed. This is because the changes in soil carbon are too slow to be measured reliably over time scales of years (Post et al. 2001). It is observed that changes in soil carbon are a function of management practices, environmental conditions, crop cultivation, and depth of measurement. Another issue related to carbon sequestration measurement is the question of permanence, i.e., for how long the carbon accrued in the soil will stay out of the atmosphere. Therefore, this part is usually left out during GHG accounting, but some studies have shown that carbon sequestration in soils can offset a part of carbon footprint of cultivation systems significantly.

For emissions taking place from farm machines and electricity consumption, emission factors are available for most countries. According to the GHG protocol, an activity data sheet should be maintained, keeping records of different farm activities, fuel consumption, hours of operations, etc. From this activity data sheet, emissions associated with different activities can be calculated using emission factors or models. As a requirement of carrying out GHG emission inventories for countries signatory to the United Nations Framework Convention on Climate Change (UNFCCC), emissions associated with combustion of fossil fuels and electricity generation have to be calculated under their national communications to UNFCCC. These emission factors have been modified according to the types of engines and machines used on the field. However, for $tier_3$ emissions, production technique and country-specific emission factors are not available in most of the countries. In such cases, IPCC (2006) National GHG inventory guidelines provide average and default emission factors for production of fertilizers, GHG emissions from soil under different irrigation, and manure applications etc. Based on these factors, GHG emissions embodied in Urea production have been calculated (Tirado et al. 2010; Lal 2004a) as the most commonly used resource, in which emission factors for CO_2 emissions associated with different activities on farm and inputs of fertilizers and pesticides were derived on the basis of extensive literature survey. Such estimates nevertheless need to be refined and updated. Nelson et al. (2009) also calculated the onsite and offsite CO_2 emissions from different farm activities in USA during 1990–2004. Results indicated that onsite and total CO_2 emissions ranged from 23 to 176 kg C ha^{-1} yr^{-1} and from 91 to 365 kg C ha^{-1} yr^{-1}, respectively. Such region-specific emission factors are necessary for reducing uncertainties in the calculations.

6.3 Footprint Calculation

Global warming potential (GWP) of all the tiers is calculated individually using the conversion factors of IPCC (2007) corresponding to a 100-year time horizon. The formula for the calculation of GWP of $tier_i$ ($i = 1$, 2 or 3) is given by:

$$\mathrm{GWP}\,(\mathrm{tier}_i) = \mathrm{emission/removal\ of\ CH_4} \times 25 + \mathrm{emission/removal\ of\ N_2O} \times 298 + \mathrm{emission/removal\ of\ CO_2}$$

where GWP is in kg CO_2-e ha^{-1}.

Emissions are taken as positive while removal as negative. Values are given in kg ha^{-1}.

Carbon footprint is calculated by adding the GWP of all tiers. The final representation of the carbon footprint of agricultural systems can be made as spatial or yield scaled carbon footprints according to the formulae given below:

$$\mathrm{CF_s} = \sum_{\mathrm{i=1}}^{3} [\mathrm{GWP}(\mathrm{tier}_i])$$

$$\mathrm{CF_y} = \frac{\mathrm{CF_s}}{\mathrm{Grain\ yield}}$$

where CF_s is the spatial carbon footprint. Units are (kg CO_2-e ha^{-1}); CF_y is yield scaled carbon footprint. Units are (kg CO_2-e Kg^{-1} yield).

These that two units differ by the factor of yield, which is the prime motive of cultivation. Spatial carbon footprints are helpful in comparing agricultural practices that are already under high yielding conditions. Under such cases, the better practices emit less per unit area under cultivation without declining the yield. Yield scaled carbon footprints are considered a better indicator for intercomparison of different cropping systems (Linquist et al. 2011, 2012).

7 Case Studies

In the last few years, there has been an increase in number of case studies of carbon footprinting of cultivation systems and food. Some of them considered all three tiers, but none of the studies defined them. Similarly, there was no mention of boundary selection. Table 3 presents different carbon footprinting studies for crop cultivation. Even though it is remarked that GHG emissions from soil are highly sensitive to environmental conditions and management practices, none of the carbon footprinting studies was based on actual measurements. Furthermore, CH_4 emissions were considered only for rice cultivation; for the rest, only N_2O emissions were covered (Gan et al. 2011a, b, c, 2012). In light of many studies demonstrating that crops other than rice act as significant CH_4 sinks or sources, it becomes essential to monitor CH_4 fluxes under such systems. In an attempt to calculate the yield scaled carbon footprint of barley, Gan et al. (2012), calculated that nearly 26 % of the GHG emissions were contributed by farm operations. Gan et al. (2012) however, did not measure CH_4 emissions. In most of the studies, $tier_3$ emissions, particularly of fertilizer application alone, contributed from 45 to 85 % of the total yield scaled carbon footprints (Gan et al. 2011a, b, 2012), whereas

Table 3 Carbon footprints of some agricultural systems and cultivation practices

	Agricultural system (crop)	Region	Activity and tiers	Estimation protocol	Carbon footprint (kg CO_2-e kg^{-1})[a]	Reference
1	Canola-mustard (sowing to farm gate)	Canada	$Tier_1$ and $Tier_2$: soilborne N_2O, land preparation, pesticide and fertilizer spray, harvester $Tier_3$: fertilizers and pesticides (factory to farm)	Rochette et al. (2008); Lal (2004a)	0.548–0.966	Gan et al. (2011a)
2	Durum wheat (sowing to farm gate)	Canada	$Tier_1$ and $Tier_2$: soil borne N_2O, land preparation, pesticide and fertilizer spray, harvester $Tier_3$: fertilizers and pesticides (factory to farm)	Rochette et al. (2008); Lal (2004a)	0.383–0. 533	Gan et al. (2011c)
3	Barley (sowing to farm gate)	Canada	$Tier_1$ and $Tier_2$: soilborne N_2O, land preparation, pesticide and fertilizer spray, harvester $Tier_3$: fertilizers and pesticides (factory to farm)	Rochette et al. (2008); Lal (2004)	0.252–0.456	Gan et al. (2011b)
4	Spring wheat (sowing to farm gate)	Canada	$Tier_1$ and $Tier_2$: soilborne N_2O, land preparation, pesticide and fertilizer spray, harvester, changes in soil C $Tier_3$: fertilizers and pesticides (factory to farm)	Gregorich et al. (2005); Rochette et al. (2008); EF from previous studies	0.357–0.140	Gan et al. (2012)
5	Potato (farm to table)	Sweden	$Tier_1$ and $Tier_2$: soilborne N_2O, soil borne CO_2, changes in soil C Electricity	SEPA (2009); IPCC (2006); EC (2007); Swedenergy, (2009); Andrén et al. (2004)	0.12	Roos et al. (2010)

(continued)

Table 3 (continued)

	Agricultural system (crop)	Region	Activity and tiers	Estimation protocol	Carbon footprint (kg CO_2-e kg^{-1})[a]	Reference
6	Rice (sowing to farm gate)	Italy	Soilborne CH_4 and N_2O	EF from previous studies	2.90	Blengini and Busto (2009)
			Fertilizer, farm machines (land preparation, harvester, grain processing), irrigation, pesticide and fertilizer spray, harvester			
7	Biofuels (ethanol, methyl ester, FT diesel) (Cultivation to power generation)	Europe, Canada, S.E. Asia, Brazil, USA	$Tier_1$ and $Tier_2$: no demarcation between tiers	IPCC (2007); Croezen et al. (2008)	In kg CO_2-e 70-140 MJ^{-1}	Hoefnagels et al. (2010)
			Soil borne CH_4, N_2O, CO_2 Change in soil carbon Power generation $Tier_3$: fertilizers and pesticides (factory to farm)			
8	57 farms with different organic, conventional and integrated farming (sowing to farm gate)	Scotland	$Tier_1$ and $Tier_2$: soilborne N_2O; Land preparation; Harvesting; Fertilizer, pesticide spray	Lal (2004a)	In $\times 10^2$ kg CO_2-e ha^{-1}; Organic farming: 7.49; Conventional farming: 16.06; Integrated farming: 12.38; Leguminous crops: 4.50; Potato: 19.44; Cereals: 11.16–15.69	Hillier et al. (2012)

EF Emission factor
[a] Units of carbon footprints are kg CO_2-e kg^{-1} unless stated otherwise

changes in soil carbon could turn the carbon footprints of wheat cultivation negative, which was otherwise 0.34 kg CO_2-e kg^{-1} (Gan et al. 2012). On basis of this result, PAS 2050 is recommended to include changes in soil carbon in their upcoming guidelines for carbon footprint calculation for agricultural systems.

Practicing different permutations of conventional tillage and no tillage in rice–wheat systems showed that although no tillage led to significant reductions in cumulative GWP of rice cultivation under continuously no-tilled systems, it also reduced the yield; hence the yield scaled GWP was increased, resulting in a higher carbon footprint compared to the conventional practice. On the contrary, during wheat cultivation, the conventional practice acted as a net sink of CH_4, thereby leaving a negative carbon footprint of −8.11 to 125.2 kg CO_2-e kg^{-1}. Under no-tillage practice, emission of GHGs increased along with yield; hence, the carbon footprint became positive (Pandey et al. 2013).

Food as a commodity has independently become an important candidate of carbon footprinting. Kim and Neff (2009) showed that carbon footprint calculators for food items had different scopes and calculations were based on different emission factors. Hence, they could not address effectively the diet-related preferences. Pathak et al. (2010) calculated the carbon footprints of Indian food items, taking into account cultivation of crops, processing, transportation, and kitchen preparations. The average emission factors they used did not address CH_4 emissions from non-rice crops, different management conditions, and changes in soil carbon. They calculated the average daily carbon footprint of 723.7 g CO_2-e for an Indian adult male.

8 Sources of Uncertainty

According to the GHG protocol, sources of uncertainties should be mentioned when reporting carbon footprints. For agricultural practices, nonavailability of activity-specific emission factors is an important source of error. In addition, the associated land use changes and alternative scenarios under different agricultural practices are not easy to predict confidently. For example, in the case of no-tillage cultivation, the stubble left over the soil could have been used as cattle fodder. Loss of fodder under no-tillage practice might put pressure on the cattle rearing; hence, there may be requirements to arrange extra land to compensate for the fodder demand. Land use change to compensate for fodder demand and shifts in yield should be considered. Because agriculture is largely affected by the climate, long-term monitoring and calculation are required to generate better footprint estimates, and how it modulates with changes in different components. Although, soil carbon sequestration is regarded as a ‘win–win strategy,’ there are certain controversies over quantification and assessment of sink capacity reliability (Lehmann 2009). It is argued that sequestration must be able to keep the carbon out of the atmosphere for a relevant time period, conventionally assumed to be at least 100 years. Determination of the degree of permanence of sequestered carbon has not yet been established convincingly; however, scientists have adopted fractionation of the carbon pool physically and chemically (Post et al. 2001; Rovira and Vellejo 2007).

9 Conclusions

Carbon footprinting has appeared as a strong and popular indicator of the GHG intensity of any activity or organization. Due to its important role in raising awareness regarding responsibility toward global warming, scientists and policy makers are trying to use it as a management tool. However, its application over the agricultural sector is still limited. A standard methodology is required to address the emissions associated with soil, carbon sequestration in soil, emissions associated with farm equipments, and other relevant activities. Due to widespread differences in agricultural activities over the world, it is essential to have guidelines on the selection of boundaries. In addition, there is also an immediate need for uniformities in GHG estimation techniques. The lack of sector-and region-specific emission factors for important agricultural inputs add to the uncertainty. The standard method must address how to deal with alternative scenarios and land use changes. The number of carbon footprinting studies of agricultural systems is increasing, but due to widespread differences, their comparison remains difficult. Nevertheless, such studies represent the contribution of cultivation practices in a better way than merely focusing on soilborne GHG emissions, carbon sequestration, or energy intensity individually.

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Methodology for Determining the Carbon Footprint of the Construction of Residential Buildings

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Abstract With the increasing activity in the building sector in the last decade, construction has become a major consumer of natural resources. This resource consumption has been traditionally accounted for through life cycle assessment and similar approaches. In this chapter, a methodology to apply the carbon footprint indicator to a building project is proposed in order to predict the emissions generated by the construction work. The methodology takes into account the resources used and the waste generated. Thus, a number of factors involved in the calculations are first defined, followed by the methodology to determine the carbon footprint for each of the elements into which it is divided (i.e., energy, water, food, mobility, construction materials, and waste). Finally, the methodology is applied to a case study corresponding to the urbanization and building construction of a representative building type in Andalusia (Spain) when the building is in the planning stage.

Keywords Carbon footprint · Emissions · Construction · Building · Resources · Consumption · Waste

1 Introduction

Within the industrial sector, construction activity, including its associated industries, is the largest consumer of natural resources such as timber, minerals, water, and energy. In the European Union, the construction of buildings consumes 40 % of the total consumption of materials, 40 % of primary energy, and generates 40 % of the total waste, making it particularly responsible for the ongoing deterioration of

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the environment due to the expansion of urban land (Baño Nieva and Vigil-Escalera del Pozo 2005). Therefore, in the pursuit of improving the environmental performance of buildings, it is necessary to assess this through indicators, so that the weight of the environmental impacts can be qualified and quantified throughout their life cycle, from the extraction of raw materials to demolition. The tools that analyze these impacts generally follow the methodology of life cycle assessment (LCA) (Zabalza Bribián et al. 2011; Malmqvist and Glaumann 2009).

Although several methodologies of environmental assessment can be applied to the construction sector, such as emergy analysis (Meillaud et al. 2005) and material flow analysis (Sinivuori and Saari 2006), there is a current tendency to use simpler methodologies because they can be more easily understood by society. Among these, the ecological footprint and the carbon footprint constitute the most prominent methodologies.

The EF indicator was introduced by Mathis Wackernagel (Chambers et al. 2004), who measured the EF of humanity and compared it with the carrying capacity of the planet. According to its definition, the EF is the amount of land that would be required to provide the resources (grain, feed, firewood, fish, and urban land) and absorb the emissions (CO_2) of humanity (Wackernagel and Rees 1996; WWF 2008). By comparing the EF to the amount of land available, Wackernagel concluded that human consumption of resources currently stands 50 % above the global carrying capacity (WWF 2010). It is now considered one of the most relevant indicators for the assessment of impacts on the environment and can also be used in conjunction with other indicators, such as the carbon footprint and water footprint (Galli et al. 2012).

The carbon footprint is largely used in the business environment for its utility in energy planning and as a marketing tool. Furthermore, its compatibility with the Kyoto Protocol has provided a major incentive for its application. This indicator measures the total amount of greenhouse gas (GHG) emissions caused directly and indirectly by an individual, event, organization, or product, and is expressed in equivalent units of mass of CO_2 (Weidema et al. 2008). The Kyoto Protocol is considered equivalent to the category of Global Warming Potential of LCA methodologies, and is usually calculated according to the GHG Protocol and PAS 2050 methodologies (Pérez Leal 2012).

Although these indicators suffer from known deficiencies because they represent a simplification of reality that certain researchers consider extreme (van den Bergh and Verbruggen 1999), they still enjoy remarkable reception by society and by political bodies. This success is due, first, to their production of results that remain understandable by non-scientific society, and second, to their ease of application in decision-making and environmental policy (Bare et al. 2000).

This chapter aims to bring all previous knowledge related to the carbon footprint indicator into the residential building sector in order to analyze the phase of construction of buildings, to establish a methodology for calculation, and hence to determine the advantages and disadvantages that this indicator yields in the analysis of environmental impact on the building sector.

In the following sections that introduce the methodological analysis for building construction, a construction project is presented and analyzed in terms of building type, m^2 built, location, and time needed to finish the construction works. Afterwards, the whole methodology is explained using flowcharts for each element (i.e., energy, water, mobility, food, construction materials, and waste generation) and defining the auxiliary data necessary for the calculations. Finally, the results from applying the methodology to the case study are shown and expressed in kg CO_2 eq per year.

2 Case Study

For the determination of which building type to analyze, the most representative types of buildings for the residential sector in Andalusia (Spain) (Mercader 2010; Mercader et al. 2010) were first studied. This study concluded that the predominant residential buildings were two-story semi-detached houses and four-story blocks of flats.

The case study chosen is a residential complex formed by two four-story blocks of flats, being the type that theoretically generates a smaller impact on the area per m^2 built (Holden 2004), although it would be necessary to apply the methodology to various dwelling types in order to compare them.

A building and urbanization project of two purpose-built blocks were studied. Each block contained four floors above ground level and two below ground level, amounting to a total of 107 dwellings, with their parking spaces, storerooms, and shops (Fig. 1). This project was initiated in the province of Huelva (Andalusia, Spain) and was completed in 2008, which is the year to be taken as reference. The total constructed area is shown in Table 1.

In the initial assumptions for the case study, it is considered that the only activity that exerts an impact on the area is that which corresponds to the construction of the residential buildings specified above. This impact will be continued for a period of 12 months, which is the time-span considered necessary for the construction. In the event that the implementation period is longer than a year, then the impact of the building process is assumed to be uniform. For example, consider that the construction lasts 18 months; therefore, during the first year, two-third of the total impact of the construction is produced, and during the second year, the remaining one-third is generated. By the time the analysis is being performed, the project is still in the design phase, and hence certain consumption data (e.g., water consumption, power consumption) remains unavailable.

In Sect. 3, the methodology for calculating the carbon footprint due to a real building construction is explained, accompanied by flowcharts, hypotheses, and formulae. Each item of the methodology (i.e., energy, water, mobility, food, construction materials, and waste generation) is analyzed separately. Finally, the results from applying the methodology to the case study are shown and expressed

Fig. 1 Type of residential building under analysis

Table 1 Constructed area of the two blocks

	Floor area (m^2)	
Constructed area	Block 1	Block 2
Ground floor	1,359.06	1,197.86
First floor	1,359.15	1,197.86
Second floor	1,363.35	1,201.53
Third floor	1,363.35	1,201.53
Total	5,444.91	4,798.78
Total area (m^2)	10,243.69	

in kg CO_2 eq per year. The construction materials are identified as the most important element in the project's carbon footprint.

3 Methodology

In order to calculate the carbon footprint of the construction of buildings, it is necessary to establish the functional unit of the study. Unfortunately, no product category rule for entire buildings has been published yet. Currently, this is under development by the International Committee for Normalization CEN TC 350. Despite this temporal inconvenience, we are convinced that the whole project is the functional unit to be used. This functional unit comprises all the processes from cradle-to-gate, which are consequence of the building (or buildings) under construction and the urbanization required for the treated zone.

The reference unit of the study will be kg CO_2 eq/year/project and kg CO_2 eq/year/m^2. However, this will be expressed as kg CO_2 eq/year throughout the chapter, as all the calculations are referred to the project under study; only in the final results do the project and m^2 factors appear. These have been chosen because they are the most descriptive units in a construction project. The year factor is used

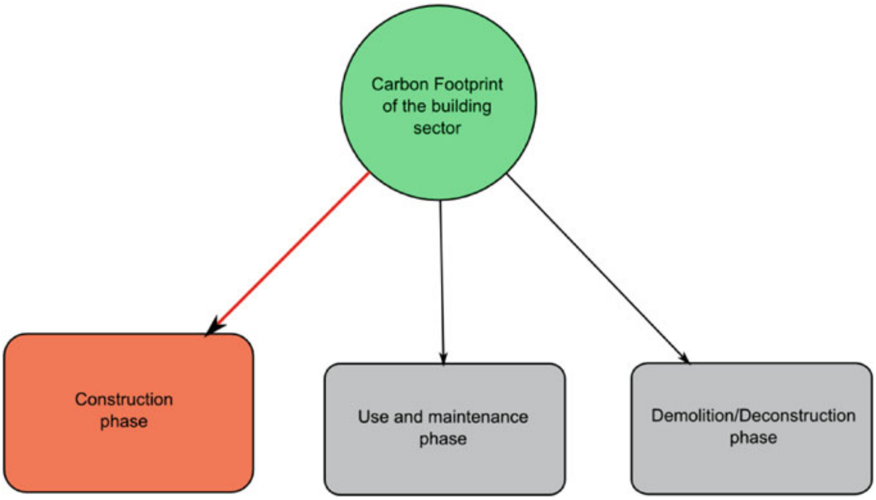

Fig. 2 Boundaries of the study

in order to ease the comparatives between the results and the planet′s CO_2 assimilation capacity.

Although the Total Carbon Footprint is expressed in kg CO_2 eq/year, the work duration factor will not appear in any formula or flowchart because the construction process of our case study lasts one year. If the construction process lasted more (or less) than a year, the total carbon footprint, or the partial carbon footprints instead, should be divided by the number of years.

This carbon footprint assessment focuses on the implementation and construction phase of residential buildings due to the complexity of the calculations; research into the other two phases of the life cycle of buildings, those of use and demolition, is not part of the present analysis (Fig. 2). Thus, the entire project, including two buildings and the corresponding urbanization in this case study, are analyzed following a cradle-to-gate methodology, given that only the construction phase is studied.

These boundaries establish a clear frontier between the three stages of a building′s life cycle. However, some of the impacts included in the methodology might be considered to be part of people′s footprint. Such is the case of food, where it has been decided to include the energy intensity of the various products based on the hypothesis that this is the energy associated with the effort of the workers, and thus it should be taken into account. Also, the mobility of workers to the worksite has been included, because it is considered to be a consequence of the construction process as well. The study follows the methodology described in the flowchart in Fig. 3.

1. Emissions-generating elements. These are the generators of CO_2 (second level of the tree of Fig. 3): direct consumption, indirect consumption, and waste

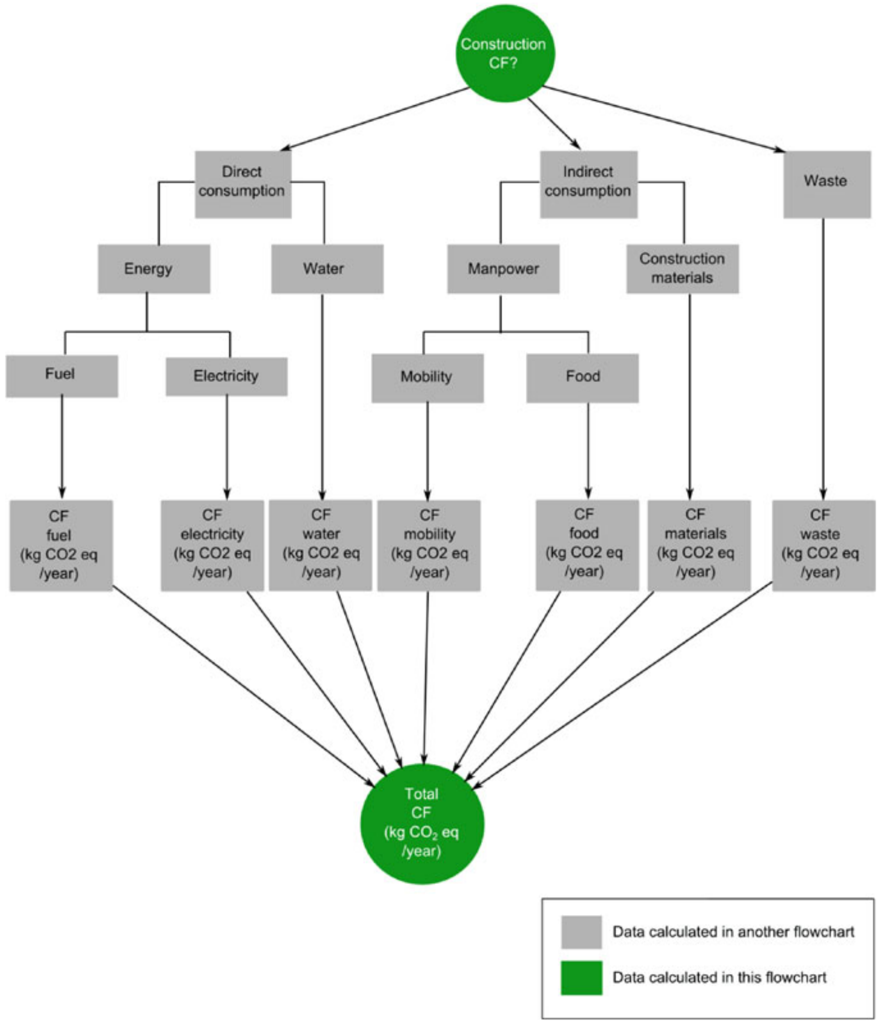

Fig. 3 Methodology flowchart. CF, carbon footprint

generation. Direct consumption is that which causes direct energy expenditure (in the form of fuel or electricity) or water consumption on the construction site. Both are located in the third level of the tree. Indirect consumption is caused by the indirect use of resources, which are in this case:

- Manpower
- Building materials consumption

The manpower in building construction comprises food expenditure by the operators, and the use of fuel for the mobility of the operators (trips to the construction site).

For their part, building materials (listed in Sect. 4.5), through manufacturing processes, transportation, and installation (see Fig. 3), consume fuel (transport of materials to the workplace) and/or energy (necessary for the manufacture of materials and commissioning). For the carbon footprint assessment of material consumption, a quantitative study is performed on the building materials, whose amount is then translated into resources expressible in terms of CO_2 emissions by using the Greenhouse Gas Protocol methodology included in SimaPro 7.3 and GaBi 4 Education. Data for primary energy consumption is also gathered because it will eventually be needed in order to determine the carbon footprint of waste generation and recycling.

LCA databases for building products have their specific limitations, and finding the most suitable data to the project under study is not simple. LCA databases contain data from studies all around the world. The most extended Spanish database (BEDEC) lacks transparency, but at the same time it is better adjusted to the construction model in Spain. Other European databases might not reflect the manufacturing process as it is in Spain. In this study, it has been considered important to use, when possible, transparent data from countries next to the project′s location.

The third factor is the impact of waste generated in the construction phase, which mostly corresponds to the so-called construction and demolition waste (CDW). Therefore, each of the emissions-generating elements uses resources (energy, water, manpower, materials) or generates waste.

2. Intermediate elements (see the key to Fig. 3). Through these elements, consumption is transformed into elements that allow us to define the various footprints that make up the total footprint of the system under study. The intermediate elements are fuel, electricity, mobility, food consumption, and extraction, transport and manufacturing of construction materials. These gray boxes comprise internal calculations developed in several flowcharts corresponding to each intermediate element (Figs. 6, 7, 8, 9, 10 and 11).

3. Partial footprints and the total carbon footprint. By means of the intermediate elements and their implied calculations, the partial footprints that are generated in the construction phase of the buildings projected are obtained. These are located at the bottom level of Fig. 2, represented by gray squares. The result of the addition of all the partial footprints is the total carbon footprint, being all of them expressed in kg CO_2 eq/year.

To apply the above methodology, a budget must be used in accordance with a building cost system. For this analysis, the Andalusian Construction Cost Database (ACCD 2008) is used. This database has been developed over the past 25 years in Andalusia and is the most widespread in this region. Its use is mandatory in public developments in Andalusia. Not only is ACCD valid as an estimation of cost, but it also provides a common method to manage information during the design and construction of buildings (Marrero and Ramirez-de-Arellano 2010). The cost

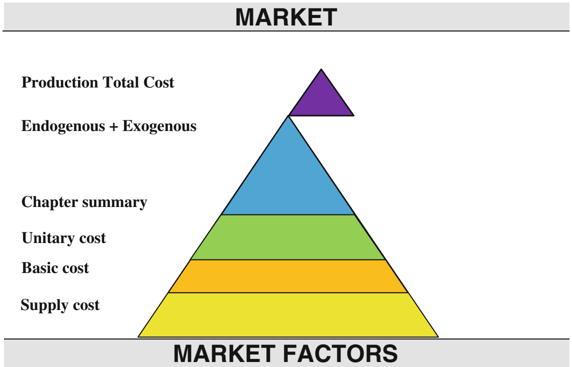

Fig. 4 Pyramidal cost structure (Marrero and Ramirez-de-Arellano 2010)

structure defined distinguishes between direct costs and indirect costs, thereby allowing a clear determination of all costs for each project type. The ACCD structure is arborescent and hierarchical, with clearly defined levels from the apex of the hierarchy down to lower levels, whereby each group is divided into subgroups of similar characteristics (Fig. 4).

For this analysis, the levels used in this structure are (Fig. 5):

1. The production total cost (PTC): covers all production costs incurred by the tasks necessary for the projected work.
2. Basic cost (BC): refers to elements that are a resource: manpower, materials, and machinery.

In our study, all costs are allocated directly, since the indirect costs (IC) are previously analyzed and integrated into the budget in a direct way. Hence, all the costs of the construction process are clearly defined. Therefore, in order to determine the PTC, it is necessary to calculate not only the direct costs of production (PDC) but also the indirect costs of production (PIC) and health and safety costs (HSC), which are usually accounted for separately. Furthermore, in order to make a detailed calculation of the materials, a budget is assumed in accordance with the ACCD for the year 2008, the year taken for this study. Therefore, the procedure for the determination of the total budget is:

1. Obtain the PTC for the construction of the blocks and the urbanization.
2. Recalculate the costs to adjust them to the ACCD (2008).
3. Integrate IC into the PTC.
4. Integrate HSC into the PTC.
5. Calculate the PTC (adjusted to ACCD 2008).

$$PTC = PDC_B + PDC_U + PIC + HSC_B + HSC_U \tag{1}$$

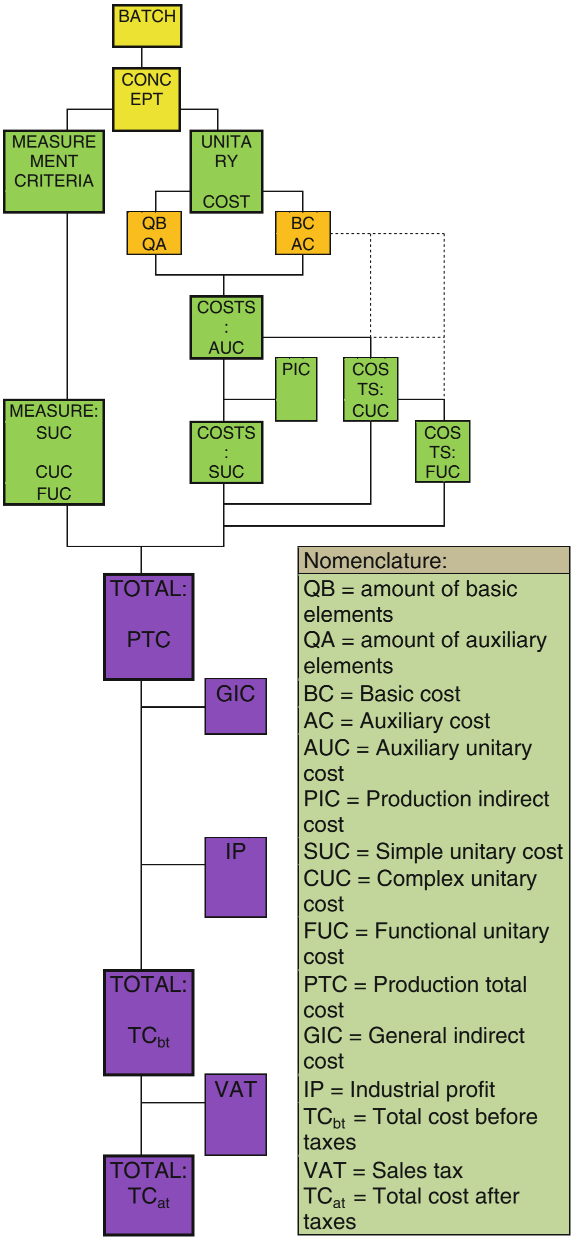

Fig. 5 Budget model (Marrero and Ramirez-de-Arellano 2010)

where

PTC Production Total Cost
PDC_B Building Production Direct Costs
PDC_U Urbanization Production Direct Costs

Table 2 Summary of the overall costs

	Cost (€)
PDC_B	5,067,139.67
PDC_U	187,613.37
PIC	380,726.02
HSC_B	51,867.43
HSC_U	938.07
PTC	5,688,284.55

PIC Production Indirect Costs
HSC_B Building Health and Safety Costs
HSC_U Urbanization Health and Safety Costs.

The overall costs are shown in Table 2.

3.1 Determination of the Emission Factors

Given that a considerable amount of the total carbon footprint will be due to energy consumption in the form of electricity or fuel combustion, an emission factor for each type of consumption is needed.

The emission factor applied for the mobility of workers (E_g) comes expressed in kg CO_2/l, and is as specified in Table 3.

For machinery, due to the higher emissions generated by their engines, the emission factor to be used will be 199.44 kg CO_2/GJ, as it is specified by the International Energy Agency for oil combustion in 2008.

The emission factor for the national energy mix (E_e) is expressed in kilograms of CO_2 per gigajoule (Table 4). The estimates of CO_2 emissions are based on the 1996 IPCC Guidelines and represent the total emissions from fuel combustion. Emissions have been calculated using the IPCC Reference Approach and the IPCC Sectoral Approach. The denominator, total primary energy supply (TPES), is made up of production + imports − exports − international marine bunkers − international aviation bunkers ± stock changes (including biofuels and other nonfossil forms of energy). For our case study, this value is 54.5 kg CO_2/GJ.

Table 3 Fuel consumption and emission coefficients of cars in Spain (IDAE 2011)

Fuel	Consumption (l/100 km)	CO_2 emissions (kg CO_2/l)
Gasoline	7.40	2.35
Gasoil	6.04	2.60

Table 4 CO_2 emissions per total primary energy supply in Spain (2000–2010)

Years	2000	2005	2008	2009	2010
kg CO_2/GJ	55.6	57.1	54.5	52.9	50.2

Source IEA 2012

Table 5 Emission factors used in the present study

Emission factor	Value	Source
Ef (fuel combustion for the mobility of operators)	2.35 kg CO_2/l (gasoline) 2.60 kg CO_2/l (gasoil)	IDAE (2011)
Eo (oil combustion for machinery)	199.44 kg CO_2/GJ	IEA (2012)
Ee (national energy mix)	54.5 kg CO_2/GJ	IEA (2012)

To sum up, the emission factors used in this methodology, except those of construction materials, which are listed in Sect. 4.5, are listed in Table 5.

3.2 Determination of the Carbon Footprint of Energy Consumption

To predict the amount of energy consumed in construction work, data provided by polynomial formulae is used (Spain MP 1970, 1981), which estimates the resources used in the work as a percentage of the total costs for 48 types of construction work (roads, canals, railways, buildings, etc.), both for public and private initiatives (Table 6).

For this case study, type 18 of these formulae is employed: "Those buildings with reinforced concrete structure and facilities that cost less than 20 % of total costs". Furthermore, as the case study is a public development, it is considered a public initiative.

The coefficients in Table 6 represent the percentage of the PTC, which does not include VAT, industrial profit, general costs, and an additional 15 % of IC. In our case, IC are allocated directly; hence, the percentages in Table 6 are increased to obtain 100 % of the costs (PTC), thereby obtaining the corrected coefficients, multiplying by 1.15, which are those used for the calculations.

Each of the initials of Table 6 refers to the following: m: manpower cost; e: energy; c: cement; s: steel; w: wood, and cr: ceramics.

Therefore, in this example, the energy consumption of the work could be estimated as 9 % of PTC.

As a hypothesis, the total energy consumption of the execution of the work is considered to be shared out between electricity and fuel consumption (Fig. 6), because this is a footprint analysis at the project design stage and therefore the consumption cannot be determined. Therefore, once the total energy consumption

Table 6 Polynomial formulae of type 18 (public initiative)

Type	m	e	c	s	w	cr	Total
18	36	8	12	12	7	10	85
18 (corrected)	42	9	14	14	8	12	100

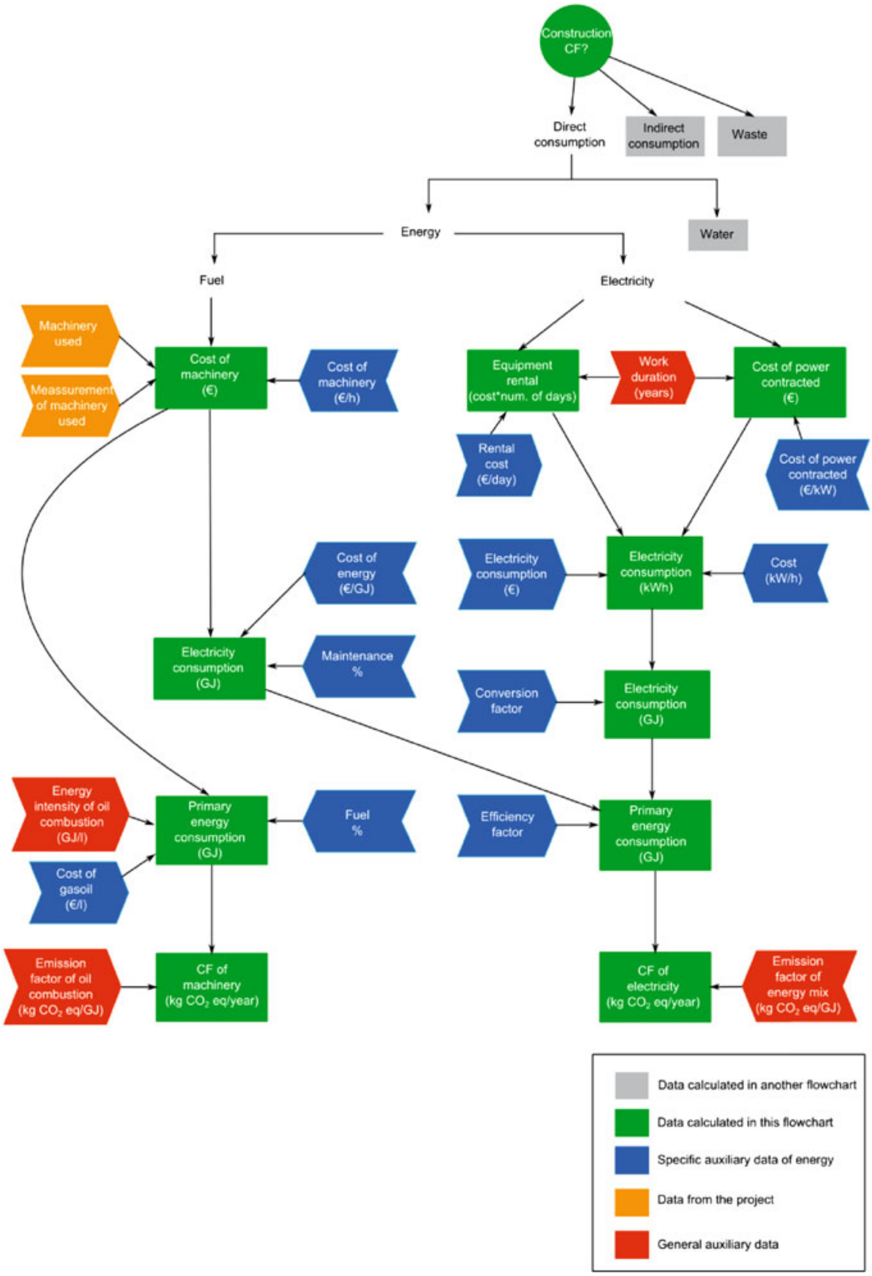

Fig. 6 Flowchart to determine the carbon footprint of energy. CF, carbon footprint

Table 7 Example of calculation of machinery consumption associated with PDC_B

	Hours	Cost (€/h)	Cost (€)	Maintenance (€) (15 %)	Fuel (€) (5 %)
Loader	272.29	23.87	6,499.56	974.93	324.98
Dump truck	1,298.44	25.60	33,240.06	4,986.01	1,662.00
Backhoe	40.93	34.98	1,431.73	214.76	71.59
Bulldozer	0.74	30.30	22.42	3.36	1.12
Vibratory roller	178.00	23.28	4,143.84	621.58	207.19
Manual mechanical tamper	311.09	3.01	936.38	140.46	46.82

and fuel consumption are determined, then the difference between these quantities can be considered to be the electricity consumption.

Once the energy cost of work construction (in Euros) is defined, the next step is to determine the fuel consumption in the work, which is due to the use of machinery. First, the calculation is carried out through measurements of the project, of the hours of machinery used, and then the economic cost of the machinery used can be calculated (Table 7). As concluded in previous studies (Sánchez-de-Mora 2012), approximately 15 % of the total cost of machinery is spent on maintenance (which is supposed to be the only activity comprised in the cost that uses primary energy generated according to the national energy mix), and another 5 % corresponds to fuel consumption. Therefore, the cost of maintenance is assimilated into energy consumption through the cost of electricity, and the cost of fuel consumption is transformed into volume of fuel by means of the average cost of gasoline, which in 2008 (the year of the project) were 1.1233 €/l for gasoline and 1.1414 €/l for gasoil, and 0.092834 €/kWh or 25.787 €/GJ for electricity.

The carbon footprint of fuel consumption can be therefore expressed as:

$$CF_f = (TC_F/C_F) * El_0 * E_f \tag{2}$$

where

CF_f	Carbon footprint of fuel consumption (kg CO_2 eq)
TC_F	Total cost of fuel consumption (€)
C_F	Cost of fuel (€/l)
EI_o	Energy intensity of oil combustion (GJ/l)
E_f	Emission factor of fuel (kg CO_2 eq/GJ)

According to ASTM D-3588-98 (2011), the density of oil at 15 °C is 0.560 kg/l, and the energy intensity of its combustion is 11.250 kcal/kg as a mean value. This results in an EI_o of 0.0252 GJ/l after converting units.

Once the fuel consumption has been determined, the electricity consumption in the construction work can be calculated. To express this data in energy consumption units, the billing model of electricity in Andalusia is used. After obtaining this information, it becomes necessary to determine the electric mix in the project location.

The emission factors obtained in Sect. 3.1 and the efficiency factor for electricity production, which is assumed to be 0.3 (IDAE 2011), are then considered.

The formula used is:

$$CF_e = (TC_E / C_E) * E_{ef} * E_e \tag{3}$$

where

CF_e Carbon footprint of energy consumption (kg CO_2 eq)
TC_E Total cost of energy consumption (€)
C_E Cost of energy (€/GJ)
E_{ef} Efficiency factor for electricity production (1 GJ/0.3 GJ)
E_e Emission factor of energy mix (kg CO_2 eq/GJ).

3.3 Determination of the Carbon Footprint of Water Consumption

The carbon footprint of water consumption is determined assuming water needs a certain quantity of energy to be carried to dwellings. Therefore, the emissions associated to this energy consumption are calculated.

In order to determine the consumption of water for the construction process, the water footprint methodology might be a good option (Hoekstra and Hung 2002). This model is based on the virtual water concept (Allan 1998) and is defined as the total volume of water employed to produce the goods and services consumed by society. In this methodology, water accounts include the withdrawal of water from rivers, lakes, and aquifers (blue water) as well as water from rainfall (green water) that is used in growing crops (Giljum et al. 2011).

However, this methodology is hard to apply for the determination of the consumption of water in our case; hence, it is estimated by comparing to similar examples and then interpolating.

The procedure, shown in Fig. 7, is:

1. Determine the ranges of water consumption and the ranges of costs in work of similar dimensions to that analyzed so that the ratio of the cost of the work to water consumption can be established.
2. Define the average water consumption of the work analyzed, by interpolating with the data obtained in the previous section. Interpolation is based on the TPC.
3. Determine the carbon footprint. This is defined by the calculation procedure that considers the energy needed to bring water to the dwellings, which according to EMASESA (2005) is 0.44 kWh/m^3 or 0.001584 GJ/m^3, employed to conduct water to the dwellings, for drinking water, and treatment of waste water.

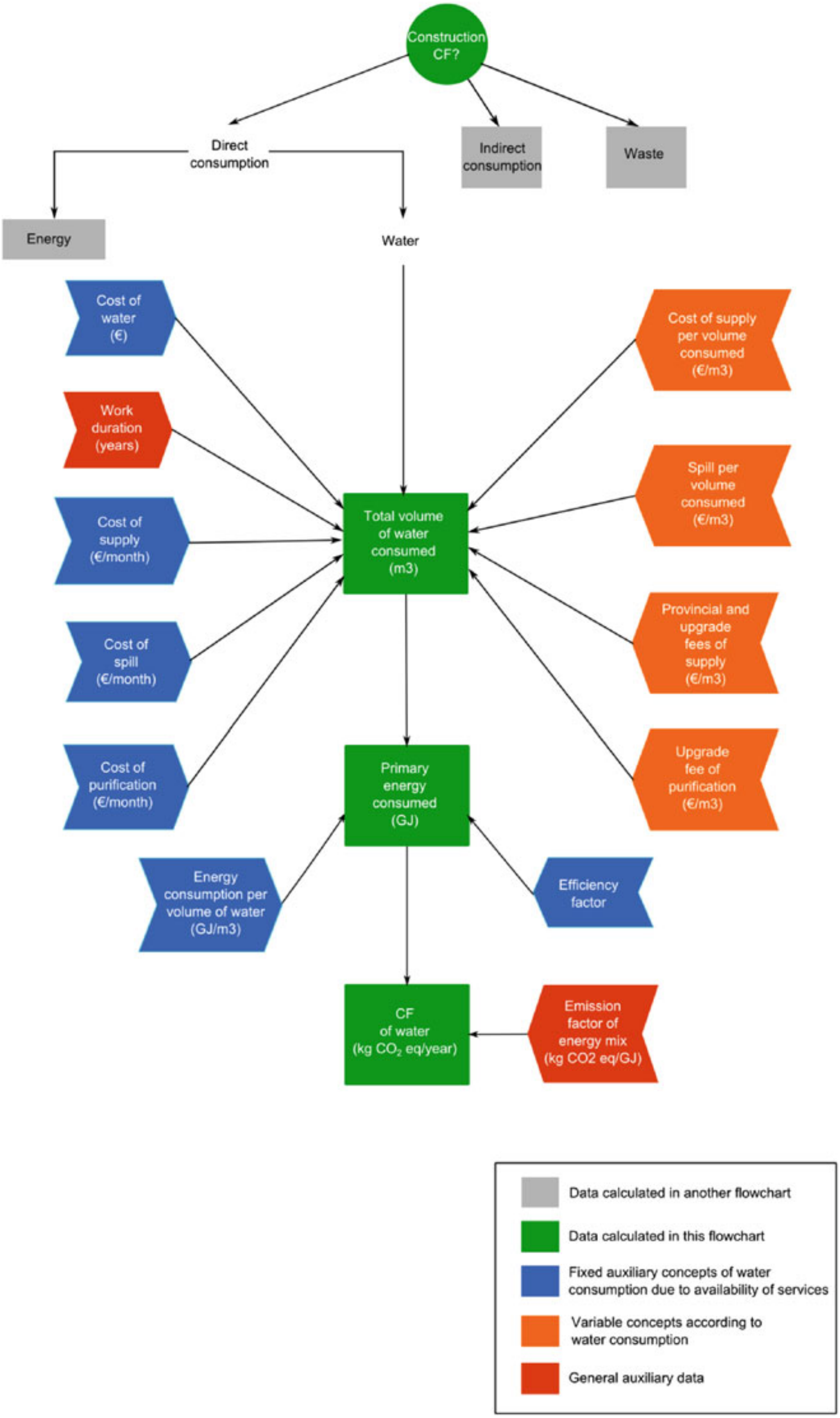

Fig. 7 Flowchart to determine the carbon footprint of water consumption. CF, carbon footprint

Therefore, the formula employed for the calculation of the carbon footprint of water consumption is:

$$CF_w = W * E_w * E_{ef} * E_e \tag{4}$$

where

CF_w	Carbon footprint of water consumption (kg CO_2 eq)
W	Water consumption (m^3)
E_w	Energy consumption per volume of water consumed (GJ/m^3)
E_{ef}	Efficiency factor for electricity production (1 GJ/0.3 GJ)
E_e	Emission factor of energy mix (kg CO_2 eq/GJ)

3.4 Determination of the Carbon Footprint of Food Consumption

The initial hypothesis of this section is that workers′ food is attributed to the carbon footprint of the building construction because this activity takes place on the worksite, in the same way as in the methodology developed by Solís-Guzmán et al. (2013) where business meals are allocated to the ecological footprint of building construction.

To this end, the total number of manpower hours for the entire work must first be calculated, which is obtained by measuring the project. Such manpower is broken down with ACCD Systematic Classification (ACCD 2008). This classification also gives the economic cost of the manpower (€/h).

The footprint is calculated using the expression:

$$CF_{fd} = CF_{me} * (N_h/h_{me}) \tag{5}$$

where

CF_{fd}	Carbon footprint of food consumption (kg CO_2 eq)
CF_{me}	Carbon footprint per meal (kg CO_2 eq/meal)
h_{me}	8 h/meal (one meal per working day is assumed)
N_h	Total number of hours worked

Therefore, it is necessary to obtain the carbon footprint of the various types of food that make up the daily meal of every worker. This carbon footprint is generated due to their required processing, or, as in the case of fish, this factor represents the fuel consumed for the capture of the fish. This translates into CO_2 emissions with the formula:

$$CF_{me} = C * EI * E_e \tag{6}$$

Table 8 Parameters for the calculation of the food footprint (Domenech Quesada 2007)

Foods	F_i %	C_i (t/1,000 €)	EI_i(GJ/t)
Meat	25	0.65	80
Fish	25	0.50	100
Cereals	12	4.69	15
Beverages	10	0.34	7
Vegetables	8	1.45	10
Sweets	6	0.70	15
Oil	5	0.71	40
Dairy	5	0.93	37
Coffee	4	0.54	75

where

CF_{me} Carbon footprint per meal (kg CO_2 eq/meal)
C Food consumption (t/meal)
EI Energy intensity (GJ/t)
E_e Emission factor of energy mix (kg CO_2 eq/GJ)

If we develop this expression:

$$C * EI = C_{me}/1000 * \sum (F_i\%/100) * C_i * EI_i \tag{7}$$

where each of the factors considered would be:
C_{me} cost per meal (assumed at a cost of 10 € per meal)
F_i % Percentage of the meal cost that each type of food represents (Table 8)
C_i Consumption in tons per 1,000 € (Table 8)
EI_i Energy intensities (Table 8)

The whole process to determine the carbon footprint of food consumption is shown in Fig. 8.

3.5 Determination of the Carbon Footprint of Mobility

In order to determine the carbon footprint related to the mobility of workers (Fig. 9), the following assumptions are made:

1. Private vehicles are established as the only means of transport, because it is assumed that the construction work is placed in a remote area away from the city center.
2. The average distance traveled by the vehicles is established. It assumes an average distance of 15–30 km.
3. The average vehicle occupancy is 1.2 people per vehicle (IDAE 2011). In order to determine the number of workers, the total number of hours worked must be

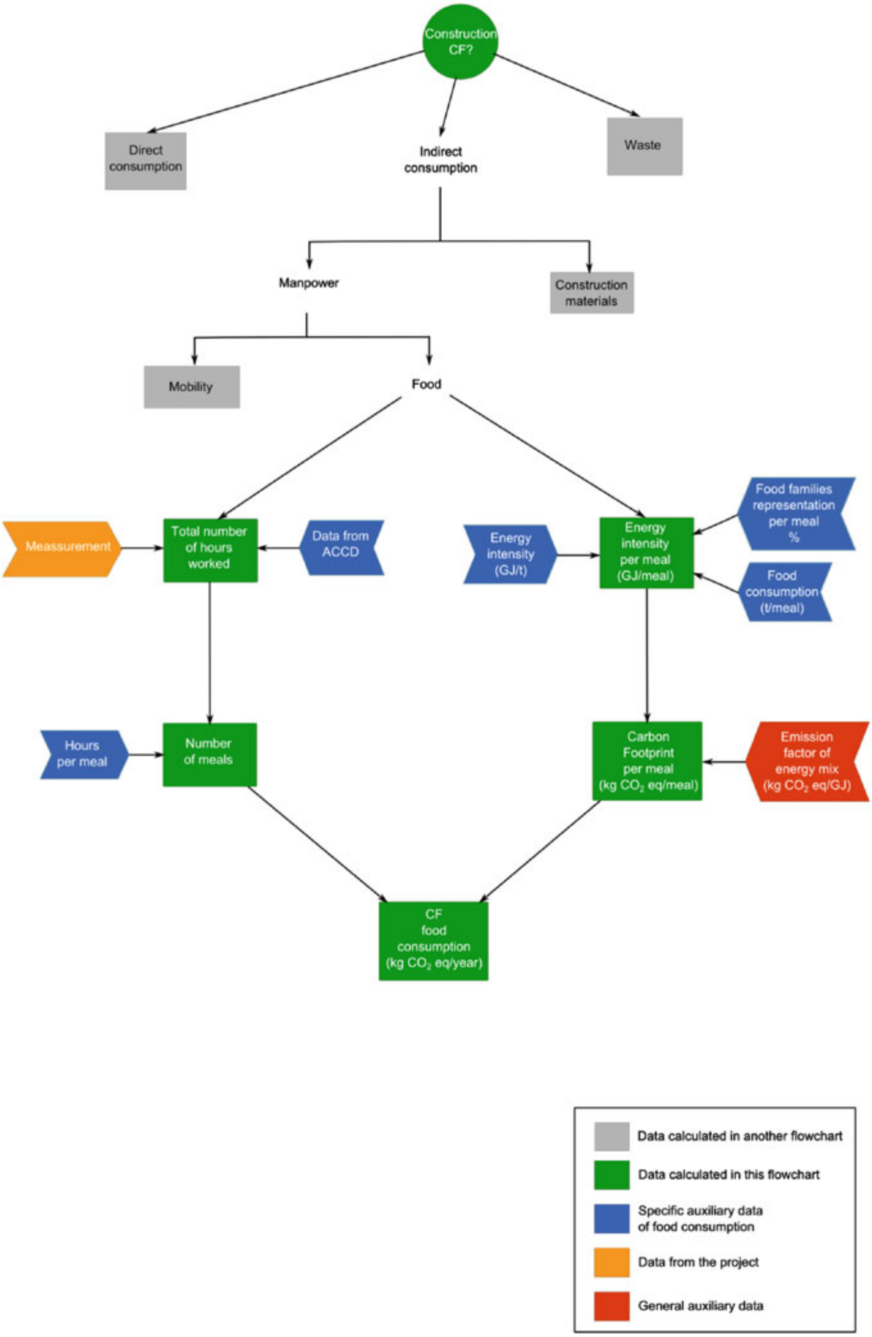

Fig. 8 Methodology for determining the carbon footprint of food consumption. CF, carbon footprint

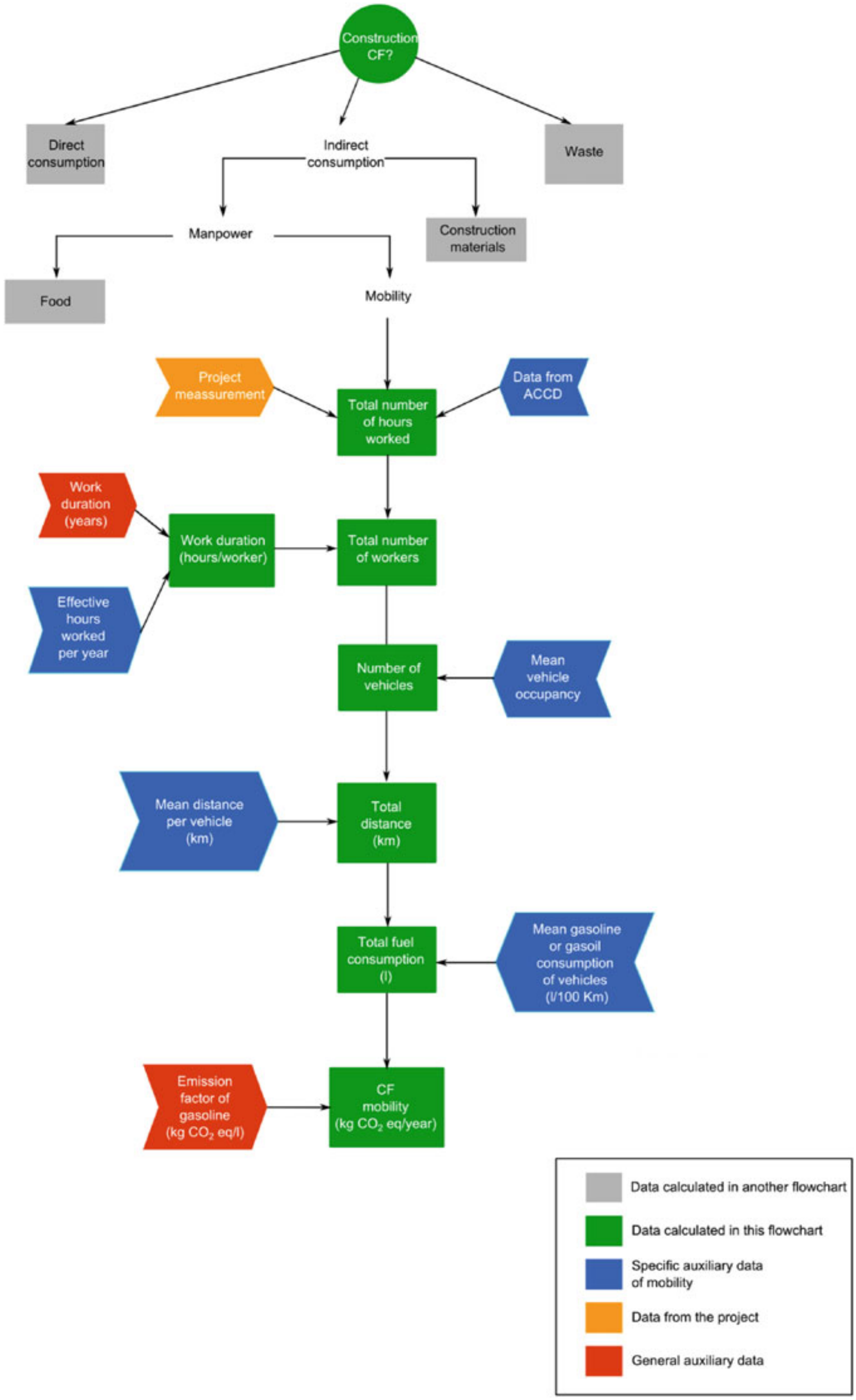

Fig. 9 Carbon footprint of mobility. CF, carbon footprint

known (calculated in the previous section on food), as well as the effective duration of the work (in hours). Both items can be obtained from the ACCD (ACCD 2008).
4. For the calculation of the fuel consumption, consumption coefficients of cars in Spain (IDAE 2011), shown in Table 3, are applied.
5. The mobility footprint is determined by following the procedure in the energy section.

3.6 Determination of the Carbon Footprint of Construction Materials

The footprint of construction materials (Fig. 10) is determined using the following expression:

$$CF_m = \sum Cm_i * Em_i \tag{8}$$

where

CF_m Carbon footprint of construction materials (kg CO_2 eq)
Cm_i Material consumption (kg)
Em_i Emission factor of material i (kg CO_2 eq/kg).

The emission factor values were obtained from various databases, (ITeC 2013; ELCD 2013a; PlasticsEurope 2013; Ecoinvent Centre 2013), by taking the most suitable values according to the origin of the data, its transparency, and comprehensiveness (Martínez-Rocamora 2012). The data for CO_2 emissions is calculated by applying the GHG Protocol methodology. These emission factors are retrieved for a batch of 32 construction materials, which represent 91.81 % of the total embodied energy of materials in this case study. The remaining materials are converted into carbon footprint through their embodied energy and the emission factor calculated in Sect. 3.1 for the national energy mix.

Based on these values, the consumption of materials (by weight) is determined through measurements of the project studied. Basic costs (BC) of the ACCD (2008) are used (see Fig. 10). In order to convert units of measurement of BC (m, m^2, m^3, etc.) into weight, the coefficients calculated by Mercader (2010) are used (Table 9).

The example shown in Table 9 corresponds to the study of the construction of our building project and features a number of the most representative materials of the work from a quantitative point of view. The grouping of BC is based on representative materials or those whose information of CO_2 emissions is available. The second column of Table 9 shows the unit in which the BC is measured. The remaining columns represent:

M_{mi} Measurement of the basic cost of the material i of the project concerned
BC_{mi} Basic cost of the material i (according to ACCD 2008)
TC_{mi} Total cost of the construction material i (€)

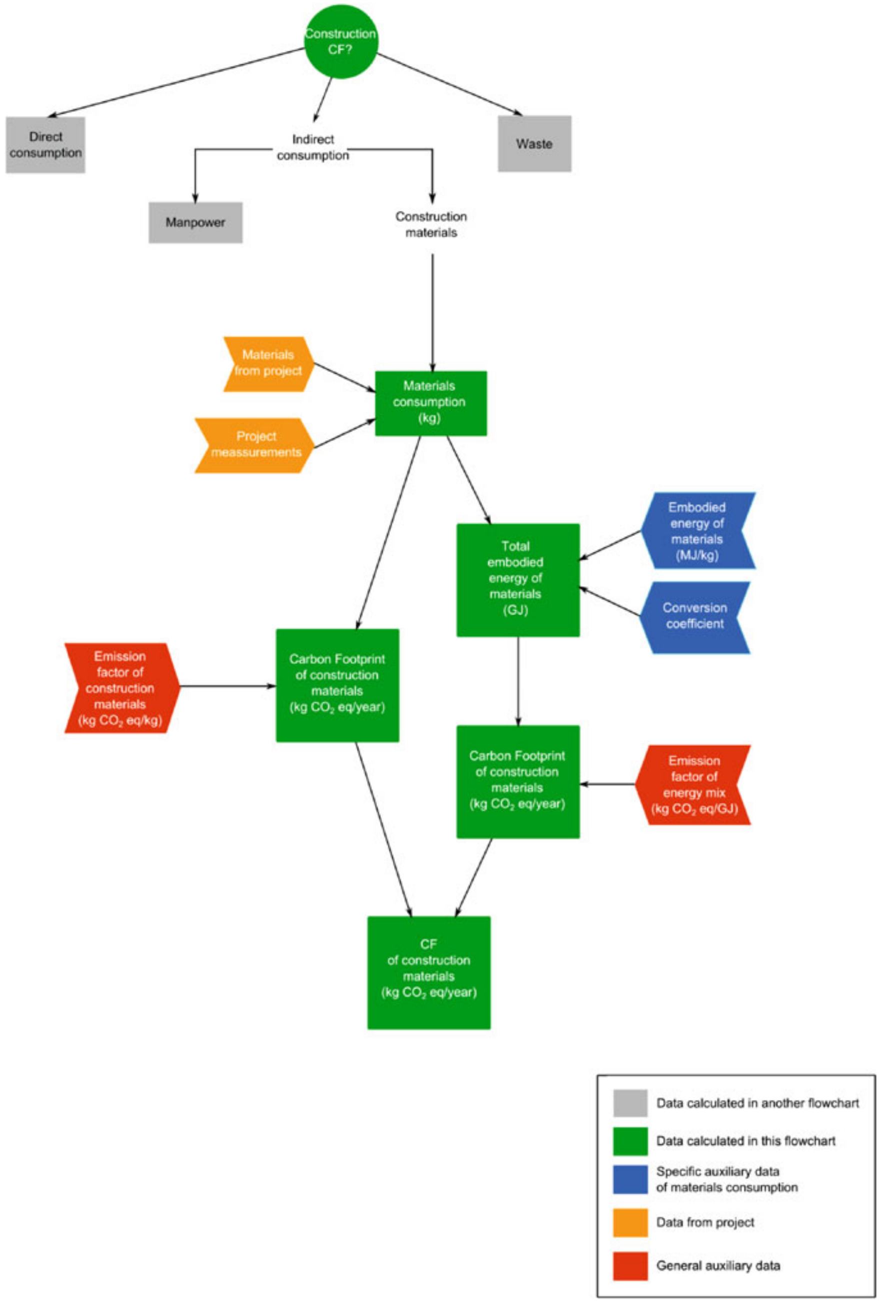

Fig. 10 Methodology for determining the carbon footprint of construction materials. CF, carbon footprint

Table 9 Examples of the calculation of carbon footprint of the most representative materials in the case study

	u	M_{mi} (u)	M_{mbi}(u)	BC_{mi} (€/u)	TC_{mi} (€)	C_{ci} (kg/u)	Cm_i (kg)	Em_i (kg CO_2 eq/kg)	CFm_i (kg CO_2 eq/year)
Steel B 500S	kg	234,915.31	223,728.87	0.77	180,884.79	1.00	223,728.87	1.26	281,898.38
Concrete HA25/B/40	m^3	1,271.37	1,234.34	69.32	88,131.37	2,500.00	3,085,849.51	0.1013	312,596.55
Brick 24/11.5/9 cm	mu	239.61	226.05	98.28	23,548.87	1,550.00	350,373.11	0.219	76,731.71
Gypsum board 13 mm thickness	m^2	21,253.45	20,241.38	4.55	96,703.20	10.00	202,413.81	0.36	72,868.97
Cement II/AL 32.5 N	t	173.07	164.83	92.54	16,015.81	1,000.00	164,827.65	0.899	148,180.06
Coated aluminum sliding door	m^2	327.60	327.60	69.60	22,800.96	20.00	6,552.00	2.39	15,659.28

$$TC_{mi} = M_{mi} * BC_{mi} \tag{9}$$

M_{mbi} Measurement of the material i, which is integrated into the building. It relates to M_{mi} through a loss coefficient, which takes into account those materials that are not integrated into the building.

C_{ci} Conversion coefficient of the unit measure of the basic cost into weight (kg). For this purpose, those coefficients calculated by Mercader (2010) are used.

Cm_i Consumption of the material i (kg)

$$Cm_i = M_{mbi} * C_{ci} \tag{10}$$

Em_i Emission factor of the material i. Em_i values come from the sources referenced above.

EEm_i Embodied energy of the material i (GJ).

E_e Emission factor of energy mix (kg CO_2 eq/GJ).

CFm_i Carbon Footprint of the material i (kg CO_2 eq)

$$CFm_i = Cm_i * Em_i \tag{11}$$

or

$$CFm_i = Cm_i * EEm_i * E_e \tag{12}$$

By performing a similar analysis with all the materials measured in the design project, the consumption and the carbon footprint of the materials are obtained.

3.7 Determination of the Carbon Footprint of Waste

The types of waste generated throughout the life cycle of a building are varied in content and origin. By focusing on the construction phase of the building, one must consider, on one hand, the municipal solid waste (MSW) generated in the workplace, and second, the construction and demolition waste (CDW) generated during this phase. Municipal solid waste can be broken down into four types: organic matter, paper/cardboard, plastics, and glass. In the case of the construction and demolition waste, two types of waste are considered in accordance with the management models that exist in the CDW treatment plants in Andalusia: excavated earth and mixed CDW. Mixed CDW groups the remains of materials generated during the execution of the work unit and the packaging used in the transport of the materials. In new construction work, excavated earth may represent over 80 % of CDW, while the mixed CDW is distributed among the remains of materials and packaging (Solís-Guzmán et al. 2009).

The procedure is based on the energy intensity (EI) of the production of the material from which the waste is made (embodied energy data collected in Sect. 3.6), with a deduction of the percentage of energy that can be recovered by recycling. Some of the waste is organic, excavated earth, or mixed CDW. The carbon footprint of waste is calculated by using the formula,

$$CF_x = \sum G_i * E_e, \tag{13}$$

where each of these terms is:

CF_x Carbon footprint of waste

G_i Waste generation (t)

EI_{xi} Energy intensity of the production of the material from which the waste is made (GJ/t). For these values, the energy intensities of the materials to be recycled must be known. The data is summarized in Table 10. Although it is known that there is no direct correspondence between embodied energy and GHG emissions, and given that we have no data source for emission savings when recycling the various waste, we are forced to use the energy intensity data and convert it into emissions using the emission factor of energy mix obtained in Sect. 3.1.

$\%R_{xi}$ Recycling rate of waste i. In the case of organic waste, nationwide information (OSE 2008) is used, by determining the percentage given in Table 10 (13 %) for composted organic waste. For the other flows, (paper, plastic, and glass), data from the Regional Government (Andalusia ME 2009) on recycling rates in Andalusia is used. For excavated earth, 50 % reuse on site and 80 % recycling on treatment plant is estimated, although all material can be recycled. For mixed CDW, a recycling rate of 15 % (GERD 2009) is considered, which is well below the national and European objectives.

$\%SE_{xi}$ Percentage of energy saved by recycling

E_e Emission factor of energy consumption (kg CO_2 eq/GJ).

In short, the procedure shown in Fig. 11 is as follows:

1. Determination of the generation of MSW and CDW. These calculations are either based on statistical data (Spain ME 2001; Andalusia ME 2009) or on a software tool (Ramirez-de-Arellano Agudo et al. 2008; Solís-Guzmán et al. 2009).
2. Calculation of the carbon footprint of the waste.

In Sect. 4, this methodology is applied to the case study described in Sect. 2. Each individual carbon footprint (i.e., energy, water, food, mobility, construction

Table 10 Parameters for the calculation of conversion rates

	Organic	Paper	Plastic	Glass	Earth	Mixed CDW
EI_x (GJ/t)	20	30	43.75	20	0.10	5
$\%R_x$	13	50	40	40	80	15
$\%SE_x$	100	50	70	40	90	90

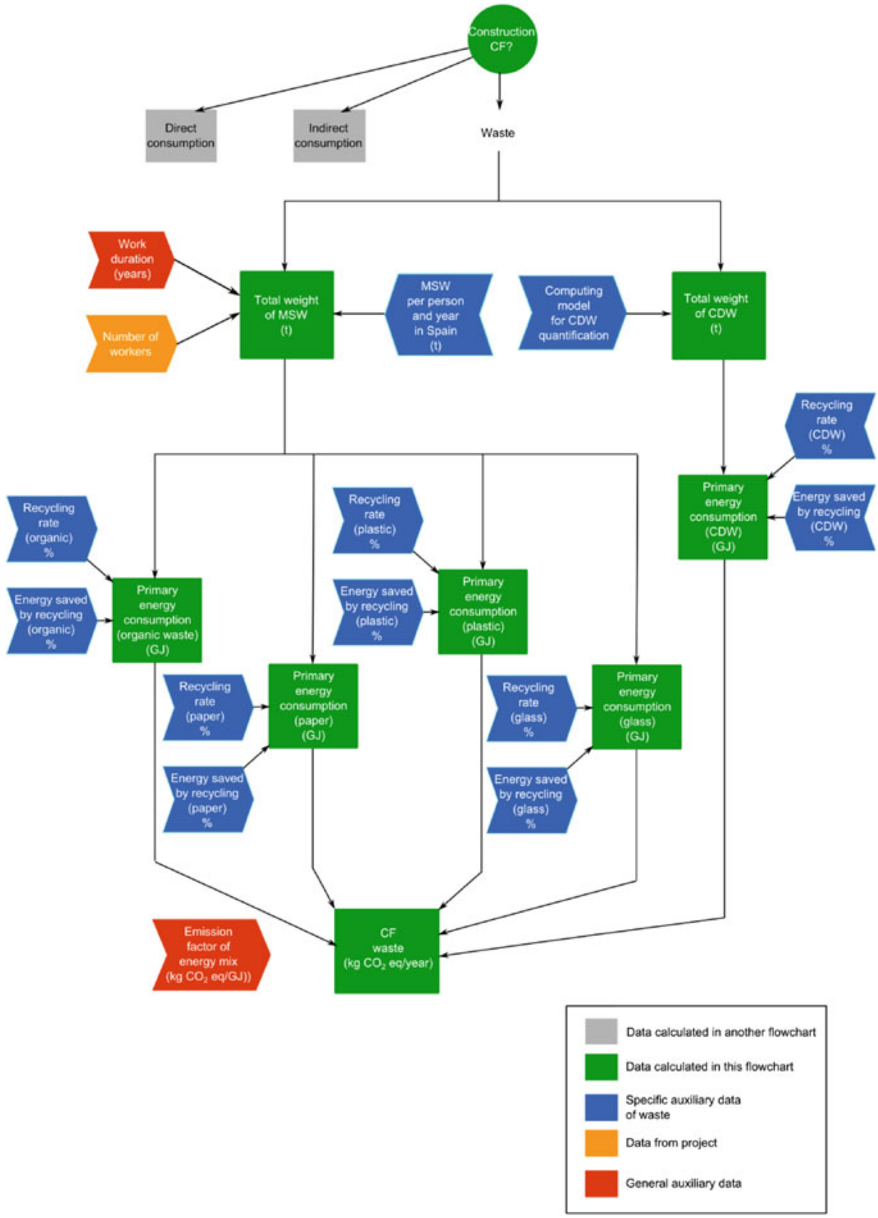

Fig. 11 Flowchart to calculate the carbon footprint of waste. CF, carbon footprint

materials, and waste) is calculated, and finally they are all summed up in a total carbon footprint of the whole construction process of the buildings included in the project under study.

4 Results

4.1 Carbon Footprint of Energy Consumption

The cost of machinery fuel and maintenance and its corresponding energy consumption is determined by the project quantities and costs. The results appear in Table 11.

By means of polynomial formulae (Table 6), the percentage of overall costs that correspond to the energy consumption is computed. Because the energy consumption of machinery is already calculated, the difference between energy consumption and fuel consumption is therefore the electricity consumption.

The electricity consumption in GJ is obtained from the billing model used by the electricity supplier. In order to obtain the electricity footprint, it is necessary to determine the source of electricity in Spain (IEA 2012). The results appear in Table 12.

4.2 Carbon Footprint of Water Consumption

The results of consumption are 2,599.48 m^3 of water, thereby resulting in a carbon footprint of water consumption of 748.03 kg CO_2 eq (Table 13).

Table 11 Overall costs and carbon footprint of machinery

Machinery	Cost (€)	Primary energy onsumption (GJ)	Carbon Footprint (kg CO_2 eq/year)
Building	167,708.63		
Urbanization	16,588.42		
Indirect costs	71,152.76		
Total cost	255,449.81		
15 % (maintenance)	38,317.47	4,953.07 (energy mix)	269,942.13
5 % (fuel)	12,772.49	281.99 (fuel)	56,240.67

Table 12 Carbon footprint of electricity

Energy total cost (€)	511,945.61
Electricity total cost (excluding machinery maintenance and fuel) (€)	460,855,65
Price of electricity (€/GJ)	25.787
Electricity consumption (GJ)	17,871.63
Efficiency factor	0.30
Primary energy consumption (GJ)	59,572.09
Emission factor for energy mix (kg CO_2 eq/GJ)	54.5
Carbon footprint of electricity consumption (kg CO_2 eq/year)	3,246,678.94

Table 13 Carbon footprint of water consumption

Water total consumption (m^3)	2,599.48
Energy consumption per volume of water consumed (GJ/m^3)	0.001584
Energy consumption (GJ)	4.1176
Efficiency factor	0.30
Primary energy consumption (GJ)	13.725
Emission factor for energy mix (kg CO_2 eq/GJ)	54.5
Carbon footprint of water consumption (kg CO_2 eq/year)	748.03

Table 14 Total cost of manpower

Task	Manpower hours	Cost (€)
Building	98,686.05	1,470,946.35
Urbanization	4,280.57	62,590.07
Building health and safety	604.46	8,752.26
Urbanization health and safety	10.93	158.29
Indirect costs	15,836.82	264,474.95
Total	119,418.84	1,806,921.92

4.3 *Carbon Footprint of Food Consumption*

First, the total number of manpower hours worked for the entire project is calculated, obtained by measuring the project. Such manpower is broken down according to the ACCD Systematic Classification (ACCD 2008). The manpower costs (€/h) are also obtained in this classification. The results appear in Table 14.

The primary energy from the different foods that make up the daily meals of the workers is then obtained by using the data in Table 8. The results are shown in Table 15.

4.4 *Carbon Footprint of Mobility*

Following the guidelines outlined in Sect. 3.5, the carbon footprint of mobility is obtained as expressed in Table 16.

Table 15 Carbon footprint of food consumption

Total number of hours worked (h)	119,418.84
Hours per meal	8
Number of meals	14,927.355
Energy intensity per meal (GJ/meal)	0.407305
Emission factor for energy mix (kg CO_2 eq/GJ)	54.5
Carbon footprint per meal (kg CO_2 eq/meal)	22.198
Total carbon footprint of food consumption (kg CO_2 eq/year)	331,359.25

Table 16 Carbon footprint of mobility

Total number of hours worked (h)	119,418.84
Hours per worker in a year (h/worker)	1,533
Number of workers	77.90
Mean vehicle occupancy (workers/vehicle)	1.2
Number of vehicles	65
Distance per vehicle (km)	30
Total distance (km)	1,950
Gasoline consumption (l/100 km)	7.4
Emission factor (kg CO_2/l)	2.35
Total carbon footprint of mobility (kg CO_2 eq/year)	339.105

4.5 Carbon Footprint of Construction Materials

In a previous study, considerable differences in the data for embodied energy and GHG emissions of construction materials from the various LCA databases were detected (over 60 % in some cases), as can be observed in Fig. 12. These discrepancies were mostly due to the use of different flowcharts and methodologies and distinct recycling rates; however, the sensitivity of the model to changes of LCA databases is proved (Martínez-Rocamora 2012).

As mentioned in Sect. 3.6, the emission factors are retrieved for a batch of 32 construction materials which represent 91.81 % of the total embodied energy of materials in this case study. The remaining construction materials are converted into carbon footprint through their embodied energy and the emission factor calculated in Sect. 3.1 for the national energy mix. The results are shown in Table 17.

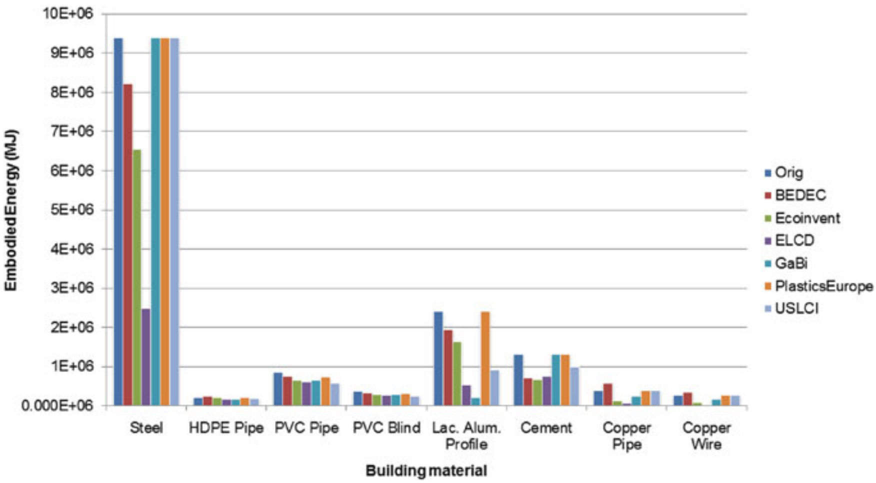

Fig. 12 Comparative analysis of the embodied energy of 8 construction materials from various LCA databases

Table 17 Carbon footprint of construction materials

Carbon footprint (91.81 %) (kg CO_2 eq/year)	6,463,263.60
Embodied energy (8.19 %) (GJ)	6,816.52
Emission factor of energy mix (kg CO_2 eq/GJ)	54.5
Carbon footprint (8.19 %) (kg CO_2 eq/year)	371,500.34
Total carbon footprint of construction materials (kg CO_2 eq/year)	6,834,763.94

The individual contribution of each construction material to the carbon footprint, sorted by quantity, is shown in Table 18.

4.6 Carbon Footprint of Waste

The generation of MSW and CDW are determined through statistical databases and tools. Conversion rates are calculated using the methodology proposed. In the case of CDW, a software tool enabled the result of 22,400 m^3 of excavated earth (of which 50 % is reused) and 1,920 m^3 of mixed CDW to be obtained. The results are shown in Table 19.

4.7 Total Carbon Footprint

The total CF of the whole construction process of the two buildings projected and the urbanization of the area is 11,250,501.85 kg CO_2 eq. Tables 19 and 20 show the overall results, expressed in kg CO_2 eq/year/project and kg CO_2 eq/year/m^2, respectively. In Table 21, the constructed area considered is that of blocks, not the built land. Therefore, the data used is 10,243.69 m^2 (Table 1).

Moreover, a sensitivity analysis should be performed to observe the behavior of the variables. For example, two models of CDW management are compared. In the first scenario, the excavated soil is not reused and the waste is neither separated nor recycled, and therefore the carbon footprint is 596,448 kg CO_2 eq. In a second scenario, 50 % of the excavated soil is reused and the remaining 50 % goes to a treatment plant, which recycles 80 % out of it. Other types of CDW are 15 % recycled, as in Sect. 4.6. The resulting carbon footprint is 473,077.44 kg CO_2 eq in this second scenario. We therefore conclude that the indicator is sensitive to changes in its variables.

Due to the complexity of the building process, with numerous elements involved (water and energy supply, machines, workers from different professional sectors, waste generation and recycling, and building materials among others), it is not easy to establish solid boundaries and not to trespass them and include impacts belonging to people′s or other sectors′ carbon footprints. In fact, other similar

Table 18 Contribution of each construction material to the total carbon footprint of construction materials

Construction material	Emission factor (kg CO_2 eq/kg)	Carbon footprint (kg CO_2 eq/year)	Source database	Source study
Mortar binder	13.73	2,060,886.18	BEDEC	–
Concrete	0.098	1,408,600.96	Base Carbone	SNBPE (2012)
Adhesive paste	13.73	1,032,737.51	BEDEC	–
Bricks	0.219	296,333.89	Ecoinvent	Kellenberger et al. (2007)
Steel	1.03	241,962.78	ELCD	WSA (2011)
Asphalt	0.25	199,027.91	BEDEC	–
Terrazzo	0.22	170,725.85	BEDEC	–
Cement	0.899	169,682.31	ELCD	ELCD (2013b)
Plasterboard	0.36	158,428.34	BEDEC	–
Painting	2.95	150,179.99	BEDEC	–
Tar-epoxy	7.09	108,819.31	BEDEC	–
Modified bitumen	6.67	95,132.88	BEDEC	–
Cement tiles	0.18	86,301.07	BEDEC	–
Tiles	0.57	57,397.60	BEDEC	–
Brass	4.66	52,394.99	GaBi	PE International (2013a)
PVC	3.24	47,384.25	PlasticsEurope Eco-profiles	Ostermayer and Giegrich (2006)
Aluminum	2.39	28,656.51	ELCD	EAA (2013)
Gravel	0.00335	26,282.99	ELCD	ELCD (2013c)
Gypsum	0.108	20,259.36	ELCD	ELCD (2013d)
Copper	2.933	18,932.72	Base Carbone	NIES (I2013)
Crushed stone	0.008	15,691.48	BEDEC	–
HDPE	2.50	6,576.28	PlasticsEurope Eco-profiles	Boustead (2005)
Sand	0.00242	5,119.77	ELCD	ELCD (2013e)
Polyester resin	4.46	3,870.03	GaBi	PE International (2013b)
Bentonite	0.01	886.26	BEDEC	–
Methacrylate	15.00	873.15	BEDEC	–
EPS	3.39	118.75	PlasticsEurope Eco-profiles	Boustead (2006)
Rubber pavement	0.000215	0.47	BEDEC	–
Rest of materials		371,500.34	–	–
TOTAL		6,834,763.94		

Table 19 Carbon footprint of waste

	Organic	Paper	Plastic	Glass	Earth	Mixed CDW	
G (t)	17.71	8.45	4.43	2.83	13,440	1,920	
EI_x (GJ/t)	20	30	43.75	20	0.10	5	
$\%R_x$	13	50	40	40	80	15	
$\%SE_x$	100	50	70	40	90	90	
Carbon footprint (kg CO_2 eq/year)	16,794.18	10,361.81	7,605.20	2,591.15	20,509.44	452,568	
Total carbon footprint of waste (kg CO_2 eq/year)							510,429.78

Table 20 Carbon footprint per year

	Carbon footprint (kg CO_2 eq/year/project)
Energy	3,572,861.74
Water	748.03
Food	331,359.25
Mobility	339.11
Materials	6,834,763.94
Waste	510,429.78
TOTAL	11,250,501.85

Table 21 Carbon footprint per year per m^2

	Carbon Footprint (kg CO_2 eq/year/m^2)
Energy	348.79
Water	0.07
Food	32.35
Mobility	0.03
Materials	667.22
Waste	49.83
TOTAL	1,098.29

approaches considered less elements of the construction process in order to avoid double accounting (see Bastianoni et al. 2007).

Also as explained in Sect. 3, LCA databases for building products have their own limitations, and finding the most suitable data to the project under study is not simple. LCA databases contain data from studies all around the world, and here it has been considered important to use data from countries next to the project′s location. Also, the most extended Spanish database (i.e., BEDEC) lacks transparency, and other European databases might not reflect the manufacturing process as it is in Spain, which limits the calculation′s precision.

5 Conclusions

1. Footprint studies are primarily focused on an urban scale, thereby making it difficult to extrapolate information to the scale of individual buildings. Furthermore, the definition of the measurement units of the indicator for buildings is complicated due to the peculiarities of construction activity. Moreover, the dependence of analysis on charts and graphs necessitates a periodic review thereof.
2. An in-depth study into the innovative aspects of research is necessary, such as research into the impacts caused by water consumption, the study of the embodied energy and GHG emissions of building materials, and that of waste generation.
3. The difficulty of establishing the overall costs of a project as adjusted to a standard cost base, in this case ACCD, is evident because most construction companies often have their own cost databases. Furthermore, for the calculation of the overall costs, it has become necessary to determine the direct costs and indirect costs in full, with the subsequent difficulty of integrating these costs into the methodology of calculation of the indicator.
4. The inclusion of the time factor has been shown to be critical because it determines hypothesis testing throughout the entire methodology. Furthermore, the assumption of carbon footprint per year as the calculation unit allows for a greater generalization of results.
5. The effect of consumption of construction materials is highly significant. For this type of activity, mobility carries no decisive impact. Other sources leading to the carbon footprint are machinery, electricity, and food. Finally, the footprint of water usage has little appreciable effect in this study. All these results require further review toward the improvement of the current model (Ostermayer and Giegrich 2006).

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Carbon Footprint of Food Products

Elin Röös, Cecilia Sundberg and Per-Anders Hansson

Abstract The food system has been identified as one of the major contributors to climate change. The main sources of greenhouse gas emissions are nitrous oxide (N_2O) from soils, methane (CH_4) from enteric fermentation in animals, and carbon dioxide (CO_2) from land use change, such as deforestation. Emissions also arise from manure management, mineral fertilizer production, rice cultivation, and energy use on farms and from post-farm activities such as processing, packaging, storage, distribution, and waste management. With increasing awareness of climate change, calculating the carbon footprint (CF) of food products has become increasingly popular among researchers and companies wanting to determine the impact of their products on global warming and/or to communicate the CF of their products to consumers. This chapter discusses issues that are especially relevant when calculating the CF of food products, such as the choice of functional unit, which is challenging owing to the multifunctionality of food. Other issues concern how to include emissions arising from indirect land use change and removal of CO_2 from the atmosphere by carbon sequestration in soils into CF calculations. Causes of the large uncertainties associated with calculating the CF of food products and ways to handle this uncertainty are also discussed and examples of uses and results of CF of food products are presented. Despite the large uncertainties, it is clear that the differences in CF between different types of food products are very large. In general, the CF of livestock-based products are much larger than those of plant-based products. CF information on food products may be useful in business-to-business communication, for professionals in the retail sector and in public procurement.

Keywords Functional unit · Land use change · Methane · Nitrous oxide · Uncertainty

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1 Introduction

The food chain in the Western world is highly industrialized. Manpower has been replaced by mechanical power, fueled by fossil energy. As a consequence of farm industrialization and production of mineral fertilizers using fossil natural gas, the amount of food produced has greatly increased, making nutritionally rich food products available to large populations. However, this 'green revolution' is placing serious stress on ecosystems. Agriculture is resource-intense, requiring large amounts of land, water, and finite resources such as fossil fuels and phosphorus. Losses of nitrogen from the agricultural system contribute to global warming, and to eutrophication and acidification in surrounding ecosystems. Biodiversity is threatened by the use of pesticides and in some areas by the rationalization and intensification of agriculture, which is causing the disappearance of the traditional mosaic agricultural landscape that is home to a number of red-listed species. Expansion of agricultural land into natural forests, scrubland, and savannah in other areas poses a serious threat to many endangered species. Post-farm stages in the food chain include processing, packaging, transport, storage, food preparation, and waste management, all requiring the use of energy.

The food system has been identified as one of the major contributors to climate change (EC 2006). In Sweden, it is estimated that approximately 25 % of greenhouse gas (GHG) emissions from private consumption are related to the activity of eating (SEPA 2008). In the European Union (EU), the corresponding figure is estimated to be 31 % (EC 2006), while EU member states have reported values in the range 15–28 % (Garnett 2011). Unlike GHG emissions from the energy and transport sector, the emissions from agriculture are not dominated by carbon dioxide (CO_2) from fossil fuel combustion, but by methane (CH_4) and nitrous oxide (N_2O). The CO_2 from land use change, such as deforestation driven by the demand for agricultural goods, is also a major contributor to GHG emissions from the food system (Figs. 1, 2).

CH_4 emissions arise predominantly from enteric fermentation in ruminants, but also from manure management and rice cultivation. Enteric fermentation is the

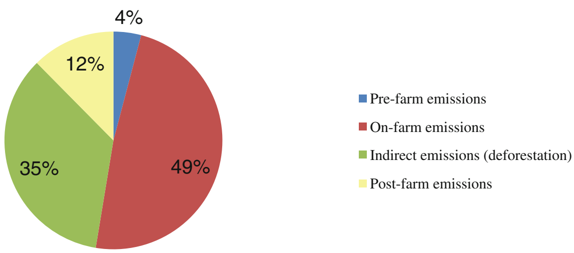


Fig. 1 The contribution of emissions of greenhouse gases in the food system (CCAFS 2013). On-farm emissions are further subdivided in Fig. 2. Pre-farm emissions are dominated by emissions from fertilizer production. Post-farm emissions include refrigeration, storage, packaging, transport, retail activities, production, waste disposal, etc.

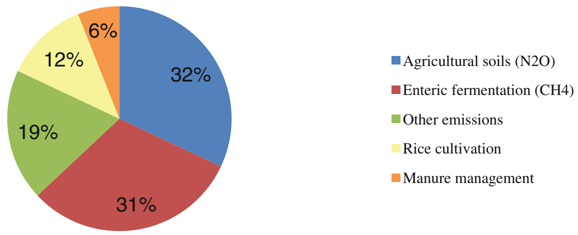


Fig. 2 Contribution of global on-farm (direct) emissions of greenhouse gases from agriculture (CCAFS 2013)

process by which the carbohydrates in the feed are broken down by microorganisms in the rumen, a multi-chambered stomach of animals such as cattle and sheep, into simple molecules that can be taken up by the blood. This enables ruminants to digest feed rich in cellulose, such as grass and straw, which is not possible for humans or other monogastric (single-stomach) animals, such as pigs and poultry. However, CH_4 is formed in the rumen as a by-product of the fermentation process and released to the atmosphere with the exhaled air. CH_4 has a 25-fold higher global warming potential (GWP) than CO_2 in a 100-year perspective and 72-fold higher GWP in a 20-year perspective (IPCC 2007). N_2O is an even more potent GHG, with 298-fold higher GWP than CO_2 in a 100-year perspective. N_2O is formed through nitrification and denitrification, two biological processes that are naturally occurring in all soils to a greater or lesser extent, depending primarily on the amount of available nitrogen, but also on soil carbon and water content, pH, and temperature. Due to the high rates of mineral and organic fertilizers applied to agricultural soils, N_2O emission from soils is a major contributor to climate change within food production. N_2O is also formed and released during the production of mineral fertilizers and during storage of manure in aerobic conditions (Smith et al. 2007). Global emissions of GHG from agriculture are illustrated in Fig. 2.

With the increased awareness of climate change, spurred by the release of the Fourth Assessment Report of the IPCC in 2007, calculating the carbon footprint (CF) of food and other products has become increasingly popular. The CF is now being calculated by companies wanting to learn more about the impact of their products on global warming and/or to communicate the CF of their products to consumers (e.g., Tesco 2012; Lantmännen 2013; MAX 2013). Research into the CF of food products has also increased in recent years. It has included studies of the methodological complexities in calculating the CF of food and calculations of the actual CF of different food products and foods from different production systems (Roy et al. 2009; de Vries and de Boer 2010; Nijdam et al. 2012; Röös et al. 2013).

This chapter examines some issues that are especially relevant when calculating the CF of food products, with the emphasis on the causes and consequences of uncertainty. The functions of food are multiple: it can provide different types of nutrients and/or pleasure, act as a status marker, or form part of cultural traditions. Therefore, choosing the functional unit is not easy as discussed in Sect. 2.1. Challenges as regards drawing system boundaries and allocating climate impacts arise in all CF calculations and some examples of how these challenges can be handled in food CF calculations are given in Sect. 2.2. Food production requires large areas of land, so direct and indirect effects on carbon stocks and emissions arising from land use change (e.g., deforestation) and carbon sequestration are discussed in Sects. 2.3 and 2.4. The risk of pollution swapping when focusing on one environmental aspect only, e.g., the impact on climate change, is discussed in Sect. 2.5. Section 3 provides a more lengthy discussion of different types of uncertainties and variations when assessing the CF of food products, while Sect. 4 gives some examples of uses and results. Some conclusions are given in Sect. 5 while the chapter ends by highlighting some future challenges and research needs in Sect. 6.

2 Challenges in Calculating the Carbon Footprint of Food Products

The CF describes the amount of GHG emissions that a particular product or service will cause during its lifetime, typically expressed in CO_2 equivalents (CO_2e) and including emissions of CO_2, CH_4, and N_2O. A CF can be seen as a subset of a life cycle assessment (LCA) in which only the climate change impact category is studied. LCA is a standardized method for quantifying the environmental impact caused during production, use, and waste management of a product or service.

2.1 Functional Unit

Food has several functions, but the functional unit (the reference unit used in the calculations) most commonly used in LCA and CF studies on food products is based on mass (e.g., 1 kg of the food product being studied) (Schau and Fet 2008). A reference to the system boundaries (Sect. 2.2) is sometimes included, so the functional unit can be, for example, "*the production of 1 kg of tomatoes at the farm gate.*" As in all LCA studies, it is important that the functional unit is chosen so that products can be compared fairly. LCA studies on milk, for example, commonly use a metric that accounts for differences in nutrient content, such as Energy Corrected Milk (ECM), which includes a measure of the fat and protein content in the milk (Sjaunja et al. 1990).

The basic function of food is to provide nutrients. Because the nutrient content varies between food products, the function of food is different for different foodstuffs. For example, most would agree that comparing 1 kg of tomatoes with 1 kg of meat is not a fair comparison, because the meat provides different nutrients than the tomatoes and considerably more energy. Fundamentally, different foods such as meat and vegetables could be compared more fairly using a nutritional index that includes a number of nutrients, such as proteins, carbohydrates, fats, vitamins, and minerals, and weighing these together, such as according to their recommended daily intake (Kernebeek et al. 2012; Saarinen 2012). For comparing food products that are more similar in nutrient content, a simpler functional unit could be used. For example, in the Western diet, livestock products are important sources of protein, so "*the production of 1 kg of protein*" has been used as a functional unit for comparing different livestock products and other protein-rich food products suitable for use as alternative protein sources (e.g., Nijdam et al. 2012).

Another important function of food is to provide pleasure. Food is also an important part of many cultural celebrations and can act as marker of status and class (Guthman 2003). In the affluent world with its abundance of food, these other functions of food might be more important than the pure nutritional aspect for certain products and in certain situations. Dutilh and Kramer (2000) include the emotional value of food in the functional unit to account for this aspect. Additionally, for populations in which obesity is a major health threat, the most nutrient-dense food products should perhaps not be valued highest. Rather, food products providing as little energy and as much pleasure as possible may be demanded (Tillman 2010).

An illustrative example of the importance of reflecting on the function of food concerns beverages, which can have many functions; to provide nutrients, intoxication, water, or just to wash down food. Smedman et al. (2010) argued the need to include the nutritional aspect when comparing milk with other beverages such as orange juice, soy drink, beer, etc., and developed an index (Nutrient Density to Climate Impact) which included 21 essential nutrients and the GHG emissions from a life cycle perspective from the production of the beverage. According to this index, milk was the most beneficial beverage of all those included in the study, although milk had a CF per unit mass (kg) that far exceeded that of orange juice, mineral water, soy, and oat drink. It is questionable whether using this nutritional index is relevant in all contexts. Obviously, if the beverage is to be consumed in areas with protein and micronutrient deficiency, it is highly relevant to include nutritional aspects in the functional unit. However, in areas with overconsumption of most nutrients, it could be argued that the function of milk as a beverage is rather to wash down food and provide water, in which case a more appropriate functional unit could be 1 kg or 1 L of beverage when comparing milk with other beverages.

Although the production of food is usually the prime function for keeping livestock, animals can have other important functions too. One function could be to avoid afforestation in order to preserve land for future agricultural production. In some areas, grazing livestock is kept to help preserve biodiversity by providing a grazing pressure that keeps highly competitive grasses short so that other plants can flourish. From the farmer's perspective, income is the key function of keeping livestock (Nemecek and Gaillard, 2010). In developing countries, livestock may supply various functions such as draft power, soil management, and financial insurance.

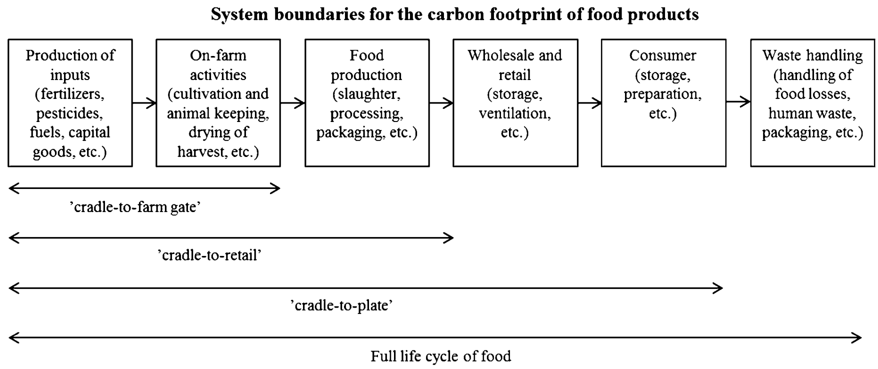


Fig. 3 Different ways of setting the system boundaries commonly used when calculating the carbon footprint of food products

2.2 System Boundaries and Allocation

In LCA and CF calculations, the system under study needs to be isolated from the natural system and the surrounding technical system. In the strictest sense, a CF study should include all emissions from the entire life cycle (Fig. 3). However, emissions beyond the point at which food ends up on the plate are very rarely included in CF calculations on food products (Munoz et al. 2008). Even 'cradle-to-plate' studies tend to be scarce, with the majority of studies being 'cradle-to-retail' or 'cradle-to-farm gate' (Schau and Fet 2008). The reasons for ending at retail are two-fold. First, the emissions from transporting the food products to the household, i.e., by foot, bicycle, car, or public transport, and from preparing the food (e.g., by either eating the food raw, frying or cooking it), as well as the amount of food wasted can vary greatly, therefore making it difficult to include these phases in a representative way. Second, although producers can somewhat influence (e.g., the energy needed when preparing the food), these post-retail steps are primarily controlled by the consumer, so it is justifiable to end the analysis before these consumer-oriented phases when looking for, for example, mitigation options for the producer. There are also several reasons why many studies end at the farm gate. One is that post-farm emissions have been shown to be small, and/or highly variable, for example, depending on transport distance, compared with emissions arising at the farm or from up-stream processes, especially for livestock products (Schau and Fet 2008). In addition, in some cases, such as if the purpose of the study is to evaluate, for example, different feeding strategies for pigs or different ways of heating a greenhouse, there is no need to include post-farm stages as these would be the same for all scenarios. The purpose of the study usually determines the phases in the life cycle that need to be included and those that can be omitted.

An interesting boundary consideration with regard to food arises when wild game meat is considered (i.e., meat from animals not controlled by humans). Many game animals such as moose and deer are ruminants and emit CH_4 from enteric fermentation. Although these emissions are smaller than those from farm animals, they are not negligible on a per animal basis (IPCC 2006a). However, emissions from wild animals are not reported in national inventories under the Kyoto protocol because they are considered part of the natural ecosystem and their emissions are not anthropogenic. Should they then be included in the CF of meat from wild game? And if game animals feed on arable crops, should part of the emissions from the cropping system be allocated to the game meat? There is no obvious answer to these questions, but one basic reason for not including these emissions is that the amount of wild game is maintained at a ‘natural’ level. If game numbers are deliberately increased, such as by feeding, it is much more difficult to justify exclusion of the emissions from the game animals from the CF of the game meat.

As in most LCA, issues regarding co-product allocation (how to divide emissions from a joint production system on the co-products) arise in CF studies of food products. Livestock production results not only in food, but also, for example, in manure, leather, and wool. During production of many plant-based food products, animal feed is produced as a by-product, such as molasses from sugar production, oilseed meal from vegetable oil production, and stillage from ethanol production. The allocation problem is solved using classical LCA strategies, such as system expansion and economic and physical allocation.

One of the most widely researched areas when it comes to the allocation problem in food CF is the allocation of emissions between milk and meat in milk production systems that inevitably also produce meat and calves as by-products. Flysjö et al. (2011) investigated six different ways of allocating emissions between milk and meat for dairy farms in Sweden and New Zealand, comparing allocation based on the mass, protein, economic value, and amount of feed needed to produce milk and meat, respectively. Furthermore, two ways of system expansion were investigated. In the first of these, it was assumed that meat originating from the milk production system replaced suckler beef on the market, while in the other it was assumed that dairy beef replaced a mixture of suckler beef, pork, and poultry. The CF values obtained varied between 0.63–0.98 kg CO_2e/kg ECM for milk from New Zealand and between 0.73 and 1.14 kg CO_2e/kg ECM for Swedish milk depending on how allocation was made. This considerable variation shows the importance of using the same allocation method when comparing different food products.

2.3 Land Use Change

Agricultural production differs from industrial production in the very important regard that it uses and affects large areas of land. Use of land causes direct emissions of CO_2, CH_4, and N_2O, as discussed in the introduction to this chapter. Increased demand for food due to an increasing global population and an improved standard of living in many developing countries is leading to increased demand for agricultural land. This is leading in turn to the conversion of forests and scrubland into pastures and cropland, and of natural grassland into cropland. This process is called land use change (LUC) and accounts for approximately 10 % of global CO_2 emissions (Global Carbon Project 2013). These emissions of CO_2 originate from standing biomass that is either burnt on the spot, removed and burnt, or used elsewhere. Considerable amounts of CO_2 are also released from soils once they are cultivated, as the carbon bound in the soil starts to oxidize.

Currently, most of the deforestation driven by demand on the global food market is taking place in Southeast Asia, in order to make way for palm oil plantations, and in South America, in order to make way for pasture for beef production and cropland for soybean production (UCS 2011). The production of soybean is driven by the demand for soybean meal as a protein feed for livestock, especially for dairy cows, pigs, and poultry. The EU imports over 20 million tons of soybean meal annually for use in livestock production (Eurostat 2011).

Deforestation cannot be blamed solely on increased demand for food on the global market because there are also other causes, including demand for lumber and biofuels, as well as subsistence farming in some parts of the world. However, there is now a growing consensus in the scientific community that a large proportion of the emissions from LUC should be attributed to food products (UCS 2011; Houghton 2012). However, quantifying the emissions from LUC and allocating them to food products involve a number of methodological challenges (Sect. 3.4).

2.4 Carbon Sequestration in Soils

Cultivation of soils can result in either net emission of CO_2 to the atmosphere or net sequestration of carbon, depending on soil characteristics, carbon input, and management practices (Powlson et al. 2011). Soils that are rich in carbon require large inputs of organic material in order to remain at carbon equilibrium and not lose carbon when cultivated. Organic soils (i.e., soils containing 20–30 % organic matter or more) are most commonly net emitters of CO_2 when cultivated. In contrast, many mineral soils (i.e., soils containing only a few percent of organic matter) lose as much carbon as they sequester during cultivation and hence remain at carbon equilibrium and do not contribute to climate change by releasing CO_2 to the atmosphere. Natural grassland, which has fast-growing biomass and undisturbed soils, is capable of sequestering large amounts of carbon (Soussana et al. 2007). When this sequestration of CO_2 from the atmosphere is taken into account in calculating the CF of meat products from animals grazing natural grassland, the CF value obtained can be heavily reduced because the carbon uptake compensates for the emissions from enteric fermentation and feed production (Pelletier et al. 2010; Soussana et al. 2010; Veysset et al. 2011). However, inclusion of carbon sequestration in the CF of food products raises several methodological issues. Perhaps the most important of these is that sequestration of carbon is a reversible process. If management practices are changed, for example, if the grassland is later plowed under and cultivated or if biomass growth is hampered by drought, the sequestered carbon will slowly be emitted to the atmosphere as CO_2 again (Soussana et al. 2007). Furthermore, measurements of carbon sequestration are highly uncertain and it is unclear whether the sequestration process can continue indefinitely. Current knowledge in soil science states that the potential for soils to sequester carbon will cease with time as the soil reaches a new equilibrium (Powlson et al. 2011; Smith 2012). In addition, animals are not essential for retaining grassland, as the biomass could potentially be used for energy production (e.g., in a biogas reactor). Hence, there is as yet no consensus on whether and how carbon sequestration in soils should be included in the CF of food products.

2.5 Risk of Pollution Swapping

The use of CF as a sustainability indicator has been criticized by both industry and the research community because it focuses solely on the environmental aspect of global warming. Rockström et al. (2009) noted that several areas of environmental pressure need urgent attention. Agriculture and food production affect the environment in several ways. The nitrogen and the phosphorus cycles are placed under heavy stress due to the large amount of fertilizers used on fields, which cause eutrophication due to leakage of nutrients into waters and surrounding land. Ammonia emissions from manure handling cause both eutrophication and acidification, and pesticide use causes the spread of toxic substances in the ecosystem. Modern agriculture also uses considerable amounts of energy and is dependent on finite resources such as fossil fuels. In areas where irrigation is used, freshwater resources are often overused and in many areas soils are depleted or eroded. Agricultural land expansion has also been identified as the main cause of global biodiversity loss (MEA 2005). Hence, it is important to include not only the CF but more environmental categories in a full sustainability assessment, especially for food, which has such a large impact in many impact categories (Röös and Nylinder, 2013).

To investigate how well the CF functions as an indicator of the wider environmental impacts from the production of meat, Röös et al. (2013) carried out a study in which results from a large number of LCA on meat were compared with regard to how the CF correlated with other environmental aspects. It was found that in most cases there was a good correlation between the CF and the eutrophication and acidification potential. Hence, for products having a large CF, the emissions of eutrophying and acidifying substances were also high. This is explained by all these impact categories being involved with the nitrogen cycle. Hence, using nitrogen efficiently in agricultural systems, such as by applying a well-adapted amount of fertilizer at a time when plant uptake is high and by reducing losses from manure management, is beneficial for both reducing N_2O and substances causing eutrophication and acidification, such as ammonia and nitrate. However, it is important to remember that the severity of the actual impact on the ecosystem from the release of eutrophying and acidifying substances is heavily dependent on local conditions, such as proximity to streams and coasts. Röös et al. (2013) also found that energy use and land use were correlated to the CF in most cases, with the important exception of extensive beef production, which can have very low energy use but a high CF due to emissions from enteric fermentation.

In the case of impacts on biodiversity and ecotoxicity, however, care must be taken when using the CF as a sustainability indicator for food products. It is unclear how CF correlates to biodiversity, and it is probably very difficult to establish this on a general level because biodiversity is a very complex and highly site-specific impact category. Beef production can have a very negative impact on biodiversity if it leads to deforestation (Cederberg et al. 2011), but it can have a positive effect on biodiversity if grazing conserves semi-natural pastures supporting many endangered species that need open areas to thrive (Cederberg and Darelius 2001; Cederberg and Nilsson 2004). Regarding leakage of toxic substances from agriculture causing ecotoxicity effects, there is also a risk of conflict when focusing on decreasing the CF of food products. Pesticide production and use cause small GHG emissions, but pesticide use can heavily influence yield levels. Because high yields are beneficial for low CF per kg of food product, there is a risk of a reduced focus on minimizing pesticide use if the prime focus is reduction of GHG emissions.

3 Uncertainties and Variation

3.1 Introduction to Uncertainties and Variation

The accuracy and precision of the CF of food products are affected by uncertainty and variability in input data and uncertainty in the models used to calculate emissions from soils, animals, manure, and LUC, for example. Added to this is the uncertainty introduced by LCA modeling choices, such as the choice of functional unit, allocation strategies, and system boundaries.

Uncertainty arises due to lack of knowledge about the true value of a parameter. Uncertainty can be improved by more and better measurements. Consider, for example, the uncertainty in potato yield from one specific hectare of land during a certain year. The yield is commonly estimated by calculating the number of boxes filled in the field at harvest. On average, one full box has an established weight and the total yield per hectare can be calculated by multiplying the number of boxes filled by this weight and divide that the number by the number of hectares in the field. The uncertainty in this measurement could be reduced by weighing every box of potatoes coming from a specific hectare of land. That would increase the accuracy in yield measurement and reduce the uncertainty to that deriving mainly from the measuring equipment.

Uncertainty in measurement should be distinguished from variability. Variability arises from the inherent heterogeneity of a parameter. Consider the potato yield estimate again. The yield can vary considerably between and within fields, farms, and years due to a number of uncontrollable and controllable reasons, such as weather and soil conditions, access to water and nutrients, variety used, management practices, and pest attacks. Variability cannot be reduced by improved measurement because it is a property of the parameter described. However, improved measurements can help to more accurately describe the variability of a parameter, such as using a probability distribution.

3.2 Variability in Agricultural Systems

Variability in agricultural systems is very large. Varying soil conditions give rise to different amounts of GHG being emitted from the soil (Sect. 3.3) and to some extent determine what crops can be cultivated, how much fertilizer is applied, etc. Organic soils, which are very rich in carbon, give rise to very large emissions of CO_2 and N_2O (Berglund and Berglund 2010), while some soils, especially those used for permanent pasture, can take up CO_2 from the atmosphere and store it as stable carbon compounds in the soil (Sect. 2.4) (Soussana et al. 2007; Powlson et al. 2011).

Yield is an important variable when calculating the CF of food products, as the emissions from soils, machinery, and inputs used on an area of land are distributed across the output from that area. Hence, greater yield gives lower GHG emissions per kilogram of product. Yields can vary greatly even within the same area. For example, Röös et al. (2011) studied wheat production in southern Sweden and found that the yield on approximately 300 farms varied between 3.7 and 11 tons per hectare (95 % confidence interval) during a period of 8 years. The amount of nitrogen fertilizer applied varied between 49 and 357 kg per hectare. The amount of nitrogen fertilizer applied is an influential variable for the CF, as it stimulates N_2O emissions from soil and gives rise to CO_2 and N_2O emissions from the production of mineral fertilizers.

In livestock systems, yield in terms of milk, eggs, and meat produced per year and animal also influences the CF of livestock products. Livestock animals that grow rapidly or produce large amounts of milk or eggs in relation to the feed consumed are favorable from a climate perspective, as less feed needs to be produced. For meat from ruminants, the lifetime of the animal is crucial, as emissions are dominated by emissions from enteric fermentation. The longer the animal lives, the more CH_4 is released.

Variation in livestock yield is very large. From a global perspective, variation in milk yield is enormous, as shown Fig. 4. This reflects the great variation in agricultural management systems globally, from subsistence systems in which the animals provide several functions (milk and meat for food, manure for fuel, draft power, and financial insurance) to the highly industrialized and specialized systems of the developed world. However, even within regions the variation is large. For example, Henriksson et al. (2011) studied milk production in Sweden and

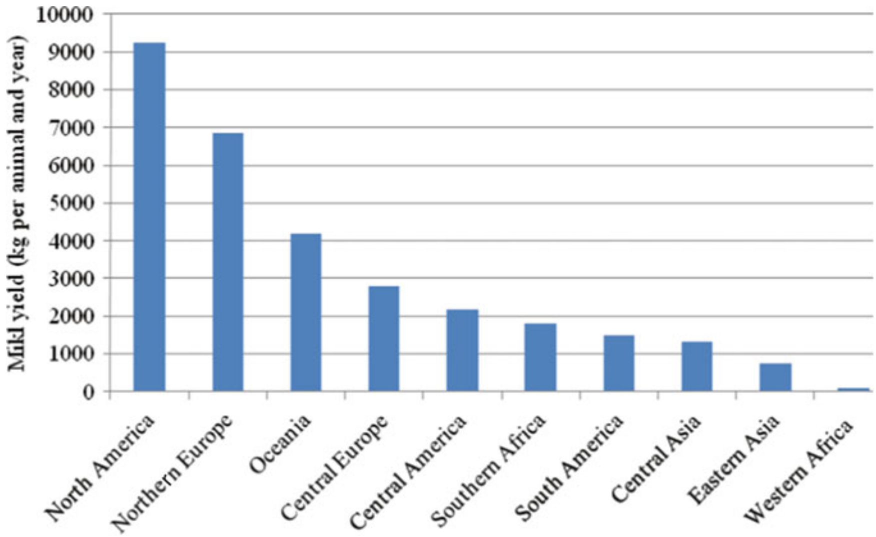


Fig. 4 Average milk yield in 2011 in different parts of the world (FAOSTAT 2011)

found that average farm-level yield varied between 6,000 and 12,000 kg ECM per cow and year.

Apart from yield levels, variability is also large when it comes to feeding strategies, manure management, fertilizer use and energy use in field machinery, greenhouses, and barns. In addition, when food ingredients leave the farm and are processed into food products sold in retail or to restaurants, food ingredients from different sources and places are mixed and are often difficult to trace back to the farm, making the CF of the finished food product uncertain for that reason. For example, wheat that is milled to flour is often a mixture of wheat from several different farms and even countries to establish an appropriate quality of the flour. Furthermore, in baking, pasta making, etc., several kinds of flour and other ingredients are commonly used in recipes.

The inherent variability in agricultural production systems makes it difficult to establish general conclusions regarding the CF of different types of food products, although a general division between livestock-based and plant-based products can usually be made for most products. This is further exemplified and discussed in Sect. 4.

3.3 Uncertainties in Emissions from Soil, Animals, and Manure

Figure 5 shows emissions from the production of pasta. The processes that contribute most to the CF of pasta are N_2O emissions from soil and the production of mineral nitrogen fertilizer (CO_2 from energy consumption and N_2O formed as a

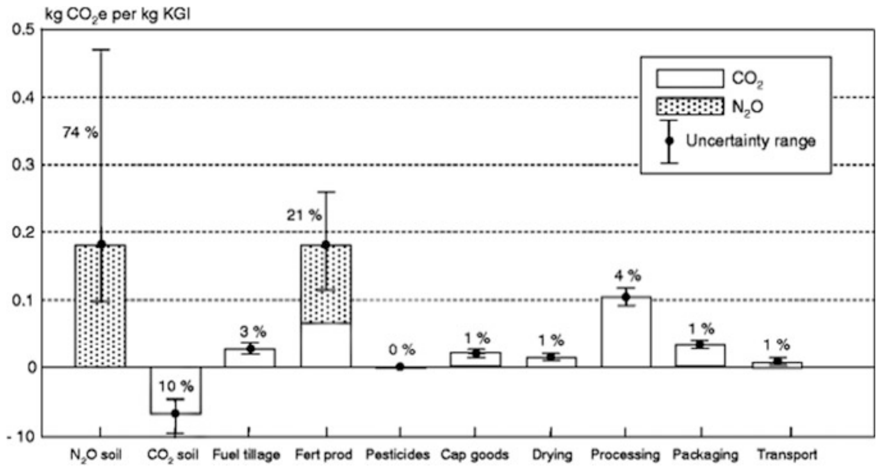


Fig. 5 Emissions of GHG from the production of Swedish pasta (KGI is short for Kungsörnens Gammeldags Idealmakaroner, which is a pasta variety made from Swedish wheat) (from Röös et al. 2011)

by-product during the production of nitric oxide). These two processes also show the largest uncertainty; the uncertainty range for N_2O emissions from soil is 74 % of the total CF and that for mineral fertilizer production is 21 %.

The large uncertainty in the emissions from fertilizer production stems from the uncertain origin of the mineral fertilizer. If the production plant is equipped with N_2O cleaning, emissions are greatly reduced. It is difficult to know at the farm level where fertilizer has been produced, hence the large uncertainty in this process. However, this uncertainty could be reduced with labeling of fertilizers or improved traceability in some other way.

The emissions of N_2O from soil, on the other hand, are difficult to model with higher precision. In the pasta example used here, as in most LCA and CF studies, the IPCC model for calculating N_2O emissions from soil was used (IPCC 2006b). This is a very simplified model of the complex soil processes giving rise to N_2O emissions. It only considers application of nitrogen through mineral and organic fertilizers and crop residues, although it is well established that the formation of N_2O depends on many factors, such as the carbon and oxygen availability in the soil, temperature, and soil pH. In the IPPC model, 1 % of applied nitrogen is assumed to be lost as direct N_2O emissions from agricultural fields, while N_2O is also lost due to nitrogen leakage and volatilization (indirect emissions). IPCC (2006b) also provides uncertainty ranges for the N_2O emission factors, which when used by Röös et al. (2011) in the study on pasta gave large uncertainty in the emissions of N_2O from soil (Fig. 5). There are several more advanced models that can be used to predict N_2O emissions, such as the Coup model (Jansson and Karlberg 2004), which models heat and water flows in a deep soil profile with plant and atmospheric exchange. However, most of these models require detailed soil and climate data that are not readily available at farm level. In addition, although advanced models are highly valuable when trying to understand the underlying processes leading to N_2O formation, they are less useful when it comes to calculating CF due to the great variability in N_2O emissions. The variability in emissions can be just as large as the IPCC uncertainty ranges (Nylinder et al. 2011), so when estimating annual N_2O emissions in order to calculate the CF of a food product, more precise estimates are not guaranteed just because a more complex model is used.

For food products originating from ruminants (milk and beef), total emissions are dominated by emissions from enteric fermentation. Such emissions are known to depend on the amount of feed consumed by the animal and the type of feed, especially its digestibility (Shibata and Terada 2010). Emissions from enteric fermentation can be measured by either enclosing the complete animal in an airtight chamber or by using tracer techniques (Johnson and Johnson 1995). Such measurements are used for developing empirical models that can be applied to predict the emissions from enteric fermentation (e.g., Moe and Tyrrell 1979; Kirchgessner et al. 1995; Mills et al. 2003; IPCC 2006a). These models take different feed characteristics as input, such as the amount of fiber, protein, fat, and cellulose. The ability of these methods to accurately predict CH_4 emissions is limited; when evaluated against different datasets of measured emissions, most equations show a mean square error of 20–40 % (Wilkerson and Casper 1995; Mills et al. 2003; Ellis et al. 2007, 2010).

Emissions from manure handling also arise due to biological processes that are difficult to control and model. Most LCA and CF studies use emissions factors provided by the IPCC, but the uncertainty in these models is substantial. For example, the emissions factor for direct N_2O emissions from manure management has an uncertainty range of −50 % to +100 % (IPCC 2006a).

3.4 Modeling Land Use Change

When modeling emissions from LUC, the concept is commonly divided into direct LUC and indirect LUC. Direct LUC is directly associated with the production of food or feed. For example, if natural forests are cleared and the land used to grow soybeans, it could be argued that the soybean grown on that land should bear some, or all, of the burden of emissions from deforestation. That would represent accounting for emissions from direct LUC. The question of 'amortization period,' i.e., the number of years after deforestation over which the emissions from deforestation should be divided, is an arbitrary choice. The period is commonly set to 20 or 30 years. The choice of amortization period can greatly influence the results (Cederberg et al. 2011).

There is still no consensus in the scientific community regarding how emissions from LUC should be included in the CF. Some studies that included emissions from direct LUC did not simply allocate the emissions from deforestation to the crops produced on the newly deforested land. Rather, they took yearly emissions from deforestation of land later used to grow a specific crop, such as soybean, and divided these emissions across all soybean produced in that region or country, irrespective of whether it was produced on newly deforested land or existing cropland (Meul et al. 2012; van Middelaar et al. 2013). This is not direct LUC emissions in its strictest sense, which would involve allocating emissions from deforestation solely to soybean grown on the newly deforested land. It could be wise to introduce different terminology, such as semi-direct LUC emissions, for this way of handling emissions from LUC (Röös and Nylinder 2013).

Emissions from indirect LUC arise when the demand for one crop causes other crops to be displaced into areas that are deforested. Emissions from indirect LUC are very difficult to estimate because they cannot be directly observed. In the field of biofuels, the issue of indirect LUC has been studied intensively (Broch et al. 2013). Advanced economic equilibrium models have been frequently used in attempts to predict how the global agricultural sector will react to the increased demand for different crops, based on actual economic data and statistics on agricultural productivity, land availability, and other constraints in different countries. The results obtained using these models have been highly variable, ranging from emissions of −90 to +220 g CO_2e per MJ fuel from LUC (Di Lucia et al. 2012). The large variation is partly a consequence of the different studies having varying scopes and using different assumptions about future development, but also because modeling such a complex system as the global agricultural market is highly challenging and greatly dependent on modeling choices.

When it comes to accounting for emissions from indirect LUC for food, apart from using economic modeling two fundamentally different ways of approaching the issue have been applied in the literature. Some authors (Leip et al. 2010; Gerber et al. 2010; Ponsioen and Blonk 2012) burden the crops that show large expansion in a region or a country with all the emissions caused by deforestation, regardless of the land use on the newly deforested land, on the basis that it is the crops that are increasing, which are pushing other crops out into the natural ecosystems or leading to cultivation of grassland. For example, Ponsioen and Blonk (2012) first allocated yearly emissions from deforestation, estimated using trend analysis based on historic data, between lumber and the cleared land based on lumber prices and the return on agricultural goods produced on the cleared land. The emissions allocated to the cleared land were then allocated to different crops based on the share of the total expansion on all types of land, existing cropland and non-cropland, for which a specific crop was responsible. Hence, crops that were not increasing in area were not burdened with any emissions from LUC, while crops that showed a large expansion, such as soybean, were burdened with emissions from LUC that were several hundred percent higher than the direct emissions from cultivating the soybean.

Another approach to indirect LUC is to consider all use of agricultural land as responsible for driving global LUC and therefore divide total emissions from deforestation globally on all products produced on agricultural land. Audsley et al. (2009) divided all emissions from global LUC that can be attributed to commercial

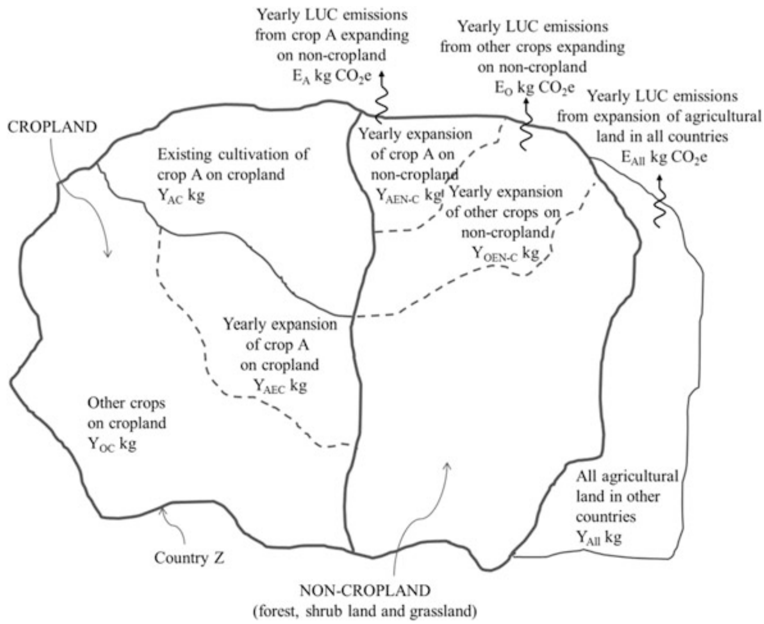


Fig. 6 Simplified example of how GHG emissions from land use change can be allocated to crops in different ways (Y is the yield of crops from different land areas and E are the annual emissions of GHG from LUC on that piece of land). Three different approaches to accounting for emissions from LUC are possible: (1) Only expansion into non-cropland of a specific expanding crop A is considered and LUC emissions are calculated for this crop as $E_A/(Y_{AEN\text{-}C} + Y_{AEC} + Y_{AC})$; that is, emissions from LUC due to the expansion of crop A into non-cropland (E_A) are divided across all crop A from country Z, regardless of where in the country crop A is grown ($Y_{AEN\text{-}C} + Y_{AEC} + Y_{AC}$). (2) The expanding crop A is considered to be responsible for all LUC because its expansion on existing cropland pushes other crops out on non-cropland. Emissions from LUC for crop A are then calculated as $(E_A + E_O)/(Y_{AEN\text{-}C} + Y_{AEC} + Y_{AC})$. Hence, all emissions from LUC, regardless of what is grown on the newly deforested land, are attributed to crop A. (3) Based on a viewpoint that all use of land is responsible for LUC, regardless of where the land is located or what is grown on the land, emissions from LUC are allocated to all crops as $(E_A + E_O + E_{All})/(Y_{AEN\text{-}C} + Y_{AEC} + Y_{A\text{-}C} + Y_{OC} + Y_{OEN\text{-}C} + Y_{All})$. That is, all emissions from LUC, both within the country ($E_A + E_O$) and outside the country (E_{All}), are divided across all crops grown globally

agriculture (58 %) evenly across all land used for commercial production. This resulted in emissions of 1.4 tons of CO_2 per hectare of land from LUC, which had to be added to direct emissions from fuel combustion and use of fertilizer. Schmidt et al. (2012) adopted the same basic viewpoint that all activities occupying land are responsible for LUC, irrespective of where they take place. However, Schmidt et al. (2012) used a more sophisticated way of allocating emissions to land that takes land productivity into account.

These different approaches to handling emissions from LUC are illustrated in Fig. 6.

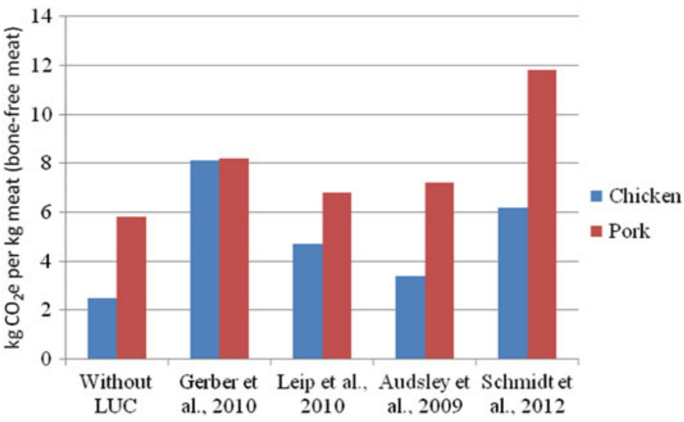


Fig. 7 Carbon footprint of chicken and pork meat calculated using different methods of accounting for greenhouse gas emissions from land use change (LUC). (Data on feed consumption from Cederberg et al. 2009.)

The method used to estimate emissions from LUC heavily influences the results. Figure 7 shows the CF of Swedish average chicken and pig meat calculated using three different ways of estimating emissions from LUC. Generally, when emissions from LUC were included, the CF is greatly increased. Chickens had a smaller CF than pig meat for all LUC methods, but the Gerber et al. (2010) method, which only assigns LUC emissions to soybean, gave very similar emissions for chicken and pig meat, due to the considerable amounts of soybean meal in the diet of Swedish chickens.

3.5 Handling Uncertainties

The first step in handling uncertainties in the CF of food products should be to try to minimize uncertainty as much as possible. One way of making CF assessments more consistent, less error-prone, and more easily comparable is to follow some kind of standard. There are some general standards or specifications in the field of CF that could be used, such as PAS 2050 (BSI 2011), the Greenhouse Gas Protocol Reporting Standard (WRI & WBSCD 2011), and the ISO CF standard (ISO 14067) that is under development. There are also standards that apply to food products specifically. PAS 2050-1 is a specification for the production of horticultural products (BSI 2012) and the ENVIFOOD Protocol aims at providing a harmonized environmental assessment methodology for food and drink products (FoodDrink Europe 2012). The global dairy industry, through the International Dairy Federation (IDF), has developed a common approach for calculating the CF of milk and dairy products (IDF 2010). Furthermore, the international Environmental Product Declaration (EPD) system contains a framework for developing

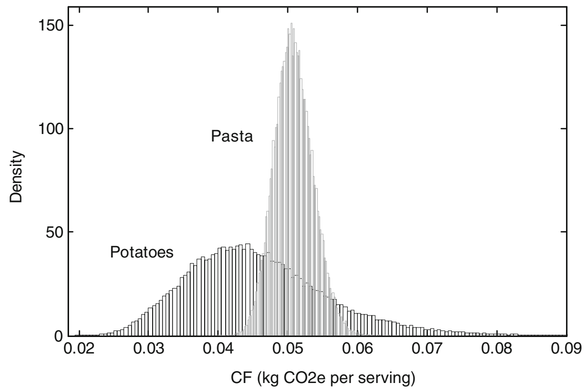


Fig. 8 Uncertainty ranges for the carbon footprint of one serving of potatoes and one serving of pasta (data from Röös 2011)

rules for specific products, that is, product category rules (PCR). Within this system, PCR have been developed for, for example, meat from mammals (EPD 2013). Reducing uncertainty by standardization means that different CF assessments will be more consistent and comparable. However, there is a risk that results are biased by the selection of methods and data. Other ways of reducing the uncertainty in LCA and CF calculations include improved data collection, validation of data, and critical review by a third party (Björklund 2002).

Uncertainty in measured data can never be reduced to zero and uncertainty due to modeling choices will always exist in CF calculations, as will uncertainty due to the great variability in agricultural systems. Therefore, when uncertainty cannot be further reduced, it is important to state the remaining uncertainty in the results. Uncertainty analysis can be used to assess the uncertainty range of the final CF. One way of performing uncertainty analysis that has been commonly applied in LCA is stochastic simulation, such as Monte Carlo (MC) simulation (Rubinstein and Kroese 2007). In MC simulation, the uncertainty in input data and/or model parameters is described using a probability distribution. The CF is calculated a large number of times, with each time randomly drawing values from the probability distribution. The outcome is a large number of possible CF values that describe the uncertainty in the final CF. Röös (2011) used MC simulation to compare one serving of cooked pasta and one serving of cooked potatoes and found that when only the deterministic CF values were used to compare the two, the potatoes were preferable from a climate perspective. However, when uncertainty in the farm gate CF values for potato and pasta and uncertainty in the preparation stage and amount of losses were included, it was not as easy to select a winner. This is illustrated in Fig. 8, where the outcomes from the MC simulations on pasta and potatoes are plotted in histograms showing possible CF values based on uncertainty and variability in input data.

Sensitivity analysis can be used to illustrate uncertainty due to choices, such as to identify the parameters that have a great or small influence on the final result, by varying input parameter individually by, for example, ±10 or 20 %. By testing how different choices regarding CF modeling and model and input data affect the results, the robustness of these results can be evaluated. Figure 7 (Sect. 3.4) presents an example of a sensitivity analysis on calculating the CF of chicken and pig meat depending on the method used to estimate emissions from LUC. From this analysis, it can be concluded that regardless of the methodology used to include emissions from LUC, Swedish chicken has a lower CF than Swedish pork, but that the differences in emissions are less pronounced for some LUC methods.

4 Examples of Uses and Results

4.1 Identification of Hotspots and Mitigation Options

One important reason for calculating the CF of food products, as in all LCA and CF calculations, is of course to identify hotspots and mitigation options. An important insight that has emerged from a number of LCA on food products is that the majority of the GHG emissions from the production and use of food originate from the on-farm and pre-farm phases. This is especially the case for livestock products, for which emissions from post-farm activities such as slaughter, packaging, transport, storage, and preparation are small in comparison with the emissions from primary production (Fig. 9).

In relative terms, emissions from post-farm phases can make a substantial contribution to the CF of root crops, cereals, and vegetables, especially for products that are transported long distances. Figure 10 shows the GHG emissions from different phases in the production of tomatoes in the Netherlands and Spain, including their transport to Sweden. Emissions from the tomatoes from the Netherlands are dominated by emissions from fossil energy used to heat the greenhouses, while emissions from the Spanish tomatoes are dominated by emissions from transport (Röös and Karlsson 2013).

However, because the difference in CF between livestock-based products and plant-based products is so large (Fig. 11), choosing a diet based on local produce has a limited effect on total emissions from all the food consumed in a typical

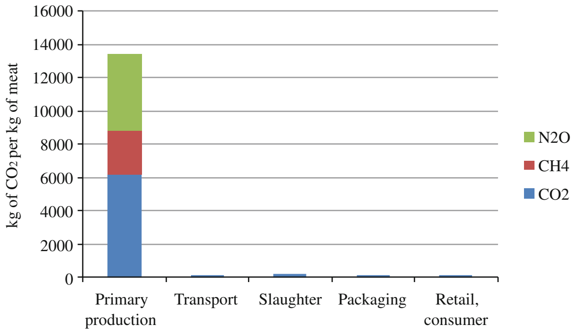
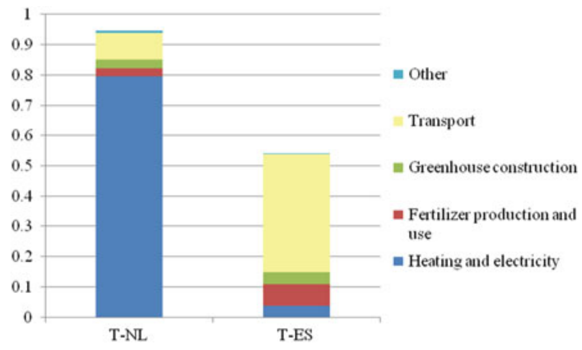


Fig. 9 GHG emissions from the production of beef meat (data from LRF et al. 2002)

Fig. 10 Carbon footprint of 1 kg of tomatoes delivered to the Swedish market from the Netherlands (T-NL) and Spain (T-ES) (data from Röös and Karlsson 2013)



Western diet (Weber and Matthews 2008; Garnett 2011). To achieve large reductions in GHG emissions from food consumption, reducing meat and dairy consumption is the most important mitigation option (Garnett 2011).

Another important insight that has come from estimating the GHG emissions associated with food production is that emissions from the food sector are strongly dominated by CH_4 and N_2O, at not CO_2 which is the dominant GHG in the transport and energy sector. Emissions of CO_2 from energy use in agriculture can be avoided by improved energy efficiency and the use of renewable energy sources; however, because energy-related emissions constitute a minor proportion of total emissions from agriculture (Fig. 2), this will not be sufficient to achieve substantial reductions in emissions. The emissions of CH_4 from enteric fermentation and N_2O from soil arise from natural biological processes that are difficult to control and the mitigation

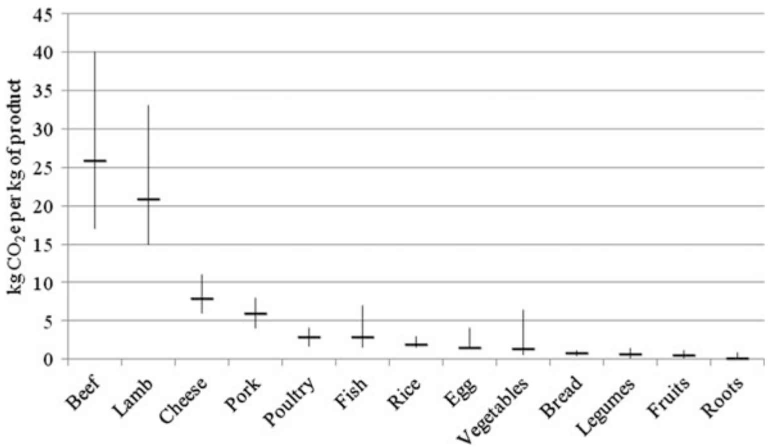


Fig. 11 Carbon footprint of different types of food products at retail. Average values estimated to be representative for food products sold on the Swedish market. Error bars show ranges of values found in the literature as a result of different production systems and methodological choices. Emissions from land use change and carbon stock changes in soils not included (Röös 2012)

potential for reducing these gases is much less. CH_4 emissions can be reduced to some extent by altering the diet fed to ruminants (Beauchemin et al. 2008), but the risk of pollution swapping is great (Shibata and Terada 2010; Flysjö 2012). N_2O emissions from soils can be reduced by optimizing nitrogen use and by N_2O inhibitors (although these are banned in many countries), but major N_2O formation in soil is inevitable. Therefore, most studies looking at mitigation options for agriculture include changes in consumption patterns as one essential option in achieving major reductions in emissions from food consumption (Beddington et al. 2011; Foley et al. 2011; Foresight 2011; SBA 2012).

Because GHG emissions from food products are dominated by emissions arising on-farm (from feed cultivation, animals, and manure), numerous studies have looked at mitigating emissions in this phase of the life cycle. For example, Ahlgren (2009) used LCA to evaluate different systems for producing tractor fuel and mineral nitrogen fertilizers from biomass, thereby reducing the use of fossil fuels in food production, and found that 3–6 % of a farm's available land was needed to produce the tractor fuel used on the farm. The study also showed potential for reduced emissions of GHG from the reduced fossil fuel use in the system. However, there were potential trade-offs with other environmental impacts, such as eutrophication, and the use of limited resources such as water and phosphorus. This is another example of the risk of pollution swapping when only emissions of GHG are considered, as discussed in Sect. 2.5.

Dairy production and the CF of milk is one of the most thoroughly researched areas of food production (e.g., Thomassen et al. 2008; Flysjö 2012). Flysjö (2012) covered several methodological aspects in calculating the CF of milk and discussed different mitigation options based on results from calculating the CF of Swedish dairy products. Another area that has received relatively great attention is comparison of different diets in livestock production in order to identify feeding strategies that contribute to reduced emissions from livestock production (e.g., Strid Eriksson et al. 2005; Pelletier et al. 2010).

4.2 Consumer Communications

Labeling food products with their CF has been proposed as a way of enabling active choices by consumers. The British supermarket Tesco was a pioneer in this area, announcing a grand ambition in 2007 of putting CF labels on all its products (Boardman 2008). However, this labeling initiative was later dropped, as it proved very time-consuming and costly to calculate the CF and as others did not follow suit, so the labeling initiative lacked critical mass (Guardian 2012). However, the CF of hundreds of products was estimated and published in a report (Tesco 2012). In Sweden, the hamburger restaurant MAX has labeled its different meals with information on the CF. The labels are presented at the point of purchase in order to illustrate to consumers the impact of different meal alternatives (Fig. 12).

Fig. 12 Example of carbon footprint label of a food product. From the Swedish hamburger restaurant MAX

Although CF labeling or some other way of informing consumers about the CF of food products is necessary to enable active choices by consumers, it is questionable whether this is an effective policy instrument that can justify the time-consuming and costly process of calculating the CF of food items. Röös and Tjärnemo (2011) point out that the attitude-behavioral gap identified as regards purchasing organic products is also applicable to carbon labeled products. In other words, although consumers have positive attitudes toward preserving the environment, sales of eco-labeled food products are still low for reasons such as perceived high price, strong habits governing food purchases, perceived low availability, lack of marketing and information, lack of trust in the labeling system, and low perceived customer effectiveness. Hence, CF information on food products might be more useful in business-to-business communication, for food professionals in the retail sector, and in public procurement. These actors have a large influence on which products are procured, marketed, displayed, and put on sale in a country. For example, in Sweden there is strong interest among different actors in the public sector in calculating the CF of total food purchases and meals in schools, hospitals, and retirement homes in order to identify hotspots and work with lowering the impact from food consumption. To facilitate these calculations, a list of average CF values representative of food items on the Swedish market has been compiled and is now in use in a number of companies, municipalities, and organizations in Sweden (Röös 2012).

The CF value of food products and the magnitude of other environmental impacts is also valuable information for organizations and authorities concerned with formulating dietary advice. Including environmental concerns in the recommendations for nutritionally sound food consumption is becoming increasingly common; for example, the new Nordic Nutrition Recommendations include a chapter on sustainable food consumption in which different food products are categorized into one of three groups: low CF (<1 kg of CO_2 per kg), such as roots, bread, and local fruits; medium CF (between 1 and 4 kg CO_2 per kg), such as poultry, rice, and greenhouse vegetables; and high CF (more than 4 kg CO_2 per kg), such as beef, pork, cheese, and tropical fruits transported by air (NNR 5 2012). Studying food consumption from both a nutritional and environmental perspective is also a rapidly growing field in research (e.g., Bere and Brug 2009; Macdiamid et al. 2012; Meier and Christen 2013).

5 Conclusions

In this chapter, the importance of the food system as a contributor to climate change, as well as the relevance and challenges of CF as a decision support tool, have been described. The main sources of GHG emissions in the life cycle of food products are N_2O from soils, CH_4 from enteric fermentation in animals, and CO_2 from LUC, such as deforestation. Emissions also arise from manure management, mineral fertilizer production, rice cultivation, and energy use on farms and from post-farm activities such as processing, packaging, storage, distribution, and waste management. With increasing awareness of climate change, calculating the CF of food products has become increasingly popular among researchers and companies wanting to determine the impact of their products on global warming and/or to communicate the CF of their products to consumers. Some issues are especially relevant when calculating the CF of food products, such as the choice of functional unit, which is challenging owing to the multi-functionality of food. Other issues concern how to include emissions arising from indirect land use change and removal of CO_2 from the atmosphere by carbon sequestration in soils into CF calculations. Causes of the large uncertainties associated with calculating the CF of food products and ways to handle this uncertainty have been discussed. Despite the large uncertainties, it is clear that the differences in CF between different types of food products are very large. In general, the CF of livestock-based products is much larger than those of plant-based products. Although informing consumers about the CF of food products is necessary to enable active choices by consumers, it is questionable whether labeling products with CF data is an effective policy instrument that can justify the time-consuming and costly process of calculating the CF of food items. CF information on food products may be more useful in business-to-business communication, for professionals in the retail sector, and in public procurement.

6 Future Challenges and Research Needs

Major changes to the food system are needed in order to sustainably feed the rising global population. The high CF of livestock-based products in relation to plant-based food products speaks for itself. Future diets must be heavily dominated by food products of vegetal origin in order to reduce emissions of GHG and other pollutants, and to protect water, land, and biodiversity. Implementation of improvements during the primary production of food, such as increased energy efficiency on farms and better nutrient management, needs to accelerate. Post-farm stages in the food chain also need to be considered, as emissions can be substantial during these phases, especially for future diets that will (hopefully) be based more on plants. Calculation of the CF will continue to be a valuable tool in preventing sub-optimization and in identifying the most effective mitigation options. More research is needed in several areas regarding calculating the CF of food products. There is a need for better methods to assess emissions from biological processes and to assess, for example, emissions from land use change and changes in soil carbon balance. More food products from different production systems need to be investigated. As food patterns need to change, assessing the CF of complete diets and optimizing these based on local resource availability, nutritional status, effect on biodiversity, and other environmental impacts, as well as cost, must be given increased attention. Finally, communicating CF results, including their uncertainty, and developing and evaluating policy instruments based on these results are research areas that require a broad interdisciplinary approach to be successful.

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The Carbon Footprint of Ceramic Products

Paula Quinteiro, Marisa Almeida, Ana Cláudia Dias, António Araújo and Luís Arroja

Abstract Nowadays it is generally recognized that human activities increase anthropogenic greenhouse gas (GHG) emissions to dangerous thresholds, leading to climate change due to an increase in global temperatures. In an industrial context, the product carbon footprint concept has been emerging as a relevant tool to support the development and implementation of GHG management strategies throughout product life cycles, in order to reduce GHG emissions along the supply chain, improve energy efficiency, and improve product competitiveness in different markets. This chapter focuses on the carbon footprint of ceramic products and has the following purposes: (1) to present general information on ceramic manufacturing, in particular a characterization of the European ceramic industry with regard to energy sources and production value, and a description of the general ceramic manufacturing process; (2) to carry out case studies in which the carbon footprint of different ceramic products (ornamental earthenware piece,

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brick, roof tile, wall and floor tile, sanitary ware) is quantified; (3) to identify improvement measures and best available techniques (BAT) to reduce the total carbon footprint of some products; (4) to analyze the specific GHG emission of each of the ceramic products studied, considering a cradle-to-gate approach; and (5) to present some methodological challenges related to carbon footprint quantification.

Keywords European ceramic industry · Best available techniques · Sanitary ware · Roof tile · Ornamental earthenware piece · Wall and floor tile

1 Introduction

The world energy mix is based on a model of fossil fuel consumption that is responsible for anthropogenic greenhouse gas (GHG) emissions. The Intergovernmental Panel on Climate Change (IPCC) in its Fourth Assessment Report (IPCC 2007) confirmed that global warming is an unequivocal fact: the global atmospheric concentration of carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) in the atmosphere has drastically increased since 1750, and nowadays provides a contribution of about 60 % to global warming.

This is evidenced by the increase in the global average air and ocean temperatures and rising sea level. To stop this trend and to reduce the current climate change development, several governmental and nongovernmental initiatives have been implemented, such as the introduction of emission trading programs, voluntary programs, carbon or energy taxes, and regulations and standards on energy efficiency and emission measurements (European Commission 2012, 2013; WRI/WBCSD 2011a).

The carbon footprint concept emerged from the ecological footprint discussion introduced in the 1990s by Wackernagel and Rees (Rees and Wackernagel 1994; Wackernagel and Rees 1996) and has become widely known over the last decade (East 2008).

Although many definitions for carbon footprinting are currently available, it is currently accepted that it refers to the sum of GHG emissions resulting directly and indirectly from a person, organization, or product (Carbon Trust 2010; Pandey et al. 2010). The product carbon footprint quantifies the GHG emissions over the product life cycle following a cradle-to-gate or a cradle-to-grave approach, as illustrated in Fig. 1. The cradle-to-gate approach includes all processes from the raw and ancillary materials extraction and energy production through product manufacturing including packing (gate of the mill), whereas the cradle-to-grave approach includes all processes from the raw and ancillary materials extraction and energy production through product manufacturing including packing, distribution, use phase, and eventually recycling, reuse, recovery, and final disposal.

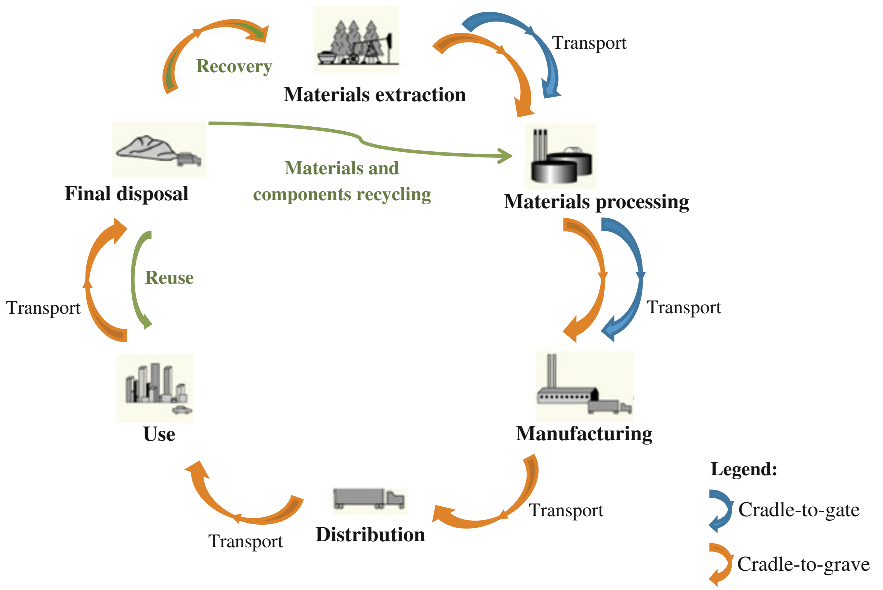


Fig. 1 Life cycle of a product following cradle-to-gate and cradle-to-grave approaches. Adapted from Remmen et al. (2007)

The GHG emissions are converted to their carbon dioxide equivalent (CO_2e) value using the global warming potentials defined by the IPCC (2007).

The product carbon footprint can be applied to:

- identify hotspots over the life cycle (i.e., main unit processes where GHG emissions occur);
- identify improvement measures for GHG mitigation, promoting energy efficiency and economic sustainability;
- communicate the carbon footprint to consumers;
- establish an opportunity for product differentiation and/or market penetration.

The ceramic industry plays a key role in sustainable development, considering its three main components: environment, economy, and society. The ceramic industry recognizes the need to mitigate GHG emissions and increase energy efficiency. These goals can be achieved by conducting an environmental impact assessment throughout the product life cycle, and therefore implementing environmental and energy improvement measures into the manufacturing process.

The carbon footprint of ceramic products emerges as a powerful tool to perform the systematic integration of energy efficiency and environmental consideration in the product design process and decision-making (European Commission 2011). In addition, it can also provide information for planning and assessing the sustainability of buildings, as it is one of the indicators included in the European standards, namely in EN 15804:2012 (CEN 2012).

Several methodologies to estimate the carbon footprint of a product have been developed. In 2011, the British Standards Institution (BSI) published the Public Available Specification (PAS) 2050, which specifies the requirements to assess the life cycle GHG emissions of goods and services (BSI 2011). The GHG Protocol Initiative convened by the World Resources Institute (WRI) and the World Business Council for Sustainable Development (WBCSD) developed a standard to quantify and report the GHG emissions throughout the life cycle of a product (WRI/WBCSD 2011b).

The International Organization for Standardization (ISO) in 2013 published the ISO/Technical Specification (TS) 14067, which specifies principles, requirements, and guidelines for the quantification and communication of the carbon footprint of a product. These three methodologies have been built based on the existing life cycle assessment (LCA) methodology established through the ISO 14040 and 14044 standards (ISO 2006a, b). ISO/TS 14067:2013 is also based on environmental labels and declarations—ISO 14020 (ISO 2000), ISO 14024 (ISO 1999), and ISO 14025 (ISO 2006c)—for communication.

PAS 2050 was applied by some companies in pilot projects to measure and report the carbon footprint of bricks (e.g., Ceram 2011; Best Foot Forward 2011). This methodology was also used to calculate the carbon footprint of an ornamental earthenware piece (Quinteiro et al. 2012a), which is, currently, the only published study that deals with this type of ceramic product.

Due to primary data (data that refers to direct measurements made along with the supply chain, from processes owned, operated, or controlled by the organization under study) confidentiality, there are only a few published studies concerning the quantification of ceramic products. Most of these available studies have been performed following ISO 14040 and ISO 14044, and are limited to the analysis of GHG emissions and the corresponding global warming impact category, such as Almeida et al. (2010a, 2011) and Koroneos and Dompros (2006), who estimated the carbon footprint of bricks; Almeida et al. (2011) and Bribilán et al. (2011), who estimated the carbon footprint of roof tiles; Almeida et al. (2010b), Bovea et al. (2010), Ibánez-Forés et al. (2011, 2013), Nicoletti et al. (2002) and Tikul and Srichandr (2010), who calculated the carbon footprint of wall and floor tiles; and Kaleseramik (2012), who calculated the carbon footprint of sanitary products. Furthermore, the application of cut-off criteria, as well as allocation procedures, is not commonly referred to in those studies.

Concerning to the content of this chapter, Sect. 2 presents general information on ceramic manufacturing, characterizing the European ceramic industry relative to its energy sources and production value, and explaining the general ceramic manufacturing process. In Sect. 3, some case studies are presented, with the carbon footprint for ornamental earthenware pieces, bricks, roof tiles, wall and floor tiles, and sanitary ware products being calculated, and, when justifiable, identifying some environmental and energy improvement measures and best available techniques (BAT). Section 4 discusses the specific GHG emission of each of the ceramic products studied, the different contribution of the manufacturing stage of the different products being analyzed for the total carbon footprint considering a cradle-to-gate approach, and a synthesis of the improvement measures and BAT studied. Section 5 presents challenges to carbon footprinting ceramic products. Section 6 presents the main conclusions of this study.

2 General Information on Ceramic Manufacturing

A brief characterization of the ceramic industry, identifying the main ceramic products manufactured and their production value, is made in this section. Moreover, the general manufacturing process of ceramic products is explained, identifying the specific manufacturing characteristics of each ceramic sub-sector analyzed.

2.1 Characterization of Ceramic Industry

The ceramic industry in the European Union (EU)-27 (an economic and political union of 27 European member states) accounts for 23 % of global ceramics production (Cerame-Unie 2012).

The ceramic industry has a wide range of product applications: structural—including bricks, pipes, wall and floor tiles, and roof tiles; refractories—such as kiln linings; table and ornamental ware (household ceramics); sanitary ware; expanded aggregates; inorganic bonded abrasives; technical—such as insulators, biomedical implants, and ceramic capacitors; among others (European Commission 2007; Rahaman 2006; Remmey 1994). This classification of subsectors has evolved in accordance with the ceramic technological evolution.

All these ceramic industry subsectors are energy intensive, namely due to the drying and firing processes, which involve firing temperatures between 800 and 2000 °C (European Commission 2007). From a generic point of view, the energy costs of the European ceramic industry represent an average of 30 % of the total manufacturing costs, where the energy mix is around 85 % of natural gas to 15 % of electricity (Cerame-Unie 2012). However, the energy sources and their percentages vary depending on the ceramic subsectors and their products, as well as on the specific country considered. For instance, in the case of Portugal, about 3 % of the brick mills operate on fuel oil, about 11 % with petroleum coke, 15 % with biomass, 70 % with natural gas, and the remaining 1 % corresponds to the use of liquefied petroleum gas (Dias 2008).

The production value of the EU-27 ceramic industry has been fluctuating over the last few years, as illustrated by Fig. 2. After the economic crisis of 2008, the production values of ceramic products, namely wall and floor tiles, bricks, and roof tiles, dropped and have been recovering slowly since 2009. In addition, some ceramic subsectors, such as table and ornamental ware, have been experiencing strong competition from new emerging markets (European Commission 2007).

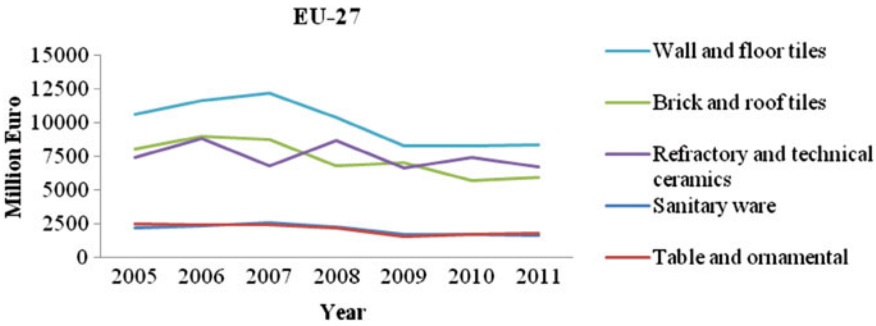


Fig. 2 Trend in the production values of the EU-27 ceramic subsectors (Eurostat 2013)

2.2 Ceramic Manufacturing

The manufacture of ceramic products is a complex interaction of raw materials, technological processes, people, and economic investments. It takes place in different types of kilns (e.g., continuously operated tunnel and periodically operated shuttle), with a wide range of raw materials and in numerous shapes, sizes, and colors. The manufacturing includes the transport and storage of raw materials, ancillary materials and additives (e.g., deflocculating agent—sodium silicate for the preparation of raw materials), preparation of raw materials, shaping, drying, surface treatment, firing, and subsequent treatment.

Figure 3 schematically shows the typical steps in the manufacturing of ceramic products. The following steps are identified: transport and storage of raw materials, ancillary materials and additives (e.g., deflocculating agent—sodium silicate for preparation of raw materials), preparation of raw materials, shaping, drying, surface treatment, firing, and subsequent treatment. However, the manufacturing operations can vary according to the specific requirements of ceramic products and raw material characteristics, as explained below.

2.2.1 Preparation of Raw Materials

The preparation of raw materials consists of mixing several raw materials and additives, with the aim of obtaining a material with a homogenous composition and an appropriate granulometric distribution. Even in the case of bricks (typically red ceramics) that use almost only clay as the raw material, two or more types of clay with different composition are used.

2.2.2 Shaping

The shaping of the ceramic product depends on the product type and the technique applied:

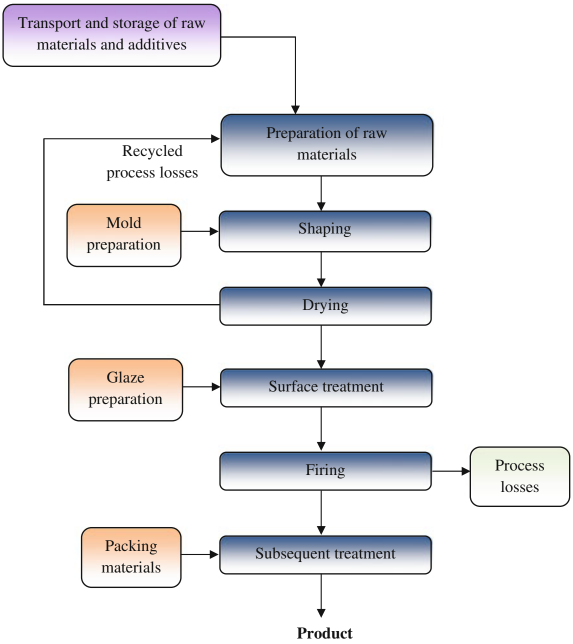


Fig. 3 General steps in the manufacturing of ceramic products. Adapted from Quinteiro et al. (2012a)

(1) slip casting process for ornamental ware, sanitary ware and refractory ceramics;
(2) dry pressing for ornamental ware and wall and floor tiles;
(3) plastic shaping for ornamental ware, bricks and roof tiles (European Commission 2007; Serrano et al. 2009).

In the slip casting process, body formation takes place in a mold made of gypsum plaster, and the mold is placed on a bench with a closed pipe system for warm water circulation. This water warms the mold and the capillary suction of the mold draws a portion of the liquid from the slip casting to form a high solid cast on the inner surface of the mold. The wall thickness increases progressively with time; when the piece has an appropriate wall thickness, the operator proceeds to the draining of the remaining slip casting, which is reintroduced in the production process. In the case of dry pressing, the powder (moisture content 5–7 % of water after the spray drier) is pressed into the molds (pressing unit process), whereas in plastic shaping, the 'extrusion paste' (moisture content 20–25 %) is formed in jigger machines.

After the shaping step, the green ware of ornamental, sanitary, and technical ware undergoes a dressing process, which consists of the removal of the surface roughness and mold marks from the ceramic.

2.2.3 Drying

The next step is the ceramic product drying. Green ware still usually contains water from the preparation of the raw materials. Therefore, to avoid tension and consequently nonconforming pieces, it is necessary to remove this water, slowly and gradually, in intermittent dryers, continuous dryers, or stoves at temperatures varying between 50 and 350 °C (European Commission 2007; Serrano et al. 2009). Heat for air drying is mainly supplied by gas burners and by hot air recovered from the cooling zone of the tunnel kilns or by using heat exchangers in shuttle kilns.

The ceramic industries, such as ornamental, bricks (less usual), roof tiles, and sanitary ware, use intermittent chamber dryers, which consist of a battery of chambers with close-fitting entry doors, usually served by rail tracks carrying kiln-cars. These kiln-cars are loaded with ceramic products. In the ornamental drying unit process, the piece is dried for a period of about 12 h. Until this unit process, unfired broken ware (nonconforming pieces without heat treatment) are reintroduced into the mixing process as a raw material. For bricks and roof tiles, the drying cycles are in the order of 16–24 h, with a temperature of about 100 °C.

For ceramic wall and floor tiles, it is common to use a vertical dryer, in which the green tiles are fed into baskets consisting of several decks of rollers. The groups of baskets move upwards through the dryers, where they meet hot drying gases. The temperature in this type of dryer is normally less than 200 °C, and the drying cycles range from 35 to 50 min. Horizontal multideck roller dryers can be also used in the manufacturing of wall and floor tiles. These tiles are fed onto different decks within the dryer, being conveyed horizontally by driven rollers. The maximum temperature in these dryers is usually higher than in the vertical option (around 350 °C), and the drying cycles are shorter (between 15 to 25 min).

2.2.4 Surface Treatment and Firing

After drying, the green ware undergoes surface treatment by glazing, engobing, and/or other decorating techniques (screen printing, gravure, and flexo space printing) (European Commission 2007; Serrano et al. 2009). Engobing is mainly employed in the manufacture of roof tiles and wall and floor tiles, whereas glazing is mainly used in ornamental and sanitary ware.

In the glazing unit process, the green ware is covered with a thin glaze layer followed by a firing cycle, which seals the porous ceramic body. The surface of the piece becomes watertight and smooth. In the case of ornamental products, before glazing the pieces undergo a preliminary firing cycle, a biscuit firing cycle. This first heat treatment gives the piece the strength and absorbency required for glazing. During biscuit firing, some pieces undergo undesirable structural changes, like local defects and cracks, and cannot be reintroduced into the manufacturing cycle. This fired broken ware is generally sent to the cement industry. After glazing, the fired piece undergoes a second heat treatment (glost firing). Some nonconforming pieces resulting from glost firing can be retouched and then submitted again to glost firing (refiring).

The wall and floor tiles can undergo the following firing cycles: (1) unglazed firing cycle; (2) double fired (less used), in which the green ware undergoes a biscuit firing cycle, glazing, and a glost firing cycle; (3) single fired glazed, in which the green ware is glazed and then goes through one firing cycle (European Commission 2007).

Sanitary green ware is also glazed and, therefore, undergoes a single firing. However, some resulting nonconforming pieces can be retouched and submitted to a new firing cycle (refiring). The bricks and roof tiles are unglazed. However, it should be noted that a small fraction of roof tiles is glazed and then submitted to a single or double firing, depending on the technology implemented in the mills.

The firing is a key process in the manufacturing of ceramic products because it encompasses the chemical and physical changes in the ceramic body, so that the final product has the appropriate characteristics to be handled (dimensions, geometry, mechanical strength, abrasion and fire resistance, and porosity). Shuttle kilns are used in ornamental and sanitary ware and in refractory ceramics, where the pieces are placed on kiln-cars on fireproof firing auxiliaries (also called kiln furniture). Tunnel kilns are used in bricks, roof tiles, sanitary ware, and refractory products, where the green ware is placed in kiln-cars, on which there are refractory decks. These kiln-cars are pushed through the kiln at set intervals. Incoming ware is preheated by hot gases from the firing zone, whilst incoming air cools the fired ware and is itself preheated for its combustion role. The roller kilns are mainly used for wall and floor tiles, as well as for table and sanitary ware.

Table 1 shows the specific temperature profiles of ceramic subsectors.

2.2.5 Subsequent Treatment

After firing, some products require additional processing to address certain features that cannot be achieved during its manufacture. This subsequent treatment can

Table 1 Ranges of temperature profiles of firing unit processes (European Commission 2007; Remmey 1994)

Ceramic subsector	Firing temperature (°C)
Ornamental ware	
• Biscuit firing cycle:	1,000–1,100
• Glost firing cycle:	1,000–1,080
Table ware	1,180–1,350
Brick	850–900
Roof tile	1,000–1,200
Wall and floor tile	1,050–1,200
Sanitary ware	1,250–1,300
Refractory and technical ceramics	1,250–1,850

include polishing, cutting, drilling, and sawing, among others (product finishing). Afterwards, the ceramic products are sorted, labeled, packaged, and delivered to distribution.

3 Case Studies

This section presents case studies that quantify the carbon footprint of ornamental earthenware pieces, bricks and roof tiles, wall and floor tiles, and sanitary ware based on the Quinteiro et al. (2012a), Almeida et al. (2010a, b, 2011) and Almeida (2009) studies, respectively.

All case studies were performed following the ISO 14040 and ISO 14044 standards, and are limited to the analysis of GHG emissions and the corresponding global warming impact category. The goal, functional unit, system boundary, data collection, multifunctionality and allocation, and carbon footprint results are presented for each case study. Moreover, some improvement measures and best available techniques (BAT) for the ceramic manufacturing industry are also identified and evaluated, such as the incorporation of more energy-efficient technologies in the manufacturing stage and the use of alternative energy sources (European Commission 2007).

3.1 Carbon Footprint of Ornamental Earthenware Pieces

3.1.1 Goal of the Study

This case study aims to estimate the carbon footprint of an ornamental earthenware ceramic piece, manufactured and consumed in Portugal. The carbon footprint hotspots are identified, improvements in environmental measures are suggested, and their feasibility, performance, and economic viability are evaluated.

3.1.2 Functional Unit

The functional unit has been defined as one ornamental earthenware ceramic piece (cubic vessel) ready to be sold, with a mass of 0.417 kg and dimensions of $10 \times 10 \times 10$ cm.

3.1.3 System Definition and Boundary

Following ISO 14040 and ISO 14044, a cradle-to-grave approach is adopted; that is, GHG emissions are considered from the extraction of raw materials, through

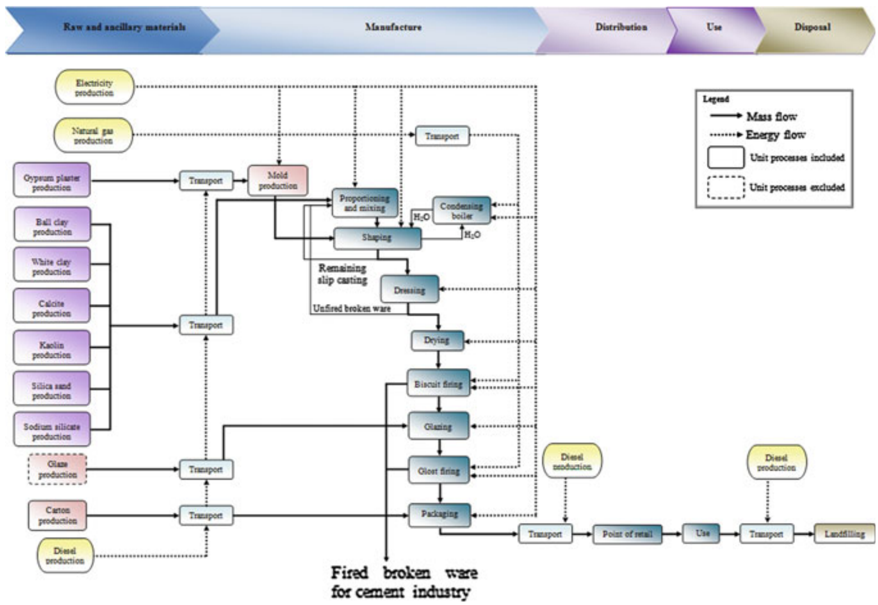


Fig. 4 System boundary for ornamental earthenware pieces. Adapted from Quinteiro et al. (2012a)

manufacturing, use, and the disposal of the used product. The cut-off criteria allow the decision of which processes should be included within the system boundary. Although ISOs do not suggest quantified thresholds, they state that the cut-off criteria should be based on mass, energy, and environmental significance. Therefore, in this study, the mass flows that represent less than 0.5 % of the functional unit were excluded from the defined system boundary. The system boundary also excludes the transport of consumers to and from the point of retail and the transport of employees to and from the manufacturing mill, as well as the production of capital goods (machinery and equipment).

As shown in Fig. 4, the following stages are considered:

- Raw and ancillary materials—which includes cradle-to-gate GHG emissions (from the raw material extraction through the production stage up to the gate of the company) for the production of the raw materials—white and ball clays, calcite, kaolin, silica sand, and sodium silicate—consumed in the manufacturing of the ceramic piece, namely in the proportioning and mixing unit processes. This stage also includes cradle-to-gate GHG emissions for the production of the gypsum plaster needed for mold production, the production of carton board used to pack the ceramic piece, the production of diesel necessary for transporting the raw materials to the ceramic mill and the GHG emissions released during this transport by the truck, and the production of electricity and natural gas.
- Manufacture—includes GHG emissions by the ceramic mill and by the operational activities such as lighting, administrative activities, heating, ventilation,

Table 2 Data sources used in secondary data collection (Quinteiro et al. 2012a)

Unit process	Data source
Kaolin production	Ecoinvent database v.2.2 (Ecoinvent 2012)
Silica sand production	
Calcite production	
Black clay production	
White clay production	
Sodium silicate production	
Cartonboard production	
Landfilling	
Gypsum plaster production	GaBi 6.0. software database (PE International 2012)
Electricity production (Portuguese mix)	
Natural gas production	
Diesel production	
Transport	• Distances: provided by the mill • GHG emissions factors: GaBi database (PE International 2012)

and air conditioning. This stage also includes cradle-to-gate GHG emissions from the auxiliary unit process mold production.

- Distribution—includes GHG emissions as a result of the transport by truck of the piece to the point of retail, and by the production of diesel used in this transport.
- Use—it was assumed that there is no energy consumption and/or GHG emissions expended during the usage of the ceramic piece.
- Final disposal—the piece was assumed to be landfilled at the end of its life cycle; this stage includes GHG emissions arising from the landfill, from the truck transportation of the piece to the landfill, and from the production of diesel used in this transport.

3.1.4 Data Collection

All data from each unit process comprised in the ceramic piece's manufacturing stage were collected at the mill that produces the analyzed piece. For the remaining unit processes, secondary data have been collected from databases (Table 2). Secondary data refers to external measurements that are not specific to the product but represent an average or general measurement of similar processes or materials (e.g., generic data from peer-reviewed publications, databases, industry reports, or aggregated data from trade associations, among others).

In the production of the ceramic piece, the CO_2 emissions arising from the consumption of energy (natural gas and electricity) and from the decomposition of calcium carbonate ($CaCO_3$) contained in the piece during biscuit firing (Aiazzi and

Aiazzi 1988). Natural gas is consumed in the condensing boiler to heat the water used in shaping and during biscuit and glost firing.

3.1.5 Multifunctionality and Allocation

The ornamental earthenware ceramic manufacturing is typically a multifunctional system because several pieces with different dimensions and geometries are manufactured in the same production line, at the same time (co-products). The data on energy consumption (electricity and natural gas) provided by the mill includes the energy needed to produce the piece under study, but also for all the other pieces manufactured during the reference year. As we are faced with multifunctional processes, it is necessary to allocate the GHG emissions due to energy consumption in the ornamental manufacturing processes to the piece under analysis. To solve the allocation problem in carbon footprint studies, a hierarchy of procedures shall be compiled (BSI 2011; ISO 2006b). Wherever possible, allocation should be avoided by unit process division or by system boundary expansion. Where allocation cannot be avoided, the GHG emissions of the process should be partitioned according to physical relationships. Where physical relationships cannot be used, the allocation should be done using other criteria, such as the economic value of the products. In this study, the application of unit process division and system boundary expansion are not feasible due to an absence of data.

The physical relationships usually employed in manufacturing processes are the mass, volume, number of items, or time of processing, as stated by the European Commission JRC (2010). However, the ornamental earthenware ceramic manufacturing process does not allow the employment of a single allocation criterion to all energy consumption flows (Quinteiro et al. 2012b).

The single mass and volume criteria seem not to be a rational choice because the energy consumption in each manufacturing stage is not always proportional to the mass or to the volume of the ornamental earthenware ceramic pieces manufactured. For example, the mass criterion is adequate for the biscuit firing cycle but not for the glost firing cycle. In the biscuit firing cycle, the pieces can touch each other, so that smaller pieces can be placed inside larger ones. Therefore, the energy consumed during the biscuit firing cycle is proportional to the mass of each ceramic piece, with the mass the being critical issue. On the other hand, during the glost firing cycle, the critical issue is the piece volume; the pieces cannot touch each other or they would vitrify together.

An allocation based on the number of items is also not applicable to all manufacturing stages, because some ornamental earthenware ceramic pieces require more energy and generate more emissions than others. For example, hard running and handling pieces are very susceptible to deformations and imperfections, requiring several firing cycles to obtain the final product.

The time of processing criterion also seems not to be a reasonable option from an operational point of view, because the mill under study produces several ceramic pieces at the same time (the mill has, on average, hundreds of different pieces in processing), and each piece has a different time of processing in each manufacturing stage. On average, the total time of processing of each piece varies between 2 to 3 weeks.

An allocation method based on the market price of the pieces was also disregarded because the market price of the pieces changes according to market demand. This allocation criterion would result in a poor time-related representativeness of the energy consumption and costs and the GHG emissions by each studied piece, as market prices change over a short time, making it necessary to reformulate the study whenever the market price changes.

Therefore, a hybrid allocation model based on the mass, volume, and/or number of pieces manufactured at the mill has been applied (Quinteiro et al. 2012b). Electricity is consumed in all the unit processes and has two components: a nonpermanent component, which occurs directly due to the piece production, and a permanent component, which represents the electricity consumed in the absence of production. This last component refers to the existence of equipment permanently in operation (e.g., stove fans) and to the mill lighting system.

The mass allocation criterion has been used to estimate the nonpermanent component of electricity and natural gas consumption during biscuit firing, the volume allocation criterion has been considered to calculate natural gas consumption in glost firing, whereas the number of pieces manufactured at the mill has been used in the calculation of the permanent component of electricity consumption. Table 3 presents the consumption of electricity (nonpermanent and permanent components) and natural gas in each unit process allocated to each piece.

As mentioned in Sect. 2.2, fired broken ware cannot be reintroduced in the production process and is sent to the cement industry. Because this material is considered waste and not a co-product, all the GHG emissions arising from the ceramic piece manufacturing stage have been allocated to the ceramic piece.

Table 3 Electricity and natural gas consumption of each unit process included in the ceramic piece manufacturing stage

	Electricity (kWh/piece)		
Unit processes	Nonpermanent component	Permanent component	Natural gas (kWh/piece)
Mold manufacture	0.007	0.011	–
Proportioning and mixing	0.059	–	–
Condensing boiler	0.001	0.011	1.06
Shaping	–	0.167	–
Biscuit firing	0.006	–	1.39
Dressing	0.003	0.045	–
Glazing	0.002	–	–
Glost firing	0.007	–	0.71
Packaging	0.002	0.033	–
Total	0.086	0.267	3.16

3.1.6 Results

The carbon footprint of the selected ornamental earthenware ceramic piece is 1.22 kg CO_2e per piece. The manufacturing stage is the main contributor to this carbon footprint, accounting for 88 % of the total carbon footprint. The raw and ancillary materials (10 %), the distribution (1 %), and the disposal (1 %) stages are the other main contributing stages to the total carbon footprint.

The unit processes that contribute more than 1 % to the total carbon footprint of the ornamental earthenware ceramic piece are shown in Fig. 5. The biscuit firing and condensing boiler (shaping stage) unit processes are the hotspots, as they contribute 18–30 % of the total carbon footprint of the ceramic piece, respectively.

The emissions of the biscuit and glost firing result from electricity consumption, natural gas burning, and the decomposition of $CaCO_3$ during biscuit firing. The $CaCO_3$ decomposition emits 0.12 kg CO_2e per piece, which corresponds to 10 % of the total carbon footprint. The shaping unit process also has significant emissions, being responsible for 10 % of the total carbon footprint of the ceramic piece. Insignificant GHG emissions (less than 1 % of the total carbon footprint) arise from the dressing, packaging, glazing, landfill, diesel production, and transportation of all the materials and products.

3.1.7 Improvement Measures and BAT

The identified hotspots in the life cycle of the ornamental earthenware ceramic piece should be preferably targeted for reducing the carbon footprint of the piece. Therefore, improvement measures and BAT to reduce the energy consumption and GHG emissions were identified and assessed, such as (1) incorporation of a gas pressure control system in kilns; (2) optimization of the lightning system; (3)

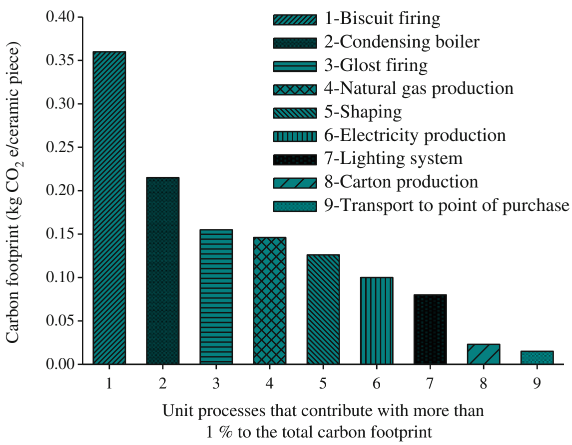


Fig. 5 Contribution of each unit process that contributes more than 1 % to the total carbon footprint of the piece

Table 4 Economic indicators to assess the implementation of the gas pressure control system in the shuttle kilns and the optimization of the lighting system

Economic indicators/improvement measures	Gas pressure control system	Lighting system
Investment	8000.00 €	2793.09 €
Cost savings	8315.00 €/year	505.00 €
Maintenance	<500 €/year	<300 €/year
Pay-back	<1 year	6 years

changing the temperature profile of the biscuit firing cycle; and (4) recovering excess heat from kilns. However, it should be noted that the type and integrative procedure of the improvement measures and BAT into the ceramic manufacturing process should be assessed cautiously. Their incorporation in the manufacture should be well-suited to the specific characteristics of the slip casting and mill installation; otherwise, they can damage the quality of the ceramic product, contributing to an increase in nonconforming pieces.

The incorporation of a gas pressure control system would result in a decrease of 10 % in both natural gas consumption and GHG emission for each biscuit and glost firing cycle, based on operating data of the mill, contributing to a carbon footprint reduction of 3 %.

With regard to natural gas costs, they would decrease by about 8 € for each biscuit and glost firing cycle.

To reduce electricity consumption at the mill, the lighting system should be optimized; for example, conventional ballasts should be replaced by electronic ones, as suggested by Sá (2008). A decrease, on average, of 2 % of the total carbon footprint of the ornamental earthenware ceramic piece is expected with the implementation of this measure.

Table 4 presents the indicators considered to assess the economic sustainability optimization improvement measures of both the gas pressure control system and lighting system. The avoided costs consist of cost savings in energy. The simple pay-back is defined as the period of time needed to recover the initial investment, dividing the initial investment costs by the annual energy costs savings. The payback is considered profitable when it is equal to or shorter than 3 years (European Commission 2006). The calculated payback for the incorporation of the gas pressure control system in kilns is usually lower than 3 years. This means that this measure emerges as the most profitable and economically sustainable because it combines the least expensive investment with the highest annual savings. Although the lighting system optimization requires a lower investment cost, its implementation would result in a payback twice that required to consider this measure profitable.

Another measure aimed to reduce the GHG emission during biscuit firing is a change in its temperature profile. The temperature profile has been adjusted considering both thermogravimetric analysis (Mansfield et al. 2010) and differential thermal analysis (Gabbott 2007) of the slip casting in order to understand its behavior when submitted to heating and cooling operations. This measure was

Table 5 Economic indicators to assess the sustainability of the thermal energy recovery system

Scenario/improvement measure	Thermal energy recovery system		
Reduction rate of natural gas consumption (%)	Investment (€)	Annual profit (€/year)	Pay-back (year)
25	.	1807.59	<3
50	4124.23	3615.19	1
75	.	5422.78	<1

applied at the studied mill and resulted in an energy efficiency of 2 % of the natural gas consumption per firing cycle. However, this measure was disregarded after conducting some experimental tests that showed that it caused an increase in broken ware and a decrease in the mechanical strength of the ceramic pieces.

To recover the excess heat from the kilns, the implementation of heat exchangers in the kiln chimneys is an option that should be considered. This BAT (European Commission 2007) would contribute to heating the water used in the shaping stage, thus reducing the natural gas consumption during this stage. However, this would require a long-term period to restructure and optimize the temperature profile of the biscuit and firing cycles, as it would cause severe changes to the kiln's atmosphere. Therefore, as an alternative, the installation of a thermal energy recovery system around the chimneys of the kilns has been analyzed. To evaluate the sustainability of this BAT, three reduction rates of natural gas consumption during the shaping stage have been simulated (25, 50 and 75 %) due to the absence of real data. The corresponding GHG emissions from the shaping stage would have the same reduction rates (25, 50 and 75 %).

These reduction rates lead to a total decrease in the GHG emissions of 2, 8, and 10 % for the ceramic piece under study, respectively. Table 5 shows the economic indicators used to assess the profitability of this improvement measure. The investment costs are the same for all scenarios. All defined scenarios present a simple payback lower than 3 years, arising as economic and environmentally sustainable options.

However, the reduction of 75 % in natural consumption appears to be the most profitable, since it results in the highest annual profit and has the lowest simple payback.

3.2 Carbon Footprint of Bricks

3.2.1 Goal of the study

This case study aimed to quantify the carbon footprint of a ceramic brick manufactured in Portugal and to identify the environmental hotspots throughout the brick's life cycle. In addition, some improvement measures are presented and discussed. The study follows a cradle-to-gate approach, considering GHG emissions from the extraction and processing of raw and ancillary materials until reaching the ceramic mill gate.

3.2.2 Functional Unit

In this case study, the functional unit is defined as one brick ready to be sold, with a mass of 4.21 kg and dimensions of 30 × 20 × 11 cm.

3.2.3 System Boundary and Data Collection

The cut-off criteria allows the decision as to which processes should be included within the system boundary. Although ISOs do not suggest quantified thresholds, they state that the cut-off criteria should be based on mass, energy, and environmental significance. Therefore, in this study, mass flows that represent less than 0.5 % of the functional unit were excluded from the defined system boundary (Almeida et al. 2010a, 2011). The distribution stage and the production of capital goods (building, machinery, and equipment) are excluded from the system boundary. The transport of consumers to and from the point of retail and the transport of employees to and from the manufacturing mill were also excluded.

Primary data (direct measurements made along the supply chain, from processes owned, operated or controlled by the organization under study) concerning brick manufacturing were collected from brick mills and quarries. Moreover, data concerning lightning and other activities, such as maintenance and cleaning, were also collected. The transport profiles (distance traveled, load state of the truck on the return journey) for the raw and ancillary materials were also provided by brick mills.

Secondary data for the raw and ancillary materials stage, such as data on clay, packing film, wood pallet, diesel, natural gas, and electricity production, as well as the GHG emissions factors for transport, were collected from the Ecoinvent database v.2.2 (Ecoinvent 2012).

As shown in Fig. 6, the system boundary includes the following stages:

- Raw and ancillary materials—consist of cradle-to-gate production of clays, packing film, and pallet. It also includes cradle-to-gate GHG emissions from the diesel production necessary for the transport of the raw materials to the ceramic mill and GHG emissions released during this transport by truck, and the production of natural gas and electricity.
- Manufacture—includes GHG emissions by the ceramic mill and by the operational activities such as lighting, administrative activities, heating, ventilation, and air conditioning.

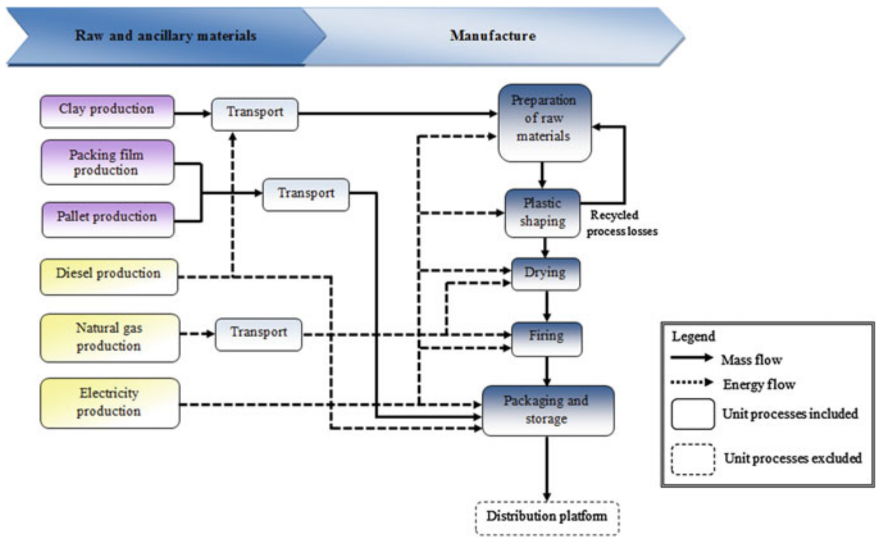


Fig. 6 System boundary for ceramic brick

3.2.4 Multifunctionality and Allocation

The manufacturing of bricks is a multifunctional system as the mill under study produces bricks with dimensions other than 30 × 20 × 11 cm (co-products). Therefore, it is necessary to allocate the GHG emissions of the manufacturing stage to the brick under analysis, as the energy consumption provided by the mill includes all the bricks manufactured. The choice of the most appropriate allocation criterion depends, among others, on the available data and the characteristics of the multifunctional system (European Commission JRC 2010). Therefore, bearing in mind these aspects and following the hierarchy of procedures to solve the allocation issues referred to in Sect. 3.1, an allocation based on the mass criterion was applied. In contrast to the manufacturing stage of ornamental earthenware pieces (case study presented in Sect. 3.1), in brick manufacturing the energy consumption in each manufacturing stage is proportional to the mass of the brick manufactured.

3.2.5 Results

The average carbon footprint of the brick manufactured in Portugal is 0.51 kg CO_2e per brick, using natural gas as the energy source for the drying and firing unit processes. The processes that contribute more than 1 % to the total carbon footprint of the brick are shown in Fig. 7. The firing process is responsible for about 60 % of the total carbon footprint (hotspot process), mainly due to the burning of natural gas in the kilns.

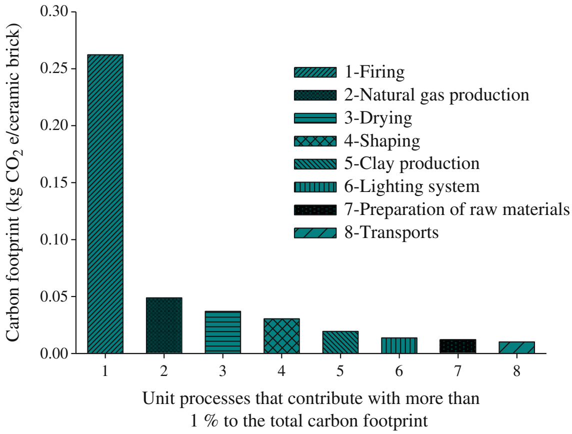


Fig. 7 Unit process that contribute more than 1 % to the total carbon footprint of the ceramic brick

Although the functional unit is one brick, to allow the comparison of the result of this carbon footprint of the brick with other published studies, it was converted to 1 kg of brick. Therefore, the carbon footprint of the brick manufactured in Portugal is 0.12 kg CO_2e per kg of brick. A higher value has been found by Koroneos and Dompros (2006), 0.20 kg CO_2e per kg of brick. These differences can be due to: (1) the definition of the system boundaries and cut-off criterion; (2) use of distinct energy sources; and (3) different manufacturing technology implementation due to the use of different energy sources and the specific composition of the raw materials. Koroneos and Dompros (2006) included distribution and use stages within the defined system boundary, whereas in the Portuguese brick study these stages were disregarded. Additionally, no information is given concerning whether the study uses any of the cut-off criterion. Also, the main source of energy used is different than that of the Portuguese brick because petroleum coke represents almost 100 % of the total energy consumption in the manufacturing stage (Koroneos and Dompros 2006). Petroleum coke is composed of a higher carbon ratio than natural gas, which means that when burned petroleum coke releases higher levels of CO_2, therefore having a higher warming potential than natural gas.

3.2.6 Improvement Measures and BAT

The switch from natural gas to biomass as the energy source in the brick industry was analyzed.

Although more than 80 % of brick kilns are fired with natural gas (Schimmel 2010), a growing number of companies have been using biomass as an alternative energy source to promote the environmental and economic sustainability of the mills (Fernandes et al. 2004). Table 6 presents the total carbon footprint of the brick either by using natural gas or biomass in the manufacturing stage. The carbon footprint of the brick using biomass as the energy source was calculated by

Table 6 Carbon footprint of the ceramic brick using different energy sources

Energy source	Carbon footprint (CO_2e per brick)
Natural gas	0.51
Biomass	0.28

considering the same functional unit, system boundary, and applying the same cut-off criterion. Primary data were collected from a similar brick mill, i.e., producing the same type of bricks, with the same implemented technology, the same kiln load capacities, and using the same raw materials, which already uses biomass as an alternative energy source.

According to Table 6, energy source switching from natural gas to biomass leads to a reduction of 55 % in the total carbon footprint of the brick. However, the use of biomass in the ceramic mills depends on long-term availability of forestry residues. Although the use of biomass appears to be suitable to reduce GHG emissions, biomass burning could lead to other environmental impacts. For instance, biomass burning generates higher emissions of particulate matter to the atmosphere than natural gas. In addition, it should be noted that the brick mill had to install a unit of biomass preparation, which requires an initial investment cost that needs to be assessed from an economic sustainability point of view.

In the calculation of the carbon footprint of the brick using biomass as the energy source, biogenic carbon (i.e., carbon that is captured and stored across the biomass growth) was considered neutral. This approach is valid because neutral biogenic CO_2 emissions are balanced by CO_2 sequestration in the forest, providing that the forest is sustainably managed (e.g. Bribián et al. 2011; Dias et al. 2007, 2012; González-García et al. 2010; Ross and Evans 2002).

3.3 Carbon Footprint of Roof Tiles

3.3.1 Goal of the Study

The purpose of this case study is to calculate the carbon footprint of roof tiles manufactured in Portugal over its life cycle, from the extraction of raw materials through to the manufacturing stage (cradle-to-gate approach). Also, the identification of the main unit processes that contribute to the total carbon footprint is the intention of this case study.

3.3.2 Functional Unit

The functional unit (i.e., the reference flow to which all flows are assigned), is a 22 × 40 cm roof tile ready to be sold with a mass of 2.50 kg.

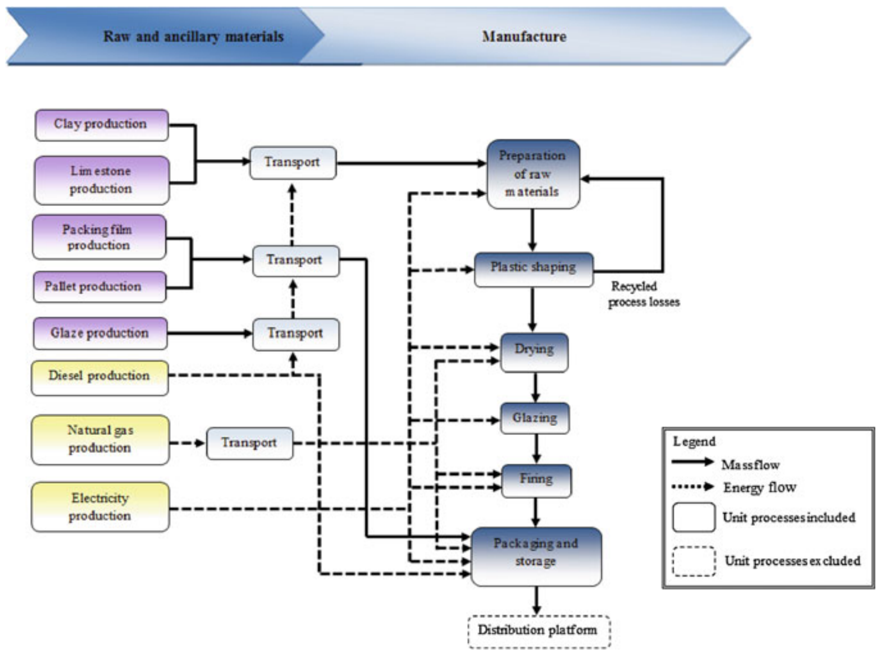


Fig. 8 System boundary for roof tiles

3.3.3 System Boundary and Data Collection

The system boundary, schematically presented in Fig. 8, includes the raw and ancillary materials and the manufacturing stage (cradle-to-gate approach). As in Sects. 3.1 and 3.2, in this study the mass flows representing less than 0.5 % of the functional unit are excluded from the defined system boundary (cut-off criterion). The system boundary also excludes the transport of consumers to and from the point of retail and the transport of employees to and from the manufacturing mill, as well as the production of capital goods (machinery and equipment).

The inventory of primary data for the manufacturing stage, including the transport profiles, consisted of data obtained from on-site measurements.

Secondary data for the raw and ancillary materials stage (i.e., data on clay, limestone, packing film, wood pallet, glaze, diesel, natural gas and electricity production), as well as the GHG emission factors for transport, were collected from the Ecoinvent database v.2.2 (Ecoinvent 2012).

3.3.4 Multifunctionality and Allocation

Roof tile manufacturing is a multifunctional system because it produces different types of roof tile. The measured primary data is related to all types of roof tiles

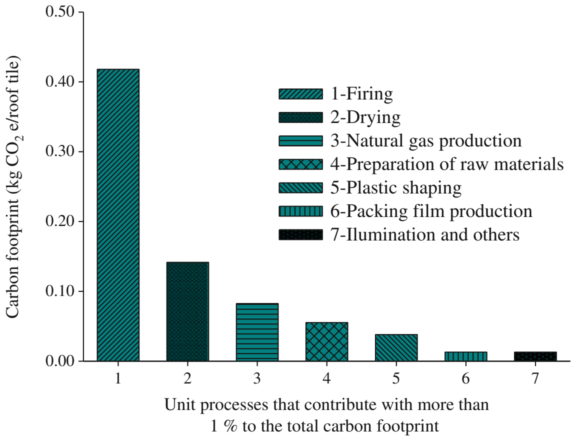


Fig. 9 Unit process that contributes more than 1 % to the total carbon footprint of the roof tile

manufactured. Therefore, in order to quantify the inputs and outputs of the roof tile under analysis, an allocation procedure based on mass criterion was applied. In this case, it was assumed that the energy consumption in each manufacturing stage is proportional to the mass of the roof tile manufactured (Almeida et al. 2011).

3.3.5 Results

The carbon footprint of the roof tile manufactured in Portugal is 0.78 kg CO_2e per roof tile. The unit processes that contribute more than 1 % to the total carbon footprint of roof tile are shown in Fig. 9. Firing emerges as the unit process that most contributes to the total carbon footprint of the roof tile, with 54 %. The burning of natural gas needed to achieve the firing temperature profile into the kilns (Sect. 2.2) is responsible for this GHG emissions hotspot.

The Portuguese roof tile mills studied have already the most suitable BAT incorporated into the manufacturing stage (e.g., recovery heat from hot flue gases from the kilns).

Although the functional unit is a roof tile, to allow the comparison of the result of this carbon footprint of roof tile with other published studies, it was converted to one kilogram of roof tile. Therefore, the carbon footprint of a roof tile manufactured in Germany is 0.31 kg CO_2e per kg roof tile (Creaton 2012), which is slightly higher than the Portuguese roof tile, which is 0.28 kg CO_2e per kg of roof tile. Both studies were performed considering a cradle-to-gate carbon footprint assessment, considering both similar raw and ancillary materials as well as the manufacturing stage.

This slight difference between German and Portuguese roof tiles can be explained by the specific features of each manufacturing process, such as load capacity of kilns and firing temperature profiles, which result in different energy source consumption rates, as well as their different transport profiles.

3.4 Carbon Footprint of Wall and Floor Tiles

3.4.1 Goal of the Study

In this case study, the carbon footprint of the wall and floor tiles is assessed from a cradle-to-gate perspective. The main unit processes that contribute to the total carbon footprint of the wall and floor tiles are also identified.

3.4.2 Functional Unit, System Boundary, and Data Collection

The data used refers to 1 m^2 of wall and floor tile as the functional unit. In this study, the mass flows that represented less than 0.5 % of the functional unit are excluded from the defined system boundary (cut-off criterion). The system boundary also excludes the transport of consumers to and from the point of retail, and the transport of employees to and from the manufacturing mill, as well as the production of capital goods (machinery and equipment).

The system boundary illustrated in Fig. 10 was, therefore, considered to comprise the following stages:

- Raw and ancillary materials—includes cradle-to-gate GHG emissions (from raw materials extraction through the production stage up to the gate of the company) for the production of the raw materials—clay, kaolin, calcium carbonate, quartz, and feldspar—consumed in the manufacture of the wall and floor tiles, namely in the preparation of the raw materials unit process. This stage also includes cradle-to-gate GHG emissions for the production of the glaze, production of carton, packing film and wood pallet, production of diesel necessary for the transport of the raw materials to the ceramic mill and GHG emissions released during this transport by truck, and the production of electricity and natural gas.
- Manufacture—includes GHG emissions by the ceramic mill and by operational activities, such as lighting, administrative activities, heating, ventilation, and air conditioning.

Primary data concerning wall and floor manufacturing were collected from mills. Moreover data concerning the lightning and other activities, such as maintenance and cleaning were also collected. The transport profiles (distance traveled, load state of the truck in the return journey) for the raw and ancillary materials were also provided by wall and floor tile mills.

Secondary data for all the unit processes considered within the raw and ancillary materials stage (Fig. 10), as well as the GHG emissions factors for transport, were collected from the Ecoinvent database v.2.2 (Ecoinvent 2012).

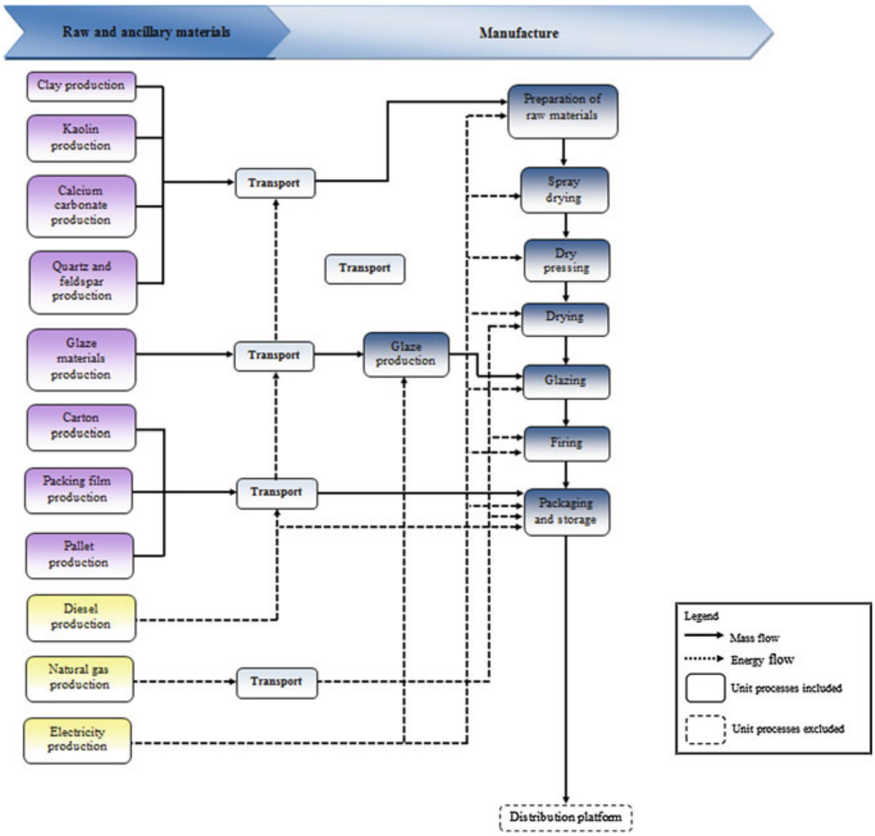


Fig. 10 System boundary for wall and floor tiles

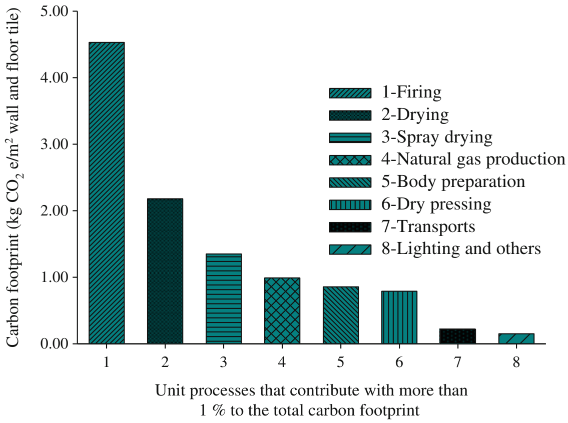
3.4.3 Multifunctionality and Allocation

In this cradle-to-gate analysis, an allocation procedure based on mass criterion is required, as wall and floor tile mills produce more than one co-product (produce wall and floor tiles with different characteristics and dimensions) (Almeida et al. 2010b).

3.4.4 Results

The carbon footprint of the wall and floor tiles manufactured in Portugal is 11.29 kg CO_2e per m^2 of tile. Fig. 11 shows the contributions of the unit processes that contribute more than 1 % to the total carbon footprint of wall and floor tiles. As observed in this figure, firing emerges as the main hotspot in terms of GHG emissions, with 41 % of the total carbon footprint. Besides, the drying unit

Fig. 11 Unit process that contributes more than 1 % to the total carbon footprint of 1 m^2 of wall and floor tile



processes assumes a relevant role because it is responsible for 19 % of the total carbon footprint of 1 m^2 of wall and floor tile.

The studied tile mills have incorporated into their manufacturing process the majority of suitable BAT. The use of fast-firing kilns, roller kilns, lead to reduced energy consumption due to the lower residence time and the reduced amount of material needed to load tiles in the kilns (IEA 2007). Moreover, electronic variable speed drives are connected to the main electric motors.

The carbon footprint result of this study is compared with other published studies concerning the quantification of the carbon footprint for wall and floor tiles.

Comparing the result obtained in this study, the carbon footprint of a wall and floor ceramic tile manufactured in Thailand, 39.43 kg CO_2e per m^2 of ceramic tile (Tikul and Srichandr 2010), it can be seen that this one is more than 3 times higher than the carbon footprint of the ceramic tile manufactured in Portugal. These discrepant results can be explained by the use of different production techniques, firing technology, and energy sources. The wall and floor tiles are manufactured in Thailand using double firing, in which the green ware goes through a biscuit firing cycle, glazing, and a glost firing process, whereas the manufacturing of the wall and floor tiles manufactured in Portugal only requires a single fired glaze, in which the green ware is glazed and then undergoes a single firing cycle. Portuguese tile mills use roller kilns, whereas Thai ceramic tile use tunnel kilns. Moreover, the manufacturing of wall and floor tiles in Thailand uses liquefied petroleum gas for the firing processes and furnace oil for the preparation of raw materials, whereas Portuguese mills use natural gas with slower emission factors.

The preparation of raw materials (gridding and spray drying) is the unit process that contributes the most to the total carbon footprint of the wall and floor tiles manufactured in Thailand, with 34 %. The preparation of raw materials is a hot-spot because it consumes electricity and also furnace oil; it also includes GHG emissions resulting from the consumption of diesel in the internal transport of raw materials. However, both biscuit firing and the glost firing unit processes assume a relevant role because they are responsible for 27 and 29 % of the total carbon footprint of ceramic tile manufactured in Thailand.

Another study carried out by Bovea et al. (2010) reported a carbon footprint of a wall and floor tiles manufactured in Spain of 8.46 kg CO_2e per m^2 of ceramic tile, which is lower than the carbon footprint of the wall and floor tile under analysis. Both studies perform a cradle-to-gate carbon footprint assessment, but there are some differences. With regard to the system boundaries, the study performed by Bovea et al. (2010) considers: (1) the distribution unit process, which is disregarded in the present study and (2) an average distance of 20 km for raw materials and glaze transportation to the ceramic mill, in contrast to the wall and floor tile Portuguese study that considers higher average distances (150 km). Concerning energy consumption, the ceramic tiles manufactured in Spain consume 78 % of natural gas and 22 % of electricity during the manufacturing stage (Bovea et al. 2010), whereas tiles manufactured in Portugal consume 85 % of natural gas and 15 % of electricity. These aspects and specific features of ceramic mills, such as different capacity of kilns and firing temperature profiles, can explain the differences presented in the studies performed in Portugal and Spain.

3.5 Carbon Footprint of Sanitary Ware Products

3.5.1 Goal of the Study

The aim of the current case study is to estimate the carbon footprint of sanitary ware manufactured in Portugal, as well as identify the hotspots that exist across the life cycle of the sanitary ware products.

3.5.2 Functional Unit, System Boundary, and Data Collection

The functional unit, which allows comparison between products without bias, refers to 1 kg of manufactured sanitary ware product. The material flows representing less than 0.5 % of the functional unit are excluded from the defined system boundary (cut-off criterion). The mold is not considered in the system boundary. For these processes, primary data are confidential and secondary data are lacking. Also, the distribution stage and the production of capital goods (building, machinery, and equipment) are excluded from the system boundary. The transport of consumers to and from the point of retail and the transport of employees to and from the manufacturing mill were also excluded.

As shown in Fig. 12, the system boundary (cradle-to-gate) includes the following stages:

- Raw and ancillary materials—includes cradle-to-gate GHG emissions (from the raw materials extraction through production stage until the gate of the company)

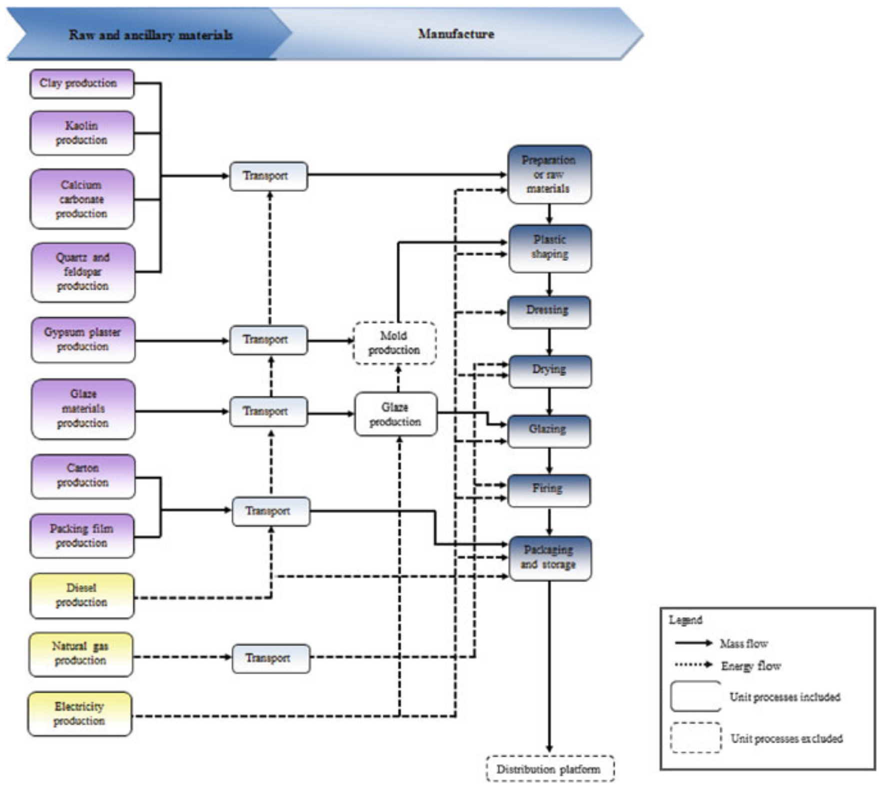


Fig. 12 System boundary for sanitary ware products

for the production of the raw materials—clay, kaolin, calcium carbonate, quartz, and feldspar, gypsum plaster—consumed in the manufacture of the sanitary ware, namely in the preparation of raw materials unit process. This stage also includes cradle-to-gate GHG emissions for the glaze material's production (raw materials used in glaze production such as aluminum oxide, silicon dioxide, magnesium oxide, among others), production of cartons, packing film and wood pallets, production of diesel necessary for the transport of the raw materials to the ceramic mill and GHG emissions released during this transport by truck, and production of electricity and natural gas.

- Manufacture—includes GHG emissions by the sanitary ware mill and by the operational activities such as lighting, administrative activities, heating, ventilation, and air conditioning.

Primary data (direct measurements made along the supply chain, from processes operated or controlled by the organization under study) including lighting and other activities (e.g., cleaning and maintenance) as well as trucks transport profiles, were collected from sanitary mills.

Secondary data for the raw and ancillary materials stage, as well as the GHG emissions factors for transports, were collected from the Ecoinvent database v.2.2 (Ecoinvent 2012).

3.5.3 Multifunctionality and Allocation

The sanitary mills produce different pieces (co-products) with different dimensions and geometries at the same time in the same production line. Therefore, in order to quantify the inputs and outputs of the sanitary ware products, an allocation procedure based on mass criterion was applied (Almeida 2009).

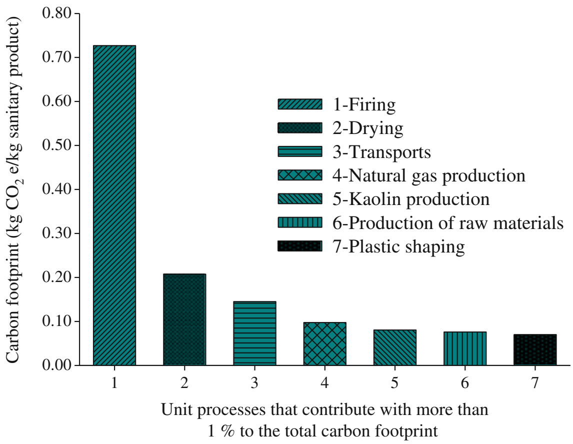
3.5.4 Results

The carbon footprint of a Portuguese sanitary ware product is 1.50 kg CO_2e per kg of sanitary product. For instance, a wash basin of 48 × 48 cm, with an average mass of 15.0 kg, has a carbon footprint of 22.5 kg CO_2e.

Figure 13 shows the unit processes that contribute more than 1 % to the total carbon footprint of sanitary ware products. The firing unit process emerges as the largest contributor to the total carbon footprint of 1 kg of sanitary product, with 49 %. The drying is the second unit process that contributes the most to the total carbon footprint of 1 kg of sanitary product, with 14 %. Both unit processes assume the primordial role due to natural gas consumption. The sanitary ware industry follows a sustainable development policy, having incorporated the majority of BAT in their manufacturing system. Therefore, an analysis of improvement measures was not carried out.

Although there is a general lack of published studies concerning the quantification of the carbon footprint of sanitary ware, we can compare the results

Fig. 13 Unit process that contributes more than 1 % to the total carbon footprint of 1 kg of sanitary product



obtained in this study with the study by Kaleseramik (2012), which estimates the carbon footprint of sanitary ware manufactured in Turkey. The carbon footprint of a sanitary ware manufactured in Turkey is 1.34 kg CO_2e per kg of sanitary product (Kaleseramik 2012), which is slightly lower than that obtained for the sanitary ware manufactured in Portugal. The system boundaries considered in both studies are similar. Therefore, this slight difference in carbon footprint results can be explained by different technologies used, as well as different transport profiles.

4 Discussion

The carbon footprint of ornamental earthenware pieces, bricks, roof tiles, wall and floor tiles, and sanitary ware products were quantified. In addition, hotspots across the life cycle of ceramic products were identified. Moreover, for earthenware pieces and bricks, some improvement measures and BAT were identified and evaluated. For the remaining products, the ceramic mills analyzed had already installed the majority of the BAT suggested for the ceramic industry (European Commission 2007).

4.1 Specific GHG Emissions of Ceramic Products

Table 7 shows the specific GHG emissions (GHG emissions per mass of product) of each ceramic product analyzed, considering a cradle-to-gate approach. The earthenware piece emerges as the ceramic product that is responsible for the highest specific GHG emissions (i.e., 2.87 kg CO_2e per kg of ceramic product) because its manufacturing leads to the highest specific energy consumption (natural gas plus electricity). This type of product requires several firing cycles for the manufacture of one piece. After the biscuit and glost firing cycles, the piece needs to be retouched, undergoing a new or even two further glost firing cycles. Therefore, to increase energy efficiency, the development of new technologies, allowing the manufacture of ornamental products with a single fired glaze, was identified as an important issue to reduce their carbon footprint and has been the subject of scientific research.

Table 7 Specific GHG emissions of the ceramic products under analysis

Product	Specific GHG emissions (kg CO_2e/kg)
Ornamental earthenware pieces	2.87
Sanitary ware products	1.50
Wall and floor tiles	0.58
Roof tiles	0.31
Bricks	0.12

The sanitary ware products ranked second with regard to specific GHG emissions. This can be explained by the process specific requirements, namely the temperature profile, in which the maximum temperature is higher than for the other products studied. Although their manufacture only requires a single firing cycle, it is common practice to retouch pieces that present some imperfections, leading to the need for a second firing cycle, which represents additional consumption of energy and, consequently, GHG emissions.

The remaining ceramic products have lower specific GHG emissions, as they are manufactured using only a single firing cycle. Moreover, brick has the lowest specific GHG emissions (i.e., 0.12 kg CO_2e per kg), because it is the ceramic product that requires the lowest temperature profile for the firing unit process. Wall and floor tiles present higher specific GHG emissions than bricks and roof tiles because their manufacture requires an additional process of spray drying that consumes natural gas and requires a higher temperature for the firing unit process (Fig. 10).

4.2 Contribution of Manufacturing Stage to the Carbon Footprint of Ceramic Products

The manufacturing stage is the stage responsible for the largest carbon footprint of all the ceramic products investigated (Table 8). The manufacturing stage of the earthenware ceramic piece represents almost 90 % of the total carbon footprint when a cradle-to-gate approach is considered. This significant contribution can be explained by the several firing cycles needed to manufacture the piece. For the remaining ceramic products, the manufacturing stage presents a contribution ranging from 73–89 % of the total carbon footprint of each ceramic product. Although the sanitary products present the second highest specific GHG emissions, its manufacturing stage has the lowest contribution to the total carbon footprint of ceramic products. This can be explained by the fact that in studied sanitary ware, the distance traveled to deliver raw and ancillary materials to the ceramic mill is significantly higher than in the other ceramic products analyzed, which results in

Table 8 Contribution of the manufacturing stage to the total carbon footprint of ceramic products, considering a cradle-to-gate approach

Product	Raw and ancillary materials stage (%)	Manufacturing stage (%)
Ornamental earthenware pieces	11	89
Sanitary ware products	27	73
Wall and floor tiles	12	88
Roof tiles	16	84
Bricks	19	81

higher GHG emissions; this contribution (10 % of the total carbon footprint of sanitary ware) is considered in the raw and ancillary materials stage.

4.3 Improvement Measures and BAT

Although the analyzed ceramic subsectors have been focusing on environmental and economic sustainability by incorporating improvement measures and BAT in their manufacturing processes, this study identifies a few further improvement measures and BAT that could be implemented for ornamental earthenware pieces and bricks.

For earthenware pieces, one of the suggested BAT consists of the incorporation of a gas pressure control system into the shuttle kilns. This measure would result in a decrease of 3 % in the total carbon footprint of a ceramic piece. Another measure is the recovery of excess heat from kilns (using heat exchangers), which could decrease the total carbon footprint up to 10 %. Both measures appear to be economically sustainable as they present simple pay-backs shorter than 3 years (European Commission 2006), as explained in Sect. 3.1. The optimization of the lighting system was also analyzed. However, although requiring a lower investment cost, its implementation would result in a simple pay-back that is two times more than what is required to consider this measure profitable. In addition to these measures, experimental tests were performed in order to optimize the temperature profile of the biscuit firing cycle. However, this measure was disregarded because it results in an increase in nonconforming ornamental pieces.

For bricks, the switch from natural gas to biomass leads to a reduction of 55 % in the total carbon footprint of brick. However, the use of biomass in brick mills depends on the long-term availability of forestry residues, as explained in Sect. 3.2. Also, the economic sustainability of this BAT still needs to be assessed.

It is not feasible to apply the switching of energy sources to the other ceramic products analyzed, due to product quality reasons. Biomass burning results in higher dust emissions than natural gas burning. Some of these dust emissions would become lodged into the kilns, increasing the number of nonconforming products during the firing cycles. Furthermore, there are some technical constraints to maintaining a constant temperature during the firing cycles.

4.4 Cradle-to-Gate and Cradle-to-Grave Assessments

In order to understand the repercussions of considering only part of the life cycle (cradle-to-gate) or the full life cycle of the product (cradle-to grave) in the carbon footprint results, a complete assessment of cradle-to-grave life cycle of bricks and wall and floor tiles was also performed, in addition to the cradle-to-grave carbon footprint study for the ornamental pieces as explained in Sect. 3.1. All the considerations explained in Sects. 3.1, 3.2, and 3.4 relating to ornamental pieces, bricks, and wall and floor tiles, respectively, are valid for the cradle-to-grave studies. Beyond these considerations, in the cradle-to-grave studies, it was assumed that bricks and wall and floor tiles are distributed within 100 and 500 km respectively, whereas ornamental pieces within 250 km. The distribution stage includes the GHG emissions from transport by truck of the ceramic products to the point of retail and by the production of diesel used in this transport.

During the use stage of these three products, it was considered that no energy consumption or GHG emissions occur. However, in practice, the cleaning of ornamental and wall and floor tiles could emit GHG, but these emissions were excluded due to the high uncertainty related to the type of detergent used, and the times and frequency of cleaning.

The final disposal (end-of-life) of ceramic products was considered to be landfill. In this stage, the GHG emissions include those arising from the landfill, truck transport of ceramic products to the landfill, and production of diesel used in this transport.

Table 9 presents the contributions of each stage to the total carbon footprint of ornamental pieces, bricks and wall and floor tiles when cradle-to-gate and cradle-to-grave approaches are applied. The carbon footprint of these ceramic products increased by 2–14 % when compared to the carbon footprint results using a cradle-to-gate approach. In the case of ornamental pieces and bricks, the distribution stage represents 2 and 3 % of the total carbon footprint of the piece, respectively, whereas in wall and floor tiles, the distribution represents 11 %. This higher contribution than the ornamental pieces and bricks can be explained by the higher distances traveled, as referred to above. It should be noted that even in the cradle-to-gate approach, the manufacturing stage appears as main hotspot, in which environmental measures and BAT should be a priority.

Table 9 Carbon footprint of some ceramic products following cradle-to-gate and cradle-to-grave approaches

Carbon footprint of products	Cradle-to-grave					
	Cradle-to-gate		Gate-to-grave			Total
	Raw and ancillary materials stage (%)	Manufacturing stage (%)	Distribution (%)	Use	Disposal (%)	
Ornamental pieces (kg CO_2e per piece)	0.13 (10)	1.06 (87)	0.02 (2)	0	0.009 (1)	1.22
Bricks (kg CO_2e per brick)	0.10 (19)	0.41 (77)	0.014 (3)	0	0.0079 (1)	0.53
Wall and floor tiles (kg CO_2e per m^2 of tile)	1.35 (11)	9.94 (77)	1.37 (11)	0	0.17 (1)	12.83

5 Challenges in Calculating the Carbon Footprint of Ceramic Products

During the quantification of the carbon footprint of ceramic products, the practitioner deals with several methodological aspects and questions, such as:

(1) specifying the cut-off criteria
(2) collecting primary and secondary data during the inventory
(3) determining how to treat multifunctional and allocation procedures

The cut-off criterion decides which processes should be included within the system boundary. In all the case studies presented, a mass criterion was applied, wherein the mass flows that represent less than 0.5 % of the functional unit were excluded from the system boundaries. However, the selection of cut-off criteria can be a challenge that needs harmonization. On the one hand, ISO 14040:2006 and ISO 1044:2006 do not define any mass, energy, and environmental criteria thresholds. This can hinder or make difficult comparisons between products. On the other hand, applying only a mass criterion can lead to the exclusion of important inputs; that is, an excluded input mass flow can encompass significant energy consumption and GHG emissions. Therefore, a general understanding of how to correlate mass, energy, and environmental significance to define an ambiguous cut-off criterion remains a challenge.

The inventory data can be one of the most labor- and time-intensive stages of carbon footprint quantification (Finnveden et al. 2009). Collecting primary data for a specific product can be a challenging task due to the confidentiality measures imposed by mills, as well as due to the absence of intermediate sampling locations along the manufacturing stage for measurements of mass, energy, and emission flows related to the manufacture of the product under analysis. Although there are databases to facilitate the inventory when primary data are not available, the majority of databases are based on average data representing the average environmental burdens for manufacturing a product (Finnveden et al. 2009), leading to a high uncertainty in the inventory. In addition, databases covering the several ceramics subsectors are still lacking.

Multifunctional processes occur when several co-products are manufactured within the same unit process. Although there is a recommended hierarchy of procedures (ISO 2006b) to attribute GHG emissions to a certain product, allocation is still scientifically a challenge. The ceramic subsector is not an exception. For some manufacturing processes, such as ornamental earthenware pieces, the application of a single allocation criterion does not seem appropriate (Quinteiro et al. 2012b); thus, it is necessary to develop and apply a hybrid approach, as explained in Sect. 3.1. This hybrid approach can be applied to other ceramic subsectors, such as sanitary ware. However, some adaptations should be performed due to the specificities of the different manufacturing processes, which involve on-site tests and measurements. Due to this “constraint,” the carbon footprint estimated in this study for the remaining ceramic products was based on a single mass allocation criterion. However, it should be noted that when a mass allocation is adopted, the inventory data should be collected for an annual temporal basis to guarantee that the GHG emissions are not under- or overestimated due, mainly, to fluctuations in the ceramic products load during firing unit processes.

6 Conclusions

The main conclusions drawn from this study are as follows:

- The product carbon footprint is a strong tool to aid the ceramic industry to better understand the GHG emission of their products and identify GHG emissions hotspot processes and improvement measures to reduce the carbon footprint of ceramic products, thereby promoting the energy efficiency and competitiveness of ceramic mills.
- Direct measurements in mills increase the accuracy of product carbon footprint results because they decrease the need to collect secondary data from databases, which represent an average or general measurement of similar processes or materials.
- The manufacturing stage emerges as the main contributor to the total carbon footprint of ceramic products, with the firing unit process being the hotspot for all the ceramic products studied.
- The ornamental earthenware piece has the highest specific GHG emissions, whereas the brick has the lowest specific GHG emissions, due to the requirement of different numbers of firing cycles and temperature profiles.
- All improvement measures and BAT should be assessed from an environmental, technical, and economic point of view. Moreover, the trade-off between improvements measures and BAT and the quality of the ceramic product should be assessed. For instance, in the performed carbon footprint calculation of the ornamental earthenware, the optimization of the biscuit firing cycle was disregarded because it leads to an increase in nonconforming pieces.

Although some core challenging questions dealing with the harmonization of the quantification of the carbon footprint of ceramic products remain, this tool is currently being used by industries for decision making, marketing purposes, and labeling as well as energy efficiency improvements.

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Carbon Footprint of Mobile Devices: Open Questions in Carbon Footprinting of Emerging Mobile ICT Technologies

Tuomas Mattila, Jáchym Judl and Jyri Seppälä

Abstract Carbon footprinting is becoming a mainstream practice in product design and marketing. At the same time, consumer products are becoming so complex that their footprinting becomes increasingly difficult. The supply chain of a typical mobile ICT device (i.e., smartphone) contains hundreds of suppliers in several continents and the product itself is composed of several complex subassemblies. The use of the smartphones also has large systemic effects (e.g., cloud computing, server load, increased consumption, and green applications), which are commonly left outside the scope of product carbon footprints. In this chapter, we argue that the parts which are most easily left out of a study are in fact the most significant for the whole product life cycle. The chapter is arranged in subchapters for each topic: components and subassemblies without emission inventory data available, energy consumption of data transfer and storage in clouds, the effect of recycling and consumer behavior, induced consumption, and the potential of green applications.

Keywords Smartphone · Cloud computing · Hybrid LCA · Consumer behavior · Information theory · Networks

1 Introduction: The Rapid Emergence of Smartphones

All new innovations typically follow a bell-shaped diffusion curve with four distinct stages: introduction, growth, maturity, and decline (Bass 1969) (Fig. 1). After introduction to the market, the product is purchased as consumers find it useful. In the early stages, marketing and external information transfer drives the increase in

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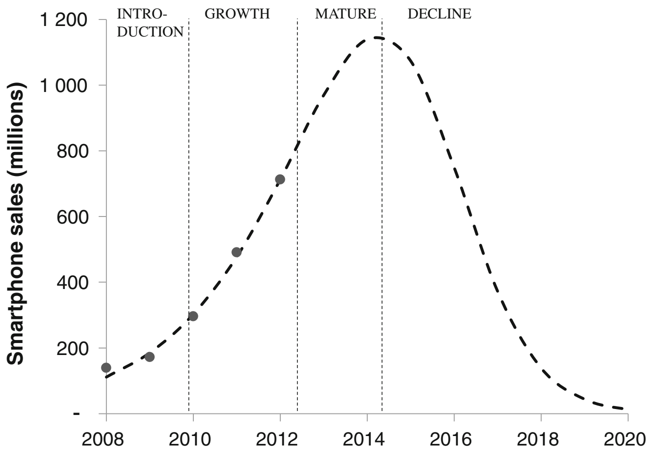

Fig. 1 World smartphone sales forecasted with a Bass diffusion model fitted to publicly available sales data (Gartner 2012; IDC 2012). The four stages represent the general stages of a product market diffusion

market. As more and more people use the product, the innovation spreads through word-of-mouth and social network effects (Peres et al. 2010). This results in a stage of exponential growth. At some point (usually around half of the market saturation), the product life cycle matures, growth ceases, and sales begin to decrease. During the decline stage, new consumers are found through cheaper versions of the product.

Most carbon footprinting studies have been done on products that are already in mature or declining markets. Examples are different shopping bags (Mattila et al. 2011), food items (Röös et al. 2010), waste disposal (Villanueva and Wenzel 2007), and paper production (Gaudreault et al. 2010). Carbon footprinting can then identify ways to improve efficiency and drive down both costs and climate impact. With mature products, companies have information collected about the production processes and consumer behavior, relatively stable supply chains, and reliable background inventories available. With emerging products, this is rarely the case.

Assessing the carbon footprint of emerging technologies presents a unique set of challenges. First, the inventory data available for emerging products are usually from products during their introduction stage, and the production processes are likely to change with the mass production of growth stage and the cost cutting concepts of decline. With new ICT products, many of the components are also so new that reliable emission inventories for the components and subassemblies (such as neodymium magnets or touchscreens) are rarely available from databases. The carbon footprint inventory of any rapidly developing product is therefore going to be incomplete and obsolete.

Second, the growth stage of an emerging product will also require the growth of the surrounding infrastructure. For the case of mobile ICT technologies, this will mean networks and cloud computing as well as the recycling of rare earth minerals. Finally, a successful emerging consumer product will also affect consumer behavior. These consumer effects during the use phase of the product can be considerable and largely unknown during the initial stages of product emergence.

In this chapter, we will illustrate the problems of assessing emerging products with the example of smartphones. Smartphones are mobile communication devices that have an inbuilt operating system and are capable of running a diverse set of applications. Introduced to the world market after 2006, they are still in the initial stages of the product life cycle with rapidly growing sales (Fig. 1). The rapid increase in their use has also influenced data transfer and the infrastructure needed for cloud computing. Section 2 will consider the problems associated with finding reliable inventories for subassemblies and components of a completely new product. Section 3 discusses the external impacts to networks and servers. Section 4 discusses the challenges of recycling. Sections 5 and 6 discuss consumer behavior from two distinct viewpoints. Section 5 discusses the effects of increased consumption due to increased market information availability and Sect. 6 presents the potential of green applications for reducing climate impacts.

Typically, when a product has matured and several carbon footprints or life cycle assessments have been made, specific product category rules (PCRs) can be made on how to assess certain types of products. Because smartphones are still so new a product, these PCRs are not available. Consequently, the available carbon footprints of smartphones (HTC 2012; Nokia 2012; Apple 2012) have been made with general PCRs for electronics (see Table 1). Consequently, they include only the energy used by the product itself, but not the effect of data transfer and distributed computing services, which form a crucial part of the product usability. In a sense they are not in line with the publicly available specifications of carbon footprints (Sinden et al. 2008), which recommend containing all relevant services needed for the use of the product in the product carbon footprint. This is caused by the rapid development in technology; therefore, more research is needed to make the new PCRs for mobile communications devices which are directly linked to the Internet, such as smartphones.

The aim of this chapter is not to provide a cookbook for assessing ICT devices but to provide a roadmap for identifying problematic issues and to suggest possible ways around them. The suggestions are based on the authors' own experiences in constructing a life cycle assessment of smartphones in the Prosuite EU project (www.prosuite.org).

2 How to Get a Reliable Life Cycle Inventory for Components and Subassemblies?

The carbon footprints of smartphones published by different manufacturers in their environmental product declarations vary by a factor of five (Table 1). The largest difference is between Apple iPhone 5 and the Nokia Lumia 920. This difference is

Table 1 Reported carbon footprints (kg CO_2 eq./phone) of three smartphones, disaggregated to life cycle stages (Apple 2012; HTC 2012; Nokia 2012)

	Apple iPhone 5	Nokia Lumia 920	HTC Sense
Production	57	11	
Use	14	3	
Recycling	2	0,2	
Transport	3	2	
Total	75	16	33

mainly caused by the production of the device itself, and it can to some extent be explained by the different material composition. The iPhone 5 is made with an aluminum and steel body, whereas the Lumia is made from polycarbonate. However, because primary aluminum has a carbon footprint of approximately 12 kg CO_2 eq/kg (Ecoinvent 2010), the material selection of the 40 g body is of minor importance, compared to the overall difference between iPhone and Lumia production emissions (46 kg CO_2 eq.). Most likely, the difference is caused by different assumptions concerning the life cycle emission inventories of the electronic components and subassemblies. For example, the emission intensity of printed wiring boards is about 280 kg CO_2 eq./kg and for a microprocessor it is around 1,000 kg CO_2 eq/kg (Ecoinvent 2010). Therefore, minor differences in assumptions regarding these components may have a large difference in the overall results.

With a complicated product such as a smartphone, the collection of the primary inventory is a considerable task. The main option for an analyst without access to full production data from the subcontractors is to perform a manual disassembly of the product. The manual disassembly consists of disassembling, weighing, and identifying as many of the components as possible. Web searches of product codes found from individual components can then be used to identify the manufacturer and possibly the material composition of each component. These are then grouped and linked to available background information on component carbon footprints.

Without access to the primary data of the subcontractors, production processes, the analyst has to rely on background data for the components. The most commonly used database, Ecoinvent 2.2, contains 122 life cycle inventories for electronic components and modules (Ecoinvent 2010). Many of them are on a general level, such as “integrated circuit, IC, logic type” or “transistor, unspecified, at plant.” The task of the analyst is then to find the most appropriate inventory item for estimating the upstream impacts of each component. Because of the limited amount of available life cycle inventories, this stage requires aggregating the individual subassemblies, which introduces further error to the assessment.

The manual disassembly and individual identification is time consuming and error prone. It is difficult to identify many of the subassemblies and a misclassification of a component might have considerable impacts on the whole analysis. For example, all of the neodymium in a smartphone is contained in the vibration unit, which on the outside will look like a small black cube connected to a rod. If it gets grouped into general electronic components, a large share of the emissions associated with the mining of very rare elements is ignored.

Some of the efforts of manual disassembly can be avoided if a teardown report of the product can be obtained. These are commercially available for many new products from vendors such as IHS ISuppli. A teardown report will show a step-by-step disassembly of the product and identification of the subcomponents. Some teardown reports also present the costs of the identified components and a bill of materials. Intended for product developers and electronics repair companies, the teardown reports can be used as a basis of the primary inventory data for an emerging product. However, teardown reports are usually available only for products that are entering the growth stage.

From the viewpoint of product carbon footprinting, in both manual disassembly and teardown reports there are two critical issues to be discovered: the number of layers in the printed circuit board (PCB) and die area of the processor. The printed circuit board and the integrated circuits of the processor are the main contributors to electronics carbon footprint (Williams 2011) and they are dependent on the layers of the wiring board and the actual "die area" of the circuits. Processors contain silicon wafers, on which integrated circuits are etched. These etched areas are then called dies and are covered in protective casing and wiring to produce processors for electronics assembly. The manufacturing of the high-purity silicon is the most energy-intensive stage of the process and determines the environmental burden of the processor. Unfortunately during disassembly, the die area can be determined from within the processor only through x-ray microscopy. Therefore, for carbon footprinting purposes, either care must be taken to obtain a teardown report with the die area identified, this analysis has to be conducted separately, or a considerable amount of uncertainty has to be tolerated in the inventory.

With new and emerging products, many of the subassemblies do not yet have life cycle inventories. Examples of these would be the touchscreens and the new generation of processors found in smartphones. A proxy for the missing inventory item has then to be used. In the life cycle assessment (LCA) literature, hybrid-LCA based on economic input–output data is often recommended to fill the gaps in the process-based LCA (Suh et al. 2004; Lenzen and Crawford 2009). Previously, the economic input–output data have often been outdated and based on very aggregated results (i.e., electronics instead of communications equipment) (Lenzen 2001). With the introduction of new multiple region input–output models (MRIO), these problems have been largely avoided. The EORA database (www.worldmrio.com) contains, for example, carbon footprints for products manufactured in China with a resolution of 123 industries and from the year 2009.

However, for the purposes of high end new electronics, even the new disaggregated models may be too aggregated. Using the EORA data, the carbon footprint of $1 worth of communications equipment from China is about 1.4 kg CO_2 eq. The bill of materials for a low end smartphone is approximately $150, amounting to 211 kg of CO_2 eq. This figure is almost four times the amount reported for the iPhone 5 and almost twenty times that reported for the Lumia. The error is caused by the aggregation of low-volume/high-value and high-volume/

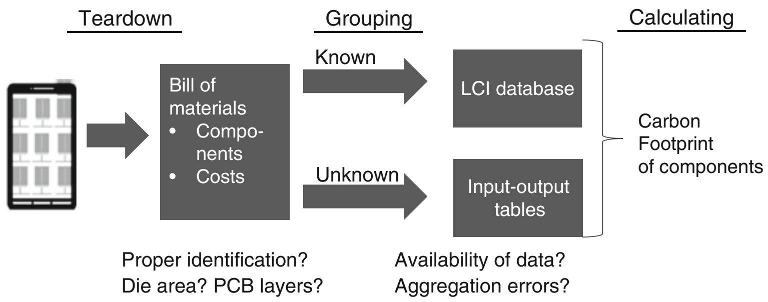

Fig. 2 An outline of the process for obtaining a carbon footprint for a new electronic device. The questions represent critical questions for minimizing the uncertainty of the study

low-value goods in the sector of communication equipment. The sector includes goods such as modems, cables, televisions, radios, and landline telephones, which have a large volume and a low value. The sector also includes smartphones; therefore, if the burdens are allocated to final products based on their economic value (as is commonly done in input–output analysis), the high-value products will get a disproportionate share of the total burden. Due to the aggregation errors in assessing high-value electronics, the focus should be on actually identified components and input–output analysis should be only used to estimate the magnitude of those components, which cannot be identified.

Figure 2 presents and overview of the recommended procedure for constructing and inventory for a new electronic device, when a company's inside information is not available. The main sources of uncertainty and critical questions are also presented. Because the main causes of uncertainty are due to human nature (identification and choices of aggregation), the process should be extremely well documented to maintain a level of transparency in the end results. Unfortunately, this is not commonly done, resulting in several factors of variation in smartphone carbon footprints, although the same methodology (ISO standardized LCA) has been used for all products (Table 1).

3 What are the External Influences to Networks and Servers?

In the published environmental product declarations of smartphones (Table 1), the use phase is dwarfed by the large emissions of manufacture. This could indicate that the role of the user is not important in the overall life cycle. However, the environmental product declarations include only the direct electricity consumed by the device during its use. In reality, a smartphone requires a considerable amount of infrastructure to provide its services. Networks and data centers are required to provide internet access and to allow phone calls. Data are more commonly stored

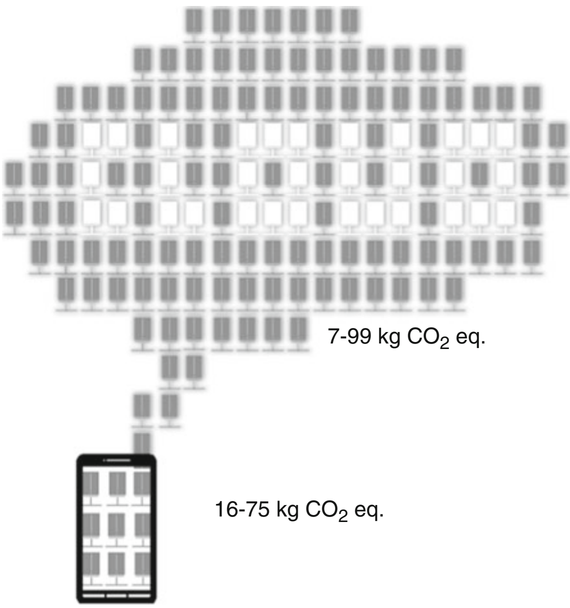

Fig. 3 The services and data storage properties of a smartphone are currently increasingly being covered outside the actual device in "cloud" storage and computing. See text for details on calculating the footprints for each part

outside the phone and, also in an increasing manner, computation is done outside the device. For example, Google provides navigation services with Google Maps, which operate on distant servers. This "cloud computing" infrastructure cannot operate without energy and forms an integral part of the life cycle energy demands of mobile ICT technologies.

Figure 3 presents an illustration of a smartphone as a product, which is only partially represented by the physical phone. Most of the services required by the smartphone user are provided by the infrastructure outside the actual product. This requires a radical rethinking of the scope in environmental product declaration. If the product is useless without the infrastructure, then the infrastructure is an essential part of the product and should be included. Unfortunately, very few studies have been done on the carbon footprint of server computing and data transfer.

The order of magnitude of data transfer can be captured with a straightforward calculation based on published results. An average smartphone produced in 2012 342 MB of data traffic per month (Cisco 2013). Using published estimates for internet traffic electricity demand (7 kWh/GB) (Weber et al. 2010), this would amount to 86 kWh over the 3 years considered in the environmental product declarations (Table 1). Assuming similar carbon intensity for electricity than used in China (Ecoinvent 2010), the carbon footprint of this data traffic would be 99 kg CO_2 eq. This figure is larger than the rest of the carbon footprint (75 kg CO_2 eq.) reported for an iPhone in the environmental product declaration (Apple 2012) and an order of magnitude higher than the figures reported for the use stage (Table 1).

In addition, the data transfer of smartphones has increased 81 % per year (Cisco 2013). With development in technology, the data transfer has increased also nonlinearly. Modern tablets produce twice as much data transfer as smartphones and 100 times the data transfer amounts of regular mobile phones (Cisco 2013). Also, an increasing amount of the data transfer is associated with cloud storage (uploading data for remote storage). With cloud storage, there is a substantial amount of "indirect" data transfer as data are also stored in backup copies, refreshed, and kept ready for immediate access.

Compared to local data storage, cloud storage will also increase energy consumption, even when data are not transferred. With local data storage on a computer hard drive, the hard drive is operating only when the computer is on and data are retrieved. In cloud storage, in order to maintain instantaneous access, the hard drives have to be on constantly. Also, the centralized hard drive racks require active cooling, which typically consumes more energy than the local hard drive, which can be passively cooled (Greenpeace 2013).

In spite of these hindrances in cloud storage and computing, there are some results that might suggest that cloud storage in a public cloud can be the most efficient way to provide services to a computer user. This is due to high utilization rate in servers and the rapid rate of updating servers to more efficient models (NRDC 2012). The range in data service provision per one office user per year ranged from 1 to 45 kg CO_2 eq. depending on the extent of outsourcing and modeling choices. The range in cloud computing was 1–15 kg CO_2 eq., making it plausible to claim that cloud computing may reduce overall emissions (NRDC 2012).

Although cloud computing may reduce the emissions of existing computing needs, it also has a great potential to increase computing needs, therefore offsetting the positive development. For example, the possibility of mobile access to video has increased data traffic considerably, with more than half of all internet video traffic now being associated with mobile devices (Cisco 2013). These aspects are problematic for carbon footprinting because they depend greatly on consumer behavior, which is the topic of the next two sections on recycling and on induced consumption.

4 What is the Effect of Consumer Behavior on Recycling?

Usually in product carbon footprinting of electronics, a full recycling rate is assumed (Apple 2012; Nokia 2012). In reality, however, only 5–12 % of the phones are reportedly recycled (Table 2). This has several implications for the product carbon footprint.

First of all, what has happened to the phones that were not recycled? Most of them have been initially stored as a spare, but eventually some of them might have ended up in waste incineration. In that case, the approximately 30 g of plastic in the phone is oxidized to carbon dioxide, resulting in an increase of 0.07 kg CO_2 eq. per phone. Compared to the overall carbon footprint of the product, the amount of (carbon dioxide) emissions from combustion of plastics is insignificant.

Table 2 Results from a global study on what people had done with their previous phone (Tanskanen 2012)

	Developed countries %	Developing countries %
Kept as a spare	40	32
Gave to friend/family	18	24
Traded for a new phone	9	16
Lost/stolen/broken	7	17
Recycled	12	5
Other	14	6

A more considerable effect is caused by the resources lost by non-recycling. The difference between primary and recycled aluminum for example is approximately 11 kg CO_2 eq./kg. The loss of aluminum from recycling would then increase the carbon footprint of an iPhone 5 by 0.23 kg CO_2 eq., which is again quite insignificant compared to the whole footprint. For steel and plastics, the difference between primary and recycled secondary materials is even less, so overall the lost resources are not a key issue for carbon footprint.

An exception might be the rare and precious metals, such as gold, silver, platinum, indium, and neodymium. Typical amounts found in a smartphone are presented in Table 3, together with an estimate of the potential emission savings from their recycling. Overall, the benefit from recycling could be approximately 1 kg CO_2 eq., which would be of same order of magnitude as the emissions caused by the recycling activity (Table 1). Most of the recycling benefits would come from the recovery of gold, silver, palladium, and copper. Of these, gold is the most significant, covering more than half of the overall benefit. No emission figures were available for neodymium, but neodymium is also not currently recovered in recycling (Reck and Graedel 2012). The potential emission saving is therefore not completely known, but would seem to be of such a scale that recycling should be better included in carbon footprint studies, preferably based on actual recycling behavior.

Table 3 Typical amounts of metals found in a smartphone (Villalba et al. 2012) and the potential benefit of their recycling (Ecoinvent 2010). All numbers are per one smartphone

	Amount in a typical smartphone (g)	Potential savings from recycling (g CO_2e)
Copper	13	62
Nickel	1.5	14
Tin	1	17
Silver	0.37	157
Gold	0.035	623
Palladium	0.015	139
Platinum	0.0005	7
Indium	0.006	1
Total	16	1,013

The benefits of recycling could be increased if whole components could be reused. Currently, the printed circuit boards, memory modules, and processors are the main cause of product carbon footprint. If the products would be designed for active disassembly and component reuse, some manufacturing of electronic components could possibly be avoided. This is a key idea in so-called cradle-to-cradle design (Braungart et al. 2007). Nokia has experimented on the idea since the year 2000, but the concepts have not gotten to mainstream production (Tanskanen 2002). If implemented, active disassembly would enable removing the valuable components and subassemblies from a recycled smartphone in two seconds. This would allow the reuse of components, which are not affected by the product age or do not develop at a very rapid pace.

Overall, the uncertainties associated with recycling play a minor role in the smartphone life cycle compared to the impacts of data transfer or the inventories for missing components. If recycling could be restructured into reuse of components, it might have a considerable effect on the overall lifecycle. Also, the overall benefits of recycling can only be quantified when data on the rare metal (especially neodymium) mining and recycling is available.

5 How to Account for the Increases in Other Personal Consumption?

Since their introduction and especially after they have reached the growth stage, smartphones have changed the society in a number of ways. In a recent survey, *Time* magazine asked people around the world how mobile technology has changed their lives. Among the responses, a few are of importance for product carbon footprint: mobile phones have made it easier for businesses to reach customers and they have made doing business more efficient (*Time* magazine, 27 Aug 2012).

Both of these effects are caused by the fact that smartphones are *information* devices, and information is a powerful tool in increasing consumption and improving markets. Most of the traditional economic models were constructed on the assumption of perfect markets and perfect information. More recently, the behavior of markets under imperfect information has become a research topic, resulting in research on information economics (Stiglitz 2002).

Under perfect information, markets operate on supply, demand, and pricing. Increased demand for a product drives up prices, which increases supply until all demand is met. Under imperfect information, the consumers are not aware of all of their options and obtaining knowledge through research costs time and money. Many operators take the benefit of this information asymmetry by having higher prices than could be maintained if customers would know the whole market. (An example would be a hotel breakfast, which has a much higher price than what is usually found in any of the restaurants within a few hundred meters of the hotel.) In this sense, by withholding information, companies are able to keep consumers in a state where they do not know about competing alternatives (Stiglitz 2002). This is why most companies participate in marketing and some large companies try to control marketing. It is also why information technology has had such an influence on consumption and production behavior.

The effects of smartphones on the economy can be explained with a simple model with two variables: information and available income. From the viewpoint of consumption, information can be seen as a resource. With no information about available products and services, there is no consumption. Also, with no consumable income, there is no consumption. When income increases, consumption increases only if the information about available consumables increases. With increasing income level, the level of consumption finally becomes limited by the availability of information. In the case that a consumer does not know of any additional products that would increase his subjective welfare ("utility") more than additional savings, the consumer will save the money for later consumption.

Smartphones and ubiquitous access to the internet are removing the constraints of information by providing a large and low-cost connection between producers and consumers. Consumers can compare options in the global market, therefore ensuring near-perfect information (assuming that search engines and filters of misinformation keep up with the development).

Some life cycle assessment practitioners have included the rebound effects of consumption to the analysis by looking at the impact on available income from the purchase of a product (Finnveden et al. 2009). This consequential LCA is usually done on money- or time-saving products in order to see whether the net benefit is lost by increases in consumption. Taking the information component into account, the purchase of a smartphone can either increase or decrease personal consumption, depending on where on the curve of Fig. 4 a consumer is. Figure 5 illustrates this issue further. The purchase and use of a smartphone will always move a

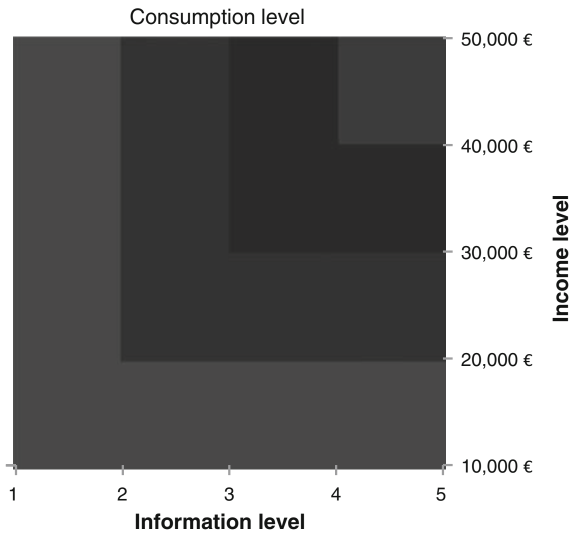

Fig. 4 A simple conceptual model for relating consumable income, market information, and consumption level

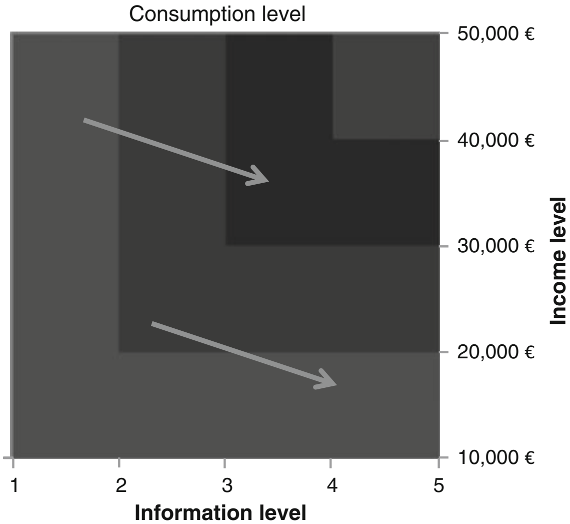

Fig. 5 An example of a smartphone either increasing or decreasing consumption, depending on the initial values of information and income levels for each consumer

consumer to a certain direction in the curve. With the purchase, money is consumed, so less money is available for consumption. At the same time, exposure to mobile media increases information about markets and consumption opportunities. If a consumer has a relatively high income but a low market information level, the purchase of a smartphone will increase overall consumption because more consumption opportunities are offered (Fig. 5, upper arrow). With lower income level and higher information level, the purchase of a smartphone may decrease consumption, as less is available for other purchases.

The effect of consumption can be considerable. The carbon footprint of an average European is 13 t CO_2 eq./capita (Steen-Olsen et al. 2012). A moderate 5 % increase would amount to 650 kg CO_2 eq., or almost an order of magnitude higher than the carbon footprint of an iPhone 5. Of course, it can be discussed whether the increased information can be allocated only to the smartphone or if the increased consumption can be tracked to improved information. However, generally it can be stated that mobile communication improves information flow and that increased information improves the functioning of the markets, which results in increased opportunities for production and consumption.

From the producer's viewpoint, smartphones and mobile internet offer other kinds of opportunities. The best documented case study is the introduction of mobile phones and internet to fishermen in Kerala, India (Jensen 2007). Prior to mobile phones, the local markets were highly inefficient. Fuel costs prevented the fishermen from circulating between docks and buyers had no information about the catch available at each dock. As a consequence, 5–8 % of the fish catch was dumped because it could not be sold; at the same time, buyers had to leave other docks without enough fish. With mobile phones, the dumping was eliminated, fishermen's profits increased by 8 %, and market prices declined by 4 %. Therefore, for producers, improved market information can result in growing economic activity. In the best case, improved information increases production, which allows higher investments, which again increases production, resulting a spiral of economic growth due to better market information. Encouraged by the results in fishing industry, some ICT companies are now participating in case studies on introducing mobile internet services to fishermen to improve both market and production efficiency. Currently, the effect of mobile technology on economic development is a key research topic in economics (Donner 2008). Although currently there are no methods for quantifying the multiplier effects caused by increased information for producers, the issue should not be forgotten in carbon footprint studies. Again, a minor increase in regional consumption levels may have a carbon footprint that is an order of magnitude higher than that of the device itself.

6 Can Green Applications Offset the Other Emissions?

Mobile computing combined with GPS and social media presents very good options for improving environmentally conscious decision making. Because the main climate impacts of consumers are caused by activities other than smartphones (i.e., food, transport, housing) (Steen-Olsen et al. 2012), it is possible to offset the emissions caused by smartphone manufacture and use with so-called green applications.

The carbon footprint of an iPhone 5 is 75 kg CO_2 eq. (Table 1). This amounts to approximately 0.5 % of annual European consumers overall carbon footprint (Steen-Olsen et al. 2012). On the other hand, it corresponds to about 350 km of car driving. Many applications have been made that may reduce the distances driven with a personal car (Table 4). For example, Avego facilitates car sharing and may reduce driving by far more than the 350 km. Other applications focus on improving fuel efficiency, and on average driving habits resulting in a 2 % improvement in fuel economy would offset the emissions of manufacturing and using a smartphone.

Table 4 A sample of green applications from MyGreenApps (U. S. Environmental Protection Agency)

Application	Description	Reduction pathway
Avego, Carticipate,	Facilitates car sharing	Reduces personal transport needs
Get There by Bike	Recommends routes for urban cycling	Helps substituting cars with bikes on urban short trips
Green Gas Saver	Monitors driving performance and gives guidance	Improves car fuel efficiency through ecodriving habits
One Stop Green Mobile App	Building energy audit	Energy efficiency retrofits
Nexia Home Intelligence for iPhone	Remote controlling of home automation	Heating, cooling, ventilation, and lighting energy reduction
Locavore	Finds nearby farms, farmers markets, and seasonal food	Reduces the emissions from food consumption

Some applications aim to reduce emissions by improving home energy efficiency. This can be achieved either through environmental education for home owners concerning energy efficiency or through improved automation. Based on the energy carbon intensity in the United States (Ecoinvent 2010), a 100-kWh electricity savings would be necessary to offset the emissions of smartphone manufacture and use. This would represent approximately 0.7 % of the annual electricity consumption of an American citizen (World Bank 2013). With most of the electricity consumption related to air conditioning, improving the efficiency of house automation can have much higher emission savings than the emissions caused by smartphone manufacture.

Finally, some applications aim to educate consumers about green purchases. For example, Locavore gives information about local and seasonal food retailers based on current location. Combined with social media, this encourages people to shift their consumption habits.

Overall, many of the so-called green applications can have emission reductions that exceed the carbon footprint of the smartphone itself. However, a key issue in the potential of green applications is that not all smartphone users will use and benefit from them. For example, a large share of future smartphone users will be under 18 years old and therefore unlikely to make decisions concerning car fuel use or house ventilation. Therefore, on a larger level, it is unlikely that individual green apps would offset the whole carbon footprint of smartphones and their external impacts.

7 Summary

Overall, the carbon footprint assessment of smartphones was found to be much broader than what could be expected from the environmental product declarations. Many of the components are so new that no reliable background information about their emissions is available. In addition, the identification of hundreds of components and subassemblies is costly and difficult. Beyond the product itself, the use of smartphones requires a considerable amount of infrastructure in data transfer and storage. Based on most calculations, the external impacts are likely to be larger than the emissions of manufacturing and using the device. With the rapid development in the smartphone market, devices are becoming obsolete quite rapidly. Contrary to common assumptions, only a minor fraction of the devices is actually recycled. This results in a loss of resources, which if recovered could offset emissions in primary material production. However, the offsets are likely to be of minor importance. Smartphones and other information devices have great potential to influence consumer behavior. On one hand, the increased information will make more efficient markets and increasing both production and consumption. On the other hand, green applications can provide environmental education rapidly to a large group of consumers, potentially even offsetting the overall impacts of smartphone manufacture.

Based on this review, our recommendation is to include a quantification of the effects of data transfer through networks and data centers to all carbon footprinting studies related to smartphones or other mobile communications equipment. The data transfer seems to have a major impact on the results, so it cannot be ignored. On the other hand, the use of the smartphone may result in considerable emission savings through green applications. However, because these aspects depend largely on consumer behavior, they aspects should be treated with a high uncertainty in the analysis.

The carbon footprinting of mobile communications equipment is still developing. Even at this stage, it is very useful to identify the hotspots in the product life cycle, but the quantitative results may not be accurate until the product category rules have been defined and updated. Until that stage, it may be useful to separate the traditionally reported device manufacturing emissions from the more recent additions of data transfer and consumer behavior. The latter should not be excluded, however, because their impact dominates the overall life cycle.

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The Carbon Footprint of Pigmeat in Flanders

R. Jacobsen, V. Vandermeulen, G. Van Huylenbroeck and X. Gellynck

Abstract Although several international carbon footprint (CF) calculation initiatives have been developed, studies that focus specifically on estimating the CF of pigmeat are rather limited. This paper describes the application of a CF methodology, based on lifecycle assessment of greenhouse gas emissions, for Flemish pigmeat production using the Publicly Available Specification methodology (PAS2050, BSI 2011), which is at present the most developed method and relevant within the agricultural and horticultural sector. Both primary and secondary data have been used to model the meat system through a chain approach. The results are reported using the functional unit of 1 kg of deboned pigmeat; they range from 4.8 to 6.4 kg CO_2 eq. per kg of deboned pig meat. A sensitivity analysis has been executed on changes in herd and feed characteristics. The results have been compared to other studies on the CF of pigmeat in the EU and with CF studies on milk and beef production in Flanders. Furthermore, two major hotspots in the CF have been defined: 1) the composition and production of feed and 2) manure production and usage. It is important to mention that the CF is a good indicator for greenhouse gas emissions, but it is not an indicator for environmental impact in general. This article helps to fill the void in the CF literature that existed around pigmeat products and to define a benchmark for the CF of pigmeat.

Keywords Carbon footprint · Pigmeat · LCA · Sustainability · Hotspots

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1 Introduction

Meat is a major part of the human diet in many countries (van Wezemael 2011). The accompanied livestock production leads to substantial greenhouse gas (GHG) emissions and subsequently climate change (Johnson et al. 2007). More specifically, livestock production accounts for half of all GHG emissions related to the human diet in Europe (Kramer et al. 1999; European Commission 2009). Limiting agricultural GHG emissions is therefore particularly relevant to facilitate more sustainable development and to achieve the stabilized GHG emissions and global mean temperature targets of the 1997 Kyoto Protocol and the 2011 Durban Accord (Dalgaard et al. 2011).

To initiate this development towards more sustainable production, there is a need to analyze the current situation, as well as the potential for improving the system (Eriksson et al. 2005). To find out where along the chain of production the improvement can be achieved, all related emissions need to be quantified. One such method is the calculation of the carbon footprint (CF) of livestock products (Espinoza-Orias et al. 2011). More specifically, a CF quantifies the climate change impact of an activity, product, or service. Within the CF, all GHG emissions (with the most important ones being carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O)) are combined and expressed as CO_2 equivalents.

Of all the major sources of world meat production, pigmeat represents the highest proportion (37 %) compared to poultry (33 %) and beef (23 %) (FAO 2008). In view of the importance of pigmeat in terms of world consumption and livestock production, a study was ordered by the Flemish Government to calculate the CF of Flemish pigmeat production. Furthermore, pigmeat provides an interesting case to study because, estimations for the CF of pigmeat are rather limited, especially compared to the carbon footprinting of milk, for which examples in the literature abound (Blonk et al. 2008b; Muller-Lindenlauf et al. 2010; Sonesson et al. 2009; Thoma et al. 2010; van der Werf et al. 2009). Moreover, most studies on the CF of agricultural products are not at all clear in terms of the methodology or standard being used for calculations, the chosen system boundaries, and system definition. Eventually, people of different backgrounds are wrongly informed and draw the wrong conclusions. These different approaches in terms of methodology do not allow a fair benchmark of CF between products and sectors, because different means of calculations are being used; hence, apples and oranges are compared. The fact that each LCA has to deal with many different issues (e.g., allocation, scope, system boundaries, data, land use; Finkbeiner 2009) makes it necessary that each of these aspects is described in a proper way. In our own study (CF methodology applied to livestock produce in, we did a literature review on the state of the art in terms of existing LCA or CF studies on pig production and found that in several cases important information was missing (e.g., Dalgaard et al. (2007) and Leip et al. (2010) did not indicate the used allocation method). This problem was also mentioned by de Vries and de Boer (2010), who had to exclude sources from their meta-analysis due to a lack of data.

This chapter presents the outcomes of an estimation of the CF of pigmeat production in Flanders. We used the most up-to-date product category rules and standards.

2 Background

The results of this book chapter were obtained through a study conducted for the Flemish government, Department of Agriculture and Fisheries. The purpose of the study was to estimate the CF and furthermore identify hotspots in the life cycle of beef, pigmeat, and milk production in Flanders; however, this article focuses solely on pigmeat production. In Flanders, the concentration of pigs for agriculture is among the highest in Europe (European Commission 2008). Hence, environmental pressure from livestock production in Flanders abounds, with a major impact on climate change by large emissions of greenhouse gases (GHGs).

Flemish pig farmers produce about 10.5 million pigs per year, which is approximately 4 % of the EU yearly production (Statistics Belgium 2010; Eurostat 2012). These pigs are distributed over 5,377 farms (Statistics Belgium 2010). Half of these farms (42 %) are specialized in pig production, and they breed 95 % of all pigs (van Liefferinge 2011). Therefore, the study focused on specialized pig production.

3 Methodology

3.1 Standard and Method Used

For the study, we used the IPCC guidelines in line with the National Inventory Report of Belgium (VMM et al. 2011). Even though the IPCC directive describes the calculation of the total amount of GHG emissions, it does not mention the allocation to a particular product. To tackle this issue, a specific methodology such as PAS2050 is needed (Espinoza-Orias et al. 2011). Currently, the Publicly Available Specification (PAS2050, BSI 2011) is the most largely used and profound method; therefore, this method was chosen. Moreover, the PAS2050 was used several times before for estimating GHG emissions within the agricultural and horticultural sector. Based on the PAS2050, specific product category rules (PCR) were developed for dairy and horticultural products. In 2012, specific PCR were developed, according to the international EPD system (Environdec 2013), for mammal meat, including pigmeat, in which slaughter activities, packaging processes and storage are the core processes, (Studio LCE 2012). The upstream chain activities, namely animal production, feed cultivation, manure management, transportation, and packaging, are seen as core activities in the current study.

Based on the IPCC 2007 (IPCC, AR4 2007), the global warming potentials (GWP) for methane and nitrogen gas emissions are defined as follows: 1 kg of methane (CH_4) equals 25 kg of CO_2 and 1 kg of nitrogen gas (N_2O) equals 298 kg CO_2.

In order to benchmark and correctly interpret the estimated CF values, it is necessary to describe the assumptions in relation to the scope and system boundaries, the functional unit used, the allocation method, and whether land-use change (LUC) has been taken into account.

3.2 Scope and System Boundaries

The scope of the study is from cradle to gate. PAS2050 states that those emission factors contributing less than 1 % of the total CF are negligible (BSI 2011). The lion's share of GHG emissions occur at farm level. Therefore, the ultimate steps in the chain (distribution, consumption, transport, and waste processing) (Blonk et al. 2008b; Campens et al. 2010), are not included in the calculations of the CF. Table 1 gives an overview of the included emission sources throughout the chain.

The system boundaries were identified based on the scope of the study. The study included the GHG emissions as shown in Fig. 1. The production of materials, energy, and transport steps are taken into account. The system boundary excludes production of capital goods (machinery and equipment), which is a similar approach to most international studies.

3.3 Functional Unit

We defined several functional units, based upon a chain approach. Calculations were hence done based upon three different functional units: 1 kg of life pigmeat, 1 kg of pigmeat after slaughtering, and 1 kg of deboned pigmeat. Each one gradually adds up to the overall CF.

3.4 Allocation Method

Another important assumption describes the allocation of GHG emissions between the various by-products emerging from a single process. International standards were followed in the determination of the allocation method. These standards, however, indicate that allocation should be avoided. If not possible, a single allocation method should be used, not a combination. However, this was not possible in this case. In terms of the slaughtering and deboning process, economic allocation was used, in which the economic value of the by-products (bones, fat, skin, hide, heart, blood, etc.) represents the market prices multiplied by the mass

Table 1 Overview of emission sources within the covered system boundaries

Name	GHG	Description
Feed mixtures (purchased)	CO_2 and N_2O	Farming, transport, processing, and land conversion is taken into account in the covered emission factors
Animal	CH_4	The IPCC method is applied (Tier 2 calculation)
Manure storage and disposal	CH_4 and N_2O	The IPCC method is applied (Tier 2 calculation)
Manure application (not used for own feed mixtures)	CH_4 and N_2O	Allocation between animal (40 %) and vegetable production system (60 %) on the basis of nitrogen uptake by plants
Energy and water consumption	CO_2, CH_4 and N_2O	Energy consumption (electricity; diesel; red diesel; gas). Water consumption (tap and groundwater)
Transport of goods	CO_2, CH_4 and N_2O	Assumptions are being made for the goods entering and leaving the farm
Processing materials	CO_2, refrigerant	Use of cleansing products and refrigerants

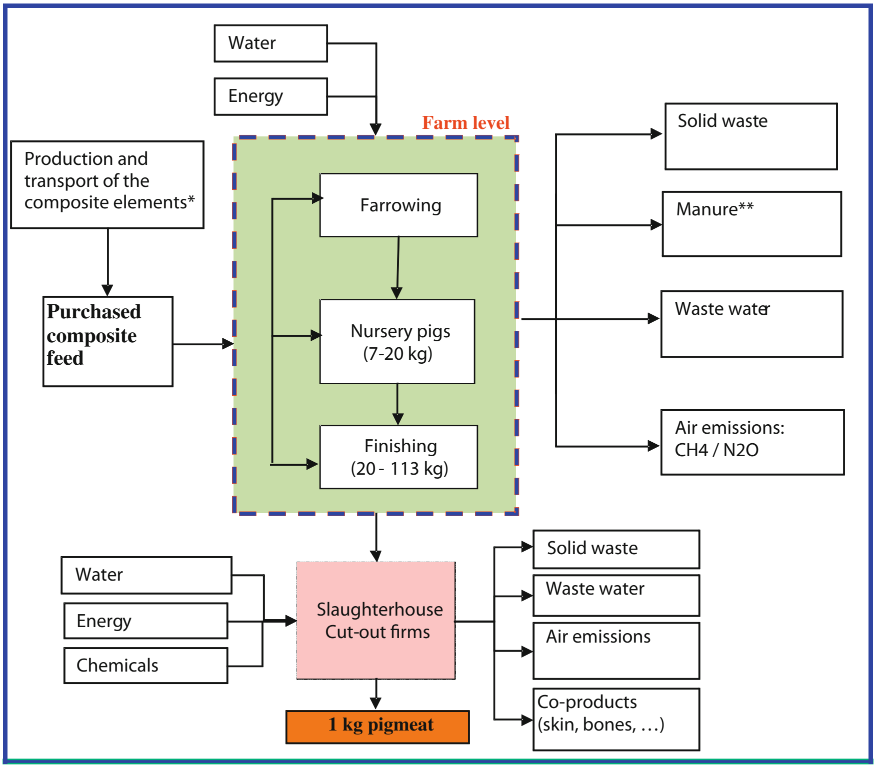

Fig. 1 The system boundaries. The boxes with *dashed lines* present a process, those with *full lines* a product flow. The *colored boxes* are the foreground system, whereas the other parts were based on generic data

fraction per incoming product. Where the byproduct has a negative economic value, the share of these by-products in the GHG allocation is considered to be zero (see Table 6). However, if one were to apply this to the manure production process, allocation to crops would be zero because of the oversupply of manure in Flanders. However, manure does contribute to the production of crops; therefore, physical allocation was used here in order to allocate the GHG production of manure between crops and animal production. Overall, in this study we used a combination of physical and economic allocation based upon several other references (Blonk 2008a).

3.5 Land Use and Land-Use Change

This chapter also takes into account the impact of LUC. Emissions of carbon dioxide due to LUC are the result of cutting down forests and using the land instead for agriculture or built-up areas, urbanization, roads etc. When large areas of (rain) forest are cut down, the land often turns into less productive grasslands with considerably less capacity for storing CO_2.

According to PAS2050, LUC should be taken into account if land conversion took place in the last 20 years. This is not the case for agricultural products in the EU, and thus LUC is zero. However, part of the feed (mainly soy and palm) is imported across the Atlantic Ocean, for which FAO statistics (2010) are used to define the LUC in the past 20 years. The total emissions from this LUC are calculated and 1/20th is attributed to each forthcoming year. The information on the emissions themselves comes from Blonk (2008a), Ecoinvent (2011), and LCA Food (Nielsen et al. 2010).

Land use as such (or carbon sequestration in the soil) is not considered in the calculations. In Flanders, considerable uncertainty remains with regard to the net effect (absorption or emissions) from land use; therefore, we opt not to include it. The international standards and guidelines for carbon footprinting also exclude it from the necessary calculations.

4 Data Sources

PAS2050 has specific rules in favour of using of primary data(BSI 2011). Whenever possible, primary data were used in the study—collected through interviews (e.g., on slaughtering and deboning data). These primary data were complemented with secondary data (e.g., on farming) and reports (e.g., on emission factors). The data were collected for the year 2009. For certain data, it was necessary to obtain more recent values, such as for the feed mixture composition, which changes daily.

An overview of the data used and its sources are given in the Annex.

4.1 Raw Materials and Husbandry Level

The data on the representative conventional pig farm were collected from the dataset of the Farmers Union (Boerenbond). The latter keeps track of economic data at the farm level. The dataset includes both farm accountancy data (such as type of farm, region, type of crops or livestock, input, production) as financial data, technical–economic data, environmental indicators (such as water, nutrients, or energy balance sheets) and a balance sheet for the farm (Boerenbond 2012). This dataset was used to select pig farms that are specialized in pig production and breed their own piglets (i.e., are closed). The average of their data was then identified in order to obtain a representative pig farm. A limited group of farms was then involved. We did not use outliers in the data; hence, the average was representing a real farm in Flanders.

Data on manure production was not available within the Farmers Union dataset and was therefore collected from reports of the Flemish Centre for Manure Processing. The average manure production per animal was linked to the number of animals on the farm (VLM 2011a).

Other data were collected from the NIR Belgium (VMM et al. 2011), Ecoinvent (2011); Statistics Belgium (2010); LCA Food (Nielsen et al. 2010), and IDF (2010).

Moreover, regarding the pig stable system, we used the most common one implemented in Flanders. During the farrowing phase, a semi-slotted floor was used, whereas for the nursery pigs and finishing phase a slotted floor was used (van Meensel, personal communication, April 24th, 2011).

Because most pig farms in Belgium do not produce their own feed compound, the latter at farm level was not included, and only concentrated feed was used for the pigs. Subsequently, the average composition of the feed concentrate was given by the Belgian Association of Feed Mixture Companies (Maes and Dejaegher, personal communication, April–October 2011). Data were delivered for different animal categories, including the origin of the composite elements (see Table 2). The composition was given for the day of October 4, 2010—(randomly chosen). The composition of feed compound varies daily according to the availability of components on the market. By not taking the annual average, one can be sure that the feed used has an appropriate composition for the animals.

The emissions associated with the production of the feed mixture are calculated based on emission factors for the resources, which were derived from Blonk Milieu Advies (Blonk et al. 2008a), LCA Food (Nielsen 2010), and Ecoinvent (2011).

Table 2 Feed components

Component	% LUC	Piglet 7–12 kg	Piglet 12–20 kg	Pig 20–35 kg	Pig 35–75 kg	Pig 75–100 kg	Sow (gestation)	Sow (lactating)
Barley	0.00	37	35	21	21	20	22	1.5
Soy meal import	75.83	5.3	14.6	13	12.5	6.4		10.8
Palm kernel import	8.15				4	5	5	
Wheat	0.00	8.7	18.1	30	35	35	34.2	35
Maize	0.00	15	12.3	11.6	13.3	9.7		10
Barley	0.00			10	7.5	15	20	6.7
Sugar beet pulp	0.00						4.5	
Soy hull import	20.16	15	10					
Blend (21 % protein)	0.00	15	7					
Sum		96.0	97.0	85.6	93.3	91.1	85.7	79.0

Note In the analysis, the numbers have been recalculated as to sum to 100 %. Soy for feed compound: 53 % originates from Brazil, 11 % from Argentina, 21 % from the US and 16 % from Canada.

4.2 Meat Processing

Data were gathered through in-depth interviews (N = 5) with the biggest slaughterhouses and cut-out firms in Flanders. The contacted slaughterhouses represent 20 % of the volume of processed pigmeat in Flanders and were seen as representative for the whole sector. Data were collected with regard to meat weight and prices, by-products and carcass, energy consumption (water, electricity, refrigerants, cleansing products), and transportation characteristics (distance, type of truck) (see Table 6).

Moreover, to fill the gap of missing data, such as price of cuts and amount of waste/meat generated through slaughtering, we contacted the Flemish Meat Federation (Febev), the Flemish Waste Authority (OVAM), and the National Institute of Statistics.

5 Data Analysis

5.1 Emissions from Fodder Production

Fodder production begins with the production of resources and transport in order to sow, grow, and harvest crops (i.e., seeds, fertilizers, pesticides, and diesel). GHG emissions associated with resource production are allocated to the crops, as well as transport to the processing plant. Emissions accompanying the processing (grinding, crushing, mixing, etc.) are also taken into account. The relevant emissions were calculated based on resource emission factors (see above) and are given in Table 3. The first column shows the annual average number of animals on the farm, divided according to their growth stage. Column 2 indicates the amount of feed consumed. Based on the information in Table 2, column 3 of Table 3 shows the resulting emissions from feed consumption. The last column indicates the importance of LUC caused by the use of soy and palm.

5.2 Emissions from Pig Production

Two types of emissions relate to pig production: energy use and animal emissions.

Because a specialized pig farm was considered, all energy consumption at the farm can be attributed to pig production. The energy consumption was not further divided into subactivities for the farm. Energy consumption per meat pig amounts to 46.4 kWh, of which 25 % is electricity and 75 % gasoline oil. Table 4 shows the related emission factors.

Pigs show a low level of methane output compared to ruminants because of a limited gastrointestinal fermentation process. The IPCC Tier 1 method (IPCC

Table 3 Consumption of fodder for the modeled pig farm per animal category (based on the exact composition, and multiplied by the number of days the pigs stay in each category) and accompanying emissions (Jacobsen et al. 2013)

Animal category	Number	kg product*	kg CO_2eq/kg product	% LUC
Sows (gestation and lactating)	219	251, 000	0.76	9.7
Piglets (<12 kg)	5,319	57,282	1.02	16.9
Piglets (12–22 kg)	5,319	92,652	1.10	19.3
Meat pigs (22–35 kg)	5,133	196,012	0.79	10.8
Meat pigs (35–75 kg)	5,133	603,116	0.79	9.9
Meat pigs (75–115 kg)	4,857	542,150	0.62	5.4

LUC = land-use change
* See Table 2 for the exact composition of the feed product.

Table 4 Emission factors for electricity and gasoline oil in Belgium and the emission value for animal exhaustion of pigs (Jacobsen et al. 2013)

Name	Value	Unit	Source
Electricity	0.40	kg CO_2eq/kWh	Energy convenant
Gasoline oil	2.66	kg CO_2eq/kg	Energy convenant
Animal exhaustion	37.5	kg CO_2/pig/year	IPCC

2006a) was used to calculate methane (CH_4) output. The IPCC Guide for National Inventories states that each pig yearly emits 1.5 kg of CH_4 (IPCC 2006b, Table 10.10). This is converted to kg CO_2 eq. taking into account the Global Warming Potential of methane (25 kg CO_2 eq/kg CH_4) and included in Table 4.

5.3 Emissions from Manure Storage and Usage

Pig manure is stored as manure slurry in a pit underneath the stables, creating methane and nitrous oxide emissions. One can use manure for crop production; hence, the emissions need to be allocated to crop and pigmeat production through physical allocation (see Sect. 3.4).

5.3.1 Methane

Methane emissions related to manure production are dependent on the excreted volatile solids, the maximum methane production capacity of the manure, and the manure storage. The excreted volatile solids are calculated by using the IPCC (2006a) formula. Moreover, the IPCC 2006 reference value for the urine fraction (2 %) and dry matter content (2 %) was used (IPCC 2006b, Table 10.17). The methane conversion factor for manure slurry (under the stables) is 19 % (IPCC 2006b, Table 10.17).

Table 5 N_{ex} per type of animal for Flanders (*source* NIR Belgium/manure database)

Animal category	N_{ex} (kg/head.yr)
Piglets (<20 kg)	2.24
Meat pigs (20–110 kg)	11.24
Meat pigs (>100 kg)	21.18
Sows (inclusive piglets <7 kg)	21.66

5.3.2 Nitrous Oxide

Stored manure provokes nitrous oxide emissions by means of a combination of nitrification/denitrification. The amount produced depends on the total nitrogen emissions by the animals (N_{ex}). This is calculated by revealing the difference between nitrogen uptake (in fodder) and the amount retained in the body or within products. The nitrogen excreted per animal category was taken from Belgium's NIR report (VMM et al. 2011, Table 5). The amount of nitrous oxide released from manure depends on the way of storage. We assume 0.1 % of the total nitrogen is converted to nitrous oxide for manure slurry stored under the stables, which is lower than the IPCC (2006b) value (0.2 %) but in line with the value for Flanders in the National Inventory (VMM et al. 2011). This involves direct nitrous oxide emissions.

Furthermore, manure releases indirect nitrous oxide emissions. These are formed through volatilized ammonia (NH_3) and NO_x. The amount of NH_3 and NO_x formed from the manure depends on the way of storage. Based upon the IPCC guidelines, we assume 25 % of total nitrogen emission originating from pig manure storage underneath stables is converted to ammonia (NH_3) or NO_x (IPCC 2006b, Table 10.22). Furthermore, we assume 1 % of total nitrogen losses are converted to nitrous oxide through indirect nitrous oxide emissions (IPCC 2006c, Table 11.3). Leaching is assumed to be 0 %.

5.3.3 Manure Usage for Crop Production

When extra manure is exported and used on agricultural land for growing crops, emissions are allocated among the crops (manure usage) and livestock (production of manure surpluses). On the pig farm we studied, all produced manure is transported to another farm. Emissions related to manure surpluses are allocated, by means of physical allocation, based on nitrogen efficiency (0.60) of the crops (Kramer et al. 1999). This means 40 % of emissions related to manure usage is allocated to pig livestock and 60 % to the crops. We assume that all pig manure is distributed on agricultural land without prior treatment. This is the most common practice in Flanders.[1] Nitrous oxide emissions involved by the usage manure are calculated by means of the IPCC 2006 methodology (IPCC 2006a).

[1] In 2010, only one-sixth of all pig manure produced in Flanders was processed and/or exported (VLM 2011b).

5.4 Emissions from Transport

Transport takes place on several levels throughout the chain of production in question (see Fig. 1).

Firstly, feed components are transported to the fodder processing plant. For those components grown in Europe, transport distances are limited (less than 1,000 km). The soy component can originate from Brazil, Argentina, and the United States; hence, maritime transport is needed. Emissions related to both types of components are taken into account in the emission factors applied and are covered by the production of purchased fodder.

Secondly, the feed is transported from the fodder processing plant to the farm. We assume an average distance of 30 km for the purchased fodder. The related emissions are covered within the farm data.

Thirdly, pigs are transported from farm to slaughterhouse. The average distance for this in Flanders amounts to 25 km (based on interviews with slaughterhouses). Half of the pigs are transported with a 23-ton truck (about 200 pigs), the other half with a 12-ton truck (about 110 pigs). The emissions accompanying transport are hence taken into account (Jacobsen et al. 2013).

5.5 Emissions from Meat Processing

Emissions originating from slaughtering are related to the consumption of electricity, fuel consumption, the use of cleansing products, water usage, and waste processing. Transport of necessary resources, such as pigs and other processing materials, are taken into account as well. Emission factors related to electricity and gasoline are derived from Table 4. Furthermore, we used the Ecoinvent database to derive emission factors (2011).

Furthermore, emissions are allocated to meat and other useful by-products. We used economic allocation (see Sect. 3.4) in two steps: at the slaughtering phase and when the carcass is deboned. After the two steps, 63 % of the pig's liveweight was retained as pigmeat (see Table 6). Using the economic value of pigmeat, in relation to the economic value of involved by-products, 92 % of the emissions were attributed to pigmeat production.

6 Results

6.1 The CF of Pigmeat

The results are presented in Fig. 2 and can be summarized as follows: 1 kg of pigmeat (after slaughtering and deboning) creates a CF of 5.7 kg CO_2 eq.

Table 6 Allocation between meat and by-products (Jacobsen et al. 2013)

Step		Total	Carcass – meat	By-products that can be sold at a positive price*	By-products that cannot be sold at a positive price**
Slaughtering	Mass (kg)	115	90.5	13.6	10.9
	Unit price (€/kg)		1.5	0.4	0.0
	Economic value (€)	141	136 (96.5 %)	5	0
Deboning	Mass (kg)	90.5	72.4	16.7	1.8
	Unit price (€/kg)		2.0	0.4	0.0
	Economic value (€)	152	145 (95.5 %)	7	0

* Such as blood, head, intestines, liver, heart, tongue, bones, and fat.
** Such as waste and losses.

Production and transport of purchased fodder, and manure production and usage, have the lion's share in the overall CF. Slaughtering and the deboning process contribute approximately 4 % of the total CF.

At farm level, fodder production is responsible for 63.4 % of the CF. This is a high percentage and is very much influenced by the used emission factors for fodder. The total impact of LUC is about 7 %. Manure storage accounts for 25.2 % of the emissions, of which 88 % is related to methane and 12 % to nitrogen gas emissions. The use of manure surpluses accounts for 3.1 % and animal methane production for 4.9 % of the life weight's CF. Energy and water contribute 3.4 %, of which electricity has a share of 33 %, gasoline oil 66.9 %, and water 0.1 %.

At slaughterhouse and deboning (cut out) firm level, an extra 0.15 kg CO_2 eq. per kg of life weight is added to the CF. The largest contribution (79.3 %) relates to energy consumption. Waste management of the by-products (about 20 kg per pig) accounts for 16.6 %, of which 61 % relates to energy consumption and the rest (39 %) originates from fossil fuel combustion. Furthermore, animal transport between farm and slaughterhouse amounts to 4 %. Production of process materials (0.1 %) is negligible.

6.2 The Sensitivity of the CF of Pigmeat

The single outcome of a CF calculation (in this case for pigmeat) should be used with caution, because it is based upon the use of specific input data. Giving a range of figures in which the CF is expected to be provides better and more realistic insight (Flysjo et al. 2011b). Therefore we conducted a sensitivity analysis[2] to define how

[2] A statistical sensitivity analysis was not carried out because not enough information was available to calculate the standard deviation on the secondary data used or on the final result.

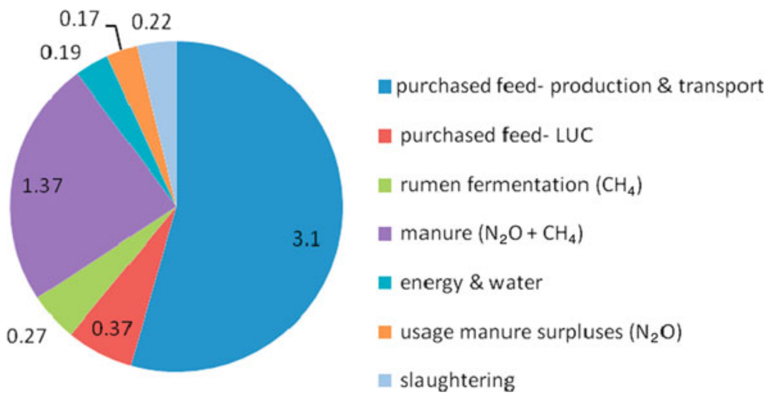

Fig. 2 The carbon footprint of 1 kg of pigmeat (after slaughtering and deboning) in Flanders (data in kg CO2 eq; Jacobsen et al. 2013) LUC, land use change

the single figure is expected to fluctuate. Table 7 shows the overview of possible trends in feed and herd characteristics and their possible impact on the CF.

First of all, effects of changes in feed and herd characteristics on the CF outcome were estimated. When the mortality rate for piglets in the first life stadium is decreased from 12 to 8 %, the CF decreases with 0.08 kg CO_2 eq. per kg of deboned meat. Greater effects are identified for the rise in piglets per sow per year from 27.6 to 30, which is leading to a fall in the CF of 0.14 kg CO_2 eq. per kg of deboned meat; and for an increase in the final weight of the meat pigs from 115 to 125 kg, leading to a decrease in the CF of 0.16 kg CO_2 eq. per kg of deboned meat. Finally, a switch in digestible energy content of feed compound also has an impact. Changing it from 85 to 95 % gives rise to a decrease in the CF of 0.14 kg CO_2 eq. per kg of deboned meat. Table 7 also shows the shift in CF when an increase in mortality rate, a decrease in piglets per sow, a decrease in final meat pig weight, and a decrease in digestible energy content is assumed. Maintaining all other parameters in the model constant, and swapping only one feed and herd parameter at a time, leads to a change in CF from minus 0.16 kg CO_2 eq. per kg life weight to plus 0.18 kg CO_2 eq. per kg life weight.

Secondly, the impact on CF by using soy beans and meal in the fodder was analyzed. The impact was expected to be substantial due to the impact of soy use on LUC. Limiting the use of soy in the piglets' feed by 10 % leads to a CF decrease of 0.02 kg CO_2 eq. per kg of deboned meat. Limiting the use of soy in the feed of the sows by 10 % can lead to a fall of 0.03 kg CO_2 eq. per kg of deboned meat. Limiting the use of soy in feed of the meat pigs by 10 % can even lead to a decrease in the CF of 0.13 kg CO_2 eq. per kg of deboned meat. Therefore, we state that adjusting the soy content in the feed concentrate of meat pigs in the latest life stage has a significant impact on the overall CF of pigmeat. However, limiting the use of soy with a high emission factor implies that a substitute product needs to be found (e.g., soy produced without LUC, locally produced protein-rich products,

Table 7 Applied sensitivity analysis: the impact of changes in herd and feed concentrate parameters on the CF of 1 kg deboned pigmeat (Jacobsen et al. 2013)

Parameter	Initial value	Min	Shift in CF	Max	Shift in CF
Mortality rate of piglets in stadium 1	12 %	8 %	−0.08	16 %	+0.08
Piglets per sow per year	27.6	25	+0.18	30	−0.14
Final weight meat pigs	115	105	+0.18	125	−0.16
Digestible energy content of fodder (% GE)	85	75	+0.13	95	−0.14
Use of soy beans and meal in the piglet feed	*	−10 %	−0.02	+10 %	+0.01
Use of soy beans and meal in the meat pig feed	*	−10 %	−0.13	+10 %	+0.12
Use of soy beans and meal in the sow feed	*	−10 %	−0.03	+10 %	+0.03
Manure management (solid vs. pit storage)	6/94	0/100	−0.01	100/0	+0.18

* See Table 2

animal by-products, etc.). Furthermore, an estimated emission factor for these alternatives needs to be added to make an overall conclusion on the impact of soy banning policies on the CF of pigmeat (see below).

Thirdly, because manure production is a huge contributor to the overall estimated CF, the impact on it of changes in manure management were estimated as well. In the original setup of the calculations, 6 % of the manure was stocked as solid manure and the rest as pit storage. A complete transition towards pit storage leads to a decrease in the CF of 0.01 kg CO_2 eq. per kg of deboned meat. A complete transition towards solid storage leads to an increase in the CF of 0.18 kg CO_2 eq. per kg of deboned meat.

Based upon this sensitivity analysis, the overall estimated CF of pigmeat production in Flanders is expected to fall between 5.5 and 5.9 kg CO_2 eq. per kg of deboned meat. No sensitivity analysis was conducted on allocation method because we applied the most common method.

7 Discussion

7.1 Relative Importance of the Results

To fully understand what this CF of Flemish pigmeat implies, it is helpful to benchmark results with both the CF of non-Flemish pigmeat and other animal products. Comparing it with other international studies is not straightforward due to the different choices made in relation to allocation, functional unit, and/or system boundaries (see introduction).

Some authors report findings on a similar CF for pigmeat, whereas others report a lower or even significantly higher CF. For example, Leip et al. (2010) found a CF of 7.5 kg CO2 eq. per kg of meat for European pigmeat, by using physical allocation and taking into account LUC. Williams et al. (2006) reported a CF of 6.4 kg CO_2 eq. per kg of deboned pigmeat. Carlsson-Kanyama reported a CF in Sweden of 6.1 kg CO_2 eq. per kg of pigmeat, including transport to the consumer. Leip et al. (2010) reported a CF in the UK of 5.7 kg CO_2 eq. per kg of pigmeat. In 2007, Blonk Milieuadvies reported a 5.0 kg of CO_2 eq. per kg of pigmeat CF in the Netherlands. However, 1 year later, Blonk Milieuadvies (Blonk 2008b) found a lower CF of 3.6 kg of CO_2 eq. per kg of pigmeat in the Netherlands. Although they used a similar methodology to the one we used, soy as a feed component was not included; this is a very important contributor to the CF in our findings. A similar CF was found by Dalgaard et al. (2007) using the ISO 14044 methodology to calculate the CF of 1 kg Danish pork delivered to the port of Harwich (UK). Without making use of an allocation method and not taking into account LUC, they found a CF of 3.6 kg CO_2 eq. per kg of pigmeat.

Next, the CF of pigmeat can be compared with the CF for beef and milk production. Within the same study, we have shown that the CF of beef produced in Flanders lies between 22.2 and 25.4 CO_2 eq. per kg of deboned beef and of milk between 1.03 and 1.36 kg CO_2 eq. per kg of milk consumed (1.5 % fat). Looking at consumption, it is known that in Flanders people consume more pigmeat than beef per annum (namely 6.8 kg of pigmeat and 5.6 kg of beef in 2010 (GfK 2013)). Because the CF per kg of beef is much higher compared to pigmeat, the contribution of pigmeat to the total CF for consumption is much lower: eating pigmeat leads to an average annual production of 38.0 kg CO_2 eq., whereas eating beef leads to an average annual production of 132.3 kg CO_2 eq. The consumption of milk makes a higher contribution to a person's CF than the consumption of pigmeat as well. On average, Flemish people yearly drink about 52.7 L (which equals 54.4 kg) of milk (GfK 2013), implying an average annual production of 65.0 kg CO_2 eq. Therefore, the importance of pigmeat consumption in the Flemish diet is to some extent limited compared to milk and beef.

When production is considered, recent reports for Flanders (Platteau et al. 2012) show an overall impact for the CF of milk production of about 2.1 billion kg CO_2 eq. (for 2 billion litres of milk); for beef production of about 5.0 billion kg CO_2 eq. (for 0.3 million tons of beef) and up to 6.3 billion CO_2 eq. for pigmeat (for 1.1 million tons of pigmeat). This shows that the importance of pigmeat production in Flanders, in terms of GHG emissions and the region's CF, is quite important and much higher than the production of milk (Jacobsen et al. 2013).

7.2 Mitigation Measures

The results of our study revealed two huge hotspots in the production chain of pigmeat for which the highest contribution to GHG emissions can be identified: fodder production and manure production (and the usage of it). Based upon those hotspots, we defined some opportunities to reduce the CF of pigmeat.

In particular, the composition of feed has a huge impact on the overall CF. Within Europe, the use of, for example, soy bean in feed concentrates has grown dramatically. However, the use of soy has a negative impact on the CF because of the negative LUC impact and the need to transport the feed components over long distances (Hortenhuber et al. 2011). Therefore, replacing overseas products by using regional products can reduce the CF by limiting the consumption of energy needed for transport. When overseas products are to some extent necessary, preference should be given to products that are produced in a sustainable way and have a limited impact on LUC. Hortenhuber et al. (2011) showed that decreasing the use of soy in feed compounds has a positive impact on the CF. In their study on milk, replacing soybean meal by 50 % with alternative regionally produced, protein-rich feed leads to a decrease of about 26 % in the GHG emissions by dairy cattle.

However, the composition of the feed does not depend upon its contribution to climate change by means of GHGs, but more on availability, price, and the characteristics of the components. Price and availability are two important economic factors influencing the final price of the feed and the possible usage by farmers. Hortenhuber et al. (2011) commented that regionally produced alternatives are not always at people's disposal within Europe. Shifting production in Europe towards these alternatives might lead to LUC effects in Europe (Steinfeld et al. 2006). The rise in demand for alternatives and limitations in the supply of the traditional products might put pressure on prices of both products. Therefore, one cannot simply suggest to ban all carbon negative components, because this would imply limiting the economic sustainability of farming practices.

The characteristics of the components will define whether they can be combined to provide a digestible and sustainable feed for the pigs. Pigs need a balanced diet meeting all of their nutritional requirements. The inclusion of certain feed components is therefore necessary, whereas the inclusion of others might have upper limits. For example, excluding soy products from the feed would increase the need for protein-rich alternatives, which might not all be as digestible for the pig as soy. Moreover, it is possible that these alternatives are very expensive, limiting the usability of the feed (as described above). This economic aspect is often neglected in other literature, as described by Verspecht et al. (2012).

Therefore, the parameters of price, availability, and characteristics of the feed components need to be considered alongside the CF to ensure that pigmeat production is not compromised in an effort to reduce the GHG emissions (Espinoza-Orias et al. 2011).

Manure production, storage, management, and usage is the second largest contributor to the overall CF. Especially in Flanders, manure production and usage creates a serious problem—not only in relation to GHG emissions, but also in terms of nutrient leakage and water pollution (Verspecht et al. 2011). By improving the nutrient efficiency, such as by processing the manure (Masse et al. 2010), several problems related to sustainability could be dealt with at the same time.

In Flanders, one type of manure management, which is the most popular, involves separation of liquid and solid components of manure. The liquid element is cleaned so that it can be disposed of as water. The solid component contains all nutrients from the manure and can be used as an artificial manure. This solid part creates a similar quantity of nitrate emissions as the storage and use of untreated animal manure would do. Some sources point out that the emissions of nitrate might even be higher (Hansen et al. 2006). In terms of methane emissions, a decrease can be expected, although the exact amount is hard to estimate and depends upon the treatment conditions (Sommer et al. 2000). In our study, not enough data was available to estimate the exact impact of manure management on the CF of pigmeat.

As was the case with changes in the feed, described above, parameters such as price and availability will also have an important influence on the mitigation possibilities in relation to manure. Veillette et al. (2012), for example, described how the system of biofiltration can substantially reduce the emission of methane; however, in Flanders, the economic viability of the system on a small scale, such as a farm, has been questioned (van Dooren and Smits 2007).

Both things exemplify the possible trade-offs between dealing with GHG emissions and other aspects of sustainability. Sustainability consists of three pillars: environmental protection, economic growth, and social equity, and a mitigation measure only has a positive effect when all aspects lead to better or higher sustainability. For example, a reduction in the CF at farm level, such as by adapting the feed diet, needs to be combined with food safety and public health, product quality, genetic diversity, efficiency, environment, animal health, and welfare in an economically viable way (Hoffmann 2011).

Moreover, it is important to stress that the CF is a good indicator for GHG emissions, but it is not an indicator for environmental impact in general. It only reports one single environmental impact; when considered alone and not placed alongside other potential environmental impacts, it could misdirect resources away from actions that are more important (Ridoutt et al. 2011). Nowadays, there is a trend of working on environmental footprinting containing more than just 1 environmental indicator. This provides a broader picture of the environmental performance of a product, company, or chain than solely the CF relying on GHG emissions (Jacobsen et al. 2013).

8 Conclusions

The CF of pigmeat estimated in our study using the PAS 2050 methodology (BSI 2011) ranges from 5.5 to 5.9 kg CO_2 eq. per kg of deboned pig meat. The main hotspots were found in fodder production, accounting for the greatest proportion of the total CF. Furthermore, manure management is another important hotspot in the production chain. These hotspots clearly reveal where measures can be taken in order to decrease GHG emissions throughout the chain. The contribution of transport and processing to the overall result is rather small compared to other levels in the chain.

Our study helps to fill the void in the CF literature that existed around pigmeat products. Moreover, the chapter reports on the methodology and assumptions that have been used, the chosen system boundaries, and the system definition. This makes it possible to follow a similar method and estimate the CF of pigmeat in other regions, allowing better and fairer comparisons (Flysjo et al. 2011a), thus assisting the definition of a benchmark for the CF of pigmeat. This in turn will stimulate the search for opportunities to reduce the CF of pigmeat within the framework of international targets, such as the 2011 Durban Accord (Dalgaard et al. 2011).

Flanders is required to implement European policy measures with regard to agriculture. From this perspective, our study will assist Flemish policy makers in achieving their aims for the period 2012–2020. During this period, GHG emissions for EU sectors that do not fall under the transferable emission system have to decrease by 15 %. Therefore, this study helps to reveal hotspots in the chain and potential strategies to decrease their impact in terms of GHG emissions. However, it should be noted that an integrated sustainability approach is necessary, whereas this study focuses solely on the environmental impact of one indicator—climate change.

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Annex

Necessary Data to Calculate the CF for Pigmeat Production on Farm Level

Data	Source
Number of animals per age category	Farmers union
Weight of animals per age category	Farmers union
Number of sold animals	Farmers union
Number of gets per sow and piglets per get	Farmers union
Mortality rate per age category	Farmers union
Replacement percentage sows	Farmers union
Composition/usage of fodder per age category	Farmers union
Feed conversion per age category	Farmers union
Emission factors purchased fodder	Blonk/WUR/Eco-invent/LCA food
Gastrointestinal fermentation	NIR Belgium
Methane conversion factors from manure	NIR Belgium
N_{ex} per type of animal	NIR Belgium
Manure storage systems	Farmers union
Nitrogen losses from manure conversion into laughing gas	NIR Belgium/IPCC 2006

(continued)

(continued)

Data	Source
Manure usage for crop production	IPCC 2006/NIR Belgium/
Energy and water consumption	Farmers union
Emission factors electricity and fuels	Flemish energy convenant
Emission factor water	Ecoinvent database

Necessary Data for Resources of Fodder

Resources	EF (kg CO2 eq/kg product)	% LUC	Source
Soy meal	3.10	70	Blonk/WUR
Soy hulls	0.945	62	Blonk/WUR
Beet pulp	0.108	0	Ecoinvent
Minerals, protein core and vitamins	0.570	0	Ecoinvent
Wheat	0.466	0	Blonk/WUR
Milk powder	7.9	0	LCA food
Barley	0.281	0	Blonk/WUR
Corn	0.488	0	Blonk/WUR
Corn gluten feed	0.424	0	Blonk/WUR
Palm kernel flakes	1.12	13	Blonk/WUR
Wheat starch	0.837	0	Blonk/WUR
Wheat gluten feed	0.338	0	Blonk/WUR
Linseed flakes	0.583	0	Blonk/WUR
Rapeseed flakes	0.583	0	Blonk/WUR
Rapeseed meal	0.455	0	Blonk/WUR

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Carbon Emission and Carbon Footprint of Different Industrial Spaces in Different Regions of China

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Abstract Carbon emission has become an important research hotspot under the background of global climate change and low-carbon economy. Studies on the carbon footprint of different industrial spaces help to establish different low-carbon strategies for different regions. In order to evaluate the carbon cycle pressure of different industrial spaces in different regions, using energy consumption and land use data of each region of China from 1999 to 2008, this chapter establishes carbon emission and carbon footprint models based on energy consumption and estimates the carbon emissions from the use of fossil energy and rural biomass energy of different regions in China. By matching the energy consumption items with industrial spaces, this chapter divides industrial spaces into five types: agricultural space, living and industrial-commercial space, transportation industrial space, fishery and water conservancy space, and other industrial space. Then, the carbon emission intensity and carbon footprint of each industrial space in different regions of China are analyzed. Finally, suggestions for decreasing industrial carbon footprint and optimizing industrial space patterns in different regions are provided.

Keywords Carbon emission · Carbon footprint · Industrial space · Region · China

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1 Introduction

Since the 1990s, issues of carbon emissions and global climate change have increasingly become one of the major technological as well as important societal and political challenges; they are closely related to energy generation and use (Pierucci 2009). Anthropogenic carbon emission from traditional fossil-fuel energy consumption is one of the main causes of global warming. The strategies for greenhouse effect mitigation and carbon emission reduction are very important for each country to combat with climate change. To explore the impact of human activities on global carbon cycles, carbon emission caused by economic development and energy consumption has become one of major concerns in academic circles (Soytasa et al. 2007; Qi et al. 2004; Zhang 2006; Liu et al. 2002; Zhu et al. 2009). Because the production value of heavy industries account for large proportion of the total GDP, there is huge energy consumption in China during industrial activities. With rapid economic development, CO_2 emission in China increased more than 73 % from 1990 to 2003 to 17 million tons, and China has become the world's second largest carbon emitter (International Energy Agency 1996; Zou et al. 2009). Therefore, in the efforts to decrease CO_2 emission, carbon emissions in China and their changes have become the focus of all countries across the world.

1.1 Research on Carbon Emissions

In essence, the impact of human economic and energy activities on regional carbon cycles is largely achieved by changing the industrial space pattern. The alteration of industrial space structure and the regional differences will change the pattern and intensity of human energy consumption and further affect the rate of regional carbon cycle. Therefore, carbon emissions from industrial activities have also become the concern of scholars at home and abroad. For example, Schipper et al. (2001) analyzed the carbon emission intensity of nine manufacturing sectors of 13 International Energy Agency countries using factor decomposition method; they explained the main reasons for growth in carbon emissions since 1990 and made evaluations combined with the targets of Kyoto Protocol. Chang et al. (1998) studied the industrial carbon emission of Taiwan based on the input–output approach and decomposition model. Casler et al. (1998) used model method to analyze the structure of U.S. carbon emissions and showed that the use of alternative energy was the major factor causing carbon emission decline. Chen et al. (2009) analyzed the embodied carbon emissions from final consumption and industrial process of all industrial sectors in China based on input–output analysis. Yu et al. (2009) and Wei et al. (2009) used input–output analysis to compare the carbon leakage and transfer of different industries in the study of carbon emissions embodied in international trade. In addition, some scholars carried out research on the relationship between different industries and carbon emissions (Zhang 2005; Tan et al. 2008).

Judging from the recent research, carbon emissions from energy consumption were mainly calculated on national and provincial scales. Houghton (2008) made a comparison of carbon emissions between China and the world from 1850 to 2005 and found China accounts for 5.9–8.4 % of global energy-related carbon emissions in different years. BP (2010) and World Bank (2009) indicated that China has the highest CO_2 emissions at 6.468 billion tons in 2007 and accounts for 20.85 % of the world's total carbon emissions. Li et al. (2010) calculated the total carbon emission in China and analyzed its changes of the increasing rate from 1953 to 2007; they found there was a low growth stage during the period of 1953–1980, a steady growth stage during the period of 1981–1996, and a rapid growth stage during the period of 2001–2007. By using the relative statistical data of different industries and the method of IPCC greenhouse gas inventory, Sun et al. (2010) calculated carbon emissions from 1995 to 2005 in China.

Based on the method recommended by IPCC, Geng et al. (2011) estimated carbon emissions from energy consumption of Beijing, Shanghai, Tianjin, and Chongqing in 1990, 1995, 2000, and 2004–2007; they found that coal combustion was the leading cause of total CO_2 emissions. By adopting the carbon emission calculating method for all kinds of energy proposed by IPCC in 2006, Hong (2011) calculated the amount of carbon emissions in Shandong province, China, and found it increased 2.63 times from 1997 to 2008. By using fossil energy consumption data of different provinces and the carbon emission data announced by Oak Ridge National Laboratory's CO_2 information analysis center in the United States, Yue et al. (2010) calculated carbon emissions from 1995 to 2007 in China at provincial level.

Based on the study of carbon emission, the study of low-carbon economy and its relationship with energy consumption also became a hot topic. For example, Kawase et al. (2006) used an improved Kaya identity to study factor decomposition on carbon emissions, and carried out scenario forecast on carbon emission reduction targets of different countries. Shimada et al. (2007) established a future regional scale, low-carbon economic scenario analysis method. Zhuang (2005, 2007) analyzed the possible paths and potential for low-carbon development in China's economy. Gomi et al. (2010) studied carbon emission and the future low-carbon economy development of Tokyo City using the scenario analysis method.

The above studies provide important theoretical references to low-carbon economy planning based on industrial carbon emission reduction. However, most of these studies focus on the impact of industrial structure on carbon emissions, without considering carbon emission intensity and its discrepancy of different industrial spaces. Industrial activity is always associated with a certain space. Therefore, research on the carbon emission of different industrial spaces will be of great importance for analyzing and comparing per space carbon emission intensity of different industrial activities, and further taking reasonable measures of industry regulation and space pattern optimization to finally reduce regional carbon emissions.

Generally, carbon emissions in China have been widely studied, but these studies mainly focused on the national or provincial level. China can be divided into several different typical regions, such as Northeast China, Northwest China and so on, among which there are quite different industry development and natural conditions, so research on carbon emission based on regional scales is still needed.

1.2 Researches on Carbon Footprint

Carbon footprint was put forward based on the concept of ecological footprint. It is the measure of the amount of direct or indirect CO_2 emissions caused by an activity (or accumulation of a product in life cycle) (Wiedmann et al. 2007). There are two views on the comprehension of carbon footprint, as follows.

One view defines the carbon footprint as the carbon emission of human activities (Wiedmann et al. 2007; Lee 2011)—that is, to measure it by emission amounts. In this view, Christopher et al. (2008) calculated the household carbon footprint in USA by using the input–output model. Gary et al. (2008) founded that the carbon footprint produced on Christmas day accounts for 5.5 % of the whole year in England. Chambers et al. (2007) evaluated Hurricane Katrina's carbon footprint on U.S. gulf coast forest. Based on apparent consumption, Qi et al. (2010) estimated the carbon footprint in China from 1992 to 2007 and found it increased nearly twofold over that time period.

The other viewpoint regards the carbon footprint as part of the ecological footprint—that is, the ecological carrying capacity required for absorbing CO_2 emissions from fossil fuel combustion (Wiedmann et al. 2007; Global Footprint Network), which measures in area. In this view, Kenny et al. (2009) compared and analyzed the performance of six carbon footprint models for use in Ireland. Based on global average net ecosystem production (NEP) of forest and grass, Xie et al. (2008) made an analysis of ecological footprint (carbon footprint) brought by fossil energy and electricity consumption in China.

As the measurement of impact and pressure of human activities on the environment, carbon footprint has become the new focus in the field of ecology in recent years. For example, in the "Living Planet Report" (World Wildlife Fund 2008) for calculating ecological footprint, carbon footprint as a separate category includes not only the direct carbon emissions caused by fossil fuel combustion but also indirect carbon emissions brought by foreign imports. The results showed that the global ecological footprint per capita was 2.7 hm^2, in which carbon footprint was 1.41 hm^2, thus demonstrating that carbon footprint was an important factor causing human ecological impact. Sovacool et al. (2010) carried out an assessment and analysis on 12 metropolitan carbon footprints and put forward policy proposals to reduce carbon footprint. Kenny et al. (2009) compared and analyzed the performance of six carbon footprint models for use in Ireland. Schulz (2010) took Singapore as the case, estimated the direct and indirect greenhouse gas emission footprint of a small and open economic system, and suggested that indirect pressures of urban systems should be included in discussions of effective and fair adaptation and mitigation strategies. Some Chinese scholars carried out beneficial exploration on carbon footprint studies from the angle of carbon footprint accounting (Huang et al. 2009), carbon footprint per capita and carbon footprint products (Guo 2009), and the infection and inductivity of carbon footprint (Lai et al. 2006).

Overall, carbon footprint research is still in its early stages and further development is needed, especially in the field of regional differences in carbon footprint of various human energy activities. The studies of carbon sink were usually carried out on a small scale and the research results were quite different from each other. In China, some research was done on the national scale, but it adopted the global average carbon sink value (Xie et al. 2008), which cannot precisely represent the actual situation of different regions in China. Therefore, how to precisely evaluate the ecological carrying capacity of absorbing CO_2 emission measured in area still needs further research.

1.3 The Purpose of This Study

To deeply explore the mechanism of industrial carbon emission and its environmental impact in the study of anthropogenic carbon emissions, not only carbon emission from industrial activities should be considered; the analysis on carbon emission intensity of different industrial spaces and its carbon footprint effects are also needed. From the view of industrial spaces, this chapter establishes a carbon emission model based on energy consumption. Through matching industrial spaces with energy consumption items, the carbon emission intensity and carbon footprint of different industrial spaces in different regions of China are discussed. Finally, advice for decreasing the industrial carbon footprint and optimizing industrial space pattern are put forward. The objectives of this study are: (1) to calculate and compare carbon emissions from total energy consumption of different industrial spaces in different regions of China; (2) to estimate carbon sinks of terrestrial ecosystems of different industrial spaces in different regions; (3) to calculate and compare the carbon footprint based on carbon sources and carbon sinks in different regions and their temporal changes; and (4) to explore the strategies for reducing the carbon footprint in different regions of China.

2 Data and Methods

2.1 Data Sources

Presently, the main sources of energy are fossil energy, electricity, biomass, solar, hydraulic, wind and nuclear energy, and traditional energy; of these, fossil energy is the main cause of carbon emissions. Therefore, this chapter only calculates the carbon emissions from major traditional high-carbon energy sources, including fossil energy and rural biomass energy. Industrial energy consumption; land use data; crop yield; and output values of farming, forestry, animal husbandry, and fishery of provinces, municipalities, and autonomous regions in China from 1999 to 2008 were adopted. Energy consumption data are from the "China Energy Statistical Yearbook;" the land use data, crop yield, sown area, output value, and other data are from the "China Statistical Yearbook;" and the standard coal consumption of electrical supply is from CEINET industry database. Because of the lack of relevant data in the Tibet Autonomous Region, Taiwan Province, Hong Kong, and Macao Special Administrative Region, no data sources or results in this chapter include these areas.

2.2 Methods

In this chapter, the carbon emission of energy consumption (fossil energy and biomass energy) was first estimated. Then, through the matching relationship between carbon emission items and land use types, the carbon emissions of five industrial spaces of different provinces were analyzed. Through estimation of carbon sinks of different vegetation types, the carbon footprint of different industrial spaces was obtained. Furthermore, the carbon emission and carbon footprint of different regions in China were further analyzed (Fig. 1).

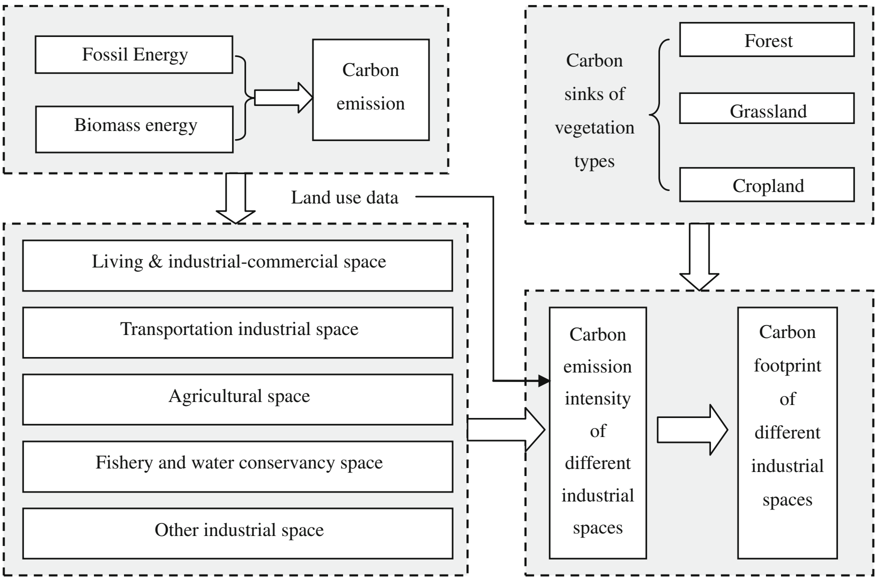

Fig. 1 Theoretical framework of this study

2.2.1 Calculation Method of Carbon Emissions

By establishing energy carbon emission model to calculate the annual carbon emissions from major energy consumption in different provinces, municipalities and autonomous regions, we have Formula 1:

$$Ct = \sum (Ch + Cb) \tag{1}$$

where Ct is the total carbon emission; Ch is the carbon emission from fossil energy consumption; and Cb is the carbon emission from rural biomass energy consumption. It should be noted that the carbon emissions of China mainly come from energy consumption, which includes fossil energy, electricity, and biomass energy. Because the electricity was produced mainly by coal combustion, which belongs to fossil energy, the carbon emission from electricity consumption was not calculated to avoid double counting. The calculation method of fossil energy is as follows:

$$Ch = \sum Qh_i \times \mathrm{NCV}_i \times \left(Cf_i \times \frac{1}{1000} \times \frac{12}{44} + Mf_i \times \frac{1}{1000} \times \frac{12}{16} \right) \tag{2}$$

where Ch is the total amount of carbon emission from fossil energy consumption; Qh_i the fossil energy consumption type i; NCV_i is the net calorific value of energy; Cf_i is the default CO_2 emission factor; and Mf_i the default CH_4 emission factors. Given values of NCV_i, Cf_i, Mf_i from IPCC are used. The unit conversion coefficient is 1/1000, with 12/44 and 12/16 being the conversion coefficients of carbon content in CO_2 and CH_4, respectively. $Cf_i = A_i \times B_i$, A_i is the default carbon content; B_i the default carbon oxide factor.

$$Cb = \sum Qb_i \times Db_i \times Eb_i \tag{3}$$

where Cb is the carbon emission from rural biomass energy consumption; Qb_i is the energy consumption type i (firewood, biogas, straw); Db_i is the carbon emission coefficient. The average of coal carbon emission coefficient from domestic scholars is adopted here (Table 1); Eb_i is the standard coal coefficient.

Table 1 Transfer coefficient of carbon emission of coal (tC/t)

Carbon emission coefficient (tC/t)	References
0.702	Wang and Feng (2006)
0.756	Wang and Feng (2006)
0.726	Wang and Feng (2006)
0.7476	Xu (2006)
0.7329	Tan and Huang (2008)
0.651	Gao (1994)
0.703	Wang (2006)
0.7193837	He and Kang(2008)
0.717235	This chapter (average)

Table 2 The matching relationship between industrial spaces and carbon emission items

Industrial spaces division	Land use type	Energy consumption items (energy balance table)	
Living and industrial-commercial space	Urban built-up land	Construction	
		Wholesale and retail, hotels and catering service	
		Urban residential consumption	
	Rural settlements	Rural residential consumption	
	Independent mining land	Industry	
Transportation industrial space	Transportation land	Transport, storage, postal and telecommunications services	
Agricultural space	Cultivated land	Farming	Farming, forestry,
	Garden land		Animal husbandry,
	Wood land	Forestry	Fishery and water conservancy
	Grass land	Animal husbandry	
Fishery and water conservancy space	Water body	Fishery	
	Water conservancy infrastructure	Water conservancy	
Other industrial space	Unused land	Other	
	Special use land		

2.2.2 Carbon Emission Intensity of Different Industrial Spaces

To further estimate carbon emission and carbon footprint of different industrial spaces, based on energy consumption items of the energy balance table and land use classification system, we cite the study of Li (2009) as a reference. On the basis of merger, decomposition, and appropriate adjustments, we established the corresponding relationship between different industrial spaces and carbon emission items (Table 2).

Note the following: (1) The industrial space here not only means the industry itself, but refers to the spatial extent of industrial activities sustained by land; (2) Carbon emissions of different industries were merged in order to combine the divided industrial spaces with land use data, and carbon emission per space was calculated; (3) Living and industrial-commercial space mainly refers to human living and production space or residential space; (4) Given that rural energy use is mainly centralized in the rural residential areas, carbon emission from rural energy consumption was incorporated into living and industrial-commercial space; (5) Other sectors in the energy balance table cannot be easily subdivided further and were thus incorporated into other industrial spaces; (6) Agriculture, forestry, and animal husbandry are mainly for carbon absorption with little human carbon emission and were thus incorporated into the agricultural space. Carbon emission intensity of industrial space is calculated as follows:

$$Cp_i = Ct_i / S_i \tag{4}$$

$$Cp = \sum Ct_i / \sum S_i \tag{5}$$

where Cp is the carbon emission intensity of provincial industrial space; Cp_i the carbon emission intensity of various industrial spaces (t/hm^2); i is the type of industrial space; S_i is the type i industrial space land area; Ct_i is the type i carbon emission amount.

2.2.3 Calculation Method of Carbon Footprint

In this chapter, carbon footprint is defined as the productive land (vegetation) area needed for absorbing carbon emissions, which means the ecological footprint of carbon emissions. Because the energy carbon emission calculation includes the carbon emissions from rural biomass energy, agricultural vegetation was regarded as part of the carbon footprint. NEP reflects the carbon fixation capacity of vegetation—that is, the carbon absorption amount of per hectare vegetation per year (Xie et al. 2008). Here, NEP indicators were adopted to reflect the carbon absorption of different vegetation, and the area of productive land needed in absorbing carbon emissions (carbon footprint) was further calculated. The method is as follows:

$$\mathrm{CF} = Ct \times \left(\frac{P_f}{\mathrm{NEP}_f} + \frac{p_g}{\mathrm{NEP}_g} + \frac{P_a}{\mathrm{NEP}_a} \right) \tag{6}$$

where CF is the carbon footprint (hm^2) brought by the total amount of carbon emissions (Ct); p_f, p_g and p_a is the total carbon absorption proportion of forest, grassland, and farmland, respectively; and NEP_f, NEP_g, and NEP_a is NEP of forest, grassland, and farmland, respectively. The NEP results of forest and grassland of Xie et al. (2008) were used in this study. The NEP of farmland was estimated by following method:

$$\mathrm{NEP}_a = C_S / S = \sum_i C_d / s \tag{7}$$

where i is the crop type i; C_S the total carbon absorption of crop during growth period; S is the cultivated land area; C_d is the carbon absorption of a certain crop during the whole growth period, such that $C_d = C_a D_w = C_a Y_w / H$; C_a is the carbon absorption rate; Y_w is the economic output; Dw is the biological yield; H is the economic coefficient. The economic coefficients and carbon absorption rates of China's main crops can be found elsewhere (Li 2000; Zhao et al. 2007).

Based on the analysis of total carbon footprint, per space carbon footprint of different industrial spaces can be obtained by carbon footprint of certain industrial space divided by the corresponding industrial space land area.

3 Carbon Emission and Carbon Footprint of Different Industrial Spaces in China

3.1 Carbon Emissions of Different Industrial Spaces

The total amount of carbon emission from energy consumption of China in 2007 was 1.65 GtC (1Gt = 10^9 t), in which the carbon emissions from fossil energy and rural biomass energy consumption were 1.46 and 0.19 GtC, respectively, and the proportions were 89 and 11 %. The largest amount of regional carbon emission was in Hebei province (0.14 GtC). The regions in which total carbon emission amounts exceeded 100 million tons were Shandong, Liaoning, and Henan provinces, which were mainly associated with the high energy consumption of these regions; the smallest amount was in Hainan province, of only 4.85MtC (1Mt = 10^6 t). In addition, carbon emissions in western China, such as Qinghai and Ningxia, was also relatively small (Fig. 2).

There were differences in carbon emission constitution in various regions of China. Overall, carbon emissions from fossil energy were the main contributor to regional total carbon emissions. However, the constitution of carbon emissions was quite different in each region. Carbon emissions from fossil energy occupied a large proportion in eastern China, mostly more than 90 %. However, in western China, carbon emissions from rural biomass energy made up a relatively large proportion, in which Guangxi and Sichuan even reached 30 %. This was mainly related to the different energy consumption structure of different regions; in western China, the proportion of rural energy use was relatively high.

Among the five industrial spaces, carbon emission of living and industrial-commercial space was the highest, for (1.47 GtC, which accounted for 90 % of total carbon emissions), followed by that of transportation industrial space (accounting for 7.3 %). Carbon emission amounts from other types of industry were relatively small (Table 3). Therefore, energy consumption was mainly concentrated in the fields of production, living, and transportation.

There were significant regional differences in the constitution of carbon emissions of industrial spaces. In general, carbon emissions of most regions were mainly constituted of carbon emissions from living, production, and transportation

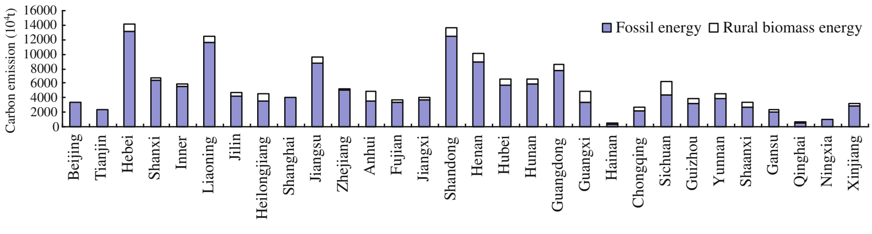

Fig. 2 Carbon emissions from energy consumption in different regions

Table 3 Carbon emissions of different industrial spaces

Industrial space	Carbon emissions (10^6 t)		Land area (10^6hm^2)	Carbon emission intensity of industrial space (t/hm^2)
	Total	%		
Agricultural space	30.74	1.87	505.46	0.06
Living and industrial-commercial space	1467.54	89.12	26.61	55.16
Transportation industrial space	120.19	7.30	2.42	49.65
Fishery and water conservancy space	3.18	0.19	36.80	0.09
Other industrial space	25.11	1.52	259.20	0.10
Total	1646.77	100	830.49	1.98

industrial space. The carbon emission proportion of living and industrial-commercial space in central and western China was higher than that in some developed provinces of eastern China. For instance, the proportion of Henan, Anhui, Hebei, Jiangxi, and Shanxi provinces were all more than 93 %, and that of Hebei was even as high as 95.7 %; the proportion of Beijing and Shanghai were relatively low at 75.1 and 69.4 %, respectively. It indicated that the energy consumption of production, living, and industry and mining was higher in central and western China than that in eastern China. The carbon emission proportion of transportation industrial space in the developed regions such as Beijing and Shanghai was high, with Shanghai at 24.6 %, while that in central and western China was low, with Hebei at only 2.8 %. This demonstrated that in the economy, transportation, and population concentrated areas, due to the development and concentration of transportation and other industries, with limited industrial space, the carbon emission intensity was relatively high.

The carbon emission intensity of industrial spaces of China in 2007 was 1.98 t/hm^2, in which the carbon emission intensities of living and industrial-commercial space and transportation industrial space were 55.16 and 49.65 t/hm^2, respectively. Carbon emission intensity of the other three types of industrial space was lower, with that of agricultural space being only 0.06 t/hm^2 (Table 3). There were large regional differences in the carbon emission intensity of industrial spaces. Generally, carbon emission intensity of central and eastern China was significantly higher than that of the western region. The highest was in Shanghai at 49.68 t/hm^2, whereas the lowest was in Qinghai at 0.083 t/hm^2—a difference of nearly 600 times (Fig. 3). In addition, the carbon emission intensities of living and industrial-commercial space, transportation industrial space, other industrial space and agricultural space in Shanghai were 128.01, 521.79, 41.43 and 0.95 t/hm^2, respectively, all of which were the highest of the country. Therefore, Shanghai had high carbon emissions while the land resources of various types of space were scarce and intense, resulting in high carbon emission intensity and carbon density. Furthermore, various types of industrial spaces in Beijing, Tianjin, Jiangsu, and Zhejiang also had high carbon emission intensity.

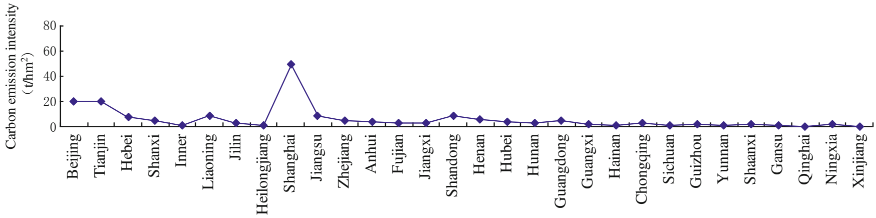

Fig. 3 Carbon emission intensity of industrial spaces in different regions

3.2 Carbon Footprint of Different Industrial Spaces

Carbon footprint caused by industrial activities of China in 2007 was 522.34×10^6 hm^2, while the area of productive land was only 493.65×10^6 hm^2, which brought about ecological deficit of 28.69×10^6 hm^2 (Table 4), equivalent to 3.46 % of the country's total land area. It meant that the productive land area was not sufficient to compensate for carbon footprint of industrial spaces, and the compensating rate was 94.5 %. The primary reason was that China's carbon emission from energy consumption evidently exceeded the carbon absorption of productive land in 2007. This calculation method included the carbon absorption of farmland. Although there was carbon deficit of industrial activities in China, the deficit was not large. Therefore, generally the annual carbon emission from energy consumption of industrial activities can be absorbed by the country's productive land.

As to various regions, the carbon footprint of Hebei province was the largest at 44.71×10^6 hm^2; the smallest was that of Hainan province, with only 1.54×10^6 hm^2 (Fig. 4). Regional differences in the carbon footprint were basically in accordance with that of carbon emission from energy consumption (Fig. 2). Moreover, due to the large differences in productive land area of various regions, the ecological deficit varied significantly. The ecological deficit of Hebei was the

Table 4 Main results of different industrial spaces

Industrial space	Carbon footprint (10^6 hm^2)	Productive land area (10^6 hm^2)	Ecological deficit (10^6 hm^2)	Land area (10^6 hm^2)	Per area carbon footprint (hm^2/hm^2)
Agricultural space	9.75	–	–	505.46	0.02
Living and industrial-commercial space	465.49	–	–	26.61	17.50
Transportation industrial space	38.12	–	–	2.42	15.75
Fishery and water conservancy space	1.01	–	–	36.80	0.03
Other industrial space	7.96	–	–	259.20	0.03
Total	522.34	493.65	28.69	830.49	0.63

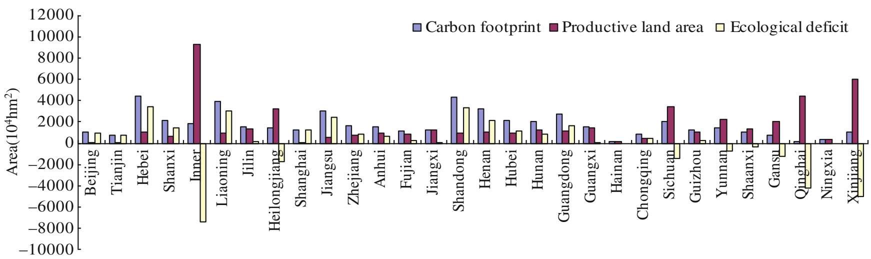

Fig. 4 Carbon footprint and ecological deficit of industrial activity in different regions

highest, reaching 34.31×10^6 hm^2. Shandong, Liaoning, Jiangsu, Henan, and Guangdong provinces also had high ecological deficit. Some regions that possessed large areas of productive land had ecological profit (the ecological deficit is negative), such as Inner Mongolia, Heilongjiang, Qinghai, Xinjiang, Sichuan, Gansu, and Yunnan provinces, among which Inner Mongolia had the highest ecological profit at 74.34×10^6 hm^2 (Fig. 4). The ecological profit was mainly due to the high vegetation coverage of those regions. Therefore, in the provincial level, some regions with low energy consumption and high vegetation coverage can fully compensate for their own carbon emission from energy consumption.

Per area carbon footprint of industrial space in China was 0.63 hm^2/hm^2 in 2007. Different industrial spaces had large differences in per area carbon footprint: living and industrial-commercial space was the highest (17.5 hm^2/hm^2), followed by transportation industrial space (15.75 hm^2/hm^2); agriculture space was the lowest with only 0.02 hm^2/hm^2 (Table 4). Per area carbon footprint of different industrial spaces of various provinces and regions varied significantly. Per area carbon footprint of industrial space of various provinces and regions showed a declining trend, from central and eastern China to the western part (Fig. 5f). Per area carbon footprint of Shanghai was the largest, up to 15.76 hm^2/hm^2; followed by Tianjin and Beijing, North China, and the eastern coastal areas, with generally more than 1 hm^2/hm^2; again followed by South China. Per area carbon footprint of the northeast and western regions was relatively low; the lowest was in Qinghai Province, with only 0.03 hm^2/hm^2. In addition, research found that in the 30 provincial administrative units studied in this chapter, there were 14 provinces with per area carbon footprint greater than 1 and 16 provinces with per area carbon footprint less than 1. The latter ones included South China, northeast and southwest regions (which have better ecological environments), and the underdeveloped western regions, indicating that there were about half of the provinces in China in which per area carbon footprint of industrial space was less than the area of the region itself.

There were also large regional differences in per area carbon footprint of different industrial spaces. Generally, the per area carbon footprint of different industrial spaces all showed a declining trend from the east to west of China (Fig. 5a–e). The largest per area carbon footprint of fishery and water conservancy

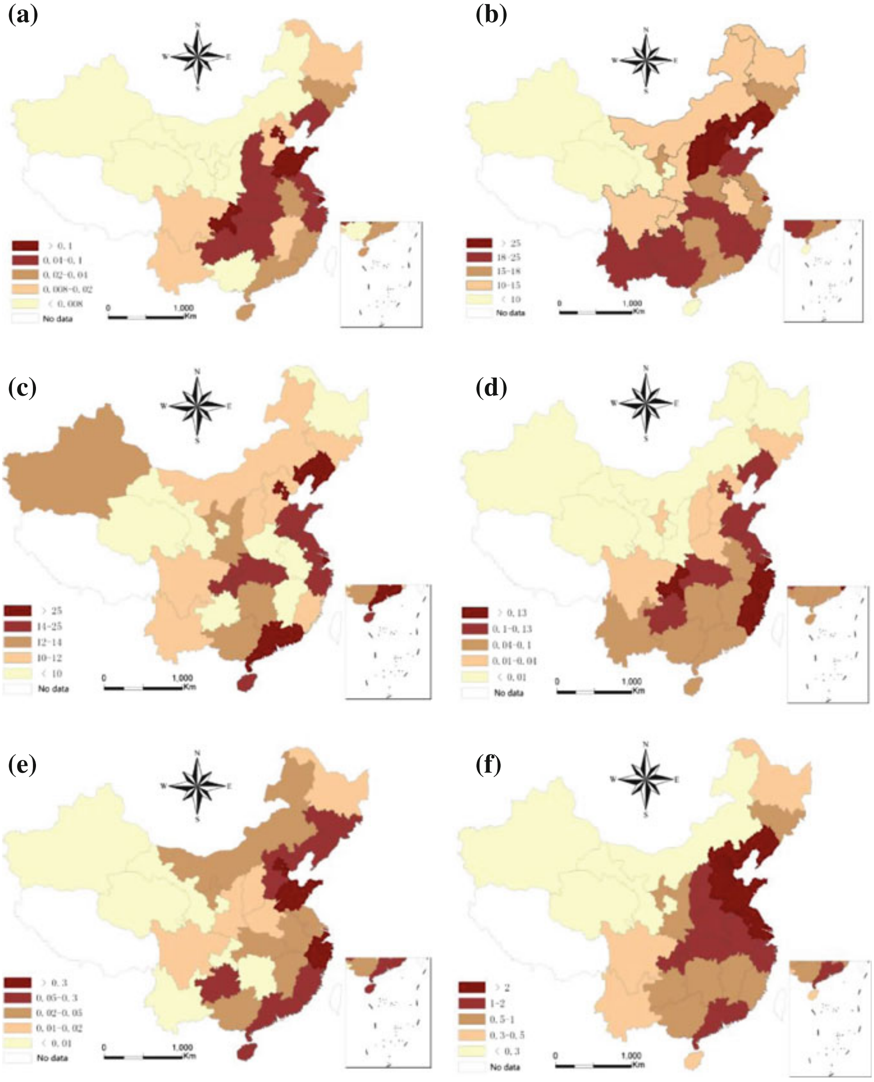

Fig. 5 Distribution of per unit area carbon footprint of different industrial spaces in different regions. **a** Carbon footprint of agricultural space, **b** Carbon footprint of living and industrial-commercial space, **c** carbon footprint of transportation industrial space, **d** carbon footprint of fishery and water conservancy space, **e** carbon footprint of other industrial space, **f** carbon footprint of industrial space of various regions

space was that of Fujian Province (0.29 hm^2/hm^2). The largest per area carbon footprint of other types of industrial space were all in Shanghai, in which that of living and industrial-commercial space, transportation industrial space, and other industrial space were 40.6, 165.51, and 13.14 hm^2/hm^2 respectively. These values for Shanghai were not only far ahead of the provinces and regions of the country, but also far greater than the national average of various lands. Moreover, in the types of living and industrial-commercial space, transportation industrial space, and other industrial space, Beijing, Tianjin, and the eastern developed areas also had high carbon footprint intensities. In contrast, in Qinghai, Xinjiang, Inner Mongolia and other western regions, and Hainan province, per area carbon footprint of various industrial spaces were all relatively low.

The results indicated that, on the one hand, the economically developed eastern regions had high energy consumption, resulting in high carbon emissions; on the other hand, the eastern regions, especially municipalities with land shortage due to the industrial space concentration, the carbon emission intensity of various industrial spaces was high, which led to a high carbon footprint. Instead, due to the larger land area and less energy consumption, the carbon footprint intensity of various industrial spaces in the western regions was lower. For instance, the lowest values for living and industrial-commercial space and transportation industrial space were in Hainan (5.16 hm^2/hm^2) and Qinghai (4.28 hm^2/hm^2), respectively, which were 1/8 and 1/39 the values for Shanghai. The carbon footprint for agriculture space and fishery and water conservancy space in the western regions was even lower. For example, in Qinghai Province, because there was little carbon emission from energy consumption of these two types of land, the carbon footprint was almost negligible.

4 Carbon Emission and Carbon Footprint of Different Regions in China

4.1 Methods for Carbon Sink Estimation in Different Regions

Productive land mainly includes woodland, grassland and, agricultural land. However, because carbon emissions absorbed by agricultural vegetation will be decomposed in the short term and released into the atmosphere (Fang et al. 2007), in this chapter the energy carbon emission calculation does not include the carbon emissions from rural biomass energy. Therefore, carbon absorption from agricultural ecosystems was not considered here.

The results for the carbon footprint are greatly affected by carbon absorption from the productive land; the productive land carbon sink includes carbon absorption both from vegetation and soil covered by vegetation (Fang et al. 2007; Pan et al. 2003; Lal et al. 2002). Therefore, it is crucial to determine the value of the carbon sink from vegetation and soil before calculating the carbon footprint.

As for vegetation, many scholars have studied the carbon sink (the ability to absorb carbon) from vegetation in different regions of China. Lai et al. (2010) collected more than 800 related research achievements in recent years, which cover almost all kinds of vegetation in China, according to the comprehensive analysis (Lai et al. 2010). The value of carbon sinks for different vegetation types in China are indicated in Table 5.

According to the carbon sink values for different vegetation types in Table 6 and the vegetation type map of China that was made in the 1980s, we produced the vegetation carbon sink map of China shown in Fig. 6.

Different vegetations can be classified into different land uses, such as woodland and grassland. By intersect analysis between Figs. 1 and 2 in software ArcGIS9.3, the average vegetation carbon sink value of woodland and grassland in different regions can be calculated (Table 6). Because the distribution of vegetation did not change significantly from 1980s to 2000s, the carbon sink value of woodlands and grasslands in Table 3 can well be used for different years in this chapter.

Unlike vegetation carbon sink, there is limited detected data for soil carbon sink in China, so it is difficult to make a relatively accurate assessment of soil carbon sink. Pacala et al. (2001) reported that soil carbon sink accounts for about two-third of vegetation carbon sink in the United States. In Europe, soil carbon sink accounts for approximately 30 % of the whole ecosystem (Janssens et al. 2003); Piao et al. (2009) calculated that soil carbon sink in China was 75.4 Tg year^{-1}, whereas vegetation (forest, shrub, grass) carbon sink was 105.2 Tg year^{-1} from 1980 to 2000. Fang et al. (2007) calculated that soil carbon sink in China was 41.2–70.8 Tg year^{-1}, whereas vegetation (forest, shrub, grass) carbon sink was 96.1–106.1 Tg year^{-1} from 1981 to 2000. According to their studies, this chapter adopted the average value of other researchers, with soil carbon sink accounting for 65 % of vegetation carbon sink.

4.2 Changes of Carbon Emissions in Different Regions

As indicated in Fig. 7, the amount of carbon emissions in the six different regions are much different from 1999 to 2008. Eastern China always had the largest amount of carbon emissions; it increased from 236.77×10^6 t in 1999 to 603.47×10^6 t in 2008—an increase of nearly 155 %; Following Eastern China, northern and central/southern China also had large amounts of carbon emissions: respectively 201.02×10^6 t in 1999 to 451.13×10^6 t in 2008 (an increase of nearly 124 %), and 157.64×10^6 t in 1999 to 377.77×10^6 t in 2008 (an increase of more than 140 %. Northeast China had a medium carbon emissions, with an slower increase compared with other regions: 142.55×10^6 t in 1999 to 252.56×10^6 t in 2008 (77 %); Carbon emissions in southwest and northwest China were low, but the rates increased at 110 and 156 %, respectively.

Overall, carbon emissions from energy consumption in the six regions of China increased obviously. The spatial distribution pattern of carbon emissions among the six regions did not change greatly from 1999 to 2008. Eastern China always had the largest amount of carbon emissions; carbon emissions in northern and

Table 5 Carbon sink value for different vegetation in China

Vegetation	Carbon sink t $(hm^2)^{-1}$ $year^{-1}$	Vegetation	Carbon sink t $(hm^2)^{-1}$ $year^{-1}$
Boreal, temperate mountain deciduous coniferous forests	0.271	Tropical evergreen broadleaf sclerophyllous coastal scrub, coppice	0.082
Temperate mountain evergreen coniferous forest	0.428	Tropical evergreen broadleaf shrub succulent coral reefs, coppice	0.211
Temperate steppe sandy evergreen coniferous woodland	0.178	Sub-tropical mountains, subalpine evergreen scoriaceous scrub, coppice	0.162
Temperate evergreen coniferous forest	0.229	Temperate, subtropical subalpine deciduous scrub	0.062
Subtropical and tropical evergreen coniferous forest	0.442	Temperate alpine dwarf shrub tundra	0.063
Subtropical and tropical mountain evergreen coniferous forest	0.428	Temperate, subtropical subalpine cushion-shaped dwarf shrubs, herbaceous vegetation	0.090
Temperate deciduous broadleaf-evergreen conifer mixed forest	0.585	Temperate grass and forb steppe	0.021
Temperate, subtropical deciduous broadleaf forest	0.407	Temperate needlegrass steppe	0.021
Temperate, subtropical deciduous microphylla forest	0.585	Temperate mountain needlegrass steppe	0.021
Temperate deciduous microphylla woodland	0.324	Temperate dwarf needlegrass, semi-dwarf shrub steppe	0.021
Subtropical limestone deciduous-evergreen broadleaf mixed forest	0.729	Temperate mountain dwarf grass, semi-dwarf shrub steppe	0.021
Subtropical mountain yellow-soil evergreen-deciduous broadleaf mixed forest	0.729	Temperate, subtropical alpine steppe	0.021
Subtropical evergreen broadleaf forest	0.729	Subtropical and tropical shrub savanna	0.060
Tropical rain forest of evergreen broadleaf forest	0.729	Temperate meadow	0.077
Subtropical evergreen sclerophyllous broadleaf forest	0.624	Temperate and subtropical alpline meadow	0.077
Subtropical Bamboo	0.883	Temperate herbaceous swamp	0.389
Tropical semi-evergreen broadleaf forests	0.762	Temperate alpine herbaceous swamp	0.389
Tropical evergreen broadleaf forests and secondary vegetation	0.271	Cultivated vegetation	0.000
Temperate, subtropical deciduous shrub, coppice	0.174	Bare land/ice/desert	0.000
Subtropical and tropical acid soil evergreen, deciduous broadleaf shrubs, coppice and meadow	0.418	Desert	0.000
Subtropical and tropical limestone evergreen, deciduous shrubs, coppice	0.195		

Table 6 Mean carbon sink value of woodlands and grasslands in different regions of China

Region	Mean carbon sink value t $(hm^2)^{-1}$ year^{-1}	
	Woodland vegetation	Grassland vegetation
Northern China	0.25	0.03
Northeast China	0.30	0.15
Eastern China	0.42	0.08
Central and southern China	0.46	0.06
Southwest China	0.35	0.05
Northwest China	0.19	0.04
Total studied area of China	0.34	0.05

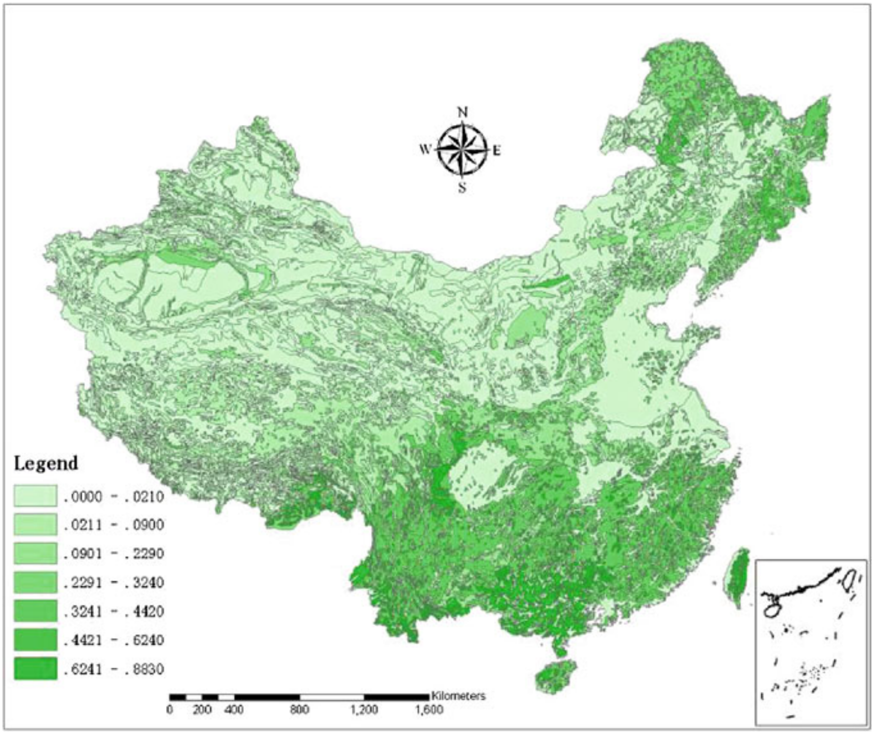

Fig. 6 Distribution of carbon sinks for different vegetations in China

central/southern China were also high, with high increasing rates; and carbon emissions in Southwest China and Northwest China were low, but with high increasing rates.

Because the area of each region is different, which can greatly influence its total amount of carbon emissions, this chapter also made a study of carbon emission density from 1999 to 2008 in order to make the comparison among different regions more scientific and precise, as indicated in Fig. 8. Similar to the amount of carbon emissions, carbon emission density in Eastern China was also much higher

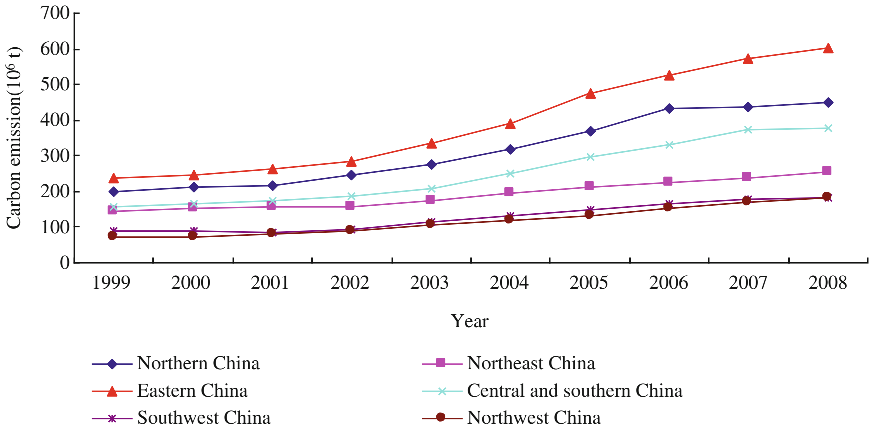

Fig. 7 Carbon emissions in different regions of China from 1999 to 2008

than other regions: the density increased from 2.93 t/hm^2 in 1999 to 7.46 t/hm^2 in 2008. Except for Eastern China, carbon emission density in Northeast China was higher than other regions from 1999 (1.8 t/hm^2) to 2003 (2.2 t/hm^2). However, from 2004, carbon emissions in Central and Southern China began to increase rapidly and become higher than other regions except Eastern China, reaching 3.72 t/hm^2 in 2008. Carbon emissions in Northern China were higher than in Southwest China and Northwest China; its carbon emission density increased from 1.31 t/hm^2 in 1999 to 2.95 t/hm^2 in 2008. Carbon emission density in Southwest China was only higher than in Northwest China; its density increased from 1.08 t/hm^2 in 1999 to 2.46 t/hm^2 in 2008. In Northwest China, the density was only 0.23 t/hm^2 in 1999 and 0.59 t/hm^2 in 2008.

Overall, the developed regions usually had high carbon emission density. The average carbon emission density in mainland China increased rapidly from 1999 to 2008. It can be seen that energy consumption accelerated the growth of economy and also led to high carbon emissions in China.

4.3 Changes in the Carbon Footprint in Different Regions

The carbon footprint caused by energy consumption has increased greatly, but the productive land did increase significantly (woodland and grassland) in different regions of China. As indicated in Table 7, the productive land increased only between 0.22 % (Northeast China) and 2.03 % (Southwest China) from 1999 to 2008, while energy consumption increased 77 % (Northeast China) to 156 % (Northwest China). This led to a great increase of carbon footprint from 1999 to 2008. Northern China had the largest carbon footprint, with an area of 1267.82 × 10^6 hm^2 in 1999, which increased to 2697.67 × 10^6 hm^2 in 2008;

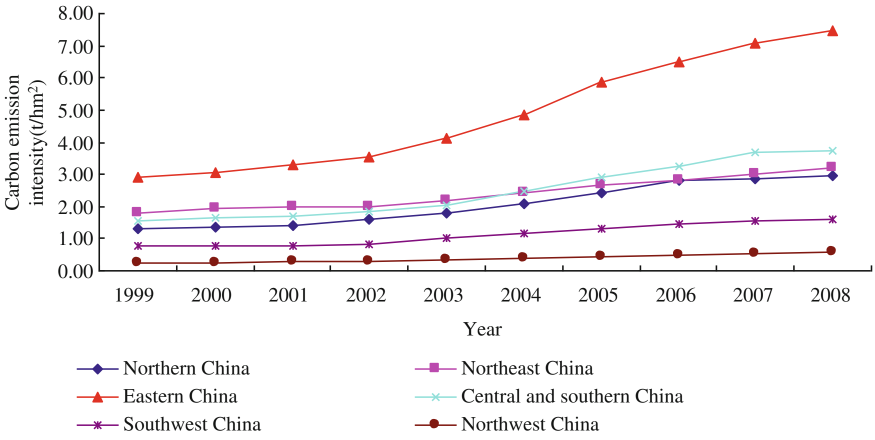

Fig. 8 Carbon emission intensity in different regions of China from 1999 to 2008

according to the area of its productive land, Northern China could only absorb 7.65–3.65 % of its carbon emissions from 1999 to 2008. Southwest China had the next largest carbon footprint, with an area of 190.45 × 10^6 hm^2 in 1999 that increased to 396.52 × 10^6 hm^2 in 2008; the vegetation (forest, shrub, grass) and soil in this region could only absorb 35.66–17.48 % of its carbon emissions from 1999 to 2008. Although carbon emissions were the lowest in Northwest China, where there is the highest area of productive land, the value of the carbon sink is much lower than other regions (Table 6). This made it also have large area of carbon footprint, with an area of 616.93 × 10^6 hm^2 in 1999 and 1531.45 × 10^6 hm^2 in 2008; productive land in this region could absorb 21.66–8.82 % of its carbon emissions from 1999 to 2008; Carbon emissions in Eastern China were the highest and it had the lowest area of productive land compared with others. However, because the value of the carbon sink was high, carbon footprint here was much lower than in Northwest China, with the area of 346.9 × 10^6 hm^2 in 1999 and 883 × 10^6 hm^2 in 2008; the productive land here could only absorb 8.43–3.35 % of its carbon emissions; The area of productive land in central and southern China differed not much compared with that in Northeast China, and carbon emissions here were much higher than in Northeast China (Fig. 8). However, because the value of the carbon sink is much higher (Table 7), the area of the carbon footprint in the two regions did not differ too much; the vegetation (forest, shrub, grass) and soil in the two regions could absorb 13.78–7.81 % and 21.82–9.23 % of their carbon emissions from 1999 to 2008, respectively.

Overall, the carbon footprint in China increased 125.46 % from 1999 (2529.61 × 10^6 hm^2) to 2008 (5703.19 × 10^6 hm^2). Vegetation (forest, shrub, grass) and soil in China could absorb 16.43 % of its carbon emissions from energy consumption, but the percentage decreased to 7.38 in 2008. Northern China had the largest carbon footprint and the lowest percentage of carbon absorption, which is the region under the maximum ecological pressure. Following Northern China,

Table 7 Comparison of carbon footprint and productive land in different regions of China from 1999 to 2008

Item	Region							Total studied area
	Year	Northern China	Northeast China	Eastern China	Central and southern China	Southwest China	Northwest China	
Productive land area (10^6 hm^2)	1999	96.94	41.33	29.25	46.44	67.92	133.65	415.54
	2000	96.78	41.33	29.25	46.42	68.09	133.56	415.43
	2001	96.93	41.32	29.24	46.45	68.15	133.83	415.91
	2002	97.42	41.35	29.29	46.58	68.36	134.12	417.12
	2003	98.20	41.50	29.55	46.95	68.97	134.78	419.96
	2004	98.46	41.46	29.59	47.01	69.22	134.90	420.63
	2005	98.39	41.44	29.63	47.04	69.29	135.01	420.81
	2006	98.48	41.42	29.68	47.05	69.35	135.01	420.99
	2007	98.46	41.42	29.63	47.02	69.33	135.00	420.86
	2008	98.46	41.42	29.57	47.00	69.30	135.00	420.76
Increasing rate (%)	–	1.57	0.22	1.10	1.22	2.03	1.01	1.26
Carbon footprint (10^6 hm^2)	1999	1267.82	300.04	346.90	212.85	190.45	616.93	2529.61
	2000	1324.80	324.72	363.27	225.98	191.51	628.44	2652.21
	2001	1366.90	328.94	387.97	235.24	187.50	713.76	2767.08
	2002	1517.30	328.74	418.49	251.40	203.25	785.46	2973.78
	2003	1691.51	365.59	487.98	279.46	250.70	889.74	3389.05
	2004	1912.75	406.12	574.28	340.09	290.29	995.65	3920.87
	2005	2208.97	444.04	696.84	398.50	320.34	1126.87	4546.01
	2006	2581.28	469.07	768.43	446.00	356.56	1291.98	5087.49
	2007	2618.13	499.23	837.95	501.65	382.76	1418.95	5467.76
	2008	2697.67	530.38	883.00	509.14	396.52	1531.45	5703.19
Increasing rate (%)	–	112.78	76.77	154.54	139.21	108.20	148.24	125.46
The percentage of carbon absorption accounting for carbon emissions (%)	1999	7.65	13.78	8.43	21.82	35.66	21.66	16.43
	2000	7.31	12.73	8.05	20.54	35.55	21.25	15.66
	2001	7.09	12.56	7.54	19.74	36.34	18.75	15.03
	2002	6.42	12.58	7.00	18.53	33.63	17.08	14.03
	2003	5.81	11.35	6.06	16.80	27.51	15.15	12.39
	2004	5.15	10.21	5.15	13.82	23.84	13.55	10.73
	2005	4.45	9.33	4.25	11.81	21.63	11.98	9.26
	2006	3.82	8.83	3.86	10.55	19.45	10.45	8.27
	2007	3.76	8.30	3.54	9.37	18.11	9.51	7.70
	2008	3.65	7.81	3.35	9.23	17.48	8.82	7.38

Northwest China, Eastern China, Northern China, Central/southern China also faced a great and increasing ecological pressure from 1999 to 2008. Southwest China experienced less ecological pressure, with the lowest area of carbon footprint and the highest percentage of carbon absorption compared with others.

As indicated in Table 8, per unit area carbon footprint caused by energy consumption in the studied area of China was 3.04 hm^2/hm^2 in 1999, which increased to 6.86 hm^2/hm^2 in 2008. There were significant regional differences. From 1999 to 2008, Northern China always had the highest per unit area carbon footprint of 8.29–17.65 hm^2/hm^2. Following Northern China, the per unit area carbon footprint in Eastern China and Northeast China were also high with the value between 4.29 and 10.92 hm^2/hm^2 and between 3.79 and 6.7 hm^2/hm^2. Southwest China always had the lowest per unit area carbon footprint of 1.69–17.65 hm^2/hm^2. Compared with other regions, central/southern China and Northwest China had a similar medium per unit area carbon footprint from 1999 to 2008.

The results indicated that the per unit area carbon footprint was determined not only by the amount of carbon emissions but also by its land area and the ability of carbon absorption brought by vegetation and soil. Mainly influenced by three factors, Northern China presented the highest per unit area carbon footprint and, followed by Eastern China and Northern China. Central/southern China and Northwest China had a similar medium per unit area carbon footprint. Southwest China always had the lowest per unit area carbon footprint.

5 Discussions and Policy Implications

5.1 About Carbon Emissions

The carbon emission results in this chapter were slightly higher than that of other Chinese scholars in recent years (Table 9), for two main reasons. First, the carbon emission calculations in this chapter included carbon emission from fossil energy and rural biomass energy consumption; the total amount of carbon emission would be 1.46 GtC if we only included that from fossil energy. Second, the calculations in this chapter were based on data from the year 2007; therefore, it is reasonable that the results had a certain degree of growth compared to 2003–2005.

Compared with results from abroad, the carbon emissions in 2007 in China in this chapter (1.647 GtC) were relatively low. For example, the carbon emissions in China collected by the CDIAC was 1.783 GtC in 2007.

Table 8 Comparison of per unit area carbon footprint in different regions of China from 1999 to 2008

Item	Region							Total studied area
	Year	Northern China	Northeast China	Eastern China	Central and southern China	Southwest China	Northwest China	
Region land area (10^6 hm^2)	–	152.88	79.18	80.86	101.59	112.57	304.42	831.51
Per unit area carbon footprint (hm^2/hm^2)	1999	8.29	3.79	4.29	2.09	1.69	2.03	3.04
	2000	8.67	4.10	4.49	2.22	1.70	2.06	3.19
	2001	8.94	4.15	4.80	2.32	1.67	2.34	3.33
	2002	9.92	4.15	5.18	2.47	1.81	2.58	3.58
	2003	11.06	4.62	6.03	2.75	2.23	2.92	4.08
	2004	12.51	5.13	7.10	3.35	2.58	3.27	4.72
	2005	14.45	5.61	8.62	3.92	2.85	3.70	5.47
	2006	16.88	5.92	9.50	4.39	3.17	4.24	6.12
	2007	17.13	6.30	10.36	4.94	3.40	4.66	6.58
	2008	17.65	6.70	10.92	5.01	3.52	5.03	6.86

Table 9 Comparison of results from other authors

Author	Carbon emission (GtC)	Year	Reference
Wei Yiming	1.37	2004	Wei et al. (2008)
Xiao Lian	1.127	2003	Xiao (2008)
Liu Hongguang	1.13	2004	Liu and Liu (2009)
Xu Guoquan	1.28	2004	Xu et al. (2006)
Liu Qiang	1.505	2005	Liu et al. (2008)
Wei Baoren	1.282	2005	Wei (2007)
Chen Qingtai	1.3-2.0	2020	Chen (2004)
CDIAC	1.783	2007	CDIAC (2010)
This chapter	1.647	2007	

5.2 *About Carbon Footprint*

It should be noted that the carbon emissions of industrial spaces in this chapter placed more weight on the carbon intensity analysis of industrial activities, so as to understand the spatial carbon emission density caused by different industrial activities. The land only sustains space rather than being a source of carbon emission. Therefore, carbon emissions do not mean the emission from the land itself, but the carbon emissions from industrial activities sustained by the land.

In different regions, the large land area of certain industrial spaces would probably make the result of per area carbon footprint of the corresponding industrial space a little too small. For instance, the per area carbon footprint of Anhui province was 1.12 hm^2/hm^2, ranking 13th in China, which was high; however, due to the relatively large area of living and industrial-commercial space and transportation industrial space, the per area carbon footprint of the two types of space ranked 26th and 28th in the country, respectively. The total carbon footprint of Xinjiang was low, yet because of the small area of transportation industrial space, per area carbon footprint of transportation industrial space in Xinjiang was relatively high, ranking 12th in China. The results indicated that based on the calculation method of the carbon footprints of industrial spaces in this chapter, the carbon footprint of regional different industries was affected by the structure of regional industrial land.

5.3 *Uncertainty Analysis*

The main disadvantages and error sources are as follows: First, the division of industrial spaces was based on energy carbon emission items and land classification system. Because the correspondence between the data should be considered, some of the industrial spaces were not subdivided. Thus, there was inevitably some error in the corresponding relationship between industrial spaces and carbon emission items. Also, in different regions, there might be small differences in the total amount of carbon emissions; however, the large area of certain industrial spaces might make the per area carbon footprint result a little too small, and vice versa. In addition, due to the difficulty in combing time-series data of land and energy at the provincial level, this chapter only studied the regional differences in carbon footprints from energy consumption of different industrial spaces. The variation characteristics of carbon footprint of industrial space in various provinces and regions on temporal dimension were not analyzed.

5.4 Policy Implications

In order to reduce regional carbon emission intensities and carbon footprints, the following measures can be considered. First, the use of fossil energy is the primary reason causing carbon emission. Therefore, innovations on traditional energy structures and the use of clean energy are the main ways to reduce regional per area carbon emissions and carbon footprints. Second, the central and western regions should minimize the energy consumption of living, industry, and mining spaces, particularly to reduce the use of rural biomass energy, to lower the carbon emission intensity of living and industrial-commercial space. The eastern regions should adopt clean energy in the transportation industry as much as possible, in order to reduce the carbon pollution of transportation sector. Third, efforts should be made to strengthen the ecological management and protection of the regions with ecological profit, as well as to enhance the carbon fixation efficiency of productive land, which can effectively reduce regional carbon emission level and intensity. Also, the key to reduce carbon emission intensity and carbon footprint is to adjust industrial space pattern and regulate industrial activities (such as construction industry, transportation industry, etc.) with high carbon footprint. Finally, efforts should be made to consider carbon footprint effects in the industrial space arrangement and planning and introduce the concept of carbon emission reduction. In this way, it may be possible to reduce carbon pollution of the high-carbon-emission spaces through industrial regulation, while minimizing carbon emission intensity of industrial space by improving energy efficiency and energy structure.

The consumption of fossil energy is the primary cause of carbon emissions. In particular, coal was the main source of energy in China, and it can lead to great carbon emissions. Therefore, traditional energy structure must be innovated and the use of clean energy should be increased. Energy consumption in different industries varied greatly, so the focus should be on energy-intensive industries, such as the steel and non-ferrous metal industry, cement industry and so on. China should adjust its industrial structure, not only adjust among the primary industry, the secondary industry, and the tertiary industry, but also the specific industry structure of the three industries. For example, some energy-intensive industries can be decreased, while some low-pollution industries are increased. Furthermore, the efficiency of energy use in China is low, so improving energy efficiency may be an effective way to reduce carbon emissions, which is also a challenge for China.

In terrestrial ecosystems, carbon emissions will mainly be absorbed by vegetation (especially forest and grass) and its covered soil, as discussed in this chapter; therefore, some measures should be taken to increase the carbon absorption. Firstly, the areas of productive land should be protected, especially woodlands, which have the highest production compared with other land use types. Secondly, ecological management should be strengthened to enhance the carbon fixation efficiency of productive land, such as prohibiting the behavior of deforestation and overgrazing, to make soil less disturbed. Thirdly, according to the local climatic and soil environment, more vegetation should be planted, which can adapt to its local natural environment and absorb carbon more effectively.

6 Conclusions

Using energy consumption and land use data for each region in China, this chapter established carbon emission and carbon footprint models based on energy consumption and estimated the carbon emission amount of fossil energy and rural biomass energy for different regions of China in 2007. By matching the energy consumption items with industrial spaces, industrial spaces were divided into five types: agricultural space, living and industrial-commercial space, transportation industrial space, fishery and water conservancy space, and other industrial space. Then, the carbon emission intensity and carbon footprint of each industrial space and in different regions was discussed. Finally, suggestions for decreasing industrial carbon footprint and optimizing industrial space patterns were put forward. The main conclusions are as follows:

(1) Total carbon emissions from energy consumption in China in 2007 were approximately 1.65 GtC, in which the proportion of carbon emission from fossil energy was 89 %. The carbon emission intensity of industrial space in China in 2007 was 1.98 t/hm^2, in which, the carbon emission intensities of living and industrial-commercial space and of transportation industrial space were 55.16 and 49.65 t/hm^2 respectively; they were high-carbon-emission industrial spaces, among others.
(2) The carbon footprint caused by industrial activities of China in 2007 was 522.34×10^6 hm^2, which brought about an ecological deficit of 28.69×10^6 hm^2. Therefore, the productive lands were not sufficient to compensate for the carbon footprint of industrial activities; and the compensating rate was 94.5 %. Regarding the regional carbon footprint, several regions have ecological profit, although others do not. In general, the present ecological deficit caused by industrial activities was small in 2007. The per area carbon footprint of industrial space in China was approximately 0.63 hm^2/hm^2 in 2007, in which that of living and industrial-commercial space was the highest (17.5 hm^2/hm^2). The per area carbon footprint of different industrial spaces all showed a declining trend from the east to west of China.

(3) Carbon emissions from energy consumption in different regions of China all increased significantly from 1999 to 2008. Eastern China always had the largest amount of carbon emissions; in Northern, Central, and Southern China, it was also high. There was also a high increasing rate. The carbon emissions in Southwest China and Northwest China were low, but the increasing rate was high from 1999 to 2008. The rankings for carbon emission density was Eastern China > Northeast China > Central and Southern China > Northern China > Southwest China > Northwest China from 1999 to 2003, but from 2004 Central and Southern China began to have higher carbon emission densities than Northeast China, although the ranking for other regions did not change.

(4) The carbon footprint increased significantly since the rapid increase of carbon emissions, but the areas of productive land did not significant change in the different regions of China. Northern China had the largest carbon footprint and the lowest percentage of carbon absorption. Following Northern China, Northwest China, Eastern China, Northern China, and Central/southern China also faced a great and increasing ecological pressure from 1999 to 2008. Southwest China presented less ecological pressure, with the lowest area of carbon footprint and the highest percentage of carbon absorption compared with others. Mainly influenced by regional land area, Northern China presented the highest per unit area carbon footprint, followed by Eastern China and Northern China. Central/southern China and Northwest China had a similar medium per unit area carbon footprint. Southwest China always had the lowest per unit area carbon footprint.

(5) China faced great ecological pressure brought by carbon emissions. Some measures should be taken for both reducing carbon emissions and increasing carbon absorption. Efforts should be made to strengthen the ecological management and protection of the regions with ecological profit and enhance the carbon fixation efficiency of productive land, which can effectively reduce regional carbon emission levels and intensities. The key to reducing carbon emission intensity and carbon footprint is to adjust industrial space pattern and regulate industrial activities (such as the construction industry and transportation industry) with high carbon footprints.

The following two aspects should be further investigated in future research. First, different industrial spaces should be further divided in order to precisely calculate the carbon emission of different land use and industrial spaces, thus providing theoretical support for low-carbon economy planning based on the optimization of industrial space pattern. In addition, the research on the carbon emissions of industrial activities and land use should be further extended. On the one hand, the varied mechanisms of carbon emissions of industrial activities sustained by land should be further discussed. On the other hand, the carbon flux and carbon metabolism of different land use types and the carbon emission effect of land use type conversion should be deeply explored, so as to establish comprehensive carbon cycle model that includes both natural carbon emissions and socioeconomic carbon emissions on the regional scale.

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Eco-Design and Product Carbon Footprint Use in the Packaging Sector

Esther Sanyé-Mengual, Raul García Lozano, Jordi Oliver-Solà, Carles M. Gasol and Joan Rieradevall

Abstract Packaging products are common in all industrial sectors and in the market place. However, packaging design needs to be optimized while avoiding superfluous designs that do not consider the environment in their design. Directive 94/62/EC established a framework in order to harmonize the environmental requirements for packaging as well as to determine targets for recycling and recovering packaging waste. In this chapter, the eco-design projects of different sectors are presented in order to show the different strategies that are used to improve the environmental performance of packaging products. The carbon footprint of the products is quantified and used as an environmental indicator. Common strategies to reduce the carbon footprint of packaging are optimizing the volume (and therefore reducing the transportation requirements), using renewable materials, and optimizing the end-of-life management.

Keywords Eco-design · Carbon footprint · Industrial ecology · Packaging · Sustainable manufacturing

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1 Introduction

Packaging has an extended presence in markets because they have turned into basic elements for distributing and selling products. Packaging has the function of protecting and maintaining the product during the distribution and retail processes. Moreover, packaging has evolved as a new piece of the product, in which design and marketing play an important role. However, the environmental burdens of products are sometimes increased due to the packaging design (Fig. 1). For example, an informatics device can have different types of packaging (multi-packaging systems) that can increase the product volume more than 20 times, therefore increasing the environmental impact of the distribution stage. Moreover, multimaterial packaging are common in stores, such as in food retail or multi-packaging systems.

For example, packaging has an important role in the food sector, where it helps to avoid product losses during distribution and increases the lifespan of the product during the consumption stage. According to the Food and Agriculture Organization of the United Nations (FAO), an important part of food waste is generated during distribution in developing countries, whereas in Western Europe food distribution has low values of food waste, partly because of better food packaging design.

The environmental performance of this sector has recently been analyzed, not only as a product (e.g., Ross and Evans 2003; Zabaniotou and Kassidi 2003) but also as part of the entire lifecycle of a food product (e.g., Koroneos et al. 2005; Sanyé-Mengual et al. 2013; Torrellas et al. 2008). The packaging used for distribution represents one of the highest contributing elements for the life cycle of a tomato consumed in Barcelona (Sanyé-Mengual et al. 2013), as well as for a tomato produced in the Canary Islands (Torrellas et al. 2008). Furthermore, packaging increases the global energy consumption, thus making processed food a highly energy-intensive product (Garnett 2003).

Moreover, food-related packaging is the most common waste in households (Garnett 2003). According to INCPEN (2001), packaging represented a quarter of the household waste production in the UK, and 70 % of this packaging was food-related. This fact is narrowly associated with the retail stage, where packaging is

Fig. 1 Examples of packaging designs that increase the environmental burdens of the product: multipackaging systems, volume increase, and multimaterial packaging

also a key aspect. When comparing different types of food stores, packaging of a standard purchase in a retail park has an impact 2.5 times higher than in a municipal market due to three main reasons: the overuse of primary packaging (overpacking), the total amount of materials, and the higher presence of multi-material packaging (Sanyé et al. 2012).

In this context, EU Directive 94/62/EB and the subsequent directives (European Council 1994, 2004, 2005, 2009a) established a framework for environmental requirements in packaging production, as well as determined recovery and recycling targets for waste packaging. Based on these requirements a new packaging product can enter the market only if the manufacturer has taken all measures to reduce its impact on the environment without degrading its essential functions. Other legislation also aimed to establish a framework for better managing waste packaging, such as Decision 97/129/EC on the identification system for packaging materials (European Council 1997).

The main strategies to optimize packaging design for this legal framework were based on "packaging optimization" in order to reduce the waste packaging. The four strategies most used for this purpose are as follows (Hanssen et al. 2002, 2003):

(1) Optimize packaging to reduce the waste of products
(2) Optimize packaging to maximize the recycling of packaging materials
(3) Optimize packaging to minimize transport work and loss of efficiency in transport and distribution
(4) Optimize packaging by minimizing material consumption

This chapter aims to show the eco-design and product carbon footprint (PCF) methodologies in the packaging sector. The use of eco-design and carbon footprint methodologies are introduced (Sect. 2). Different packaging products from different sectors (Sect. 3) are assessed along with the eco-design and PCF methods in order to improve their environmental performance. The common issues regarding the implementation of PCF accounting in packaging systems and their materials are presented (Sect. 4). The eco-design methodology is applied to five different packaging systems: a multipurpose industrial packaging (Sect. 5), a detergent bottle (Sect. 6), a technical packaging for lighting products (Sect. 7), and two food packaging products (Sects. 8 and 9). Finally, a comparative assessment among the results is performed in order to show the main differences among sectors (Sect. 10).

2 Eco-Design and Carbon Footprint in Packaging

Eco-design is the integration of environmental aspects into the design process in order to improve the environmental performance of the entire lifecycle of a product (EU Directive on Eco-design) (European Council 2009b). This tool provided to be useful in the improvement of packaging products in order to meet the legal requirements. Common eco-design strategies implemented in the packaging sector are related to material selection (e.g., use of renewable or biodegradable

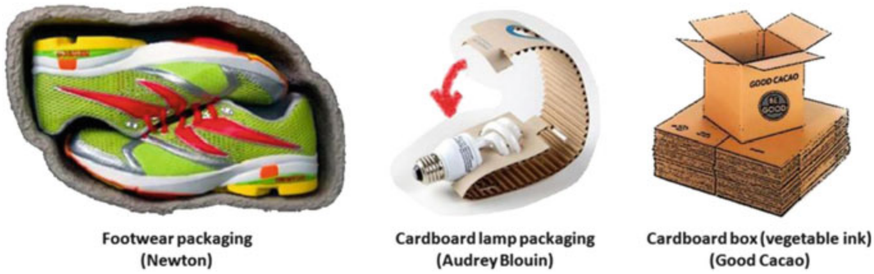

Fig. 2 Case studies for optimizing packaging volume and selecting renewable materials: footwear packaging (Newton), cardboard lamp packaging (Audrey Blouin), and cardboard box printed with vegetable ink (Good Cacao)

materials) (Fig. 2), optimization of the volume (i.e., to decrease the transportation impact) (Fig. 2), and multifunctionality of the packaging in order to increase its lifespan as well as to attract the customer (Fig. 3). As usually applied in eco-design projects, other packaging case studies also focused on consuming local materials (e.g., González-García et al. 2011).

On the other hand, product carbon footprint (CF) (BSI 2011; ISO 14067) is used as a communicative tool for companies to show the customer the environmental performance of their products in terms of greenhouse gas (GHG) emissions. This

Fig. 3 Multifunctional designs for packaging products: **a** Packaging convertible into a spoon (SpoonLidz), **b** cardboard pack convertible into a handle (Hangerpak), and **c** paper bag convertible into a handle for clothes (Muji)

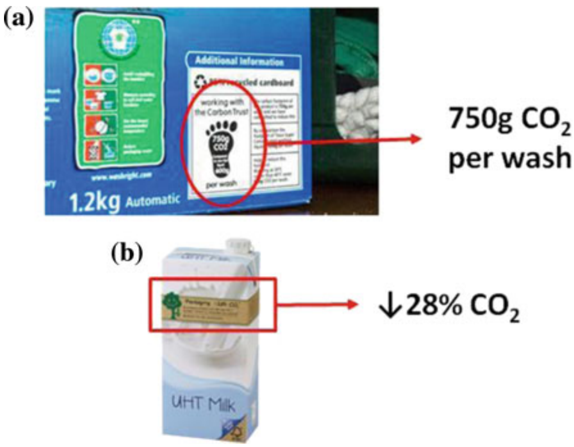

Fig. 4 **a** Packaging used as a communicative channel for consumers: carbon footprint of food products in Tesco supermarkets (UK). **b** Carbon footprint of packaging improvements of the new design Combibloc EcoPlus of the company SIG

tool is useful and understandable by the general public because climate change and global warming issues have been explained by the mass media and CO_2 units are already used in consumable products (e.g., vehicles and emissions per km).

Moreover, packaging can become a communicative channel for the company when used as a platform to inform the customer about eco-labeling, marketing, design, and aspects of the company. In this sense, PCF has been used as a tool for environmental communication to the user not only about the product (Fig. 4a) but also about the packaging itself (Fig. 4b).

3 Case Studies and Methodology

Five eco-design projects in the packaging sector are presented in this chapter. The projects were implemented in different sectors, from industrial to food packaging, and included both primary and distribution packaging (Table 1). All the projects were realized by a collaborative team made of research entities and the company involved.

The projects were performed within the development of the Catalan Ecodesign program (Catalan government). The Catalan Ecodesign program 2004–2006 was a pioneer experience in Catalonia that aimed to disseminate the eco-design methodology among the Catalan business network. The project was driven by the Catalan administration through the Centre for the Enterprise and the Environment, jointly the collaboration of the association from the business confederation of the county of Terrassa, the involved companies, and the Institute of Environmental

Table 1 Characterization of the case studies: economic sector, packaging, and type of packaging

Sector	Packaging	Type of packaging
Industrial	Multiuse packaging	Distribution
Chemical products	Detergent packaging	Primary
Technical products	Lighting packaging	Distribution
Food	Meat tray	Primary
Food retail	Delicatessen	Primary—distribution

Science and Technology). Therefore, it is an interdisciplinary project developed by a cooperative network within the administration, companies, and the university.

The goals of the Catalan Eco-design project are to encourage eco-design as an eco-efficient and innovative tool, to facilitate the incorporation of eco-design strategies in the business processes, to develop eco-design tools for economic sectors (such as guides and software), to train professional in product environmental prevention techniques, to communicate and to disseminate the program results in order to boost environmental improvements in the Catalan industry, and to create the Catalan agency of eco-products in cooperation with other administrations and institutions.

The eco-design methodology is detailed in González-García et al. (2011). The main steps are definition of the product, evaluation of the product, definition and selection of the strategies, and design and validation of the prototype. Regarding the qualitative assessment of life cycle criteria (QALCC) (CPRAC 2012) stage, the lifecycle stages included and the aspects evaluated are described in Table 2.

The quantitative evaluation method used was the life cycle assessment (LCA) (ISO 2006). Three indicators were used to assess the environmental performance of the product. First, the normalized CML value was used to show the global environmental performance of the product and its improvements. This indicator is obtained through the CML 2 Baseline method (Guinée et al. 2000) for the classification and characterization steps. This method includes 10 indicators that assess different environmental aspects: abiotic depletion potential, acidification potential, eutrophication potential, global warming potential (GWP), ozone layer depletion potential, human toxicity potential, ecotoxicity (fresh water, marine, and terrestrial) and photochemical oxidation.

Second, the product carbon footprint (BSI 2011; ISO 14067 2013) was used to show the contribution to the GWP of each product (see Sect. 4). This indicator was chosen as a well-known and understandable indicator for companies (i.e., CO_2 trade, climate change awareness, mass media publications, and eco-labeling). Finally, the cumulative energy demand (CED, MJ) (Hischier et al. 2010) showed the global energy consumption. Moreover, the packaging improvements were also evaluated through some indicators related to packaging design. The weight, the volume of the packaging, and the transport volume (number of units per truck capacity) were assessed as design aspects.

Regarding the PCF implementation, the PCF methodological specifications were followed in this chapter. According to the PAS 2050 (BSI 2011) method, the time

Table 2 Life cycle stages and aspects of the packaging products included in the qualitative assessment of life cycle criteria

Life cycle stage	Evaluated environmental aspect
Concept	Dematerialization
	Multifunctionality
	Optimization of the function
Materials	Elimination of the toxic compounds
	Use of recycled material
	Reduction of material use
	Reused material
	Use of renewable resources
Processing	Optimization of waste generation
	Reduction of water and energy consumption
	Energy savings
	Use of renewable energy
Distribution	Optimization of volume
	Use of recycled materials in secondary packaging
	Use of reusable secondary packaging
	Use of low-impact fuel
Use	Communication to user
	Information about the material
	Durability
End of Life	Reutilization potential
	Recyclability potential
	Energy valorization potential
	Reduction of the final waste volume

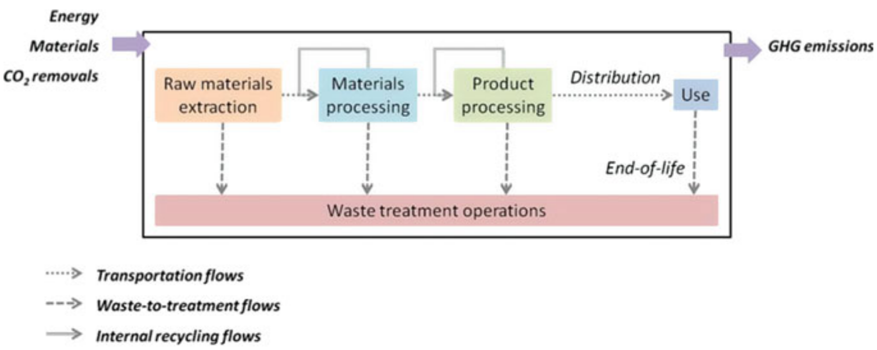

Fig. 5 Life cycle stages of a packaging product from a cradle-to-cradle approach. Processes and flows considered in the product carbon footprint accounting

period chosen for the assessment was 100 years. The last IPCC coefficients were used for the conversion from air emissions to CO_2 equivalent units. A cradle-to-cradle approach was considered for the PCF accounting. The system boundaries and the common processes of the packaging materials are described below (Fig. 5).

4 Overview of the PCF of Packaging Systems

The PCF is commonly used in the market (see Sect. 1). However, LCA and indicators such as GWP are more broadly used in the literature when accounting for the environmental burdens of packaging products. Two main packaging sectors are found in the literature: industrial packaging and food packaging.

Gasol et al. (2008) quantified the environmental burdens of two different options for distributing electrical cable or optic fiber. A wood pallet and a wood spool were analyzed from a cradle-to-grave perspective following the IPCC (2007) method for accounting the GWP. The GWP value obtained for a wood pallet was of 8.18 kg CO_2 eq, whereas the wood spool accounted for 87.1 kg CO_2 eq. Manuilova (2003) analyzed the direct emissions of industrial packaging for chemicals from a life cycle perspective. Considering a functional unit of 1.000 L of chemicals contained, the direct emissions for the different products were 61 kg CO_2 for a bulk container, 70 kg CO_2 for a composite drum, 53 kg CO_2 for a plastic drum, and 52 kg CO_2 for a steel drum.

In the field of food packaging, several studies have included the packaging as part of the life cycle of a food product, such as for beer (Hospido et al. 2005) or the banana supply chain (Svanes and Aronsson 2013). In table 3 recent studies about food packaging are compiled in order to show the GWP of different packaging systems. Most of them apply the LCA methodology for the calculations, apart from Svanes and Aronsson (2013), in which the PCF (ISO 14067) is followed. Also related to the food sector, Sanyé et al. (2012) analyzed the packaging related to food purchases, comparing two different retail options: municipal markets and commercial parks.

In a previous work, the common materials of packaging products (e.g., thermoplasts) were analyzed and their PCF accounted in order to address the use of certain materials. The PCF per kilogram of the material (in terms of CO_2 equivalent) was obtained for polyethylene (PE) (high density—HDPE, and low density—LDPE), polypropylene (PP), polyvinylchloride (PVC), polyethylene terephthalate (PET), corrugated cardboard, and wood (softwood). For each material, the largest GHG emitted and the main contributing processes were identified (Table 3). Local data from companies and the Spanish mix were used as foreground data, whereas background data were obtained from the Ecoinvent 2.2 database (Ecoinvent 2007; Frischknecht et al. 2004).

The PCF of the materials analyzed ranged from 0.065 to 3.77 kg CO_2 equivalents. The least impact materials are the renewable ones: wood and cardboard. Both are mainly used for secondary packaging purposes, although in some sectors they have a higher presence (e.g., industrial packaging). Thermoplasts are largely used in the packaging sector. PCF depends mainly on the country because electricity is the main contributing process to the environmental burdens. Within them, polyethylene and polypropylene are the least impacting materials (Table 4).

Table 3 Recent studies on the global warming potential of food packaging products by study, packaging, global warming potential (GWP), approach, and method

Study	Packaging	GWP ($g\ CO_2$ eq)	Approach	Method
Pasqualino et al. (2011)	Juice 1L aseptic carton	113	Cradle-to-grave	IPCC (2007)
	Beer 330 mL aluminum can	826	Cradle-to-grave	IPCC (2007)
	Water 1.5L PET bottle	78	Cradle-to-grave	IPCC (2007)
González-García et al. (2011)	Wine—wood box	314	Cradle-to-gate	IPCC (2007)
Madival et al. (2009)	Strawberries—PLA clamshell	171	Cradle-to-grave	IMPACT 2002+
	Strawberries—PET clamshell	198	Cradle-to-grave	IMPACT 2002+
	Strawberries—PS clamshell	165	Cradle-to-grave	IMPACT 2002+
Toniolo et al. (2013)	Sliced meat—PET tray	78.3	Cradle-to-grave	ReCiPe 2008
	Sliced meat—Multilayer tray	82.4	Cradle-to-grave	ReCiPe 2008
Humbert et al. (2009)	Baby food—glass jar	174	Cradle-to-grave	IMPACT2002+
	Baby food—glass pot A	125	Cradle-to-grave	IMPACT2002+
	Baby food—glass pot B	149	Cradle-to-grave	IMPACT2002+
Svanes and Aronsson (2013)	Banana packaging	80	Cradle-to-grave	Product carbon footprint ISO 14067
Albrecht et al. (2013)	Wood box for fruit and vegetables (15 kg)	2920	Cradle-to-grave	CML method
	Cardboard box for fruit and vegetables (15 kg)	3250	Cradle-to-grave	CML method
	Reusable plastic tray for fruit and vegetables (15 kg)	430	Cradle-to-grave	CML method

Table 4 Product carbon footprint (PCF) of different packaging materials, most emitted greenhouse gases, and main contributing processes to global warming

	PCF ($kg\ CO_2$ eq/kg)	Greenhouse gases	Main contributing processes
HDPE	1.65	CO_2, CH_4	Electricity consumption
LDPE	2.27	CO_2, CH_4	Electricity consumption
PP	2.02	CO_2, CH_4	Electricity consumption
PVC	2.66	CO_2, CH_4	Electricity consumption
PET	3.77	CO_2, CH_4	Electricity consumption
Corrugated cardboard	0.957	CO_2, CH_4	Raw material obtaining
Wood	0.065	CO_2, CH_4	Electricity consumption

5 Packaging for the Industrial Sector

An industrial box for different purposes was selected as a representative product of the industrial sector. A company produces and distributes packaging products and their designs may accomplish conditions for containing different products (e.g., weight resistance). The TriBox industrial box is mainly made of two materials (Fig. 6): cardboard and wood. The box is made of triple-channel cardboard that makes an envelope, and it is reinforced with wood pieces. The box is also reinforced with two wood pieces in the cover. Finally, the set is integrated with a pallet. This product was designed for processing, internal logistics, storing, and distribution purposes.

The company proposed a design briefing based on two key objectives: to obtain a monomaterial product and to facilitate the end-of-life management of the product. The quantitative assessment highlighted the importance of the end-of-life stage due to the difficulty for disassembling both materials (i.e., for recycling, reusing), which accounts for more than the 60 % of the CML normalized impact. Materials extraction and processing had also an important role in the carbon footprint ($\approx$50 %) and energy ($\approx$85 %) indicators, where the cardboard processing was the main contributing process. The PCF of the initial Tribox is of 16.13 kg of CO_2 (Fig. 7).

According to that, the implemented strategies were based on design for disassembly, to reduce the amount of materials and the number of different materials. These strategies aimed to facilitate the end-of-life management while optimizing the environmental impact of the materials selected. The new Tribox design is composed of the following main elements (Fig. 7): a cardboard box made of DC cardboard, a cardboard cover for the box (DC cardboard), corner reinforcement pieces (DC cardboard), and a nonintegrated pallet (wood). Although wood and cardboard are also the materials used for this design, the box can be easily disassembled and, therefore, the materials can be separated for being recycled or recovered at the end of life. Moreover, the wood pallet can now be reused while

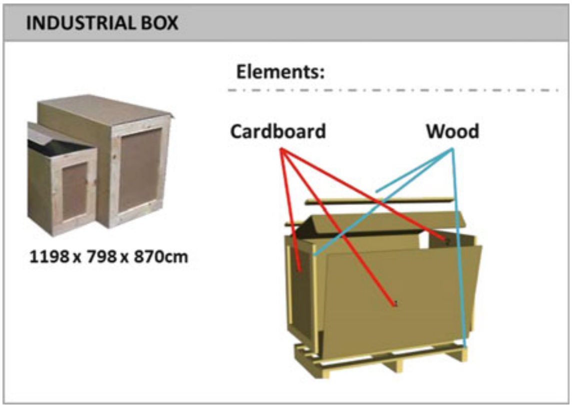

Fig. 6 Initial product, image, and elements of the industrial packaging (*Source* Emabamat)

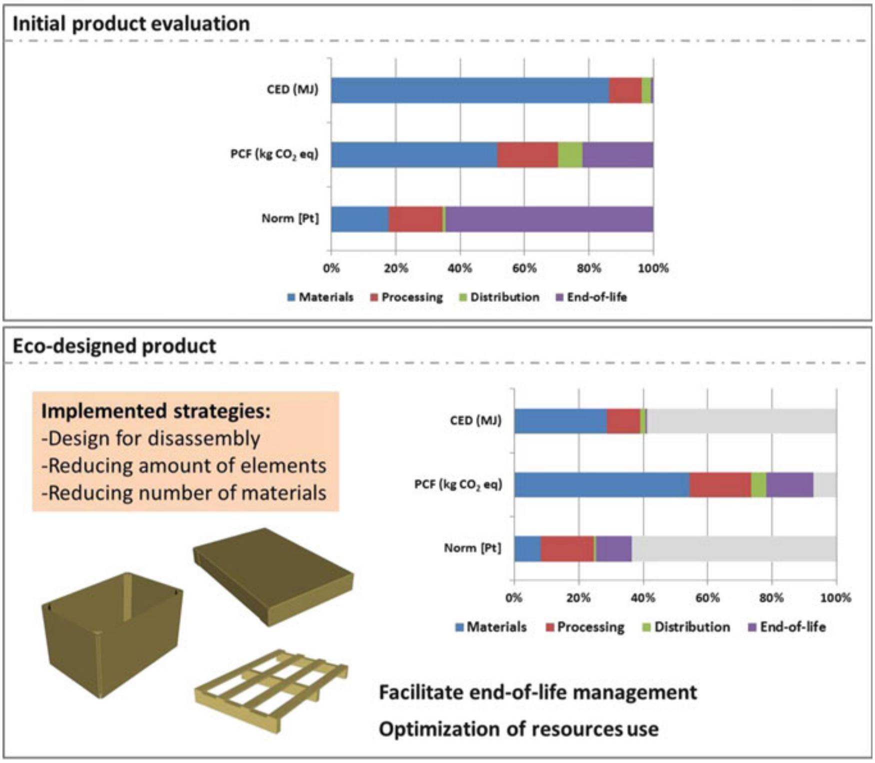

Fig. 7 Initial product evaluation of the industrial packaging: Quantitative assessment by life cycle stage. Eco-design product: implemented strategies and qualitative validation (*gray* shows the reduced amount for each indicator). The cumulative energy demand (CED), product carbon footprint (PCF) and normalized CML impact (Norm) are assessed as indicators

enlarging its lifespan. Finally, the amount of materials and the number of elements were optimized for reducing the environmental impact of the materials extraction and processing stage.

The weight of the product is reduced by almost 35 % due to the optimization of materials used in the box design. This positively affects the environmental issues of the product because the transportation requirements are reduced. The environmental indicators showed reductions from 7.2 % (carbon footprint) to 63.5 % (CML normalized). The facilitation of the end-of-life management contributes significantly to the reduction of the environmental impact (Table 5).

Table 5 Quantitative indicators for the eco-designed industrial packaging regarding design (weight, volume, and transport volume) and environmental improvements (CML norm, product carbon footprint [PCF], and cumulative energy demand [CED])

	Design	Environmental		
	Weight [kg]	CML norm [Pt]	PCF [kg CO_2eq]	CED [MJ]
Initial	25.57	2.01E−11	16.13	603.02
Eco-design	16.71	7.31E−12	14.96	247.73
Variance (%)	−34.65	−63.5	−7.2	−58.9

6 Packaging for Chemical Products

For the case study of chemical product packaging, a detergent bottle was selected. The company aims to improve the environmental performance of the packaging as well as to differentiate the product from their competitors. Moreover, the resulting eco-design strategies are expected to be implemented in other products of the company.

The packaging is a standard bottle for detergent with a volume of 2 L. There are three elements that compose the packaging: a cap (PP), which includes a measuring cup; a bottle (HDPE), with an oval base that includes a handle to facilitate its transportation and usage; and a label (PP) that includes advertising and information about use, toxicology, and environmental issues (Fig. 8). The bottle is obtained through a blowing molding, while the processing used for the cap is injection molding and flexography for the label.

As a result of the qualitative assessment, the distribution and the concept stages were identified as the critical ones. First, there is a need to optimize the packaging for distribution. Second, the packaging is not considered to be innovative in their sector. On the other hand, the technologies used for the processing are identified as optimal for the design and the materials used. However, the quantitative

Fig. 8 Initial product, image, and elements of the technical packaging for a detergent bottle (*Source* KH Lloreda)

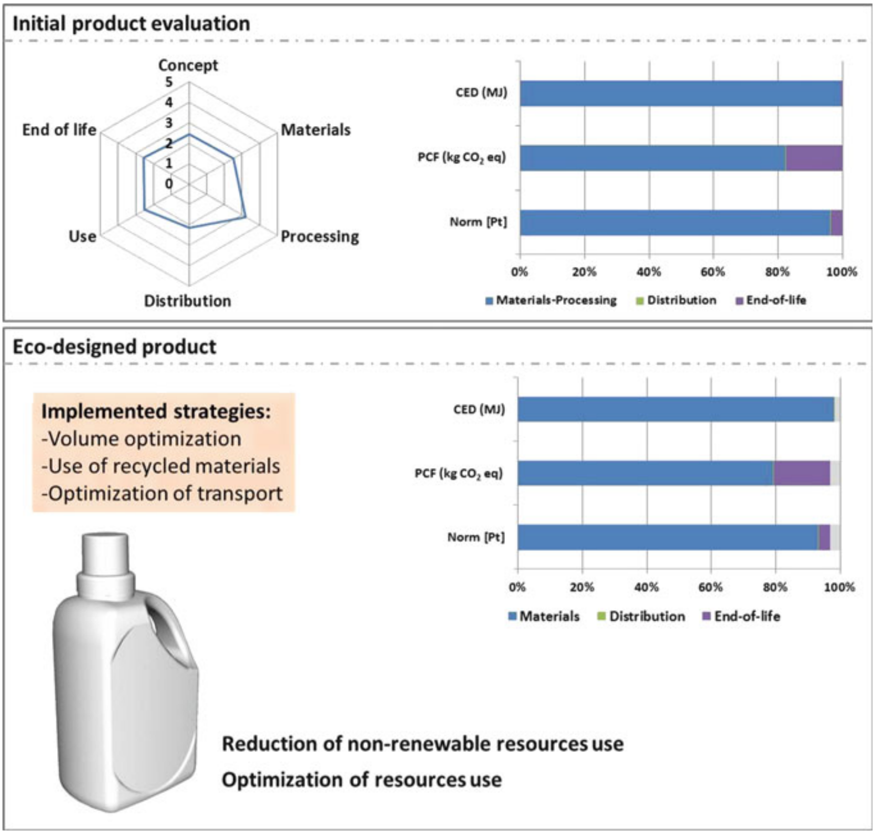

Fig. 9 Initial product evaluation of the detergent bottle: Qualitative and quantitative assessment, by lifecycle stage. Eco-design product: implemented strategies and qualitative validation (*gray* shows the reduce amount for each indicator). The cumulative energy demand (CED), product carbon footprint (PCF), and normalized CML impact (Norm) are assessed as indicators

assessment focused the attention on the materials and processing stages, which accounted for more than the 80 % of the environmental burdens. The environmental impact corresponds mainly to the HDPE bottle, which has the highest weight of the entire packaging. However, the carbon footprint of the packaging highlighted also the contribution of the disposal of the product in a sanitary landfill to the GHG emissions. The detergent packaging obtained a carbon footprint of 322.57 g of CO_2 (Fig. 9).

The resulting strategies for the eco-designed products, therefore, focused on optimizing the use of materials and improving the distribution issue. First, the shape and design of the bottle was modified. The volume was changed into a smaller but wider bottle (volume reduction of 20 %), with a functional handle that occupies less space. Second, the HDPE for the bottle is changed to recycled HDPE

Table 6 Quantitative indicators for the eco-designed detergent bottle regarding design (weight, volume, and transport volume) and environmental improvements (CML norm, product carbon footprint [PCF], and cumulative energy demand [CED])

	Design			Environmental		
	Weight [g]	Unit volume [cm^3]	Transport volume [u/truck]	CML norm [Pt]	PCF [g CO_2eq]	CED [MJ]
Initial	80	3917	36	5.58E−14	322.57	10.45
Eco-design	80	3132	48	5.41E−14	312.49	10.28
Variance (%)	0	−20	+25	−3.1	−3.1	−1.6

in order to reduce the consumption of nonrenewable materials. Finally, the design modification resulted in an optimization of the distribution stage (Fig. 9).

Although the weight and materials use is not reduced, the other design indicators resulted in positive outcomes. First, the volume of the product is optimized (20 % lower). As a result, the transportation is optimized as 25 % more product can be transported per truck. On the other hand, the environmental impacts were reduced up to 3.1 %, both for the global indicator (normalized CML) and the PCF, while the energy consumption was reduced by 1.6 % (Table 6).

7 Packaging for Technical Products (Lighting Sector)

As packaging for technical products, the packaging system for a lighting product was selected. The product was chosen as representative of the packaging used in the company as well as a multipackaging system for a lighting compounded by various parts.

The selected packaging is composed of three different packaging related to each part of the lighting: screen, mast, and base (Fig. 10). The screen is blocked by six pieces (expanded PE) situated in the corners and the sides of the screen. Then, the product is thermo-shrink-wrapped and packed in a cardboard box. Second, the mast is protected with longitudinal block pieces (expanded PE) and thermo-shrink-wrapped. Finally, the base is protected with two block pieces in the sides and is packed in a cardboard box. The main function of the packaging is to protect the different elements of the lighting during the transportation and storage of the product. Moreover, the packaging is expected to differentiate the products of the company from the competitors, and the logo in the different pieces is used for this purpose.

The use and materials lifecycle stages were the least rated in the qualitative assessment. First, the lifespan of the packaging should be adapted to the product, and more information about the materials should be provided to the customer. Second, the use of different materials is perceived as a negative environmental aspect of the product. On the other hand, the processing and the distribution are

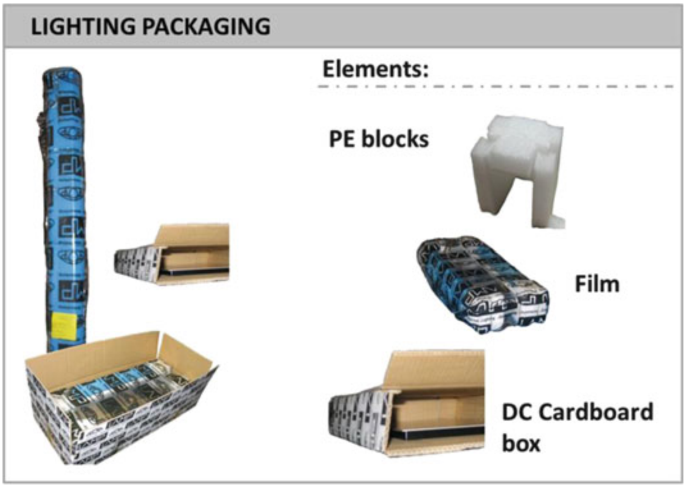

Fig. 10 Initial product, image, and elements of the technical packaging for a lighting product (*Source* Lamp)

considered the most environmentally friendly stages due to the optimization of the process and the fact that secondary packaging is avoided.

The materials extraction and their processing is pointed out as the most contributing lifecycle stage of the packaging (>72 %). Specifically, the PE blocks and film are the most impacting elements, even though cardboard is the most used material. Despite its low contribution to the carbon footprint and the energy consumption indicators, the end-of-life stage has an important role in the global environmental indicator by accounting for approximately 25 % of the impact. Finally, the PCF of the packaging is 4.61 kg of CO_2 and the distribution of the product contributes with 7 % (Fig. 11).

The strategies implemented in the new design are focused on reducing the amount of resources used, reducing the number of materials, and reducing the consumption of nonrenewable materials. The most impacting elements (PE blocks) were eliminated and substituted by elements made of renewable materials (cardboard). The new design is mainly composed of cardboard elements, and the different materials can be disassembled easily while facilitating end-of-life management (Fig. 11).

Regarding the design aspects, the weight of the packaging was reduced by 4 % and the volume by 36 %. Moreover, the facing area was increased by 8 % (in the eco-design product, it was 2.11 m^2). These improvements optimized the environmental requirements of the distribution stage as well as the use of resources in the packaging itself. The environmental indicators obtained important reductions, from 35.3 to 52.8 %. The energy consumption is the most reduced indicator; the change from plastic to cardboard implies a reduction of fuel consumption. The PCF is reduced by 35.3 % and the distribution is still the second most contributing lifecycle stage (Table 7).

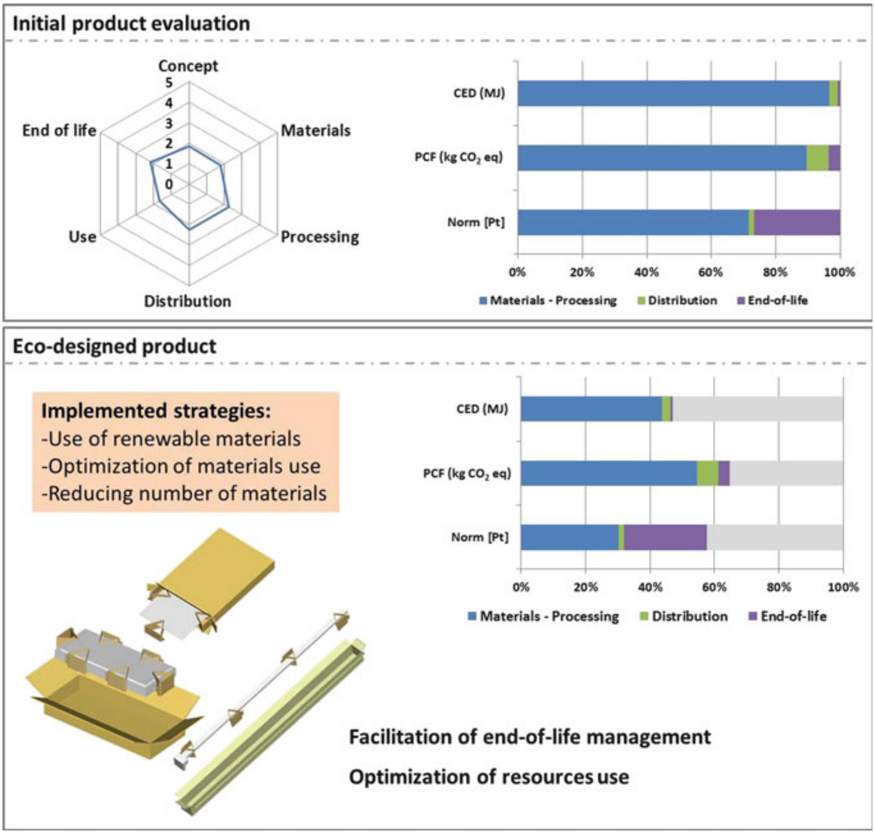

Fig. 11 Initial product evaluation of the technical packaging for a lighting product: Qualitative and quantitative assessment, by lifecycle stage. Eco-design product: implemented strategies and qualitative validation (*gray* shows the reduce amount for each indicator). The cumulative energy demand (CED), product carbon footprint (PCF), and normalized CML impact (Norm) are assessed as indicators

Table 7 Quantitative indicators for the eco-designed lighting packaging regarding design (weight, volume, and transport volume) and environmental improvements (CML norm, product carbon footprint [PCF], and cumulative energy demand [CED])

	Design		Environmental		
	Weight [kg]	Unit volume [cm^3]	CML norm [Pt]	PCF [kg CO_2eq]	CED [MJ]
Initial	2.30	43875	2.57E − 12	4.61	162.70
Eco-design	2.21	28080	1.48E − 12	2.98	76.75
Variance (%)	−4	−36	−42.3	−35.3	−52.8

8 Packaging for the Food Sector

A minced meat tray was selected for the food sector packaging case study. The company produces meat products and retails to supermarkets within Spain and Portugal. Prior to the study, the enterprise changed some cardboard packaging to trays in order to reduce the material amount per product while maintaining the functionality.

The minced meat tray was selected among different products as a representative multilayer product. The multilayer tray has a volume of 740 mL, of which 370 mL are controlled atmosphere gases; it contained 400 g of minced meat. The packaging is made of a transparent material composed of three layers: PET, EVOH, and PE. The packaging is composed of three elements (Fig. 12). First, a film (multilayer O-PET/PE/EVOH/PE) seals the tray, holding the protective atmosphere until the caducity of the product. Second, the tray itself is a transparent multilayer plastic made of PET (80 %), which gives shape to the product; EVOH (3 %), which seals; and PE, which guarantees the sealing of the film. Finally, a label made of coated paper contains information about the product, the logotype of the enterprise and quality labels.

The function of the packaging is to maintain the product in perfect condition for 12 days, 2 days of which correspond to the transportation stage and the other 10 days to the retail and use stages. Unlike traditional packaging, this type of packaging has the particularity that it almost doubles the lifespan of the packed meat. The packaging uses controlled atmosphere technology for improving the quality conditions of the product. For this purpose, the internal air of the packaging is eliminated and substituted by injected gases (CO_2 and O_2) that conserve the content beyond the normal lifespan of other refrigerated products. For an effective

Fig. 12 Initial product, volume, image, and elements of the minced meat tray (*Source* Arcadié)

protective atmosphere packaging, the material used should be as impermeable as possible to gases and water vapors to prevent migration.

In the qualitative assessment of the packaging (QALCC), the materials, use, and end-of-life lifecycles stages obtained the lowest punctuation. The multilayer materials, the longer lifespan of the packaging compared to the product, and the difficulties for its end-of-life management are the critical points. Regarding concept, attention is paid to the need for reducing the resource use of the packaging. Processing is the most rated stage due its optimal design. On the other hand, the quantitative assessment (LCA) highlighted that the most contributing lifecycle stages of the minced meat tray are the materials extraction and transportation (89 % of the normalized impact). The distribution of the product is the second most important stage, with contributions of approximately 25 % in the energy indicator and the carbon footprint. Within the distribution, the distribution packaging for the trays (cardboard boxes) is the main contributor. The amount of material per functional unit is high due to the low capacity of this secondary packaging. The PCF of the initial product accounts for 178.4 g of CO_2 per product (Fig. 13, Table 6).

According to the assessment results, eco-design strategies focus on the materials selection and design (e.g., optimization of materials use in relation to the lifespan of the packaging). The feasibility assessment and the potential compatibility of strategies resulted in a prototype design that included two of the proposed improvements. The new design varies the characteristics of the multilayer tray, while maintaining the other elements in order to ensure the function of the packaging (i.e., product production and sealing, and communication of the product). Moreover, with this selection, the company maintains the image of the product. The new tray has a new design that gives structure to the product while reducing the materials amount. This strategy accounts for a reduction of 15 % of the plastic. Second, the plastic is substituted by recycled material (Fig. 13).

The analyzed indicators showed that the strategies implemented account for a reduction between 8.6 and 50.9 %. Main reductions are done in energy consumption as the use of recycled plastic avoids the extraction of raw plastic from oil sources. The PCF is improved by 35.9 %, mainly due to the reduction of nonrenewable materials use. However, other environmental indicators obtained lower reductions than the PCF, and the normalized CML value decreases only 8.6 %. Regarding design, the eco-design packaging is 12 % lighter (Table 8).

9 Packaging for Food Retail

A delicatessen product was chosen for the food retail case study. Candy Glam Rings are candy jewelry created and sold by a specialized patisserie. The product was selected because it is a referent of the company image.

The rings are presented in a transparent box (like a showcase) and encapsulated in a case. The aspect of the packaging resembles that used in jewelry and

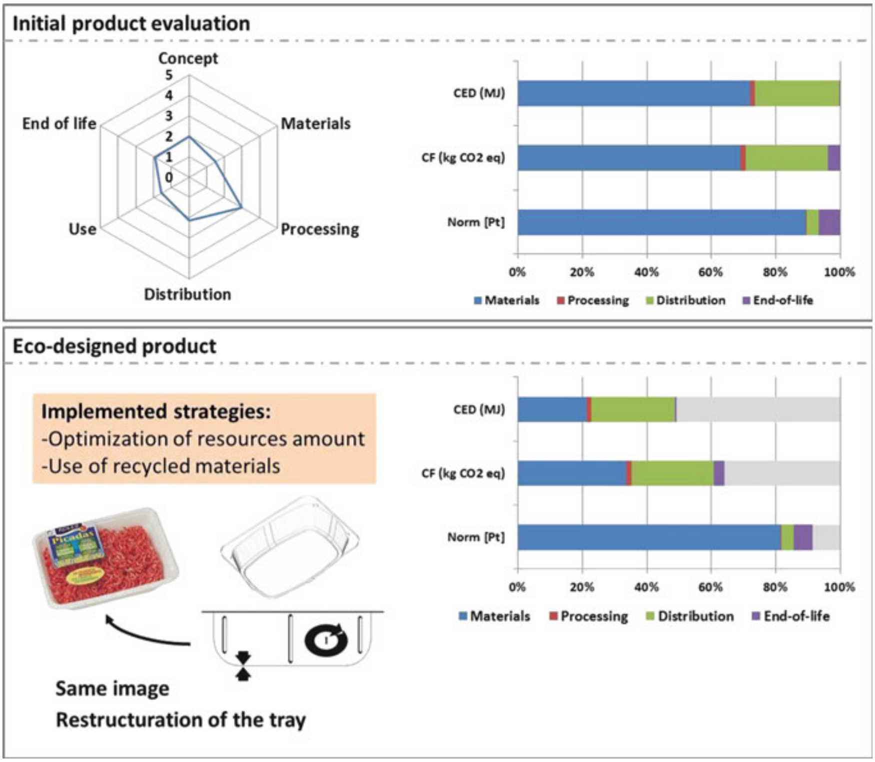

Fig. 13 Initial product evaluation of the minced meat packaging: qualitative and quantitative assessment, by lifecycle stage. Eco-design product: implemented strategies and qualitative validation (*gray* shows the reduce amount for each indicator). The cumulative energy demand (CED), product carbon footprint (PCF), and normalized CML impact (Norm) are assessed as indicators (*Source* Arcadié)

Table 8 Quantitative indicators for the eco-designed minced meat packaging regarding design (weight, volume, and transport volume) and environmental improvements (CML norm, product carbon footprint [PCF], and cumulative energy demand [CED])

	Design	Environmental		
	Weight [g]	CML norm [Pt]	PCF [g CO_2eq]	CED [MJ]
Initial	20.36	8.59E−13	178.4	4.50
Eco-design	17.92	7.85E−13	114.4	2.21
Variance (%)	−12	−8.6	−35.9	−50.9

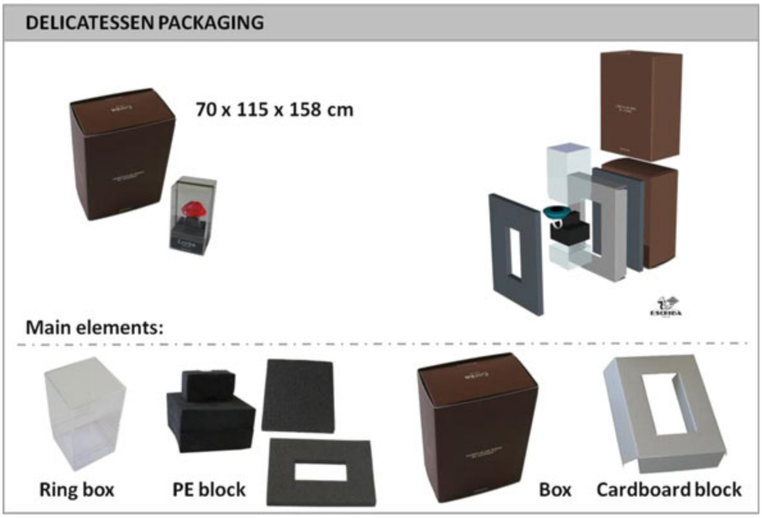

Fig. 14 Initial product, volume, image, and elements of the delicatessen packaging (*Source* Escribà)

perfumery, in order to differentiate the image of the product from other products of the company. The packaging is composed of multiple elements and made by different materials (Fig. 14). There are two main parts of the packaging: the showcase for the ring and the external case. The ring is placed in a soft block (PE) that fits in the transparent box (PS). This internal box is labeled (paper) and is sealed (PE). The external case is made of cardboard and has different block pieces made of cardboard and polyethylene (PE) in order to protect the ring showcase. The functions of the packaging are to protect the product and to show a high-end product image.

The worst result of the qualitative assessment was given to the concept of the packaging because it is not multifunctional despite its lifespan. Moreover, the materials and distribution stages were identified as potential areas to implement strategies. Regarding material, although the use of renewable materials is extended (cardboard), the amount of resources is large considering the product. Second, the transportation requirements of the product are considered as an important contributor to the environmental impacts (Fig. 15).

In the quantitative assessment, the materials extraction and processing were also identified as the most contributing lifecycle stages (40–65 %). Regarding materials, the polystyrene of the transparent box and the polyethylene blocks of the ring are the most impacting materials. Moreover, the processing of the cardboard (external case) has an important role due to the presence of this material in the packaging. The PCF of the product accounts for 708 g of CO_2 and most of the emissions are produced during distribution, mainly by airplane, as the product is sold around the world (Fig. 15).

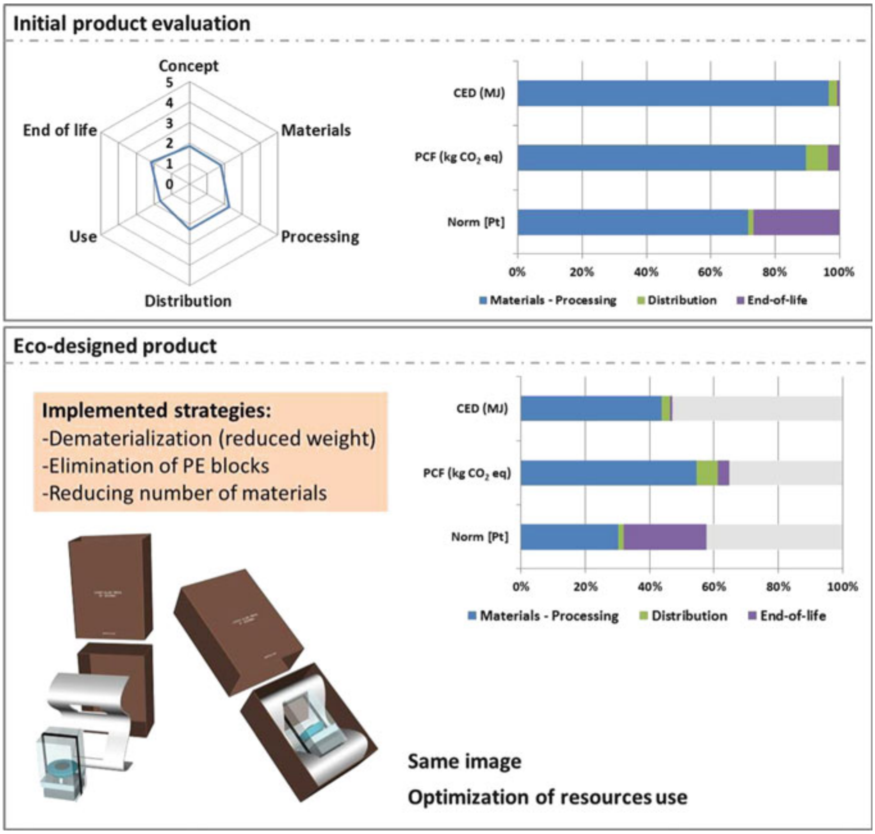

Fig. 15 Initial product evaluation of the delicatessen packaging: Qualitative and quantitative assessment, by lifecycle stage. Eco-design product: implemented strategies and qualitative validation (*gray* shows the reduce amount for each indicator). The cumulative energy demand (CED), product carbon footprint (PCF), and normalized CML impact (Norm) are assessed as indicators

The eco-design product was based on the optimization of the resource use (Fig. 15). First, the most impacting elements (PE blocks) were eliminated. Second, the packaging was dematerialized in order to reduce the weight of the product. This strategy was applied to the external cardboard case, which was lightened. Third, attention was paid to the reduction of the number of materials implemented in the design. In this sense, the internal blocks were changed for one mono-material block. Finally, the strategies aimed also to facilitate the end-of-life management of the product. However, some strategies were rejected as the luxurious image of the product must be maintained.

From the design perspective, the unit volume was optimized and reduced by 11.4 %, although the weight of the product was only reduced by 0.51 %. However, considering the small weight and volume of the packed product, the design could be more optimized. Regarding the environmental burdens, the global impact

Table 9 Quantitative indicators for the eco-designed delicatessen packaging regarding design (weight, volume, and transport volume) and environmental improvements (CML norm, product carbon footprint [PCF], and cumulative energy demand [CED])

	Design		Environmental		
	Weight [g]	Unit volume [m^3]	CML norm [Pt]	PCF [g CO_2eq]	CED [MJ]
Initial	118.67	1271.9	4.31E−13	709.46	16.28
Eco-design	118.06	1126.5	4.26E−13	699.37	16.50
Variance (%)	−0.51	−11.42	−1.1	−1.4	+1.3

(CML) is reduced 1.1 % while the energy consumption is increased by 1.3 %, as the use of cardboard is also accounted as renewable energy. The PCF is the indicator with highest reductions due to the optimization of the volume for transportation (Table 9).

10 Conclusions

The eco-design implementation in different packaging products resulted in a better environmental performance of the packaging. Regarding the design parameters, most of the case studies reduced their weight and volume. As a result, when quantifying the transport capacity, this was increased significantly and, consequently, the transport requeriments also decreased. Second, all of the case studies achieved a reduced PCF (from 1.4 to 35.9 % of reduction), a reduced environmental impact (CML norm, from 1.1 to 63.5 %), and a reduced energy consumption (from 1.6 to 58.9 %, apart from food retail case study) (Table 10).

Among the sectors analyzed, the size and weight of the packaging determine the absolute values of the PCF. Packaging systems for larger products obtained the greatest values: industrial packaging (16 kg CO_2 eq.) and technical packaging (4.6 kg CO_2 eq.). However, both packaging types had a longer lifespan related to the other case studies analyzed. First, industrial packaging is a multipurpose

Table 10 Improvement indicators [variance, %] for the eco-designed products regarding design (weight, volume, and transport volume) and environmental improvements (CML norm, product carbon footprint [PCF], and cumulative energy demand [CED])

	Design			Environmental		
Variance [%]	Weight	Unit volume	Transport volume	CML norm	PCF	CED
Industrial	−34.65	–	–	−63.5	−7.2	−58.9
Chemical	0	−20	+25	−3.1	−3.1	−1.6
Technical	−4	−36	–	−42.3	−35.3	−52.8
Food product	−12	–	–	−8.6	−35.9	−50.9
Food retail	−0.51	−11.42	–	−1.1	−1.4	+1.*3*

packaging that can be re-used in different areas of the company. Second, the technical packaging is designed not only for distribution but also for storage. However, the PCF of the single-use packaging cases primarily depends on the design and the materials used. The food retail packaging got the highest PCF value (709 g CO_2 eq.), even though it contained the smallest product (a candy ring). The design of the box is presumptuous in order to show a high-end product image and to make it similar to real jewelry. Therefore, a higher amount and variety of materials are used than what is actually needed for protection purposes.

In relative values (PCF per mass unit), food packaging accounted for the largest PCF results. First, the meat tray's PCF was of 8.8 g CO_2 eq. per gram of packaging, due to mainly the technical materials of the multilayer for food preservation. Second, the PCF of the food retail packaging resulted in 6.0 g CO_2 eq. per gram of packaging because of the luxurious design and the use of different materials, as mentioned above. Regarding the other sectors, differences depend on the type of material used in the packaging. The chemical packaging analyzed is made of thermoplasts and obtained a PCF per gram of packaging of 4 g CO_2 eq., while the technical packaging combined both plastic and renewable materials and had a PCF of 2.0 g CO_2 eq. per gram of packaging. Finally, the PCF of the industrial packaging resulted in the lowest value per gram of packaging (0.6 g CO_2 eq.), as most of the materials are from renewable sources (cardboard and wood).

Regarding the affectation of the eco-design process, the PCF is mainly reduced due to the optimization of the volume and therefore the improvement in transportation requirements, as the GHG emissions of transportation are the most contributing ones. The PCF is also largely improved when changing from plastic or nonrenewable materials (e.g., high density polyethylene, HDPE) to renewable ones (e.g., cardboard or wood), as the oil consumption is reduced. Lastly, the optimization of the end-of-life management of packaging products also decreased the PCF significantly due to the emissions in landfilling.

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Carbon Footprint of Crop Production and the Significance for Greenhouse Gas Reduction in the Agriculture Sector of China

Ming Yan, Kun Cheng, Ting Luo and Genxing Pan

Abstract World agriculture is facing a great joint challenge of ensuring food security and mitigating greenhouse gas emissions under climate change. Characterizing the carbon footprints of crop production by life cycle analysis is be critical for identifying the key measures to mitigate greenhouse gas emission while sustaining crop productivity in the near future. In this chapter, the carbon footprints of bulk crop production; individual staple crops of rice, wheat, and maize; as well as vegetable crops from China were analyzed using data from either statistical archive or of questionnaire survey for quantification of all carbon costs in a whole life cycle. Although the overall carbon footprint of crop production sector of China is much higher than that of the UK and USA, rice and wheat have significantly higher carbon footprints than maize. The nitrogen- fertilizer-induced footprint was shown to be the biggest contributor to the total carbon footprint for all the crops (more than 60 %), leaving a big space for mitigation of luxury emissions of N_2O with nitrogen use in excess. Although the carbon footprint has quickly increased since 1970s, crop production did not show a positive response to increasing carbon cost. While reducing nitrogen chemical fertilizer use is apparently a key option to cut down the highly carbon- intensive agriculture, substitution of rice or wheat with maize would offer a final option to ensure both high cereal production and low carbon cost in China's crop production sector. There is an urgent need to depict the variation of carbon footprints for different cropping and farming systems, climate conditions, and the threshold of nitrogen luxury emissions for a certain crop.

Keywords Carbon footprint · C cost · Crop production · Vegetables · Carbon management · Fertilizer · GHGs emission · Life cycle analysis

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1 Introduction: General Issues of China's Crop Production

Agriculture is a key sector in the global economy, which is critical for providing the food and fiber demanded by the huge population of 7 billion people in 2010. However, the agriculture sector contributes significantly to global warming from direct and indirect carbon emissions associated with crop production. To ensure food safety for the still fast-increasing population, world agriculture has been tackling the trilemma of high productivity, low greenhouse gas (GHG) emission, and adaptation to climate change (Smith et al. 2013). World agriculture emitted 5,100–6,100 Mt CO_2-eq year^{-1} and contributed approximately 30 % to the global total anthropogenic emissions, being the second greatest emitter after fossil fuel consumption (Smith et al. 2008). With the joint challenges of food security and climate change faced by the global society in the coming decades, world agriculture has been increasingly concerned with global solutions to mitigate climate change (FAO 2010).

The trilemma is even more critical for China, which has long struggled with a safe solution to adequate food supplies (Brown 1994). China preserved a total cropland of approximately 130 Ma and produced 0.59 billion tons of cereals in 2012. Rice, wheat, and maize were cultivated in an area of croplands of 29.8 Mha, 24.2 Mha, and 32.5 Mha, respectively, in 2010, overall making up 68.4 % of the total harvest area (160.7 Mha) and 80 % of the total arable land (109.9 Mha) in China. The yield for rice, wheat, and maize crop production were 195.8 M tons, 115.2 M tons, and 177.2 M tons, respectively, in 2010 (DRSES-SBSC 2011). Vegetables, melons, and fruits were planted in areas of 19.0 and 2.4 Mha, respectively, in 2010, accounting for 11.8 and 1.5 % of the total harvest area (160.7 Mha) in China. The yields for vegetable, melon, and fruit production were 65.1 M tons and 8.5 M tons, respectively, in 2010 (DRSES-SBSC 2011). However, the sustainability of China's agriculture is increasingly of concern in the light of excess use of nitrogen (N) fertilizer, soil degradation and pollution, and the vulnerability to climate change (Guo et al. 2010; Ju et al. 2009; Liu et al. 2010; Ye and Van Ranst 2009; Pan et al. 2011a). The high yield on the cost of high inputs for food production would certainly impact the greenhouse gas emissions from agriculture; the use of synthetic N fertilizers, for example, could cause a potential yearly emission of 400–840 Mt CO_2-eq, equivalent to 8–16 % of China's energy-related CO_2 emissions in 2005 (Kahrl et al. 2010). Contributing 9.2 % to the national total anthropogenic GHG emissions, China's agriculture emitted 686 Mt CO_2-eq in 2007 (Chen and Zhang 2010). China has committed to reduce GHG emissions by 40–45 % per unit of gross domestic product (GDP) until 2020 on the baseline of 2005 (Xinhua Net 2009). Mitigation in agriculture could offer a significant reduction in national total GHG emissions as well as cobenefits for crop production and ecosystem functioning. Recently, low carbon approaches have been encouraged by incentives under the national climate change mitigation strategy (Anonymous 2009; Anonymous 2012; NDRC 2012).

In this chapter, to offer basic formation on greenhouse gas emission intensity and the factors in China's agriculture, the carbon footprint will be characterized for staple and vegetable crops, along with a description of the methodology. Finally, a discussion is provided on policies and perspectives on future trends.

2 Methods for Quantifying Carbon Footprint in Agriculture

2.1 Rational of Accounting Approach

There have been many studies on the carbon footprint of agriculture since 2005. To characterize the GHG emissions of human activities in the production sector of industry, transportation, and human lifestyle as well as social activities, the concept of carbon footprint (CF) has been generally based on the accounting of all greenhouse gas emissions directly and indirectly caused in the whole life cycle of a product or an activity (BP 2005; POST-UK 2006; ETAP 2007; Wiedema et al. 2008; Finkbeiner 2009). Subsequently, the carbon intensity of overall greenhouse gas emissions could be assessed on the CO_2 equivalent of a product over the whole course of production (Woolf et al. 2010). Generally, the CF of crop production accounts for all carbon costs through individual inputs for crop production up to the farm gate (harvest) together with the emission factors for these inputs (St Clair et al. 2008; Hillier et al. 2009).

2.2 Procedure of Carbon Emission Accounting

In our studies, carbon footprint accounting was performed by basically following the protocol described by Hillier et al. (2009), in which the total carbon cost (CO_2-eq) was assumed to be the sum of emissions due to the energy consumption associated with chemical input and mechanical operations for spraying and tillage, harvesting, strapping, transportation, and irrigation, and the direct emissions of N_2O emissions from cropland due to N fertilizer application and CH_4 emissions from rice cultivation. Individual carbon cost of management activities or of agrochemical inputs can be calculated separately using the following formula:

$$CF_i = AI_i \times EF_i \tag{1}$$

where, CF_i is the GHG emissions induced by an agricultural input (in CO_2-eq); AI_i is the amount of agricultural input applied (in kilograms for fertilizer, pesticide, and plastic film, in liters for diesel oil due to machinery operation, or in kilowatt-

Table 1 Emission factors used for carbon footprinting

Emission source	Abbreviations	Emission factor or scaling factor	Literature
N fertilizer	$EF_{fertilizer}$	6.38 t CO_2-eq t^{-1} N	Lu et al. (2008)
P fertilizer		605.33 kg CO_2-eq t^{-1} P_2O_5	West and Marland (2002)
K fertilizer		441.03 kg CO_2-eq t^{-1} K_2O	
Pesticide	$EF_{pesticide}$	18.08 t CO_2-eq t^{-1} pesticide	West and Marland (2002)
Insecticide		1.32 t CO_2-eq t^{-1} insecticide	Hillier et al. (2009)
Herbicide		23.10 t CO_2-eq t^{-1} herbicide	
Fungicide		11.59 t CO_2-eq t^{-1} fungicide	
Plastic film	EF_{film}	2.5 t CO_2-eq t^{-1} film	Energy Source, China (2009)
Diesel oil for machinery	$EF_{machinery}$	2.63 kg CO_2-eq L^{-1}	BP China (2007)
Electricity for irrigation	$EF_{irrigation}$	0.92 kg CO_2-eq kw^{-1} h^{-1}	
Labor	EF_{labor}	0.9 kg CO_2-eq day^{-1} $person^{-1}$	Yang (1996)
Direct N_2O emission from N fertilizer	EF_{N_2O}	Dry cropland, 0.01 t N_2O-N t^{-1} fertilizer-N	IPCC (2006)
		Rice paddy, 0.0073 t N_2O-N t^{-1} fertilizer-N	Zou et al. (2007)
CH_4 emission from rice land	EF_c	1.30 kg CH_4 ha^{-1} day^{-1}	Yan et al. (2005)
	SF_w	0.52	
	SF_p	0.68	
	SF_m	1.0	

hours for electricity due to irrigation); i is the agricultural input, such as, fertilizer, pesticide, machinery operation, or irrigation, etc; and EF_i is the individual carbon intensity (in CO_2-eq per unit volume or mass) when manufactured and/or applied of individual agricultural input. The reference emission factors used in the estimation are listed in Table 1.

Direct N_2O emissions from N fertilizer can be estimated using Eq. 2:

$$CF_N = F_N \times \delta_N \times \frac{44}{28} \times 298 \tag{2}$$

where, CF_N represents the carbon footprint due to direct N_2O emissions from application of synthetic N fertilizer (in CO_2-eq ha^{-1}); F_N and δ_N is respectively the quantity (in kg ha^{-1}) of N fertilizer applied for crop production and the default emission factor of N_2O emission per unit of N fertilizer applied (in kg N_2O-N kg^{-1} N fertilizer); *44/28* is the molecular weight of N_2 in relation to N_2O; and *298* is net global warming potential (GWP) in a 100-year horizon (IPCC 2006).

For rice production, methane is produced with waterlogging in paddies, which could be estimated using the equation

$$CF_{CH_4rice} = EF_d \times t \times A \times 25 \tag{3}$$

where, $CFCF_{CH_4rice}$ represents the annual methane emissions from rice cultivation (in CO_2-eq); EF_d is a daily emission factor, (in kg CH_4 ha^{-1} day^{-1}); t is cultivation period of rice (in days); A is the size of rice farm (in ha); and 25 is the relative molecular warming forcing of CH_4 in a 100-year horizon (IPCC 2006). The factor of EF_d can be estimated using data from literature:

$$EF_d = EF_c \times SF_w \times SF_p \times SF_m \times SF_{s,r} \tag{4}$$

where EF_d is the adjusted daily emission factor for a particular rice area; EF_c is the baseline emission factor for continuously flooded fields without organic amendments; SF_w is the scaling factor to account for the differences in water regimen during the cultivation period; SF_p is the scaling factor to account for the differences in water regimen in the pre-season before the cultivation period; SF_m is the scaling factor, which should vary for both type and amount of organic amendment applied; $SF_{s,r}$ is the scaling factor for soil type, rice cultivar, etc., if available.

In particular, labor was taken into account in the counting to avoid bias from machinery operation in China for operations of fertilizing, tillage, and harvesting performed with labor in many cases. This is estimated with the following equation:

$$CF_{labor} = N \times EF_{labor} \tag{5}$$

where N is the total number of days for labor input and EF_{labor} (in CO_2-eq day^{-1} $person^{-1}$) is the carbon dioxide respired by an adult per day.

As generally accepted, the GHG emissions with disposal or treatment of crop residues were not considered in this study. In addition, CO_2 emission due to soil respiration, being a very small contribution to global CO_2 emission (Bellarby et al. 2008), was not considered in the CF analysis.

2.3 Data Used for CF Accounting

2.3.1 Statistical Data

Data from retrieved from China Rural Statistical Yearbook series of 1993–2007 was used. This data included cropland area, total production, and the various inputs of fertilizers, pesticides, diesel, plastic films, and electricity involved in crop production, which were recorded annually. Because the crops in this data set covered all the crops cultivated and the production was represented as a bulk sum of rice, wheat, corn, beans, potato, vegetable, fruits, cotton, oil, and sugar crops, an overall carbon footprint of the crop production sector of China was determined (Cheng et al. 2011).

2.3.2 Farm Survey Data

For assessing the footprint of different crops under different management practices and different cropping/farming systems, data can be obtained via surveys to farmers about the input for their production of a single crop growing season under local conditions. This is usually done with a questionnaire sheet (Table 2) for data input via face-to-face interview with the farmers who manage the crop production (Chen et al. 2011; Yan et al. 2013). A dataset (Table 3) was then established for further quantification processes and statistics with carbon footprinting software.

Table 2 A data input sheet used in questionnaire survey to farmers

___ Province ___ City (GPA position) Investigator: ___ Date: ___				
Crop: ___	Farm size: ___	Yield: ___ (kg)	Location: ___	Interviewee: ___
Agro-chemical inputs	Fertilizer type	N %, P_2O_5 %, K_2O %	Amount (kg) and times	Expense
	Pesticide type	Amount (g or mL) and times		
	Others			
	Film			
	……			
Machinery operation	Please choose √	Power (kw/h) or diesel oil (L)	Work hours and times	
	☐ Seeding			
	☐ Tilling			
	☐ Spraying agricultural chemicals			
	☐ Harvest			
	☐ threshing			
	☐ Transport			
	☐ Others			
	….			
Irrigation	☐ Pumping			
	☐ Well irrigation			
	☐ Others			
Labor		Persons	Days or hours	
	☐ Seeding			
	☐ Weeding			
	☐ Fertilizing			
	☐ Spray pesticide			
	☐ Harvest			
	☐ Others			
Dispose of straw: ☐ Burning ☐ Returning straw to field ☐ Others				

Table 3 Examples of data coding in a dataset for quantification calculation

Code of region	Code of farm surveyed	Code of crop	Code of cropping system	Code of input	Code of grain yield
1: Humid 2: Semiarid 3: Arid 4: Boreal	1: Single household 2: Aggregated 3: Company owned	1: Rice 2: Wheat 3: Maize	1: Rice-wheat rotation 2: Double-rice rotation 3: Wheat-maize rotation	1: Fertilizer 2: Pesticides 3: Plastic film 4: Farm operation 5: Irrigation	1: Rice 2: Wheat 3: Maize

3 Overall Carbon Footprints of China's Crop Production

3.1 General Feature of Carbon Footprint

Using the statistical data retrieved from the China Rural Statistical Yearbook series throughout the period of 1993–2007 (DRSES-SBSC 2008), Cheng et al. (2011) conducted a basic estimate of the overall CF of crop production in China. The study showed an overall carbon cost of 0.44 Pg CO_2-eq on average annually for production of all crops, including rice, wheat, corn, beans, potato, vegetables, fruits, cotton, oil, and sugar crops in the time span. The work indicated an overall carbon intensity of 2.3–3.4 t CO_2-eq ha^{-1} yr^{-1} for cultivated lands and of 0.5–0.4 t CO_2-eq for per ton crop harvested on average of the whole time period of 1993–2007 (Cheng et al. 2011). China's total emissions from energy consumption were estimated to be 7.5 Pg CO_2-eq in 2005 (Anonymous 2012) and 8.4 Pg CO_2-eq in 2007 (Chen and Zhang 2010). The overall carbon emissions from CF of crop production estimated here corresponded to approximately 8 % to the nation's total emissions. Because agriculture (including livestock production) contributed 14 % to the total GHG emissions of the nation, crop production made up more than half of the overall sector.

3.2 Change in CF with Agricultural Development

This work also traced the dynamics of CF during the time period of 1993–2007. Although the total CF showed a linearly increasing trend with increasing crop productivity, mainly with the green revolution using new varieties and chemicals, the carbon intensity from croplands exerted a linear increasing but an exponential decrease from per ton of harvest since 1993. In a consistently upward trend, total CF of China's crop production increased from 346.1 Mt CO_2-eq in 1993 to 516.3 Mt CO_2-eq in 2007, by 49 % over the time span of 1993–2007.

Looking at the contribution of different inputs to the overall CF, on average, two-thirds of the total CF was from agrochemical inputs (Cheng et al. 2011). In particular, N fertilization averaged 55 % of the total CF, which was very closely linearly correlated with the overall carbon intensity both in terms of lands cultivated (see Fig. 5 in Cheng et al. 2011). Kahrl et al.(2010) developed a specific emission factor for China's N fertilizer manufacturing and application (5–31 t CO_2-eq t^{-1} N), and argued that large use of synthesized N fertilizers could led to total emissions of 400–840 Mt CO_2-eq in 2005, equivalent to 8–16 % of China's energy-related CO_2 emissions in that year. In their work, sales of all N fertilizers were taken into account for the higher estimation. In general agreement with their findings, the figures of N fertilizer-induced emission here also suggest that a reduction in N fertilizer use in China's crop production will offer a great option to reduce the national total GHG emissions. A reduction in N fertilizer use by 10 % could bring about a reduction in total carbon emission by 5 %, both in terms of land cultivated and mass produced.

Another big proportion was by irrigation energy consumption, which made a mean contribution of 22 % on average. The other inputs such as plastic film use and crop management performance by machinery use were less than 10 % of the total CF for crop production, although they also showed a significantly increasing trend. This first work demonstrated the high CF of China's bulk crop production, which has been characterized by high N fertilization and with an increasing carbon cost for increasing crop production. The high proportion of energy cost for irrigation highlighted the drought impact on China's crop production, which is increasingly critical due to an increasing drought frequency under the climate change conditions (Pan et al. 2011a; Lv et al. 2011).

Our work (Cheng et al. 2011) also showed that carbon cost or CF was greatly reduced with increasing gross harvest yield per hectare [C intensity (t CO_2-eq t^{-1}) $= 0.21 \times$ Yield (t ha^{-1}) $- 0.40$, $R^2 = 0.84$, $p < 0.01$]. Gross crop production was shown in a logarithm increasing function with total carbon cost [Harvest yield (t ha^{-1}) $= 11.38 \times \ln$ CF (t CO_2-eq ha^{-1}) $+ 9.83$, $R^2 = 0.96$, $p < 0.01$]. Also, crop production failed to increase beyond a high CF over 0.8 t CO_2-eq ha^{-1} (Fig. 1), indicating a problem of luxury carbon cost of approximately 0.2 t CO_2-eq ha^{-1} with the effort to keep up crop yield with continuously increasing inputs. In other words, increasing inputs is not a practical option to sustaining high yield over a given production capacity threshold. Similarly, Burney et al. (2010) argued that approaches for yield improvements should be cautious for climate change mitigation, as all efforts would not reduce GHG emissions.

4 Carbon Footprint of Staple Crop Production in China

Assessment of the carbon footprint of different staple crop production was done using farm survey data because there were no data in the statistical bureau specifically for different crop production. Questionnaire surveys were conducted to obtain data for the individual inputs used for crop production in representative regions of China's crop production during 2010–2012. A dataset was established

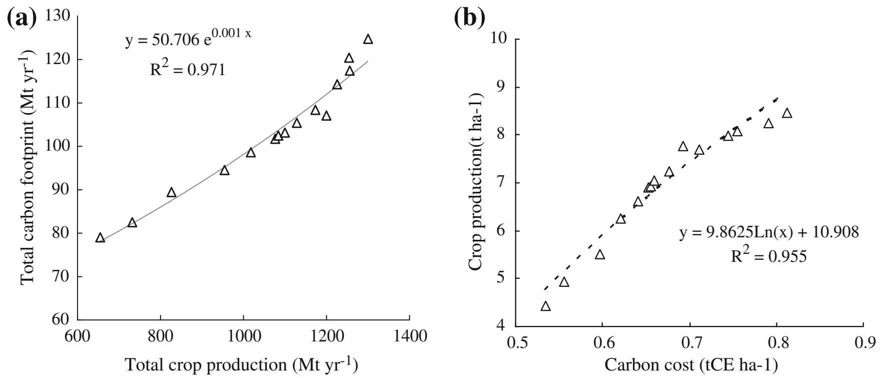

Fig. 1 Total CF in correlation with total harvested crop yield (**a**) and crop yield in correlation to CF (**b**) of China's crop production using agro-statistics data for 1993–2007 (Cheng et al. 2011)

of inputs of chemicals for fertilizers and pesticides and machinery operation for staple crops of rice, wheat, and maize crop from more than 130 household-managed farms over the representative crop production regions; it was used for quantification using the above-mentioned methodology.

4.1 CF of Staple Crop Production

Quantization using the farm survey data allowed a basic estimation of CFs for different crop production for the past years by the individual household farms, with varying size and crop productivity under different management practices.

The estimated mean total carbon emissions for crop production studied here ranged from 2,240.7 ± 131.9 kg CO_2-eq ha^{-1} and 326.9 ± 18.3 kg CO_2-eq t^{-1} for maize to 5,795.8 ± 117.6 kg CO_2-eq ha^{-1} and 769.0 ± 20.2 kg CO_2-eq t^{-1} for rice production in 2010. For rice, wheat, and maize; total cultivated croplands were 29.8 Mha, 24.2 Mha, and 32.5 Mha, respectively, in 2010 (DRSES-SBSC 2011), possessing 68.4 % of the total harvest area (160.7 Mha) and 80 % of the total grain cropland (109.9 Mha) in China. The yields of rice, wheat, and maize crop production were 195.8 M tons, 115.2 M tons, and 177.2 M tons, respectively, in 2010 (DRSES-SBSC 2011). Accordingly, the mean carbon intensity could be predicted on averaged as 3.7 t CO_2-eq ha^{-1} for overall croplands and 0.58 t CO_2-eq for grain production per metric ton in 2010, respectively. This result corresponded to the high values in the estimated range for bulk crop production reported by Cheng et al. (2011) during 1973–2007. This could be explained by the higher yield and the high input, shown in increasing trend of China's agriculture (Cheng et al. 2011). Hillier et al. (2009) reported a mean CF of 1.6 t CO_2-eq ha^{-1} yr^{-1} for crops in conventional farms in the UK based on survey data collected in 2006. The

CF of these three crop productions seemed much higher than the reported CF of 0.7–0.9 t CO_2-eq ha^{-1} and 0.27–0.42 kg CO_2-eq kg^{-1} of durum wheat grown under various cropping systems in southwest Saskatchewan, Canada (Gan et al. 2011). The higher carbon (C) intensity here demonstrated a high C cost of China's agriculture for achieving a high yield for food security of the nation (Liu and Zhang 2011). This could again be challenged by climate change with the increasing C cost during the period of 1993–2007, as shown in the work by Cheng et al. (2011).

4.2 *Difference in CF Between Major Crops*

For the three major staple crops surveyed, the CF was averaged (mean ± standard error) of 5,795.8 ± 117.6 kg CO_2-eq ha^{-1}, 3,000.4 ± 185.8 kg CO_2-eq ha^{-1}, and 2,240.7 ± 131.9 kg CO_2-eq ha^{-1}, with carbon intensity in the range of 769.0 ± 20.2 kg CO_2-eq t^{-1}, and 645.6 ± 32.6 kg CO_2-eq t^{-1}, and 326.9 ± 18.3 kg CO_2-eq t^{-1} for rice, wheat, and maize, respectively (Table 4). Clearly, rice production showed the highest C intensity, whereas maize showed the lowest in terms both of land use and grain production. Using the total cultivated croplands and the total year of grain produced, a yearly total C emission from cultivation and production could be estimated approximately as 150–172 Tg CO_2-eq of rice, 73–73 Tg CO_2-eq of wheat, and 58–73 Tg CO_2-eq of maize in 2010, with the rice production being the biggest carbon emitter in the sector of crop production.

Overall, N fertilizer contributed to the total CF by 46 %, 80 %, and 75 %, respectively, for rice, wheat, and maize production and a big portion of energy cost for irrigation and methane emission for rice (Fig. 2). Meanwhile, mechanical operation made up 8 %, 15 %, and 14 %, respectively, for rice, wheat, and maize production. However, a marginal proportion (2–6 %) was occupied by the inputs with pesticides, phosphorus fertilizer, and labor operations for all three crops.

In our analysis, there were relatively small changes in the proportion of individual inputs with the different crops, except for irrigation. For maize crops particularly, input of plastic films contributed 3 % to the total CF. The mean N use rates in the survey data was (mean ± SE) 269.1 ± 9.6 kg N ha^{-1},

Table 4 Carbon footprint (CF) of crop production of rice, wheat, and maize estimated using survey data from 123 household farms over China (mean ± standard error)

Crop	Farms surveyed	Farm size (ha)	Mean yield (t ha^{-1})	CF in cropland (kg CO_2-eq ha^{-1})	CF in production (kg CO_2-eq t^{-1})
Rice	17	0.1–12	7.6	5,795.8 ± 117.6 [a]	769.0 ± 20.2 [a]
Wheat	48	0.1–12	4.8	3,000.4 ± 185.8 [b]	645.6 ± 32.6 [b]
Maize	58	0.1–2	7.0	2,240.7 ± 131.9 [c]	326.9 ± 18.3 [c]

Different letters indicate significant differences between crops at $p < 0.05$

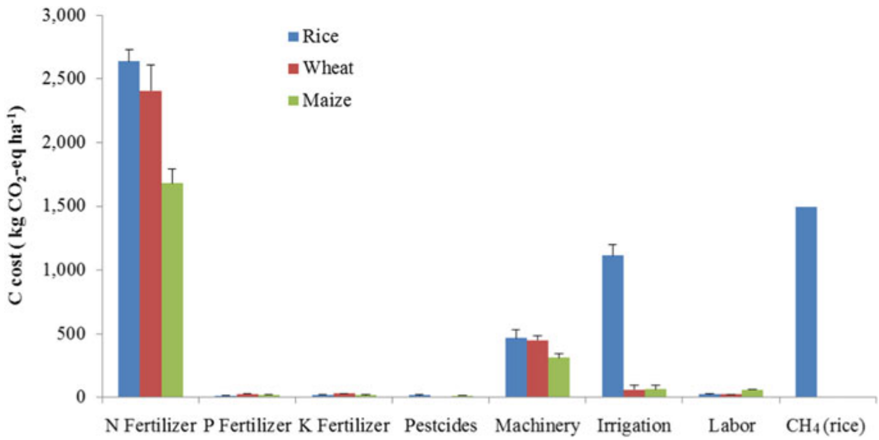

Fig. 2 Carbon cost of individual inputs for rice, wheat, and maize crop production. Data from questionnaire farm survey conducted during 2010–2011

217.5 ± 18.1 kg N ha^{-1}, and 152.0 ± 9.9 kg N ha^{-1}, respectively, for rice, wheat, and maize. CF was shown to be very significantly linearly correlated to N fertilizer use rate for both wheat and maize, but less significantly for rice due to the significant contribution by methane and irrigation-induced emission. However, yield was not observed in a linear positive response to N fertilization in these household-managed farms, reflecting a problem of N in excess. According to the survey data, a high yield (5,000–9,000 kg ha^{-1}) was achieved with N fertilizer use at rates of 200–300 kg N ha^{-1}, although >300 N ha^{-1} inputs did not statistically increase yield when a local conventional yield was approximately 6,000 kg ha^{-1} (Fig. 3). This had been a common problem in China's crop production with the household management system and thus is also a problem for luxury emissions from agriculture.

The total annual N fertilizer-induced direct emissions from the staple crop production could be estimated as 92, 60, and 45 Tg CO_2-eq, giving an overall value of approximately 200 Tg CO_2-eq for the major staple crop production. This value was much less than the estimate by Gao et al. (2011) of 313 Gg N_2O-N in 2007 and by Liu and Zhang (2011) of 403 Tg CO_2-eq for the overall cropland. Therefore, the carbon footprint of China's crop production is largely N-dependent, so reducing N overuse could be a key measure to lower the CF of major crop production in China.

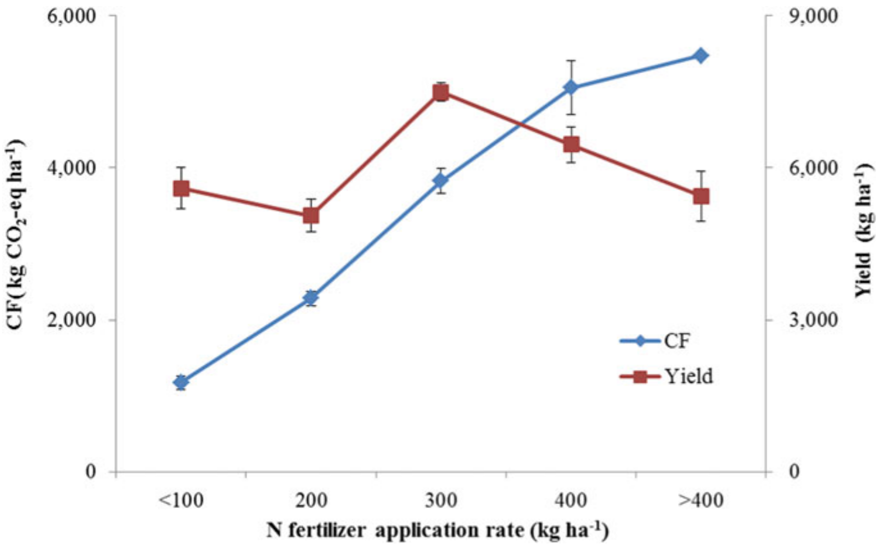

Fig. 3 Variation of carbon cost and yield with nitrogen (N) fertilization of all the farms surveyed. CF, carbon footprint.

5 Carbon Footprint of Vegetable Production

5.1 General Feature of the Carbon Footprint of Vegetable Crops

For assessing the carbon footprint of vegetable crop production in China, a provincial-wide farm survey by questionnaire were done across Jiangsu, China in 2010 following a similar procedure as that for staple crop production. Farmers were individually visited and input data were recorded for individual vegetable crops, including Chinese cabbage (*Brassica chinensis* L.), tomato (*Solanum lycopersicum*), cucumber (*Cucumis sativus* Linn), water spinach (*Ipomoea aquatica*), and amaranth (*Amaranthus spinosus* L.). A similar dataset was thus established for accounting use. The yield of the biomass or harvested fruit was much higher for vegetable production than for crops in terms of unit of land used, and inputs were much higher here (Chen et al. 2011; Yan et al. 2013). The estimated carbon cost on average for vegetables ranged from 3,880.4 ± 3,063.5 to 6,032.4 ± 366.3 kg CO_2-eq ha^{-1}. However, there was no significant difference in CF between the five types of the vegetable crops in terms of per hectare (Table 5).

As shown in Table 5, the carbon intensity showed a more divergent pattern. Chinese cabbage (496.6 ± 274.7 kg CO_2-eq t^{-1}) had the highest carbon intensity, whereas water spinach (51.1 ± 41.5 kg CO_2-eq t^{-1}) had the lowest. However, there was no significant difference in carbon intensity among amaranth

Table 5 Carbon (C) footprint on average (mean plus standard deviation) for the vegetable crops surveyed

Vegetable	C cost (kg CO_2-eq ha^{-1})	C intensity (kg CO_2-eq t^{-1})
Tomato	6,032.4 ± 366.3 a	168.5 ± 96.7 bc
Chinese cabbage	4,719.0 ± 2,135.8 a	496.6 ± 274.7 a
Cucumber	4,667.3 ± 2,356.2 a	159.0 ± 136.9 bc
Water spinach	3,880.4 ± 3,063.5 a	51.1 ± 41.5 c
Amaranth	5,668.7 ± 3,174.2 a	385.9 ± 249.9 ab

Different letters indicate significant difference between crops at $P < 0.05$

(385.9 ± 249.9 CO_2-eq t^{-1}), tomato (168.5 ± 96.7 CO_2-eq t^{-1}), and cucumber (159.0 ± 136.9 kg CO_2-eq t^{-1}). Clearly, for the carbon intensity of the harvest, vegetable production seemed to have a lower carbon footprint than crop production.

5.2 Proportion of Different Inputs to Total Carbon Cost

Statistics of the input data demonstrated a somewhat different figure of the contribution of different sources to the total CF for the vegetable crops compared to the grain crops as in the above sections. On average of the all vegetables surveyed, 76 % of the total C cost was allocated to nitrogen fertilizer use of inorganic and organic forms (Fig. 4). This is particularly due to high direct emissions of N_2O from N fertilizer, which was estimated using the default factor of 1 % (IPCC 2006). This figure was similar to the estimate of 76 % for food crop production in the UK (Hillier et al. 2009). However, the proportion of 76 % estimated here seems much higher than our previous estimate of 57 % for the overall CF of bulk crop production (Cheng et al. 2011). Among the others, the ground preparation, crop protection, P and K fertilizers, and machinery operations contributed less than 5 % on average to the total carbon footprint (Fig. 4), although this value varied for

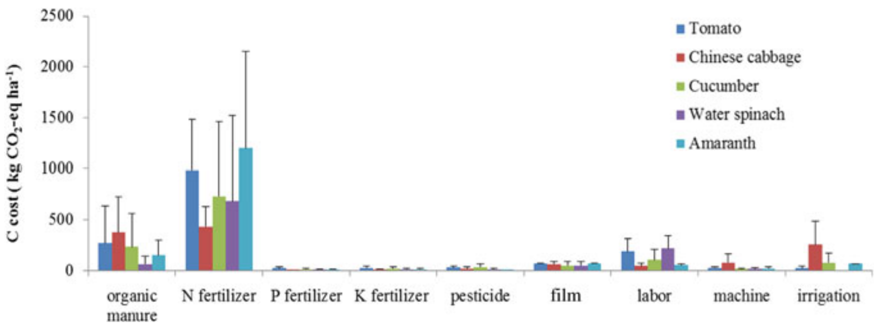

Fig. 4 Carbon cost of individual inputs for vegetable crop production in Jiangsu, China

different vegetable crops. This finding may suggest that mitigation of greenhouse gases emissions from vegetable production may be focused on reducing N fertilizer, although N fertilization in much excess had been already in debt (Zhang et al. 2010). While N fertilizer occupied a dominant proportion, farm operations with labor inputs made a bigger contribution for tomato and water spinach and irrigation made a bigger contribution for Chinese cabbage to their total C cost. In particular, crop management or maintenance may influence not only the total C cost but also the proportion of the different inputs to the overall C cost.

6 From Carbon Footprint to Carbon Management: Future of China's Crop Production

China is experiencing an economic transition from high carbon cost to low carbon for the commitment to reduce GHG emissions per unit of GDP by 2025 on the baseline of 2005 (CCSNARCC 2011), which is a global challenge. Increasing crop productivity will be still a general demand in agriculture development. However, this transition will be further challenged in a trilemma of sustaining high productivity, reducing carbon emissions, and adapting to climate change (Smith et al. 2013). China's ambition to keep stable the production of 0.4 billion ton of grain may have unforeseen difficulty, especially under the vast deterioration of soil fertility due to pollution and soil degradation under intensified cultivation, and especially under the impact of climate change.

The study of CF demonstrated a decreasing trend of intensity with crop productivity, which was not further increased when inputs (carbon costs hereby) were intensified at a background yield of 9 tons after 2003. Rice, a high yield grain crop and of key importance for grain production in China, was already very high in carbon intensity due to irrigation, methane, and N_2O emission. Water stress due to the increasing drought frequency in Northeast and South China (Pan et al. 2011a; Lv et al. 2011) could limit the rice productivity. However, maize was shown to have the lowest carbon intensity, which could be suitable to produce increasing areas of mainland China due to the climate change (Yang et al. 2011). With the help of new verities, crop management practices, and conservation tillage, maize cultivation could reach a high grain yield of over 10–12 ton per hectare. Therefore, to purse a safe and high productivity of China's crops, improving the cropping regionalization and extending maize to potentially suitable lands offers an option to sustain high grain production while stabilizing GHG emissions.

All the different case studies in this chapter revealed that N was the biggest GHG emission contributor in China's agriculture (Table 6). The negative impacts of N fertilizer overuse have been very well addressed. Particularly for GHG mitigation in agriculture, optimum management of N fertilization is urgently required to avoid luxury emissions.

Table 6 A comparison of approximate carbon footprint (CF) of China's crop production (2007–2010)

Sector	Total land used (Mha)	Carbon intensity in land use (tCO_2-eq ha^{-1})	Carbon intensity in harvested product ($kgCO_2$-eq t^{-1})	N fertilizer proportion to total CF (%)
Overall crop production	160	2.9	403	55
Staple crop production	110	2.2–5.6	327–769	46–80
Vegetable crop production	20	3.9–6.0	51–497	59

This study also indicated that changes in food consumption could help to establish a lower carbon intensity of crop that is production in China. First, if all rice were replaced by maize, then a total of almost 90 Tg CO_2-eq could be avoided without tradeoffs. In addition, consumption of water spinach instead of Chinese cabbage, a vegetable crop not commonly high in nutrition quality but requires a large amount of water from irrigation, would give a reduction in carbon intensity by more than 90 %. Thus, improving diet structure would offer a key option to reduce the carbon footprint of crop production in the future.

In addition, the carbon intensity of crop production could also vary greatly with farm management conditions. Crop yield was lower but carbon intensity was much higher in fragmented farms than in scaled-up farms (Yan et al. 2013). Scaling-up household farms will be another way to sustain high crop productivity with the benefits of reductions in carbon emissions.

Finally, are there any technical measures to sustain crop productivity but reduce GHG emissions in the field? Our studies on biochar soil amendment and biochar fertilizer have indicated a positive answer. Biochar soil amendment could help to increase crop yield in rice paddies by 0–5 % but in dry croplands by 5–25 %, while reducing GHGs emissions by 25–45 % under biochar amendment of 20–40 t ha^{-1} (Zhang et al. 2010, 2012a, b; Joseph et al. 2013). Fortunately, the positive effects by biochar could be sustained for a number of years (Zhang et al. 2012a, c). Because biochar from pyrolysis of crop residue is incentivized by the state to avoid in field burning, production and application of biochar is under development in China (Pan et al. 2011b). This new technology and product input to croplands could be a "new green from black" revolution (Lehmann et al. 2006); thus, it is a priority measure to cut the high carbon footprint of China's crop production.

A number of research opportunities have emerged for carbon management in agriculture. Among these could be the variation of carbon footprint with different cropping and farming systems, with climate conditions and the threshold of N luxury emission for a certain crop. Also, the characterization of carbon intensity in terms of vegetable nutrition value is critical for the assessment of vegetable crops.

It is anticipated that carbon footprinting and carbon management will be further supported in China to better address the carbon cost and improve carbon use for sustainable agriculture and quality of life.

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Carbon Footprint of the Solid Waste Sector in Greater Bangalore, India

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Abstract Every day, Bangalore generates approximately 3,000–4,000 tonnes of waste. A major fraction (72 %) of total waste is organic or wet waste, which degrades in the natural environment. This study is focused on the estimation of the carbon footprint of household waste generated in Bangalore city. The results from two theoretical estimation methods, mass balance approach and default methodology, are compared with the measurement results derived from experimental values. Experiments revealed an emission of 0.013 g CH_4/kg of organic fraction of municipal solid waste and 0.165 g CO_2/kg, which is much lower compared to the Intergovernmental Panel on Climate Change method (0.036 kg CH_4/kg of waste) or theoretical approaches (0.355 kg CH_4/kg, 0.991 kg CO_2/kg of waste). From the elemental composition and general theoretical chemical equation of aerobic and anaerobic degradation of waste amounts, total methane and carbon dioxide were estimated to be 670,950 and 1,870 tonnes per day (tpd) by the mass balance approach, which are considerably higher than the 87.32 tpd of methane emission determined using the default methodology. These values are still higher than the

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experimental estimated values of methane and carbon dioxide. The total carbon footprint of municipal solid waste generated from the city is 361 kg/day of CO_2 equivalent in the environment.

KeyWords Municipal solid waste · Carbon footprint · GHG emissions · Waste treatment · Bangalore

1 Carbon Footprint of Solid Waste

Carbon footprint (CF) refers to the direct or indirect emissions of carbon dioxide (CO_2) and other greenhouse gases (GHGs) expressed in terms of carbon dioxide equivalents (Wiedmann and Minx 2007). This constitutes a vital environmental indicator to understand and quantify the main emission sources and is an effective tool for energy and environmental management. GHGs get into the atmosphere either due to natural sources or anthropogenic activities. The contribution from natural sources is minimal and is neutralised due to the natural environmental processes, but a large quantity is generated from anthropogenic sources, which is accumulating in the atmosphere. Intergovernmental Panel on Climate Change (IPCC) lists 17 GHGs with different global warming potentials in a 100-year time horizon (IPCC 1996). The United Nations Framework Convention on Climate Change (UNFCCC) considers only carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O), hydrofluorocarbons (HFCs), perfluorocarbons (PFCs), and sulfur hexafluoride (SF_6) in accounting national GHG inventories (UNFCCC 1997).

Carbon footprint assessment for a region h elps to determine the impact of human activities on the environment and global climate. The major sectors and activities included in the inventory for estimating the carbon footprint are listed in Table 1. This chapter focuses on the quantification of the carbon footprint in the domestic solid waste sector. Mismanagement of municipal solid waste is a vital source of anthropogenic GHGs, such as methane (CH_4), biogenic carbon dioxide (CO_2), and nonmethane volatile organic compounds (NMVOCs) (Ramachandra 2009). Among these, methane is considered to be a potent GHG, having a global warming potential (GWP) that is 25 times greater than that of carbon dioxide. The concentration of atmospheric methane is annually increasing at 1–2 %, which necessitates the quantification of the carbon footprint in the waste sector for planning appropriate mitigation measures.

A major fraction (72–79 %) of solid waste generated in Indian households is organic (Jha et al. 2008; Thitame et al. 2009; Ramachandra 2009, 2012). The quantity and composition of emission mainly depends on the quantity of organic waste and method of solid waste disposal. Indiscriminate disposal of waste without treatment (segregation of organic fraction and generating either energy or compost) produces GHGs, thus contributing to the carbon footprint. Methane is

Table 1 Carbon footprint sector description

Sector	Activities included
Energy	Emissions of all greenhouse gases resulting from stationary and mobile energy activities, including fuel combustion and fugitive fuel emissions
Industrial processes	By-product or fugitive emissions of greenhouse gases from industrial processes not directly related to energy activities, such as fossil fuel combustion
Solvent and other product use	Emissions, of primarily nonmethane volatile organic compounds, resulting from the use of solvents and N_2O from product uses
Agriculture	Anthropogenic emissions from agricultural activities, except fuel combustion, which is addressed under Energy
Land-use change and forestry	Emissions and removals of CO_2, CH_4 and N_2O from forest management, other land-use activities, and land-use change
Waste	Emissions from waste management activities

Source IPCC/UNEP/OECD/IEA 1997

produced during the anaerobic degradation or breakdown of organic waste or carbon dioxide during aerobic degradation or burning of waste.

2 Solid Waste

Solid wastes are any non-liquid wastes that arise from human and animal activities that are discarded as useless or unwanted. These are the organic and inorganic waste materials such as product packaging, grass clippings, furniture, clothing, bottles, kitchen refuse, paper, appliances, paint cans, batteries, etc. produced in a society, which do not generally carry any value to the first user (Ramachandra 2009). Municipal solid waste (MSW) is composed of wastes generated from residences, markets, hotels and restaurants, commercial premises, slums, street sweeping and parks. Bangalore residences contribute 55 % to the total waste, which is the highest among all sources (Chanakya and Sharatchandra 2005; Ramachandra 2011; Ramachandra et al. 2012). The waste generated from hotels and eateries form about 20 %, fruit and vegetable markets contribute about 15 %, trade and commerce about 6 %, and street sweeping and parks about 3 % (Table 2). The slum areas contribute only 1 % of total, because the slum population in Bangalore is low compared to other metropolitan cities, such as Mumbai, Delhi, or Kolkata. The slum populations in Mumbai, Delhi, Kolkata, Chennai, and Bangalore are 49, 19, 33, 18 and 8 % (Census of India 2001), respectively. MSW generation for Kolkata, Chennai, Delhi, and Mumbai are 2653, 3036, 5922 and 5320 tpd, respectively. The contributions of slums to the total MSW generated in these cities are approximately 875.49, 546.48, 1125.18, and 2606.8 tpd for Kolkata, Chennai, Delhi, and Mumbai, respectively.

Table 2 Municipal solid waste generation in Bangalore

Source	Quantity (t/d)	Composition (% by weight)
Domestic	780	55
Markets	210	15
Hotels and eatery	290	20
Trade and commercial	85	6
Slums	20	1
Street sweepings and parks	40	3

Source Chanakya and Sharatchandra 2005; Lakshmikantha 2006; Ramachandra 2009

Waste management strategies of these waste sources may vary with quantity and composition of waste.

3 Quantity and Composition of Solid Waste

3.1 Current Rate of Waste Generation

Greater Bangalore is the administrative, cultural, commercial, industrial and knowledge capital of the state of Karnataka, India with an area of 741 sq. km. It lies between the latitude 12°39′00–13°13′00″ N and longitude 77°22′00″–77°52′00″ E. Bangalore city administrative jurisdiction was redefined in the year 2006 by merging the existing area of Bangalore city spatial limits with eight neighboring Urban Local Bodies (ULBs) and 111 Villages of Bangalore Urban District. Bangalore has grown spatially more than 10 times since 1949 ($\sim$69–716 sq.km) and is the fifth largest metropolis in India, currently with a population of about 9 million. Bangalore city population has increased enormously from 65,37,124 (in 2001) to 95,88,910 (in 2011), accounting for 46.68 % growth in a decade. Population density has increased from 10,732 (in 2001) to 13,392 (in 2011) persons per sq. km. The per capita GDP of Bangalore is about $2066, which is considerably low with limited expansion to balance both environmental and economic needs (Ramachandra et al. 2012a).

The spatial increase in city area and increase in population have increased the total amount of MSW from 650 (in 1988) to 1450 tpd (in 2000). The current estimates indicate that about 3000–4000 tonnes of MSW are produced each day in the city—the daily collection is estimated at 3600 tpd (Ramachandra et al. 2012). The increase in the per capita generation from 0.16 (1988) to 0.58 kg/d/person (2009) is due to the changes in consumption patterns. Changes in composition are noticed recently with the increasing quantity of waste.

Table 3 Physical composition of municipal solid waste in Bangalore

Waste type	Composition (% by weight)						
	Domestic	Markets	Hotels and eatery	Trade and commercial	Slums	Street sweepings & parks	All sources
Fermentable	72	90	76	16	30	90	72
Paper, cardboard	8	3	17	56	2	2	12
Cloth, rubber, PVC, leather	1		0.3	4	0.5	0	1
Glass	2		0.2	0.7	8	0	1
Plastics	7	7	2	17	2	3	6
Metals	0.3		0.3	0.4	0.2	0	0.2
Dust and sweeping	8		4	8	57	5	6

Source TIDE 2000

3.2 Composition of Solid Waste

Usually, municipal solid waste can be broadly categorised into organic or inorganic waste using major components of solid waste composition. Organic waste is also known as wet waste, whereas inorganic waste is also known as dry waste. Inorganic waste includes both recyclable and nonrecyclable materials, whereas organic waste includes all the waste components that can degrade in natural environments, such as leftover food, vegetables, and fruit peels. Municipal solid waste is a heterogeneous mixture of solid materials that does not have any use to society. Food waste, plastic, paper, rubber, leather, glass and textiles are the common MSW components. Sourcewise solid waste composition is shown in Table 3. Waste composition changes with the source of generation, but most of the sources generated a major fraction (>70 %) of organic waste. It is evident that Indian waste has more organic than inorganic constituents, except slums and commercial places.

Solid wastes generated in Indian cities are mainly composed of organic fractions and are biodegradable. The waste generally includes degradable (paper, textiles, food waste, straw and yard waste), partially degradable (wood, disposable napkins, and sludge) and nondegradable materials (leather, plastics, rubbers, metals, glass, ash from fuel burning such as coal, briquettes or woods, dust and electronic waste) (Jha et al. 2008; Visvanathan 2004).

Most (72–79 %) municipal solid waste is organic (Ramachandra 2009; Ramachandra 2011; Sathishkumar et al. 2001; Ramachandra et al. 2012; Sharholy et al. 2007; GOI 1995). The contribution of inorganic components is gradually changing and is likely to show further changes in the future. The biodegradable fraction is quite high, arising from the practice of using fresh vegetables in India. The plastic and metal contents are lower than the paper content and do not exceed 1 %, except in metropolitan cities. This is mainly because large-scale recycling of these constituents takes place in most medium and large cities. The composition of

MSW at generation sources and collection points determined on a wet weight basis consists mainly of a large organic fraction (70–75 %), ash and fine earth (5–8 %), paper (10–14 %) and plastic, glass and metals (each less than 3–5 %) (Ramachandra et al. 2012). Paper waste generally falls in the range of 3–7 %, when the waste reaches the disposal site (Asnani 1998). The organic fraction is high (>80 %) in many pockets within many South Indian cities, such as Chikkamagalur, and is largely represented by vegetable, fruit, packing, and garden waste (Chanakya et al. 2009). The physical composition of MSW in Bangalore is as follows: paper 8 %, textiles 5 %, plastic 6 %, metals 3 %, glass 6 %, ash fine earth and others 27 %, and compostable matter 45 % (CPCB 1999; Sharholy et al. 2008). In Bangalore, organic waste mainly consists of vegetable and fruit wastes; its percentage contribution ranges between 65 and 90 % (Rajabapaiah 1988; TIDE 2000; Ramachandra 2009; Chanakya et al. 2009). Many studies have been conducted in academic institutions to determine the waste composition. As shown in Table 4, the organic fraction ranges from 72.5 (Sathiskumar et al. 2001), 79.6 (Ramachandra et al. 2012), and 88 % (Rajabapaiah 1995).

3.3 Factors/Variables of Changes in Quantity and Composition

Waste quantity and composition depends upon various factors such as country, topography of the area, different seasons, food habits, commercial status and activities of the city (Jha et al. 2008; Thitame et al. 2009; Ramachandra 2009), and standard of living. The relative percentage of organic waste in MSW is generally increasing with the decreasing socio-economic status; rural households as well as low- and mid-income urban households generate more organic waste than urban households.

4 Solid Waste Management

Municipal solid waste management (MSWM) is associated with the control of waste generation—its storage, collection, transfer and transport, processing, and disposal in a manner that is in accordance with the best principles of public health, economics, engineering, conservation, aesthetics, public attitude, and other environmental considerations. Presently, most of the metropolitan cities and MSWM systems include all the elements of waste management. However, in the majority of smaller cities and towns, the MSWM system comprises only four activities: storage, collection, transportation, and disposal (Sharholy et al. 2008; Ramachandra 2009; Ramachandra 2011)

Table 4 Studies on waste composition (%) of Bangalore

Study area waste type	Beukering (1994) Bangalore	TIDE (2000) Bangalore	Sathish Kumar et al. (2001) IISc	CPCB (2004-05) Bangalore	BBMP (2008) Bangalore	Ramachandra et al. (2012)	Ramachandra et al. (2012) IISc
Glass	0.24	1.43	–		3	1.4	0.5
Plastic	0.48	6.23	9	22.43	12	6.2	12.7
Paper/cardboard	3.12	11.6	18		13	11	6.2
Metal	0.05	0.23	–		1	1	0.2
Organic	57.04	72	73	51.84	59	72	79.59
Other	38.08	6.5	–			6.5	0.28

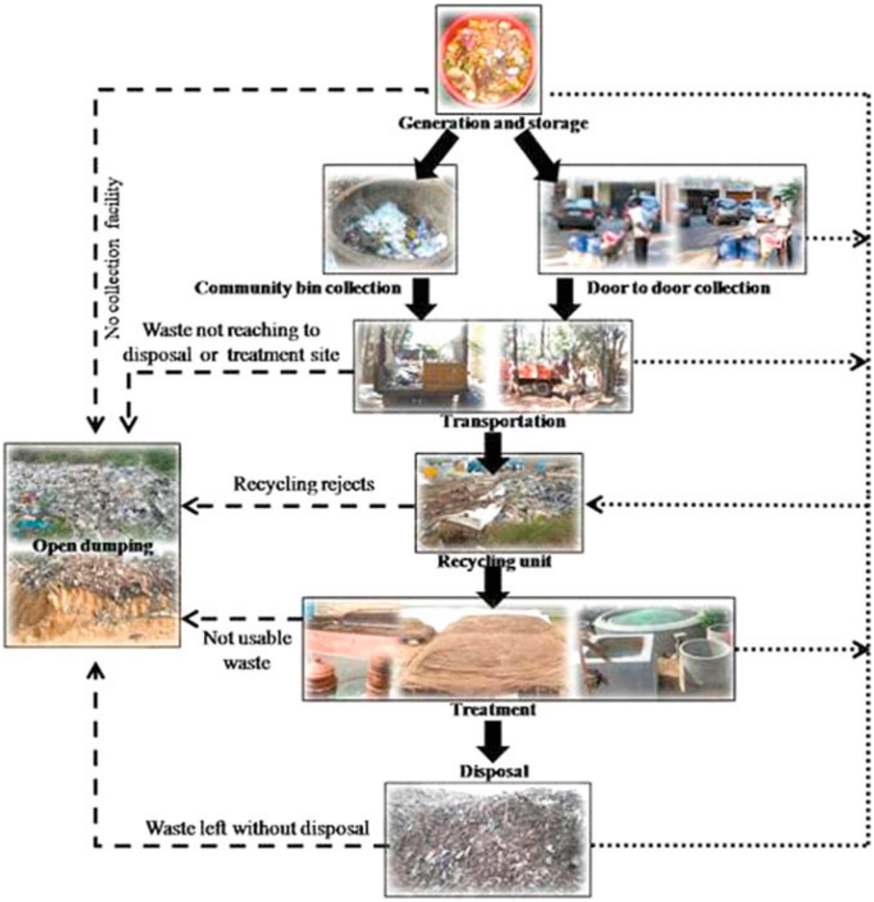


Fig. 1 Functional elements of solid waste management

A solid waste management (SWM) system refers to a combination of various functional elements (Fig. 1) associated with the management of solid wastes; details are provided in Table 5. The system, when put in place, facilitates the collection and disposal of solid wastes in the community at minimal costs, while preserving public health and ensuring little or minimal adverse impact on the environment. The functional elements that constitute the system are shown in Fig. 1.

Table 5 Wardwise auditing of functional components of municipal solid waste management

Function	Technique		Shivajinagar	Malleswaram	Koramangal	IISc	HMT	Airport road	Chickpet	Average (%)
Storage	Community bin	Percentage of covered bins	30	–	–	33	–	–	84	49*
Collection	Community bin	Percentage of area covered in commercial area	40	0	0	–	–	–	30	17.5**
	Door to door	Percentage of area covered in residential areas	100	100	100	60	100	100	100	94.29
		Percentage of waste segregated	0	0	20	5	0	0	0	3.57
Transfer		Transfer station	A	A	A	A	A	A	A	A
Transport	Truck	Truck with mesh (%)	100	100	100	75	100	100	100	96.43
		Truck with mesh and polythene cover (%)	75	40	75	0	0	0	100	41.43
Process	Percentage of waste recycled	Informal	18	18	18	18	18	18	18	18
		Formal								
	Percentage of waste composted				22					3.14
	Percentage of waste for anaerobic digestion									
	Percentage of waste incinerated									
Disposal	Sanitary landfill									
	Dump yard		85	85		85	85		85	60.71
	Quarry				63			85		21.14

A = Absent

* Only the areas having bins are taken into consideration

** Only the commercial areas have been taken into consideration, i.e. Shivajinagar, Malleswaram, Kormangala, and Chickpet

Source Ramachandra and Bachamanda 2007

4.1 Generation and Storage

Waste generation quantity and composition depends on the lifestyle of households. Segregation at the generation or source level is to divide the waste into different categories, such as organic waste and inorganic waste. In the conventional method, partial segregation of newspaper, milk pouches, etc. happens at the house level, but the rest gets mixed up during waste storage. In places with the active participation of nongovernmental organizations and the community, segregation at source/house level is in place (Pattnaik and Reddy 2009; Ramachandra 2009). However, it is still at a very preliminary stage. Informal recycling plays an important role in waste segregation and waste management (Sudhir et al. 1996). Storage of waste means the temporary containment of waste, at the household or community levels. At household level, old plastic buckets, plastic bins, and metal bins are used for storing waste; at the community level, wastes are stored in masonry bins, cylindrical concrete bins, and metallic and plastic containers (Joseph 2002; Kumar et al. 2009). Stored waste is then collected and transported to the transfer station or processing site at regular intervals.

4.2 Collection

Waste collection is the removal of waste from houses and all commercial places to a collection site, from where it will go for further treatment or disposal. Its efficiency is a function of two major factors: workforce and transport capacity (Gupta et al. 1998). Community bin and door-to-door collection are prevalent in India (Kumar et al. 2009; Kumar and Goel 2009; Pattnaik and Reddy 2009; Ramachandra 2009). Indian cities are shifting from community bin collection to door-to-door collection to improve the existing waste management system. Most of the cities are either fully or partially covered with door-to-door collection (Kumar et al. 2009). The door-to-door collection facility is only limited to 60–61 % of the present collection system in Kolkata (Chattopadhyay et al. 2009; Hazra and Goel 2009), whereas in Bangalore it has reached up to 94–100 % of total waste collected from residential areas (Ramachandra and Bachamanda 2007; Kumar et al. 2009).

4.3 Transportation

Transportation of the stored waste to final processing sites or disposal sites at regular intervals is essential to avoid bin overflow and littering on roads. Usually, light and covered vehicles with carrying capacities of around 5 tonnes per trip are used for transportation of waste (Rajabapaiah 1988; Ramachandra 2009). In small towns, bullock carts, tractor-trailers, tricycles, etc. are mainly used for transportation (Sharholy et al. 2008).

4.4 Treatment (Aerobic and Anaerobic)

Treatment is required to alter the physical and chemical characteristics of waste for energy and resource recovery and recycling. The important processing techniques include compaction, thermal volume reduction, manual separation of waste components, incineration, anaerobic digestion, and composting. The organic fraction of the waste is processed either through composting (aerobic treatment) or biomethanation (anaerobic treatment). Composting through aerobic treatment produces stable product-compost, which is used as manure or as soil conditioner. In metropolitan cities, compost plants are underutilized for various reasons, including unsegregated waste and production of poor quality of compost, thus resulting in reduced demand from end users (Kumar et al. 2009; Chattopadhyay et al. 2009; Ramachandra 2011). Vermi-composting is also practiced at few places. Biomethanation through microbial action under anaerobic conditions produces methane-rich biogas. It is feasible when waste contains high moisture and high organic content (Chanakya et al. 2007; Kumar and Goel 2009). Recyclable waste that can be transformed into new products such as plastic, rubber, glass, metal, and others are collected separately and auctioned by recycling industries (Agarwal et al. 2005).

4.5 Disposal

Waste disposal is the final stage of waste management. As in urban areas, uncontrolled and unscientific disposal of all the categories of waste, including organic waste, has lead to environmental problems, such as contamination of land, water, and air environment, in larger towns or cities, the availability of land for waste disposal is very limited (Gupta et al. 1998; Mor et al. 2006; Ramachandra 2009). In many places, a major fraction of urban wastes are directly disposed in low-lying areas or in hilly areas at city outskirts (Lakshmikantha 2006; Talyan et al. 2008; Chattopadhyay et al. 2009). In this backdrop, MSW rule 2000, Government of India (GOI) was introduced to regulate all components of waste management. Landfilling or disposal is restricted to nonbiodegradable, inert waste and other wastes that are not suitable either for recycling or for biological processing as per MSW rule 2000.

5 Mismanagement of Waste and Its Implications

Municipal solid waste management is initiated by urban local bodies to protect the environment and the society from adverse impacts of increasing waste quantity. However, mismanagement of municipal solid waste, either due to lack of adequate workforce or disregard of a vital functional element in SWM, creates serious health and environmental implications. Mismanagement in handling solid wastes include (i) mixing of organic and inorganic wastes, (ii) open solid waste dumping, (iii) unscientific/indiscriminate waste disposal practices, and (iv) burning of solid waste.

5.1 Mixing of Organic and Inorganic Waste

Segregation of organic and inorganic waste at the source level is the most critical stage regarding waste management and recycling processes. If the waste is not separated properly, it reduces the recyclability of waste and increases the volume of waste for transport and at treatment and disposal locations. Biodegradation of waste under anaerobic conditions, it releases methane; under aerobic conditions, it releases CO_2 to the environment. Apart from these, leachate from waste dumps contaminates the soil and groundwater resources.

5.2 Open Solid Waste Dumping

Bangalore generates around 3000–4000 tonnes of solid waste daily, and a major constituent is organic (72 %). The quantum of wastes generated is far greater than the capacity of the three permitted waste treatment and disposal sites at Mavallipura, Mandur, and Singehalli. Because these locations are quite far-off, many of the trucks dump at unauthorized locations such as roadsides, lake beds, vacant plots, etc. to reduce their transportation costs. The disposal of waste at private or public places in and around cities—(i.e., on locations other than the designated urban solid wastes processing sites) is termed unauthorised dumping.

The waste is collected by outsourced agencies that dispose waste in vacant places within the city as well as at outskirts/peri-urban areas. Most dumps inside the city are small and waste is dumped at the respective locations for 1–2 days. However, dumps at outskirts are large (>25 hectares) and waste is being dumped there because longer time and organic fractions are degraded with leachates getting into soil. Figure 2 provides the spatial locations of open dumps.

Wastes are dumped in public and private open lands, agricultural land, road sides, and at hilly areas with no provision for controlling gaseous emission or leachate. Most of the organic wastes are reduced by animals, and a fraction undergoes microbial degradation. Both aerobic and anaerobic degradation takes place in open disposal sites. Methane recovery attempts were reported from two open landfills in Nagpur, India (Bhide et al. 1990).

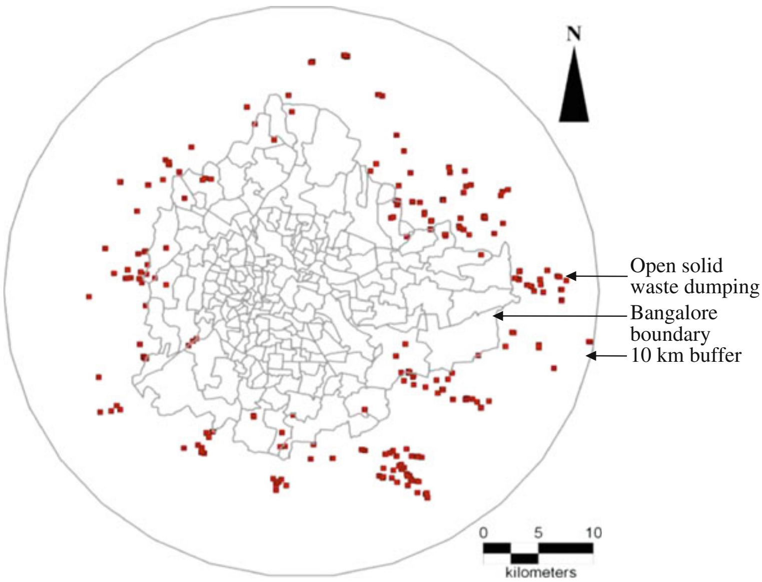


Fig. 2 Open dumpsites located around in and around Bangalore

5.3 Unscientific Waste Disposal Practice in Landfill

Sanitary landfills with options for collecting leachate and gas emissions are essential for safe waste disposal. Waste is compacted and daily covered with a layer of soil. During the final closing, the landfill site is effectively capped with a thick soil layer. All these factors lead to anaerobic conditions inside the landfill site and hence continuously produce methane gas. Waste composition and the age of landfill site are two main factors that influence the extent of methane production. However, sites earmarked for disposal of solid waste in most Indian cities do not adhere to the environment norms. Landfills with unscientific waste disposal practices are evident from the direct dumping of mixed wastes. These sites are not properly covered with soil with no appropriate collection system for leachate and gaseous emission. Also, these sites receive more waste than the capacity, disturbing the whole system at disposal site.

5.4 Burning of Solid Waste

The burning of municipal solid waste at waste disposal sites or at open dump sites is common to reduce the volume of waste or to segregate the metal items from mixed accumulated waste. Usually, this type of incomplete combustion can reduce

40–60 % of waste volume. Incomplete combustion of waste during open burning contributes to GHG emissions and other air pollutants. The carbon in MSW has two distinct origins. One is harvested biomass sources, such as yard trimming and vegetable/fruit residues, whereas the second is non-biomass sources such as plastic and synthetic rubber derivatives (EPA 2006). MSW burning results in emission of CO_2 and N_2O. The carbon stored in harvested biomass sources also is lost in the atmosphere, which can be recycled back to the system.

So the waste management practices of concern for methane emissions are open dumping, which is generally practiced in developing regions, and sanitary landfilling, which is generally practiced in developed countries and urban areas of developing countries (IPCC 1996). Aerobic waste treatment or composting of organic waste emits an almost negligible quantity of methane, as waste gets converted and increases the soil organic matter. Anaerobic waste treatment or biomethanation of waste generates significant quantities of methane, but this methane is collected and used as a source of energy.

6 Method for Determining CF of Solid Waste

Total CF of the waste sector was 23,233 CO_2 equivalent, which included municipal solid waste disposal (53 %), domestic waste water, industrial waste water, and human sewage (Garg et al. 2006). In 1990, the CF of the waste sector was 14,133 CO_2 equivalents (ALGAS 1998) which increased to 28,637 CO_2 equivalent in 2000 (Sharma et al. 2006). These estimations were based on IPCC emission factors in the absence of local emission factors. In this chapter, different techniques adopted for the estimation of CF are compared along with field experiments to assess the validity of theoretical estimates. Experimental methods have been developed to estimate the emissions from organic waste and composting processes at local levels.

There are a number of methods to estimate the carbon footprint in terms of methane emissions from solid waste disposal methods. These methods are broadly classified into (i) theoretical estimations and (ii) experimental methods. Accuracy of theoretical estimations depends on the availability of data. Usually, the available theoretical estimation methods are mass balance approach, default methodology (using degradable organic carbon content), theoretical first-order kinetics, and the triangular method (IPCC 1996; Kumar et al. 2004; Garg et al. 2006; Sharma et al. 2006).

6.1 Theoretical Estimation Methods

6.1.1 Mass Balance Approach

Mass balance approach is the simplest level of emission estimation. Its use is generally discouraged because it gives a high estimation of emissions. This method does not include any factors and does not distinguish between various types of disposal sites. In this approach, theoretical emissions are calculated using stochiometric equations as per Tchobanoglous et al. (1993). Equations for aerobic and anaerobic degradations considering complete degradation of waste are given by Eqs. 1 and 2.

$$C_{2.98}H_{0.462}O_{1.02}N_{0.099} + 2.659\ O_2 \rightarrow 2.98\ CO_2 + 0.0825\ H_2O + 0.099\ NH_3 \tag{1}$$

$$C_{2.98}H_{0.462}O_{1.02}N_{0.099} + 2.4287\ H_2O \rightarrow 1.1978\ CH_4 + 1.2143\ CO_2 + 0.099NH_3 \tag{2}$$

6.1.2 Default Methodology

This approach of emission estimation considers the degradable organic carbon content of MSW (Eq. 3) and does not include changes in the conversion of carbon to methane emissions with time (Bingemer and Crutzen 1987; IPCC 1996).

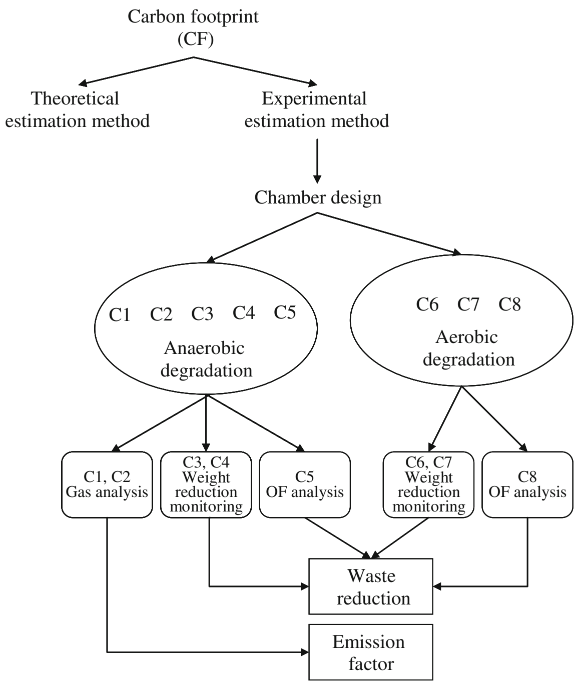
$$\begin{aligned} CH_4(\mathrm{Gg/yr}) = {} & MSW_T \times MSW_F \times MCF \times DOC \times DOC_F \times F \times (16/12 - R) \\ & \times (1 - OX) \end{aligned} \tag{3}$$

where MSW_T = Total municipal solid waste generated, MSW_F = Fraction of MSW disposed of at the disposal sites (0.6), MCF = Methane correction factor (0.6), DOC = Degradable organic carbon (0.18), DOC_F = Fraction of DOC dissimilated (0.77), F = Fraction of methane in LFG (0.5), R = Recovery of LFG (0), and OX = Oxidation factor (0).

6.1.3 First-Order Kinetics

First-order kinetics consider the availability of time series waste disposal data and other detailed informations for a disposal site to compute the methane emission as the degradable organic components degrade slowly and methane is emitted over a long period.

Fig. 3 Flow chart of experimental setup



6.1.4 Triangular Method

The triangular method considers time-dependent release of gaseous emission based on first-order decay. Total gaseous yield is computed for the organic fraction considering rapidly biodegradable waste and slowly biodegradable waste. This requires extensive waste characterization and quantification at the waste disposal site (Kumar et al. 2004).

6.2 Experimental Estimation Method

The elemental composition of the organic fraction of the MSW is presented as $C_5H_{8.5}O_4N_{0.2}$ (Bizukojc and Ledakowicz 2003). The degradable organic carbon decomposes by microorganisms under aerobic or anaerobic conditions. In aerobic conditions, carbon gets converted into carbon dioxide; in anaerobic conditions, it gets converted into carbon dioxide and methane, which are GHGs.

Methane and carbon dioxide are measured and quantified at the laboratory scale through waste degradation under aerobic and anaerobic conditions. This process involves design of a chamber, monitoring, and quantification of emissions, as shown in Fig. 3.

Fig. 4 Gas collection chamber

6.2.1 Design of Chamber

The gas analysis chamber was made up of glass with diameter of 7 and 20 cm height. The shape of the chamber was conical with a gas collection apparatus. The gas collection apparatus was round in shape, with seven openings for gas collection (Fig. 4). These openings were closed with rubber leads. In anaerobic conditions, collection openings were closed; for aerobic conditions, openings were left open. There were eight chambers; C1, C2, C3, C4, C5, C6, C7 and C8. Chambers C1, C2, C3, C4 and C5 were maintained under anaerobic condition, whereas chambers C6, C7 and C8 were maintained in aerobic conditions without any external aeration. Chamber C1 and C2 were used for gas analysis, whereas C3, C4, C6 and C7 were used to monitor weight reduction with time and C8 and C5 were used for organic fraction (OF) analysis. Anaerobic chambers were covered with paper and kept away from sunlight to give optimum conditions of anaerobic degradation. In aerobic chambers, the lead was open to provide conditions similar to open dumpsites.

6.2.2 Monitoring Duration for Waste Degradation

The experiment was started with 200 g of sample kept in each of the eight chambers (Fig. 5). Degrading samples were subjected to organic fraction analysis, gas analysis and weight reduction study. For organic fraction analysis, waste samples were collected on days 0, 3, 6, 9, 12 and 17 (Chanakya et al. 2007).

Fig. 5 Waste sample used for experiment

6.2.3 Organic Fraction Analysis

In organic fraction analysis, parameters such as temperature, moisture content, total solids, volatile solids, carbon, hydrogen, and nitrogen were measured.

pH: 3 g of dried sample was added to 15 ml of distilled water and shook for 24 h to determine pH with a pH meter.

Moisture content: To obtain dry mass, the solid waste material was weighed (W_1) and then dried (W_2) in an oven at 105 °C until the mass of the dried material became constant. The moisture content is computed by Eq. 4.

$$\%\ \text{moisture content} = ((W_1 - W_2)/W_1 \times 100) \tag{4}$$

Total solids: 5 g of sample was weighed (W_2) in an empty crucible (W_1) and dried in an oven maintained at 105 °C for 24 h (W_3). Percent of total solids (TS) was calculated using Eq. 5.

$$\%\ \text{TS} = ((W_3 - W_1)/(W_2 - W_1) \times 100) \tag{5}$$

Volatile solids (VS): This was measured in accordance with APHA (1975). Approximately 2–3 g of an oven-dried sample was weighed (B) in an empty crucible (A) and heated to 550 °C for 1 h in the muffle furnace (C). Percent VS was calculated using Eq. 6.

$$\%\ \text{VS} = ((B - C)/(B - A) \times 100) \tag{6}$$

CHN analysis: Carbon, hydrogen, and nitrogen (CHN) was analyzed with the help of CHN analyser (LECO elemental analyser). After finding the CHN of the sample, the elemental composition of waste under the study was determined with the help of the equation given by Tchobanoglous et al. (1993). Elemental composition was used for determining the theoretical estimation (mass balance approach) of gaseous emission during waste degradation in the aerobic and anaerobic processes.

Fig. 6 Gas collection from compost plant

6.2.4 Gas Analysis: *Gaseous Composition*

Gas analysis was carried out using a gas chromatograph (ProGC, Mayura Analytical Pvt.Ltd., India) equipped with flame ionisation and thermal conductivity detectors. All hydrocarbons are separated by a Heysep-R column having a mesh size of 80/100 and dimensions 2 m × 1/8in. Detection is done by FID detector. Analysis of gas was done on every 14th day. After 15 days, with the decrease of gas production with time, samples were collected at the gap of 7 days. Gas was collected in 10-ml syringe and then subjected to gas chromatography.

Quantification of gaseous emission: Quantification of gaseous emission was done using the water displacement method. In the water displacement method, samples were connected through a burette filled with potassium dichromate solution. As gas enters and passes through the burette, it displaces filed potassium dichromate solution in the burette. The volume of solution displaced is equal to the volume of gas produced from waste samples. After the 30th day of the experiment, the volume of gas was measured in chambers.

6.2.5 Weight Reduction Study

Chamber samples were weighed on a balance on days 0, 3, 6, 9, 12, 17 and 30 to check subsequent reductions in the total waste quantity kept on day 0.

Analysis of composting process: Composting is an aerobic process of organic waste treatment. During the composting process, waste gets converted into compost or manure. Particularly, the carbon content of waste gets converted to humus or emitted into the environment as carbon dioxide. To assess emissions from composting, the experiment was carried out in the compost unit that is successfully implemented and managed at Vellore city. City municipal waste in Vellore has been treated since 2009 through the aerobic option (compost). There were 17 working compost pits for residential waste. Gas samples were collected (Fig. 6)

Table 6 Change in total solids of waste (%)

Sl. No.	Days	Aerobic	Anaerobic
1	3	93.45	93.07
2	6	92.74	92.99
3	9	87.67	93.88
4	12	91.92	86.80
5	17	93.25	86.66

from compost processes happening in Vellore city. Compost chambers filled at different time intervals were selected to see the difference in emissions and other properties as these pits were filled in different months and are at different stages of degradation. These experiments were done during 2010 (Jan–May) and also verified later (June 2013).

7 Results and Comparison of Different Findings for CF of Solid Waste

7.1 Change in properties of waste

The initial pH of 4.06 in waste samples suggests the initiation of degradation. Fresh organic fractions of MSW will have pH in range of 6–7 (Bizukojc and Ledakowicz 2003). The initial temperature was 28 °C and during the degradation; the temperature in the anaerobic chamber was 24.5–26 °C in all samples. Anaerobic digestion occurs under two temperature regimes: mesophilic (between 20 and 45 °C, usually 35 °C) and thermophilic (between 50 and 75 °C, usually 55° C). The sample temperature was found to range from 25 to 29.5 °C (mostly above 28 °C) on all sampling days under aerobic conditions; this is because the heat formed was easily getting dissipated into the atmosphere because the sample quantity was small and kept open. The total solid content of waste was decreasing with time (Table 6). The percentage of carbon in the preliminary sample was about 42 %, which was highest among the three elements, with nitrogen 1 % and hydrogen 6 %. Results of CHN analysis on different sampling days showed that carbon varied from 46 to 49 %, hydrogen 6–7 %, and nitrogen 1–2 % in aerobic conditions. At the laboratory scale, total wet waste reduction in aerobic degradation was found to be 25 times faster than anaerobic degradation.

Table 7 Emission from anaerobic degradation of waste (in 30 days)

Chamber	Waste quantity (kg)	CH4 (ml/kg)	CO2 (ml/kg)	Total (ml/kg)
C1	1	22.527	134.379	156.906
C2	1	12.875	33.275	46.150
Avg	1	17.701	83.827	101.528

7.2 Emission Factor (Based on the Experiment)

Total gas produced from 30 days of continuous degradation of 1 kg of waste under anaerobic condition is 101.528 ml, which consisted of 17 % of methane and 83 % of carbon dioxide (Table 7). The emission factors for gaseous emissions of methane and carbon dioxide were 0.013 and 0.165 gm/kg, respectively. (Because gaseous volume was in millilitre, it was converted into gaseous mass using gas volume and the respective density at standard temperature and pressure: Density = Mass/Volume, with the density of methane and carbon dioxide as 0.716 and 1.965 g/l, respectively.) Total daily emissions from the organic fraction of solid waste degradation in Bangalore are 31.06 and 403.52 kg of methane and carbon dioxide, respectively.

7.3 Comparison of Emissions Computed Using Different Methods of Estimation

7.3.1 Determination of Emissions by Mass Balance Approach

The elemental composition of the sample was found to be comparable with the literature. According to Reinhart (2004), the elemental composition of sample was 1.30, 0.07, and 0.17 of carbon, nitrogen, and hydrogen, respectively. From stoichiometric calculations it can be seen that 1.1978 mol of methane and 1.2143 mol of carbon dioxide is emitted from 1 mol of analysed sample ($C_{2.98}H_{0.462}O_{1.02}N_{0.099}$) under anaerobic conditions. Thus, 0.355 kg of CH_4 and 0.991 kg of CO_2 are emitted from 1 kg of waste sample. Similarly, under aerobic conditions, carbon dioxide emissions were found to be about 2.431 kg/kg of waste. Therefore, the total emissions from Bangalore solid waste using mass balance approach in anaerobic conditions is 869.75 tpd of methane and 5955.95 tpd of carbon dioxide.

7.3.2 Default Methodology

In the estimation of methane emission potential by the IPCC default method, the amount of solid waste that is available for anaerobic degradation and methane generation was assumed as 100 %. The result shows that there was about 87.32 tpd

Table 8 Comparison of emission factors

Gaseous emission	Mass balance approach (kg/kg)	Default methodology (kg/kg)	Experimental estimation method (gm/kg)
CH_4	0.355	0.036	0.013
CO_2	0.991		0.165

Note Default method of IPCC accounts for only CH_4

of methane potential for the city, which is less than estimated emission from the mass balance approach. If we compare methane emission from each kilogram of organic waste, then through this method estimated methane emission will be 0.036 kg/kg of waste.

7.3.3 Experimental Estimation Method

In contrast to the theoretical estimation method, in which methane emission potential is calculated based on the amount of waste being disposed every day, the theoretical methods overestimate emission values, necessitating quantification of methane emission at the laboratory scale. Total methane emission from Bangalore solid waste using the experimental emission factor is 31.06 kg/day, whereas carbon dioxide is 403.52 kg/day. Results are much lower than theoretically estimated values because these methods assume that all potential methane is released as it comes in contact with the environment. Also, quantified values are lower than the emission estimated at landfills in Chennai by Jha et al. (2008). Landfills with mature waste enhance the methane emissions from fresh waste under anaerobic conditions.

Table 8 shows a comparison of emission factors computed by different methodologies. It is clear from comparison of emissions computed from different methods of estimation that the theoretical estimation method overestimates the emission from waste in comparison to laboratory-estimated or field-estimated values. Still further, more accurate estimation is possible using an accurate quantity of waste for different treatment methods, as well as by knowing emission from open dump and unscientific disposal at landfill site at more controlled conditions.

7.4 Ward-Wise CF of Solid Waste Using Experimental Values

Every day Bangalore generates around 3,500 tonnes of municipal solid waste. Of that, 55 % is from household waste (Table 2), with per capita generation of 0.35 kg/day of domestic waste.

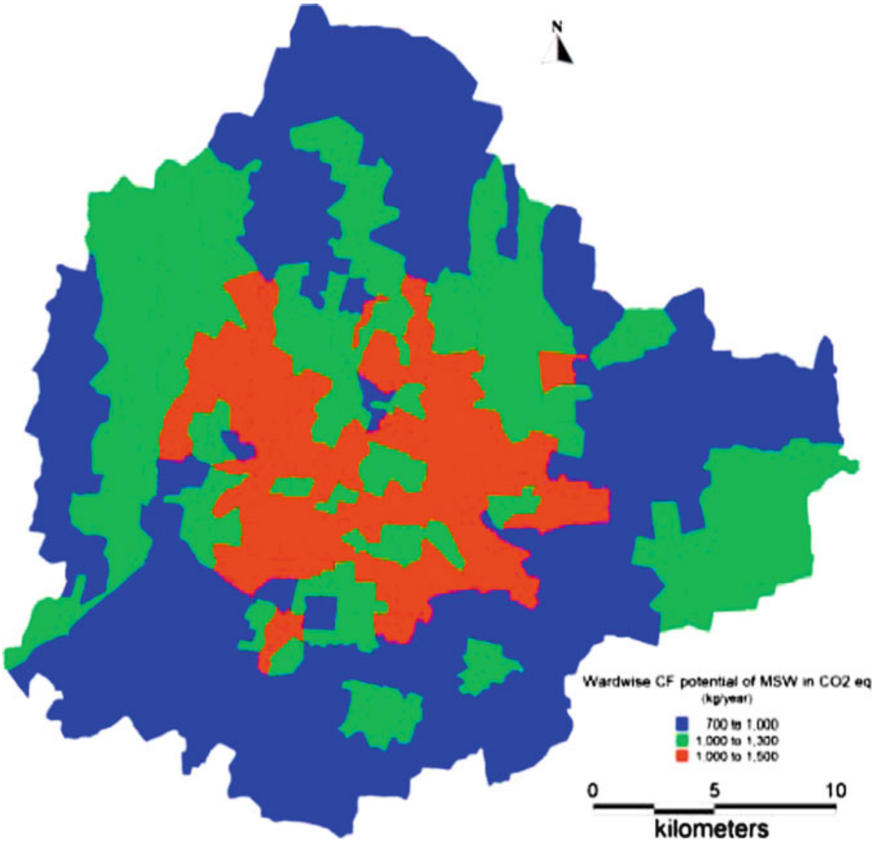


Fig. 7 Carbon footprint of municipal solid waste

7.5 Carbon Footprint of Municipal Solid Waste

Estimated methane and carbon dioxide emission from representative waste samples were used for computing annual emissions from solid waste. Total ward-wise organic waste generated is 2044 tpd. Methane and carbon dioxide emissions are 19.13 and 242.83 kg/day. Methane emission values were multiplied by 21 to compute the carbon footprint of waste. Annual carbon footprint of municipal solid waste is 644.61 kg/day of CO_2 equivalent, assuming that total waste generated in the city is reaching to waste disposal sites without any treatment. City wards where the population is less dense have less emissions than densely populated ward (Fig. 7). Most of the core city wards are densely populated, so their carbon footprint potential is more than other wards of the city. Figure 7 illustrates the pattern of open dumping, which is prevalent at outskirts.

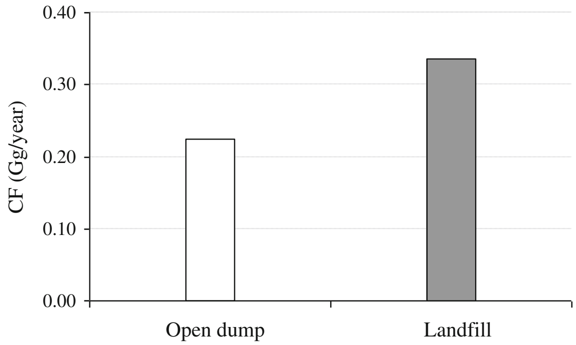


Fig. 8 Annual carbon footprint considering open dumpsites

Table 9 Emissions from air samples collected near a compost plant

Plant (residence time of waste)	Sample vol. (cm^3)	CH_4 (%)	CO_2 (%)	CH_4 ml/cm^3	CO_2 ml/cm^3
60 days	5722.65	0.02	0.23	0.0000034	0.0000401
30 days	1130.4	0.018	0.144	0.0000159	0.0001273
10 days	1844.7	0.038	0.4	0.0000205	0.0002168

7.6 Carbon Footprint of Open Dumps: Unauthorized Locations

An earlier study reported the existence of 60 open dumpsites around the city (Lakshmikantha 2006). Field work conducted in 2010 showed a considerable increase in open dumps across the city and outskirts (Chanakya et al. 2011). Quantification of dumps shows that there are 270 dumps (Fig. 2) distributed in all four zones of the city. Waste quantity is determined through the visual estimates at each location (supplemented by photographs of each site). A total of about 83,557 tonnes wastes are scattered in and around Bangalore city. The average life of an open dump is 2–3 years. Based on field investigations during 2011 and 2012, about 40 % of daily waste is being dumped at unauthorized locations (Fig. 2). Figure 8 illustrates the carbon footprint of unauthorized dumps and authorized dumps (with minimal or no treatment) based on the quantity and emission factor.

7.6.1 Characteristics of Composting and Its Emission Potential

The percentage of carbon dioxide and methane emitted from a compost plant is far less than that emitted from chambers under anaerobic condition. Among the three compost samples, samples from plant (with residence time of 10 days) show higher gas emission (as it is the new pit among the three) of 0.0000205 ml methane and 0.0002168 ml carbon dioxide (Table 9). The sample from compost of

60 days has relatively lower gas emission, due to the presence of mature waste. BMP analysis further corroborates this result, as there is no gas production from compost samples.

8 Mitigation Measures

Carbon footprint quantifications reveal that GHG emissions are mainly due to mismanagement (absence of recovery and treatment of organic fractions) of municipal solid waste. Hence, mitigation of GHG emissions (CH_4 and CO_2) from municipal solid waste involves (i) reduction of the quantity of waste, (ii) segregation of organic fractions of wastes, and (iii) treatment of waste to recover energy (biomethanation) or resources (compost —aerobic treatment). Reduction of waste generation is possible through reduced waste generation, segregation at source level, reuse, and recovery of waste. Composting and anaerobic digestion are treatment options for organic waste (which constitute 70–75 % of the total), whereas recycling is used for inorganic materials (15–18 %). Wastes that cannot be treated or recycled are ultimately disposed at disposal sites or landfills. Segregation at the source with treatment at local levels (ward levels) plays a prominent role in minimizing organic fractions getting into disposal sites.

An integrated solid waste management (ISWM) approach would aid in the mitigation of GHGs emitted into the atmosphere by open dumping or by unscientific disposal of waste in landfill site. ISWM includes source segregation, regular collection of waste, treatment of organic fractions at local levels, and disposal of only inert refuse at landfill sites. The organic fraction is the major contributor of GHGs in MSW and has to be treated for energy and resource recovery. Reduction of GHGs through biogas generation is the most common clean development mechanism approach for emission mitigation in India. Residential associations in select wards of Bangalore have successfully adopted ISWM through source segregation at household levels, recovery of recyclables, and composting of organic fractions, etc. These ventures have successfully demonstrated that sensible waste management, which includes a reduction in the carbon footprint at local levels, could be economically viable for entrepreneurs due to the market potential for composts and recyclables (bottles, plastic, paper, metal, etc.).

9 Conclusions

The direct or indirect emissions of carbon dioxide, methane, and other GHGs, expressed in terms of carbon dioxide equivalents, indicate the CF of a region, which constitutes a vital environmental indicator to mitigate global warming and consequent changes in the climate. This study indicated that the theoretical estimation of emissions from solid waste is much higher than the experimentally determined value. Total emissions from ward-wise waste of the city are 19.13 and 242.83 kg/day of methane and carbon dioxide, respectively. Reduction of waste generation is possible through reduced waste generation, segregation at source level, reuse, and recovery of waste. Composting and anaerobic digestion are treatment options for organic waste (which constitute 70–75 % of the total), whereas recycling is used for inorganic materials (15–18 %). Wastes that cannot be treated or recycled are ultimately disposed at disposal sites or landfills. Segregation at the source with treatment at local levels (ward levels) plays a prominent role in minimizing organic fractions getting into disposal site.

GHG emission factors vary with methodology. Experiments conducted reveal an emission of 0.013 gm of CH_4/kg of organic fraction of municipal solid waste and 0.165 gm CO_2/kg, which is much lower compared to the IPCC method (0.036 kg CH_4/kg of waste) or theoretical approaches (0.355 kg CH_4/kg, 0.991 kg CO_2/kg of waste). The current work provides emission factors at local levels, which could help in the accurate quantifications of emissions. Nevertheless, a comparative analysis of commonly used methods (such as IPCC) with the experimental value highlights the overestimation of GHGs from the waste sector with the techniques adopted earlier.

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