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Subramanian Senthilkannan Muthu
Editor

Assessment of Carbon Footprint in Different Industrial Sectors, Volume 2

Springer

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ISSN 2193-4614 ISSN 2193-4622 (electronic)
ISBN 978-981-4585-74-3 ISBN 978-981-4585-75-0 (eBook)
DOI 10.1007/978-981-4585-75-0
Springer Singapore Heidelberg New York Dordrecht London

Library of Congress Control Number: 2013949186

Printed on acid-free paper

Springer is part of Springer Science+Business Media (www.springer.com)

Preface

This is the continuation of Volume 1 of *Assessment of Carbon Footprint in Different Industrial Sectors*. During the compilation of the different chapters for Volume 1, I framed the contents of this volume. Volume 2 deals with the carbon footprint assessment of some other industrial sectors that were not covered in the first volume. It is needless to repeat and stress the importance of the assessment of carbon footprint in various industrial sectors, as it was covered sufficiently in Volume 1.

As discussed in the Preface of Volume 1, every industry has its unique assessment and modelling techniques, allocation procedures, mitigation methods, and labelling strategies for its carbon emissions; this second volume has also been framed with distinct chapters earmarked for each important industrial sector. However, even two volumes are unable to cover all industrial sectors in terms of their carbon footprint assessment. However, the most important and prominent sectors were covered to the maximum possible extent. I will continue my efforts in terms of collecting the information on carbon footprint assessment in other industrial sectors, and I look forward to possibly disseminating that information in the future in a new volume.

Similarly to Volume 1, each chapter in this volume discusses the assessment methodologies of carbon footprint followed in a particular industry, challenges in calculating the carbon footprint, case studies of various products in that particular industry, mitigation measures to be followed to trim down the carbon footprint, and recommendations for further research. This second volume includes the carbon footprint assessment of the sugar industry, fishing industry, wine manufacturing sector, wood industry, energy sector, recycling sector, and food sector (with a case study of beef in Flanders). Also included is a sectorwise case study in India that deals with various industrial sectors.

The food industry is one of the important sources of anthropogenic greenhouse gas emissions. This volume has two chapters that discuss the food industry, either directly or indirectly. "A Review of Energy Use and Greenhouse Gas Emissions From Worldwide Hake Fishing" deals with the energy use and greenhouse gas emissions of the fishing industry in worldwide hake fishing. This chapter revolves around the carbon footprint quantification of hake, which is the most widely used fishing product in Spain. "A Life Cycle Assessment Application: The Carbon Footprint of Beef in Flander (Belgium)" provides a detailed carbon footprint assessment, including the location of major hot-spots responsible for creating more greenhouse gas emissions, in the mitigation measures of beef in Flanders.

"Carbon Footprint and Energy Estimation of Sugar Industry: An Indian Case Study" deals with the quantification of energy and the carbon footprint of the sugar industry. This chapter provides a case study of the Indian sugar industry in three plants to enumerate the energy needs and carbon footprint quantification details.

The energy sector is one of the important sectors contribute either to the raise or it is one of the viable sectors to reduce the global greenhouse gas emissions. Energy plays a major role in carbon footprint in both of these ways. This volume has dedicated chapters on the carbon footprint of the energy sector in different forms. "Carbon Footprint as a Single Indicator in Energy Systems: The Case of Biofuels and CO_2 Capture Technologies" discusses the carbon footprint estimation of the energy sector, with the case studies on biofuels and CO_2 capture in power plants. Apart from the quantification of carbon footprint and lifecycle inventory collection, this chapter also discusses the suitability of the carbon footprint as a single indicator for this sector. "Reduction in Carbon Footprint of Coal Fired Thermal Power Plants by Promoting CFL and LED Lights in Households, Offices and Commercial Centres" deals with the reduction in carbon footprint in coal-fired thermal power plants by promoting compact fluorescent lamps (CFLs) and light-emitting diodes (LED) as replacements for fluorescent tubes (FTs). This study highlights energy conservation along with the financial repercussions, greenhouse gas emission reductions, and reduction of other air pollutants reduction in a coal-fired thermal power plant by using CFL and LED lights instead of FTs.

"Assessment of Carbon Footprinting in the Wood Industry" is discusses the carbon footprint assessment in the wood sector. With a sound methodology, this chapter provides the details of quantification and comparisons of carbon footprint values of 14 types of wood products. Importantly, in this chapter, the use of timber products for the purpose of carbon storage and the effect of allocation methods on carbon footprinting are also discussed to a greater extent.

"Carbon Footprint of Recycled Products: A Case Study of Recycled Wood Waste in Singapore" focuses on the carbon footprint of the recycling sector. With a case study of recycled wood waste in Singapore, this chapter revolves around the details of recycling and its implications. This chapter also details the different recycling modelling approaches.

"Sector-wise Assessment of Carbon Footprint Across Major Cities in India" is a bit different from other chapters, as it deals the carbon footprint estimation of many different sectors in India. The study discussed in this chapter spans the main cities of India—Delhi, Greater Mumbai, Kolkata, Chennai, Greater Bangalore, Hyderabad and Ahmedabad—and estimates the carbon footprint of various sectors (domestic, transportation, industrial, agricultural, waste and livestock). Additionally, this chapter presents a discussion on intercity variations in the light of carbon footprint results.

"The Use of Carbon Footprint in the Wine Sector: Methodological Assumptions" discusses carbon footprint estimation in the wine industry. It also discusses the major hot-spots of carbon footprinting in this sector. This chapter also deals with the future prospectus and challenges of using carbon footprint in the wine sector.

I would like to thank all the contributors to this book for their tremendous efforts toward the successful publication of this enriched content. I am sure that readers will benefit from this book, which provides the details of carbon footprint assessment for various industrial sectors in one place. This second volume will certainly become an important reference for researchers, students, industrialists, and sustainability professionals working in this field.

Contents

A Review of Energy Use and Greenhouse Gas Emissions from Worldwide Hake Fishing

Ian Vázquez-Rowe, Pedro Villanueva-Rey, Mª Teresa Moreira and Gumersindo Feijoo

Abstract Food production has been repeatedly highlighted as one of the most important sources of greenhouse gas (GHG) emissions worldwide. Within the food sector, there is a wide range of heterogeneous products that should be analyzed individually in order to understand the potential role of each one in global warming. In parallel, the fishing industry, which is essentially part of the food sector, has been shown to represent approximately 1.2 % of the world's GHG emissions. However, the impact of individual fishing species remains widely unexplored in terms of their contributions to climate change. Therefore, this chapter focuses on calculating the carbon footprint (CF) of the most widely consumed fishing product in Spain: hake. For this, an aggregation of six different fishing fleets, which account for a high percentage of the final hake landings by the entire Spanish fleet, were analyzed. Results are presented using several methodological assumptions, including the assessment method framework and allocation. In addition, the results are also presented individually per fishing fleet, fishing gear, and hake species. Finally, the individual CFs of each hake species are used to calculate the lump sum for hake landings in Spain. The discussion of the results focuses on highlighting the main inputs contributing to GHG emissions, as well as specific improvement actions to reduce the impacts of these vessels. Furthermore, the interrelation between CF and other environmental impacts, namely the impact on stock biomass, and the influence of methodological choices on the results presented, constitute two important topics for further analysis.

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S. S. Muthu (ed.), *Assessment of Carbon Footprint in Different Industrial Sectors, Volume 2*, EcoProduction, DOI: 10.1007/978-981-4585-75-0_1,

Keywords Carbon footprint · European hake · Fuel use intensity · *Merluccius* spp.

1 Introduction

Fish species from the Gadiformes order are widely commercialized for direct human consumption (DHC) in European and North American countries (Girard and Mariojouls 2008). For instance, the main source of seafood consumption per capita is cod (i.e. gadoids) in Portugal (55 % of total seafood consumption), haddock (i.e. gadoids) in Iceland, pollack in Germany (29 %), and a wide range of Gadiformes (e.g. cod, saithe, haddock, and hake) in Sweden (28 %) (Byrd-Bredbenner et al. 2000). Moreover, a survey conducted by Welch et al. (2002) in 10 European countries demonstrated that white fish such as Gadiformes represent from 40 to 60 % of total seafood consumption in these countries. Nevertheless, important differences were observed between Spain, with white fish representing over 60 % of seafood consumption, and Germany, in which their contribution was roughly 40 % (see Tables 1 and 2).

In Spain, hake species (i.e. Gadiformes from the Merlucciidae family) have become a strategic product in the food market and one of the main sources of marine protein in an average diet (Asche and Guillén 2012; Antelo et al. 2012). For instance, 3.9 kg per capita of hake species were consumed in Spain in 2009 (Martín-Cerdeño 2010), representing 14.1 % of total seafood consumption (Table 3). This white fish is very important in the Spanish market, but the environmental impacts linked to its fishing, processing, and consumption should not be ignored. For instance, European hake (*Merluccius merluccius*), which has been fished and consumed in Spain since the seventeenth century, when it became fashionable within the upper class in Madrid, has suffered important constraints in the Northern and Southern stocks due to overfishing since the 1990s (Guillén et al. 2004; ICES 2013). This situation has led the European Union (EU) to enforce strict recovery schemes through fishing moratoria, fleet reductions, and quota restrictions by limiting total allowable catches (TACs). In recent years, the abundance of these two stocks has improved considerably (Villasante 2009; ICES 2013).

In parallel, the increasing demand for seafood in Spain in the past 30 years, with a per capita consumption that has passed from 17.7 kg per capita in 1950 to approximately 40.0 kg per capita in 2010 (126 % increase), and a strong demographic expansion since the mid-1990s, has led the industry to search for new fisheries from which to obtain regular catches of Merlucciidae species (Guillén et al. 2004; Antelo et al. 2012). Consequently, besides the Spanish fishing fleets targeting European hake in European waters, Spanish companies and skippers have deployed numerous vessels overseas to ensure that domestic demand in Spain for hake species is met. Figures 1 and 2 illustrates the segmentation in different hake species of the total Merlucciidae captures by Spanish vessels in the period 1950–2011. In line with the economic expansion in the 1960s, Spanish vessels started to exploit a new fishery off the coast of Namibia: cape hake (*Merluccius*

Table 1 Seafood consumption per capita in the EU (2007). *Source* FAOSTAT (2013)

Country	Consumption per capita (kg/year)	Country	Consumption per capita (kg/year)
Portugal	54.82	The Netherlands	19.02
Spain	40.03	Estonia	16.39
Lithuania	37.55	Germany	14.80
France	34.79	Austria	13.36
Finland	31.71	Latvia	12.59
Malta	30.18	Czech Republic	10.41
Sweden	28.50	Poland	9.54
Luxembourg	27.78	Slovenia	9.38
Denmark	24.53	Slovakia	8.03
Belgium	24.48	Romania	5.26
Italy	24.40	Bulgaria	4.20
Cyprus	22.59	Hungary	N/Av
Ireland	21.35	EU (average)	22.03
Greece	21.09	Iceland	87.40
United Kingdom	20.35	Norway	51.43

N/Av not available

Table 2 Relative contribution of white fish in selected European countries as compared to total per capita seafood consumption. *Sources* FAOSTAT (2013), Welch et al. (2002)

Country	Total seafood consumption per capita (kg/year)	Consumption of white fish per capita (kg/year)	Relative contribution of white fish to total (%)
Norway	51.43	30.40	59.1
Spain	40.03	23.02	57.5
Sweden	28.50	11.20	39.3
Denmark	24.53	9.32	38.0
France	34.79	19.83	57.0
United Kingdom	20.35	9.48	46.6
Italy	24.40	11.52	47.2
Germany	14.80	6.05	40.9
The Netherlands	19.02	8.22	43.2
Greece	21.09	12.06	57.2

N/Av not available

capensis). In fact, this new species, as shown in Fig. 3, accounted for more than 60 % of the annual catch of hake species from the late 1960s until the nationalization of the Namibian fishery in the early 1990s (Armstrong et al. 2004). This trend allowed an increase of per capita consumption of hake in Spain, peaking with 5.9 kg per year in the mid-1980s, despite demographic growth and total annual landings of hake ranging from 150,000 to 250,000 tons per year.

A strong drop in total landings was observed in the early 1990s (see Figs. 1 and 3). Ever since, landings have ranged from 45,000 to 85,000 tons, with a highly reduced contribution of cape hake. More specifically, beyond 1996, the contribution of this

Table 3 Household consumption of seafood per capita detailed for the main fishing species in 2009 (adapted from Marín-Cerdeño 2010)

Types of fish	Consumption per capita (kg/year)
Hake species (*Merluccius* spp.)	3.9
Sardine species (*Sardina pilchardus*)	1.8
Tuna species	0.5
Sole species (*Solea* spp.)	1.1
Cod species	0.7
Mackerel species (Trachurus trachurus, Scomber scombrus)	0.4
Salmon	0.7
Lubina	0.4
Dorada	0.8
Rodaballo	0.1
Anglerfish species	0.5
Other fish species	4.1
Mollusks and crustaceans	8.3
Total	27.6[a]

[a] This value refers to household consumption exclusively, and not the entire per capita seafood consumption for the Spanish population

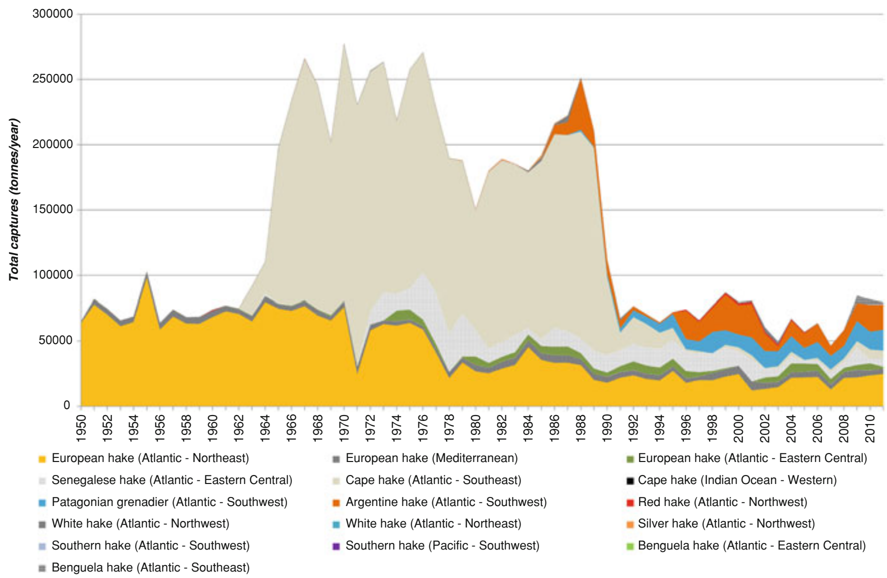

Fig. 1 Annual total Spanish hake species in the period 1950–2011 (Adapted from: FAOSTAT 2013)

particular species has been below 10 %. These circumstances, added to the reduction in quotas for European hake in the Northeast Atlantic and Eastern Central areas, which have been below 30,000 tons per year ever since the admission of Spain in the

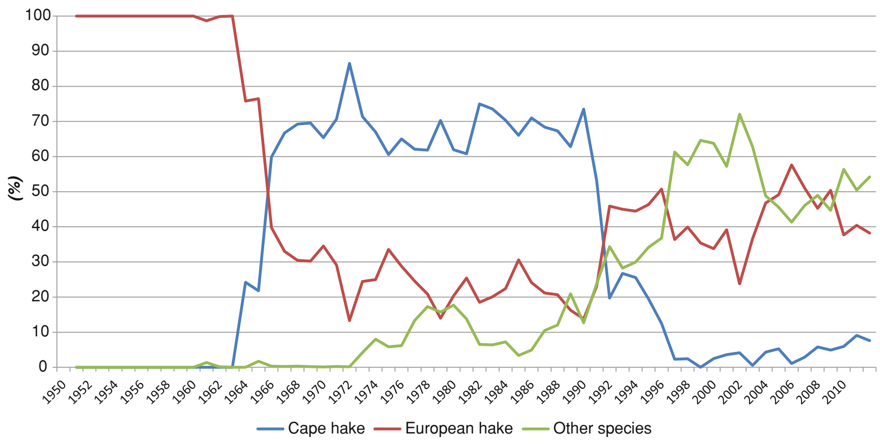

Fig. 2 Relative contribution of the main hake species in total Spanish vessel landings in the period 1950–2011 (Adapted from: FAOSTAT 2013)

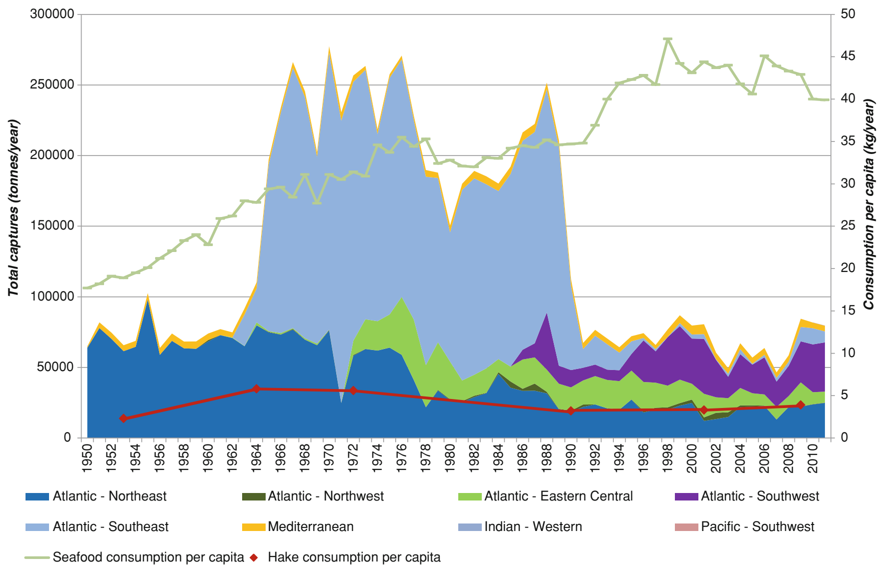

Fig. 3 Annual total Spanish hake species landings per fishing area in the period 1950–2011. Secondary axis shows per capita consumption of seafood (*green line*) and hake species (*red line*) in the same period (Adapted from: Piquero Zarauz and López 2005; Martín-Cerdeño 2010; FAOSTAT 2013)

EU, have made the annual intake of hake decrease to values between 3.0 kg per capita and 4.0 kg per capita since the 1990s (see Fig. 3). Nevertheless, it should be noted that given the globalization of world seafood trade and the incapability of Spanish fleets to meet hake demand, many vessels from other nations have started to land and export hake products to market in Spain (Asche and Guillén 2012). Moreover, as pointed out by previous publications, certain Spanish vessels have been flagged by other national fleets in order to dribble increasing quota restrictions for Spanish European hake landings (Guillén et al. 2004; Villasante 2009; Antelo et al. 2012). Therefore, Spanish fishing vessels currently fish for hake species in the Exclusive Economic Zones (EEZs) of Chile, Argentina, Mauritania, and Namibia, as well as in international waters (Guillén et al. 2004; Villasante 2009), as depicted in Fig. 3.

Consequently, the current supply chains for hake distribution and consumption have become highly complex and competitive, leading in some cases to bad practices, such as the findings recently published by García-Vázquez et al. (2012), who identified that important amounts of cape hake and silver hake commercialized in Spain were actually mislabeled. For instance, 85 % of cape hake sales in Spain between 2005 and 2010 were identified actually as being *Merluccius paradoxus*.

As previously mentioned, the strong pressures on hake stocks throughout the world due to fishing activities has led regional organizations (i.e. ICES; European Union, etc.) to enforce stricter fishing regimes through a more sophisticated fisheries management policy (García et al. 2011). For this, stock assessment by fisheries scientists has been of crucial importance to determine the maximum thresholds that should not be surpassed when exploiting a fishery (Punt and Smith 2001). Nevertheless, beyond the environmental impacts that have traditionally been monitored in fisheries, such as stock abundance, population dynamics, and fish mortality, new environmental concerns at a wider level and throughout the supply chain of products have been identified. For instance, the assessment of the greenhouse gas (GHG) emissions associated with the anthropogenic activities in fishing has been found to have a relevant contribution on a worldwide level (Tyedmers et al. 2005). More specifically, Tyedmers et al. (2005) estimated that GHG emissions linked to the production and combustion of fuel for fishing vessels' propulsion accounted for approximately 1.2 % of worldwide emissions. Furthermore, a more recent study published by the University of Santiago de Compostela in Spain (Iribarren et al. 2010a, b, 2011) estimated that 3.0 % of GHG emissions in Galicia, a fishing region in northwest Spain, were directly attributable to fishing operations at sea (Verdegaia 2010; Iribarren et al. 2011; Parker et al. 2014). However, in the latter study, the GHG emissions were calculated based on the PAS 2050:2011 standards from the British Standard Institution (BSI 2011), including all the GHG emissions linked to the lifecycle of the operations and activities that occur in the fishing stage of fishing activities (e.g. including, beyond fuel and other fossil fuels, the GHG emissions related to the production of nets, construction of the vessel, paints, or ice). This methodology is named carbon footprint (CF), which has developed as a single score indicator for monitoring climate change environmental impacts, from its parent method, Life Cycle Assessment (LCA; ISO 2006a; BSI 211).

CF is commonly used alone, rather than in a cluster of environmental dimensions (i.e. impact categories in life cycle thinking), due to the current importance that is conferred to GHG emissions and their pivotal role in climate change

(Weidema et al. 2008; Laurent et al. 2012). Hence, the use of CF has become popular in highly energy-intensive sectors, such as energy, mobility, or fishing, as well as livestock due to methane emissions from cattle (Druckman and Jackson 2009; Piecyk and McKinnon 2010; Iribarren et al. 2011; Vázquez-Rowe et al. 2013a; Ziegler et al. 2013). Nevertheless, it is important to bear in mind that in some situations, as has been proved in many cases, the analysis of CF alone can provide misleading interpretations in terms of actions to mitigate the environmental impacts (Laurent et al. 2012). This has been the case, for instance, in certain bioenergy systems, in which the increase in land use impacts, as well as toxicity, was substantially higher than the mitigations related to GHG emissions (Searchinger et al. 2008; Hertel et al. 2010; Vázquez-Rowe et al. 2013b, 2014a). Having said this, and as stated by Weidema et al. (2008) and demonstrated in fishing CF studies (Iribarren et al. 2010a, b, 2011; Vázquez-Rowe et al. 2013a), impacts on certain marine dimensions, such as seafloor, are usually highest for those fishing fleets/gears that show the highest fuel use intensity (FUI) and, therefore, highest GHG emissions. In addition, recent studies in the fishing sector suggest that fish stocks that are managed in a sustainable manner are capable of maintaining their GHG emissions at lower levels (Hornborg et al. 2012).

Gadiformes, including including species from the Merlucciidae family, are mainly demersal species. Therefore, they tend to dwell on the seafloor, feeding mainly from smaller fish organisms (Rogers et al. 1999). This feature makes them more difficult to fish than most pelagic fish, such as tuna, mackerel, or herring, which leads to more intensive and damaging gears potentially being used to capture these species. Consequently, available studies in the literature have situated the FUI intensity of fishing fleets targeting hake species between 469 L of fuel per ton of landed Patagonian grenadier (Vázquez-Rowe et al. 2013c) to roughly 2400 L/ton for European hake in the Northern Stock (Vázquez-Rowe et al. 2011a, b).

Given the strong relationship between FUI and the CF of fishing activities, as already discussed in numerous literature studies (Thrane 2004a; Vázquez-Rowe et al. 2012a, b; Avadí and Fréon 2013; Vázquez-Rowe et al. 2013a), the main aim of this chapter is to aggregate the inventories of a wide range of Spanish fishing fleets that target hake species in order to derive a first approximation to the total GHG emissions that are attributable to this important subsector in the Spanish fishing sector and, therefore, in the overall economy.

Section 2 delves into the main methods in the field of CF used to calculate the GHG emissions linked to the different hake products and fishing fleets assessed. Thereafter, Sect. 3 presents the main results of the study. The individual CF of the different products is presented using independent methodological approaches. Then, these results are used to provide a rough estimation of the GHG emissions related to the entire hake harvesting industry in Spain. Section 4 focuses on the interpretation of the main findings of the study, including the need to combine these results with other indicators to improve the environmental profile of seafood products. Finally, Sect. 5 summarizes the main conclusions of the case study.

2 Materials and Methods

2.1 Goal and Scope

The main objective of this study was to aggregate inventory data for the main fishing fleets worldwide that supply hake species for DHC in the Spanish market, in order to calculate their individual CF profile. Furthermore, the estimation of the total CF for hake fishing activities supplying the Spanish seafood market was calculated. The methodology to quantify the CF of the examined production systems was based on the ISO regulations for lifecycle approach. However, to analyze different frameworks, the results were reported using the baseline ISO framework, as well as the PAS 2050 guidelines (ISO 2006a; BSI 2011). In addition, the release of a new specification for seafood and other aquatic products was applied in the current study (BSI 2012).

Despite the important and complex interactions that occur in the Spanish hake market, this study focused on the fishing stage of the case studies, as explained in more detail in Sect. 2.2. Therefore, this study used a business-to-business (B2B) approach, also referred to as a cradle-to-gate perspective, which comprises all upstream emissions that occur until the point at which the products are delivered to a new company or organization (Iribarren et al. 2010a, b; BSI 2008, 2011). In this specific study, therefore, the GHG emissions and removals that have been included are those relating to the fishing stage of the inventoried fishing fleets. More specifically, as recommended in the PAS 2050-2:2012 supplementary requirements, all the activities—and therefore all the operational inputs and outputs that are potential sources of GHG emissions and removals—in this stage are considered, including preparation of fishing activities and transport to and from the fishing areas (BSI 2012).

The function of the system analyzed was the capture, transportation, and subsequent landing of hake species in ports worldwide, to eventually meet the Spanish domestic demand for hake. However, it should be noted, as discussed in Sect. 2.3, that not all the fleets that capture hake species rely on this group of marine organisms exclusively. However, in all the assessed fishing fleets, the capture and subsequent landing of hake species implied an important source of revenue for the vessels.

Therefore, in order to have a homogeneous functional unit (FU) to which the results can be referred to (ISO 2006a), 1 metric ton of landed hake in a given port[1] ready for delivery to wholesalers was assumed. The selection of the FU used a relevant unit of analysis that can orient stakeholders in the fishing sector in terms of environmental impacts. In addition, despite the highly variable characteristics of

[1] A wide range of different of landing ports were observed for hake distribution by Spanish vessels, including many vessels that land their cargo in international ports in Chile, Namibia, or Mauritania. For vessels or fleets in which the landing of hake species occurred outside Spanish territory, the freighting of these products was considered until the port of destination in Spain.

the fishing fleets and species assessed, as well as the way in which the fish co-products are landed, the highest level of homogeneity throughout the sample vessels was sought. However, some aspects, such as the quality and size of the landed species, were not possible to include within the selected FU because a bulk landing was being analyzed.

2.2 System Boundaries

The system boundaries of the different fishing fleets considered in this study presented essentially a common structure because they were all based on the fishing of hake species in different areas worldwide. Therefore, as shown in Fig. 4, a set of five different subsystems were common to all the fishing fleets under assessment: fuel production and use, vessel maintenance, vessel construction, provision of the fishing gear, and the emissions of cooling agents linked to the vessels' refrigeration systems. In addition, a set of remaining operational inputs were characteristic of specific fishing fleets. The latter include the use of bait, which is not used in trawling vessels but is needed by the Northern stock long lining fleet (F-3). In fact, bait for these vessels is usually European pilchard (*Sardina pilchardus*) or Atlantic mackerel (*Scomber scombrus*) supplied by purse seining vessels operating within the Spanish EEZ in Galicia (ICES Areas VIIIc and IXa). Other inputs that were not observed throughout the sample of fleets were the production of ice and oceanic freight. More specifically, oceanic freight was included for fleets that land their products outside Spanish ports and then are freighted by cargo ships to Spanish ports (mainly Vigo, in Galicia, northwest Spain).

Based on these system boundaries, we followed the guidelines provided by PAS 2050:2012-2 for the inclusion of lifecycle inputs and outputs that should be taken into account in the Life Cycle Inventory (LCI), as detailed in Sect. 2.4 (BSI 2012).

2.3 Data Acquisition

Retrieval of data for this study originated from two different individual projects developed in the past 5 years. In the first project, data for the three hake fleets located in European waters, as well as the fleet in the Mauritanian EEZ, were collected in the frame of a *María Barbeito* fellowship sponsored by the *Xunta de Galicia* during the period 2010–2011 (Vázquez-Rowe 2012). In the second project, the inventory data from the two Southern hemisphere fleets included in this review are linked to a service delivery performed for Pescanova SA, a leading global company in the processing and wholesaling of frozen products (Vázquez-Rowe et al. 2013c, d). The data from the different fleets were collected in a similar manner, permitting the joint assessment that is proposed in the current study. Data

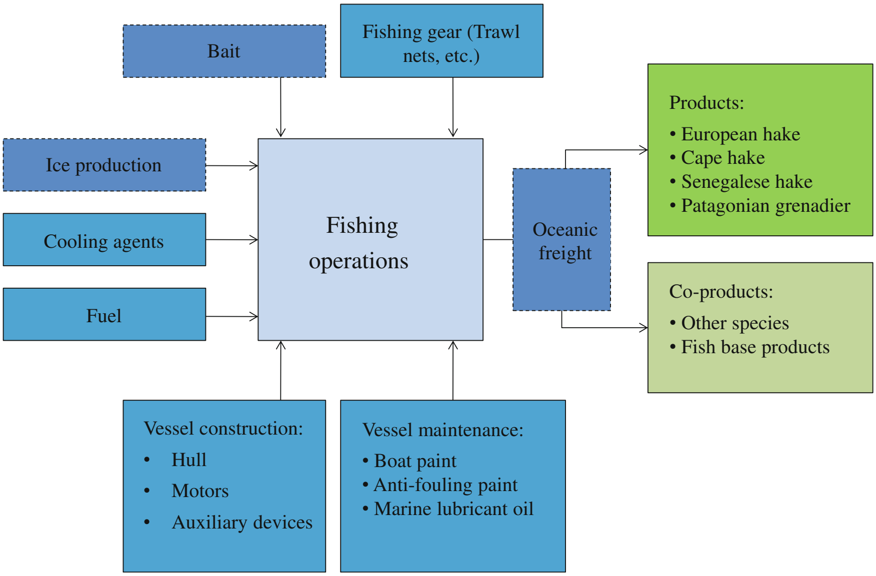

Fig. 4 Schematic representation of the system boundaries of the production systems analyzed in this case study. *Dotted lines* represent operational inputs that are only considered for certain products, whereas *solid lines* represent inputs that are common to all systems. *Blue tones* represent inputs, whereas *green tones* are linked to products and co-products

Table 4 Main characteristics of the selected Spanish fishing fleet samples

	F1	F2	F3	F4	F5	F6
Sample size	9	9	12	24	2	4
Percentage over total (%)	33.3	14.3	20.7	23.8	N/Ap	N/Ap
Year of inventory	2009	2008	2008	2008	2011	2012
Total landings (tons)	5000	16,056	3415	3776	11,220	11,856
Hake landings (tons)	346.4	2832	2072.4	571.5	11,220	11,856

F-1 trawling fleet in Mauritanian EEZ, *F-2* Galician Northern Stock trawling fleet, *F-3* Galician Northern Stock long lining fleet, *F-4* coastal trawling fleet along Galician coast, *F-5* trawling fleet off the Chilean coast (FAO Area 87; Subarea 87.3), *F-6* trawling fleet off the Namibian coast (FAO Areas 47, 48, 49 and 50)
N/Ap Not applicable; this is linked to the fact that these vessels correspond to a private processing company, not to an actual aggregation of Spanish fishing vessels in that area

from the six different fishing fleets, as shown in Table 4, were collected for 1 year of assessment. In fleets F5 and F6, the sample size was the entire population of vessels, guaranteeing a realistic assessment. For the remaining fleets (i.e. F1 through F4), the sample size ranged from 14.3 to 33.3 % of the entire population of vessels.

Despite the high variability in representativeness between fleets, they all were found to be above the standards suggested by the PAS 2050 guidelines for seafood products for random sampling, with the exception of F-2, in which these thresholds were not achieved (BSI 2012). Fishing fleets have a high level of opacity in terms of releasing their data for scientific use (see Sect. 4). This creates a situation in which actual random sampling is ultimately linked to the capability of the LCA practitioner to collect the necessary data from a significant number of individual units (i.e. fishing vessels) and to the willingness of the skippers to release their data.

Another important issue that influenced the way in which the data were collected is the period of data collection. For all fleets, a period of 1 year was performed (see Table 4 for details on the actual year of inventory). This selection may be subject to discussion (see Sect. 4.2) as it is linked to seasonal stock abundance variability, among other issues. However, it was shown to be the most feasible mechanism for data acquisition that was undertaken in the studies contributing to this aggregate review.

In total, 60 vessels were inventoried, with total landings of 28,898 tons of hake. This represents 41.2 % of the average annual hake landings of the Spanish fleet in the period of 2007–2011 (FAOSTAT 2013).

2.4 Life Cycle Inventory

Table 5 provides highly detailed input and output inventory data for all the fishing fleets that were taken into consideration for the current study. The inventory includes all input and output flows described in Fig. 4, although some subsystems are not applicable to all fleets. Diesel use represents the average bulk amount of fuel used by fishing vessels per FU, without any disaggregation based on vessel operation (i.e. gear use, cruising to and from fishing area, etc.). Other transport means were included for F-1, F-5, and F-6 because the hake species did not land directly in Spain. In these cases, the hake was briefly stored at port and then marine freighted to a Spanish port. Table 5 also presents the different co-products that land together with hake in the different fleets. In the case of F-5, we considered the headed and gutted Patagonian grenadier to be the reference product that is processed on board, whereas other processing formats for this same raw product were considered to be co-products. This was done to improve the homogeneity and, therefore, the comparability across the fleets.

2.5 Methodological Assumptions

Capital goods (i.e. infrastructure) were included in the results computation, as shown in Sect. 3.1. However, Sect. 3.2 follows the PAS 2050 recommendations to exclude these items from the system boundaries (BSI 2008).

Table 5 Detailed inventory data for the hake fishing fleets assessed. Data reported per FU: 1 ton of landed hake

Inputs							
From technosphere	Units	F-1	F-2	F-3	F-4	F-5	F-6
Materials							
Steel	kg	11.13	15.07	14.07	5.07	5.35	3.39
Diesel	t	1.74	2.10	1.31	0.50	0.44	0.37
Gear use (trawl net, etc.)	kg	3.86	7.25	–	2.39	0.04	0.24
Boat paint	kg	0.47	0.63	0.66	0.22	0.05	0.17
Anti-fouling paint	kg	1.30	1.75	1.88	0.64	0.17	0.18
Marine lubricant oil	kg	9.15	5.59	0.66	2.16	5.27	0.80
Ice	kg	–	808.0	644.1	323.1	–	–
Oceanic freight	tkm	2000	–	–	–	12200	9000
Bait (pilchard)	kg	–	–	410	–	–	–
Outputs							
To the technosphere	Units						
Products							
Senegalese hake	t	1	–	–	–	–	–
European hake	t	–	1	1	1	–	–
Patagonian grenadier (headed and gutted)	t	–	–	–	–	1	–
Cape hake	t	–	–	–	–	–	1
Co-products							
Common octopus	t	8.48	–	–	–	–	–
Pink cuttlefish	t	1.27	–	–	–	–	–
European squid	t	1.22	–	–	–	–	–
Sand sole	t	0.34	–	–	–	–	–
Common sole	t	0.73	–	–	–	–	–
Caramote prawn	t	0.35	–	–	–	–	–
Megrim	t	–	3.02	–	–	–	–
Angler *spp.*	t	–	2.05	–	–	–	–
Norway lobster	t	–	0.05	–	–	–	–
Varied species	t	–	0.49	–	–	–	0.12
Common ling	t	–	–	0.16	–	–	–
Conger eel	t	–	–	0.04	–	–	–
Atlantic pomfret	t	–	–	0.27	–	–	–
Rock fish	t	–	–	0.09	–	–	–
Fork beard	t	–	–	0.09	–	–	–
Atlantic horse mackerel	t	–	–	–	1.00	–	–
Atlantic mackerel	t	–	–	–	1.20	–	–
Blue whiting	t	–	–	–	2.45	–	–
Patagonian grenadier (blocks)	t	–	–	–	–	15.03	–
Patagonian grenadier (fillets)	t	–	–	–	–	9.81	–
Fish meal	t	–	–	–	–	0.41	–
To the environment	Units						

(continued)

Table 5 (continued)

Inputs							
From technosphere	Units	F-1	F-2	F-3	F-4	F-5	F-6
Emissions to the atmosphere							
CO_2	t	5.51	6.67	4.14	1.57	1.40	1.16
Methane (CH_4)	kg	0.09	0.11	0.07	0.02	0.02	0.02
NOx	kg	125.0	151.4	5.22	35.68	31.77	26.35
Cooling agent (R22)	kg	0.69	0.48	0.70	0.22	0.41	0.34
Cooling agent (R404A)	g	–	–	–	–	3.32	–

F-1 trawling fleet in Mauritanian EEZ, *F-2* Galician Northern Stock trawling fleet, *F-3* Galician Northern Stock long lining fleet, *F-4* coastal trawling fleet along Galician coast, *F-5* trawling fleet off the Chilean coast (FAO Area 87; Subarea 87.3), *F-6* trawling fleet off the Namibian coast (FAO Areas 47, 48, 49, and 50)

Allocation was not necessary in F-5 due to the fact that the vessels in this fleet do not target fishing species other than hake. Therefore, all environmental impacts in terms of GHG emission were attributed to the single final product: Patagonian grenadier. The other fleets all targeted multiple species. In two of these fleets, F-3 and F-6, hake was the main landed species in terms of total biomass; in F-1, F-2, and F-4, other species had a more important role regarding total landings. However, in all fleets, hake had an important role from an economic revenue perspective, being a key species to ensure the economic feasibility of the fishery.

Following the suggestions dictated by PAS 2050-2, mass allocation was performed when necessary because it was not feasible to conduct a system expansion (ISO 2006b; BSI 2012). In fact, PAS 2050-2 does not consider the use of any other type of allocation in capture fisheries, obviating some relevant seafood LCA and CF studies available in the literature that advocate for a wide range of different allocations approaches, such as economic allocation (Ziegler et al. 2003; Iribarren et al. 2010a, 2011; Svanes et al. 2011a; Ziegler et al. 2011), temporal allocation (Ramos et al. 2011), or energy allocation (Parker 2011; Svanes et al. 2011b). The recommendations posed by PAS 2050-2 are in line with a review by Pelletier and Tyedmers (2011) by disregarding the use of market-driven information (i.e. economic allocation) in biophysical systems. However, they also showed certain caution towards the systematic use of mass allocation, arguing that there may be more appropriate biophysical parameters to model material and energy flows within LCA and CF (Pelletier and Tyedmers 2011).

In the current study, as mentioned, it was decided to follow the recommendations of the ISO framework and PAS 2050 if system expansion was not feasible. Therefore, a mass allocation approach was taken into consideration. However, economic allocation is included in the discussion section as part of the sensitivity analysis (Sect. 4.2) to determine the magnitude of allocation methodological choices in the selected systems.

2.6 Impact Assessment Phase

IPCC 2001 (with a 100-year timespan) was selected as the assessment method to calculate the CF for the different fishing fleets. The selection of this specific method is in accordance with the recommendations provided by the ILCD guidelines (ILCD 2010, 2011). These recommendations were generated based on a cross-method comparison that included other commonly used assessment methods for the computation of climate change impacts, using science-based criteria to evaluate the completeness of scope, environmental relevance, certainty, scientific robustness, transparency, reproducibility, applicability, and acceptance criteria for stakeholders and scientists.

2.7 Scaling Up

The CF results calculated for each individual fishing fleet were used to estimate the GHG emissions linked to the average amount of hake species that were landed by Spanish vessels in the period 2007–2011. In particular, datasets with annual landings of the different hake species considered in the study were obtained from the FAO statistics database (FAOSTAT 2013) for the time period 1950–2011. The data were reported in tons per year by FAO fishing region. Despite the wide range of fleets that were included in this assessment, there were still some hake species and fishing fleets targeting these species for which no LCI data were available. Therefore, two different assessment analyses were taken into account. In the first analysis, considering that the sample represents 41.2 % of the average hake landing of the Spanish fleet in recent years, a rough calculation was done to extrapolate the results to 100 % of the average landings. In the second analysis, a more detailed estimation was performed by individually extrapolating the results by fishing area and hake species. The latter approach encountered a series of methodological constraints due to the fact that not all hake species and fishing areas were accounted for in the sample obtained. Hence, in this second approach, the landings of European hake in the Mediterranean and Argentine hake (*Merluccius hubbsi*), as well as the landings from other minor species captured by Spanish vessels, were disregarded due to the lack of available data. In addition, for European hake in northeastern Atlantic waters, the average value was based on data available from ICES that details the specific fishing gears used to catch hake in the period 2007–2011 in the Southern Stock (ICES 2013). In the case of the Northern Stock, the assumption was made that the proportion of hake caught with trawling and long lining corresponded to the proportion depicted in Table 4 (i.e. 42.3 % for long liners and 57.7 % for trawlers). Finally, for artisanal and small-scale vessels capturing hake, the CF value was obtained from Iribarren et al. (2010a, b, 2011).

3 Results

3.1 Carbon Footprint Values for Individual Fishing Fleets (ISO Framework)

CF results using the methodological assumptions of the ISO framework (i.e. ISO 14040 and ISO 14044) were computed. The results per FU for each fishing fleet are shown in Table 6. The highest CF value within the fishing fleets selected in this review study corresponds to the trawling fleet in the Northern Stock: 8796 kg CO_2 eq./ton of hake. The long lining fleet in this same fishing zone (i.e. F-3), in contrast, shows environmental impacts that are 27.3 % lower in terms of climate change (6.4 t/t). In an intermediate range, Senegalese hake presented a CF value of 7.8 t/t. The remaining examined fleets showed substantially lower GHG emissions per unit of landed hake, with results ranging from roughly 2.1 to 2.7 t/t. The lowest value was reported for cape hake, landed by the Spanish fleet in the Namibian EEZ.

Diesel production and consumption represented the main source of GHG emissions for all six fishing fleets (see Fig. 5). The relative contribution ranged from 65.5 % for F-6 to 88.6 % for F-2. In fact, two trawling fleets (i.e. F-5 and F-6: 66.9 and 65.5 %, respectively) presented lower relative contributions of diesel impacts than F-3 (75.6 %), a long lining fleet in the Northern Stock. On the upper range, trawling fleets in European waters (F-2 and F-4: 88.6 and 80.1 %, respectively), as well as the trawling vessels catching hake off the coast of Mauritania (F-1: 83.0 %), showed a very high impact linked to vessel fuelling.

Cooling agents, mainly R22, were the second contributor to the final CF of all the fishing fleets evaluated. The fleets with highest dependency on these substances in terms of final GHG emissions were F-1 and F-3, with average contributions of 1250 kg CO_2/FU and 1272 kg CO_2/FU, respectively. In contrast, F-4, probably due to the proximity to the coast and to the fact that vessels go back to port every 24 h, showed the lowest average contribution: 406 kg CO_2/FU. Moreover, in a similar range, two highly industrialized fleets, F-5 and F-6, showed impacts of 766 kg CO_2/FU and 611 kg CO_2/FU, respectively. However, it should be noted, as illustrated in Fig. 5, that the relative contributions of cooling agent GHG emissions to the overall impact show different tendencies, with F-6 showing the highest relative contribution (29.6 %) and F-2 showing the lowest (9.8 %).

Transoceanic freight for fleets F-5 and F-6 also involved substantial impacts, at 3.8 and 4.7 % of the total impact, respectively. For F-1, marine freighting only represented 0.3 % of the total impact. However, truck freighting of seafood products from the port of Nouadhibou (Mauritania) also occurs through North Africa into Spain, as discussed by Vázquez-Rowe et al. (2012b). Finally, the remaining operational inputs that were considered presented reduced contributions to the final GHG emissions of hake landing activities. Trawl nets and ice production were the most relevant in this group of operational inputs, but in no case were these above 1.3 %. The remaining inputs, such as paint, anti-fouling, and water use, showed minimal contributions to the final CF of these fleets.

Table 6 Carbon footprint calculations per functional unit for the selected hake species fishing fleets (ISO framework perspective)

	Diesel	Lubricants	Transoceanic freight	Cooling agents	Bait	Boat paints	Gillnets	Ice	Water	Steel	Total
F-1	6437	8.13	21.43	1250	N/Ap	2.18	27.74	0	0	4.68	7751.4
F-2	7795	4.97	N/Ap	862.8	N/Ap	2.95	52.19	72.14	0	6.34	8796.0
F-3	4838	13.12	N/Ap	1272	208.2	3.16	0.00	57.51	0	5.92	6397.5
F-4	1837	1.92	N/Ap	405.8	N/Ap	1.08	17.23	28.85	0	2.14	2293.7
F-5	1826	4.69	130.7	765.5	N/Ap	0.29	0.32	0	0	2.25	2730.0
F-6	1352	0.67	96.44	611.4	N/Ap	0.34	1.73	0	0.20	1.42	2063.9

F-1 trawling fleet in Mauritanian EEZ, *F-2* Galician Northern Stock trawling fleet, *F-3* Galician Northern Stock long lining fleet, *F-4* coastal trawling fleet along Galician coast, *F-5* trawling fleet off the Chilean coast (FAO Area 87; Subarea 87.3), *F-6* trawling fleet off the Namibian coast (FAO Areas 47, 48, 49 and 50), *N/Ap* not applicable

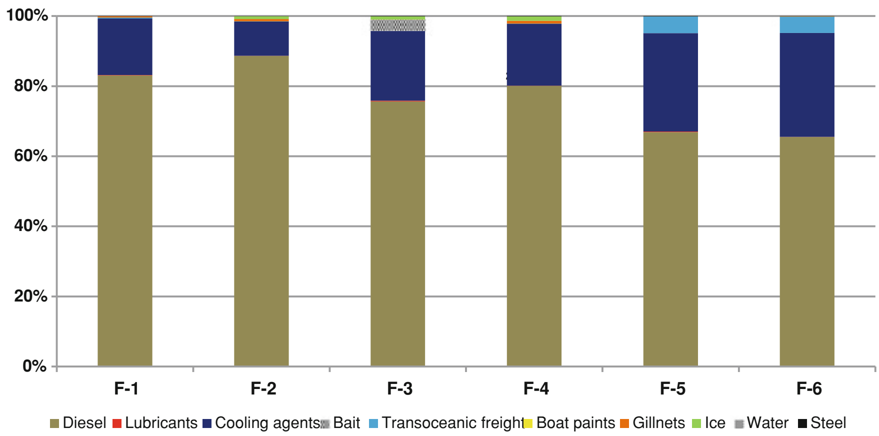

Fig. 5 Relative contribution to greenhouse gas emissions for the activities considered in each production system

3.2 *Carbon Footprint Values for Individual Fishing Fleets (PAS 2050 Framework)*

When the same fleets were assessed using the PAS 2050 standard, the results showed limited variation when compared to the results obtained with the ISO 14040/44 framework (see Table 7). In fact, the relative variations in GHG emissions observed throughout the different fishing fleets are fairly constant, ranging from 2 to 3 %. Consequently, the change in CF values between the two frameworks, considering the same methodological assumptions (i.e. allocation, FU, etc.) and the same assessment method (i.e. IPCC 2007), does not imply major differences in fleets with an important FUI. However, it should be noted that in fishing activities where the reliance on energy consumption is lower and, therefore, the relative contribution to the final CF of other operational inputs (including capital goods) increases, the range of variance could increment considerably, as shown in some studies on pelagic fisheries (Ramos et al. 2011).

3.3 *Global Carbon Footprint of Hake Landings for the Spanish Market*

As mentioned in Sect. 2.6, two different approaches were taken to calculate the total CF linked to hake landings by Spanish vessels. In the first approach, a total of 97,264 tons of CO_2e per year were attributed to the capture of hake species in the sample of 60 vessels that were available. If these results are extrapolated to the entire hake industry, a total of 236,079 t CO_2e/year can be estimated. In contrast,

Table 7 Carbon footprint calculation per functional unit for the selected hake species fishing fleets (BSI framework perspective)

	Diesel	Lubricants	Transoceanic freight	Cooling agents	Bait	Boat paints	Gillnets	Ice	Water	Total
F-1	6293	7.51	17.6	1250	N/Ap	2.04	27.29	0	0	7597.9
F-2	7621	4.56	N/Ap	862.8	N/Ap	2.75	51.34	70.99	0	8613.1
F-3	4730	12.13	N/Ap	1271.6	197.8	2.95	0	56.59	0	6271.1
F-4	1796	1.77	N/Ap	405.8	N/Ap	1.00	16.95	28.39	0	2249.6
F-5	1785	4.33	107.5	765.2	N/Ap	0.27	0.32	0	0	2662.4
F-6	1322	0.62	79.3	611.4	N/Ap	0.32	1.70	0	0.15	2015.0

F-1 trawling fleet in Mauritanian EEZ, *F-2* Galician Northern Stock trawling fleet, *F-3* Galician Northern Stock long lining fleet, *F-4* coastal trawling fleet along Galician coast, *F-5* trawling fleet off the Chilean coast (FAO Area 87; Subarea 87.3), *F-6* trawling fleet off the Namibian coast (FAO Areas 47, 48, 49 and 50), *N/Ap* not applicable

Table 8 Average annual carbon footprint for the whole Spanish hake fishing industry in the period 2007–2011

Hake species		Fishing area	Average total landings (period 2007–2011)		Estimated CF per ton of landed hake	Estimated total GHG emissions	
Common name	Scientific name		Total (tons/year)	Relative (% over total)	(kg CO_2eq./ton)	Total (kg CO_2eq./year)	Relative (% over total)
European hake	*Merluccius merluccius*	Northeast Atlantic	21,129	30.12	4305.0	90,961	41.79
		Eastern Central Atlantic	3603	5.14	7751.4	27,931	12.83
Senegalese hake	*Merluccius senegalensis*	Eastern Central Atlantic	6332	9.03	7751.4	49,079	22.55
Cape hake	*Merluccius capensis*	Southeast Atlantic	4807	6.85	2063.9	9921.2	4.56
Patagonian grenadier	*Macruronus magellanicus*	Southwest Atlantic	12,858	18.33	2730.0	35,102	16.13
White hake	*Urophycis tenuis*	Northeast Atlantic	10.2	0.01	8796.1	89.72	0.04
Benguela hake	*Merluccius polli*	Southeast Atlantic	2213	3.15	2063.9	4568.3	2.10
Total computed			*50,952*	*72.63*	*4271.7*	*217,652*	*100.0*
Argentine hake	*Merluccius hubbsi*	Southwest Atlantic	14,524	20.71	N/Av	N/Av	N/Av
European hake	*Merluccius merluccius*	Mediterranean	4431	6.32	N/Av	N/Av	N/Av
White hake	*Urophycis tenuis*	Northwest Atlantic	95.6	0.14	N/Av	N/Av	N/Av
Red hake	*Urophycis chuss*	Northwest Atlantic	104.4	0.15	N/Av	N/Av	N/Av
Silver hake	*Merluccius bilinearis*	Northwest Atlantic	1.60	0.00	N/Av	N/Av	N/Av
Southern hake	*Merluccius australis*	Southwest Atlantic	40.2	0.06	N/Av	N/Av	N/Av
		Southwest Pacific	1.00	0.00	N/Av	N/Av	N/Av
Total			*70150*	*100.0*	*N/Av*	*N/Av*	*N/Av*

when the specific fishing areas and hake species are considered, a total of 217,652 tons of CO_2e/year were estimated for the hake-fishing sector (see Table 8). However, in the latter approach, it should be noted that the landings from several species and fishing areas were excluded (see Sect. 2.7), which represented 72.6 % of the average total landings in the period 2007–2011.

The second perspective also provides a detailed calculation per fishing area and species. In the first place, European hake, which is still the main species in terms of landings in the period assessed, accounts for a total of 90,961 tons of CO_2e in the Northern and Southern Stocks in European waters, although a small percentage of the landings that occurred in Mediterranean fishing areas were not computed due to lack of data. Despite the high FUI and CF observed for European hake in this area, when the landings are weighted based on the fishing gear and, therefore, landings from artisanal and small-scale vessels are included, the average CF per ton of landed hake decreases to 4305 kg CO_2e/ton, which is substantially lower than the average CF observed when fishing hake species in Eastern Central Atlantic waters. In fact, even though Senegalese hake only accounts for 9 % of hake landings, its total CF adds up to 49,079 tons of CO_2e. However, the CF value for European hake still remains very high when compared with the landings of Patagonian grenadier (2730 kg CO_2e/t) and cape hake (2063 kg CO_2e/t).

4 Discussion

4.1 The Utility of Estimating GHG Emissions for Fisheries Managers

The decline in fishing stocks in the North Atlantic has been an important matter of concern for fisheries managers in recent decades (Christensen et al. 2003). In fact, these managers have dealt for years with the challenge of attaining sustainable fisheries in this area without jeopardizing social and economic interests in the fishing sector on both sides of the Atlantic. This situation has led to an increase in the complexity of fishery management decisions and higher risks in the final success of the decision-making process due to the ongoing depletion of many fisheries (Farmery et al. 2013). In parallel, FAO reports also indicate that seafood trade between EU and North American countries is not only related to the new trading trends of the globalized market, but also to the fact that many developed countries can no longer supply seafood products to their population from resources available in the fishing areas of their jurisdiction.

Countries such as Spain, Portugal, and the United Kingdom currently import over half of their seafood products (FAOSTAT 2013). Therefore, institutions and seafood companies have been searching for new fishing areas in which they can obtain the additional landings to meet the consumer demand back in Europe. International EU fisheries policy, which is essentially centralized through the

European Commission, has increasingly pushed for signing fishing agreements with third-party countries, mainly African countries such as Mauritania, Morocco, or Mozambique (European Commission 2013). In addition, multinational seafood companies have expanded their areas of influence by outsourcing their production to countries in other continents. These two trends should also be considered in a wider context in which aquaculture plays an important role (Klinger et al. 2013).

Special interest groups have started not only questioning the sustainability of European fisheries, but also identifying ongoing practices in other parts of the world that ultimately result in meeting the seafood demands of European grocery stores. However, the GHG emissions observed in this study show that outsourcing of hake landings is done at lower CF profiles than the fishing of European hake in European waters. Although the lack of historic data on fuel combustion is a drawback that limits the depth of the analysis that can be performed in this study, higher GHG emissions per unit of mass landed could be attributable to years of overexploitation of European hake species in European fishing grounds. In fact, as pointed out in recent ICES reports, the mortality rate of European hake in fishing areas is still above the desirable thresholds, despite recent improvements in terms of spawning stock biomass and stock abundance (ICES 2013).

Consequently, based on the descriptive results presented in this study and provided that data acquisition can be collected with a high level of detail, future actions should aim at understanding the links between GHG emissions (and, if necessary, other environmental impacts), stock abundance, and its sustainable management. Certain studies have already suggested that strong correlations between GHG emissions and management strategies may exist (Driscoll and Tyedmers 2010; Hornborg et al. 2012).

From a strictly GHG emissions perspective, it is clear that the intensity of energy use in hake fisheries is among the highest in the world, due to a series of inherent characteristics: (i) the species type (demersal bottom-dwellers); (ii) the fishing gears used (trawlers account for the highest proportion of landings); (iii) the location of the fishing areas; and (iv) the market-driven characteristics of the hake subsector (hake is a popular fish species with Spanish consumers, who are willing to pay higher prices for hake than for other fish species).

4.2 The Impact of Methodological Assumptions on the Final Results

The variation in reported CF values per fishing fleet when using the ISO standards or those suggested by the British Standards Institution (i.e. PAS 2050) have been shown to be minimal (roughly 2 %). However, in this specific case study, these results should be interpreted with care due to the overwhelming contributions of diesel production and propulsion impacts to the entire CF. In fact, in other fisheries in which fuel intensity is not as impactful, such as pelagic fisheries (e.g. tunids,

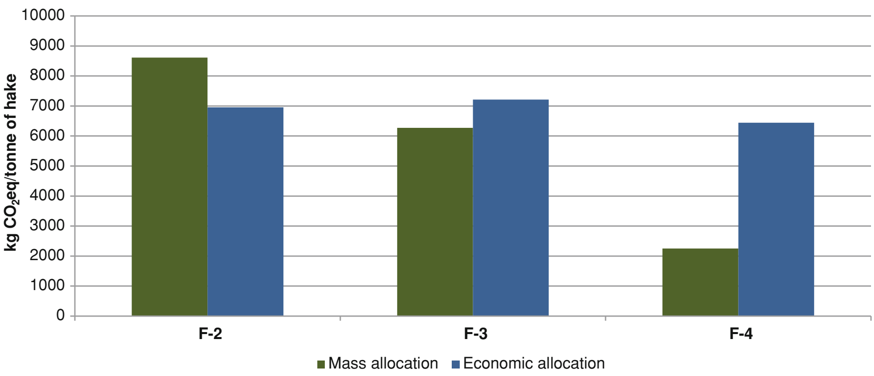

Fig. 6 The variation of a selection of fishing fleets when using mass allocation. Note: allocation for the Mauritanian trawling fleet, despite being a multispecies fishery, was not included due to a lack of solid economic data

mackerel, anchoveta), the role of other operational inputs, such as gear use or maintenance, have important contributions to the final CF profile of the captured products.

Other methodological assumptions that are transversal to the two standards, such as allocation, showed a higher change to the final CF in this particular study. It should be noted that PAS 2050-2 does not consider the possibility of performing allocations other than mass, whereas the ISO framework is more flexible, allowing economic allocation if some type of biophysical relationship cannot be established (ISO 2006b, BSI 2012). Therefore, in Fig. 6, the results obtained with the ISO framework using mass allocation are presented with results for economic allocation. The results, which only apply to multispecies fishing fleets, show that there is a relevant increase in the CF profile of hake species (in this case, always European hake) due to its higher economic values compared with the remaining species captures by the vessels. Although this statement is valid for F-3 and F-4 because most of the other landed species, such as Atlantic horse mackerel (*Trachurus trachurus*) or blue whiting (*Micromesistius poutassou*), have a much lower economic value, in F-2 the situation is reverted. The reason for this, as shown in Table 5, is the fact that megrim (*Lepidorhombus* spp.) and anglerfish (*Lophius* spp.) have a relatively high economic value in the Spanish market.

These differing results, depending on the allocation choice, demonstrate that caution must be taken when interpreting the results, especially when reporting to stakeholders or the general public. For instance, although it appears clear that the weight of hake is relevant based on economic value of the species, it should also be taken into account that the landed fish are essentially part of the same fishing effort in all these case studies. In other words, the multispecies characteristics of the fleet are not only linked to targeting different species throughout the year, but also to the mixed catch obtained within single hauls. Therefore, despite the differing economic value of the caught species, they cannot be really separated biophysically.

Two other important components of fishing activities should also be taken into account. First, fishing quotas and moratoria may have important limitations on the actual amount of profitable catch that the vessels can actually land. Second, discarding may be the preferred option for species with low or no economic value, although EU legislation is supposed to enforce stronger controls on this practice in the revised CFP. Furthermore, in many cases, the final economic value of species is unknown to the fishermen until the auction is completed at the port on arrival, with prices for one single species varying enormously on a daily basis based on a wide range of factors such as quality, size, port of landing, etc.

Based on this discussion, we argue that the monetary price of landed fish does not constitute a robust allocation perspective in most analysis because its volatility could lead to important misinterpretations. Therefore, as suggested by the different standards and in the literature, the use of economic allocation should be discouraged as a generic methodological choice to attribute environmental impacts to co-products in fisheries (ISO 2006b; Pelletier and Tyedmers 2011; BSI 2012). However, in some specific case studies, it could still be a valid methodological selection under certain conditions, as long as it can be sustained from a goal and scope perspective. For instance, a study focusing on the landings of European pilchard for midsummer celebration in Northern Portugal or Galicia (northwest Spain), which makes the price of this low-value species skyrocket for a few weeks per year in June, could justify economic allocation, provided that the catches on those days are still multispecies and not entirely devoted to pilchard landing. Consequently, the recommendation is that mass allocation represents a more solid approach to partition fisheries co-products. However, the recommendation by some authors to use other biophysical relationships, such as energy or protein content, remains highly unexplored (Pelletier and Tyedmers 2011; Svanes et al. 2011b; Vázquez-Rowe et al. 2013a, 2014b).

The correct sampling of the fishing fleets was an additional critical factor in terms of providing a cross-fleet homogeneous sampling method. Despite the recommendations in PAS 2050-2 for random sampling within fishing fleets, we found that the randomness of the sample was difficult to obtain in fishing fleets in which the study was not conducted directly through a consulting service with fishing companies. Hence, in F-5 and F-6, all the vessels of the fishing fleet were assessed due to the mentioned direct involvement of the fishing company and its interest in collaborating with the LCA practitioners. However, in the remaining fleets, the samples were obtained through direct interviews with skippers in different Spanish ports (mainly in Galicia), disabling the randomness of the studies. In some cases, skippers were unwilling to disclose data for the vessels; in other cases, the vessels were out at sea and skippers were not available for the interviews. Furthermore, in the latter case studies, an initial survey had been distributed by conventional postal mail to port authorities throughout Spanish fishing ports, but this mechanism was disabled due to the low response rate. Consequently, despite the robustness of the sampling mechanism described by the British Standards Institution in PAS 2050-2 (BSI 2012), we argue that the method lacks realistic operability in many cases because of the reluctance of many stakeholders to disclose life-cycle-valuable data to LCA practitioners. Therefore, although we advocate for the use of the sampling standards described by PAS 2050-2, we also encourage LCA practitioners to search for other sampling alternatives in case random sampling is not feasible, as long as the methods are clearly described and the researchers are willing to highlight the increased uncertainty of their results.

In addition to the sampling problem, the sampling period also appeared to be an important constraint in the fishing fleets that were assessed in this case study. Hence, despite the recommendations in the literature to use moving averages of at least 3 years when reporting LCA and CF results for fishing activities (Ramos et al. 2011), which were later included in the PAS 2050-2 specification (BSI 2012), these data were only disclosed for F-6. Therefore, we used the second alternative recommended by the BSI, which is to collect data for an entire year of assessment, although some differences were found between fleets because some fleets are subject to certain moratoria throughout the year.

4.3 Sources of Uncertainty

The sources of uncertainty behind CF results are usually varied and must be analyzed in detail in order to understand the risks of results interpretation. In the first place, the metrics behind the calculation of GHG emissions are an important source of uncertainty. Having said this, global warming metrics have been studied in depth in recent decades due to the ever-increasing importance of the mitigation of climate change. Therefore, most LCA scientists would agree that the uncertainties behind global warming metrics are among the lowest within life cycle impact assessment categories (Reap et al. 2008a, b). For instance, categories linked to toxicology or the quantification of marine eutrophication have been shown to have higher ranges of uncertainty (Reap et al. 2008a, b).

A second source of uncertainty is the depth of the LCI collected for the different fishing fleets. The collection of data for operational inputs have shown to be of high importance in previous case studies (Ziegler et al. 2003; Thrane 2004a; Hospido and Tyedmers 2005; Vázquez-Rowe et al. 2012a; Avadí and Fréon 2013). Hence, diesel consumption inputs, emissions of cooling agents, or production of ice have been modelled with care. However, other operational inputs, such as the construction of the vessel, deserve further attention in future studies in order to illustrate the full extent of their contribution to CF results. Moreover, it should be noted that emission factors used for diesel consumption and paint and anti-fouling use emissions also contribute to the potential uncertainty of these case studies.

Another important issue that should not be disregarded is illegal, unreported, and unregulated (IUU) catches. The sampling method used in these case studies (fleets F-1 to F-4) allowed us to avoid the use of official statistics by directly contacting skippers. However, we were unable to quantify the additional amount of catches that these skippers were performing above the official reports. When the CF results were scaled up to the whole hake sector (see Sect. 3.3), the use of official statistics from ICES and other organizations was needed (ICES 2013). Consequently, the uncertainties linked to IUU in this estimation definitely constitute an important issue for consideration.

Finally, regarding the characteristics of the fleets, important uncertainties can be derived from the way in which data sampling was performed. This issue, which was discussed in Sect. 4.2, is particularly relevant in fishing fleets LCA studies because most literature available to date considers, at least partially, primary data obtained directly from skippers and vessel owners, fishing associations, or fisheries managers (Vázquez-Rowe et al. 2012a; Avadí and Fréon 2013).

Regarding the scaling up of the results obtained, it should be noted that the uncertainty behind them is considerable due to the lack of representative data for roughly 27.4 % of landed hake species by Spanish vessels. Nevertheless, the results illustrate the need to tackle GHG emissions through mitigation policies and the correct handling of joint stock abundance and fuel use management in hake fishing fleets. In fact, the average CF value for hake species assessed in this study (using the second approach to scale up the results) is 4.27 t CO_2e/t of landed hake, which is substantially higher than the worldwide average for fishing: 1.7 t CO_2e/t of landed fish, as estimated by Tyedmers et al. (2005).[2] Although it should not be expected that demersal species, such as hake, should have CF values comparable to some pelagic species, this comparison serves as a basis to set specific benchmarks to achieve substantial reductions based on best practices. For instance, a series of studies using a combined method between LCA and Data Envelopment Analysis (DEA), named the LCA + DEA method, has shown that important reductions in environmental impact, including GHG emissions, can be attained by improving the eco-efficiency of these vessels (Vázquez-Rowe et al. 2010; 2011a, b; Ramos et al. 2014).

5 Conclusions

The capture of hake species at sea by fishing vessels is an important use of fuel for propulsion. In fact, when compared to other white fish species that have been analyzed from a life cycle perspective, hake appears on the upper end of the list in terms of GHG emissions per unit of landed fish. Nevertheless, important differences were identified between fishing areas and fleets, with fisheries in the northeast and eastern central Atlantic being among those with the highest environmental impact.

These differences between fisheries suggest that biotic assessment methods of stock sustainability should be assessed in combination with CF and, if necessary, with other lifecycle environmental impacts (e.g. fossil depletion), in order to shift

[2] Important methodological differences exist between the two studies. For instance, Tyedmers et al. (2005) only took into consideration fuel-driven emissions, without taking into account the GHG emissions from other life-cycle inputs in the fishing stage.

hake captures based on the tradeoffs between these two aspects. In other words, fisheries that show sustainable stocks and lowered GHG emissions in terms of fishing fleet operations should be prioritized rather than those in which these two indicators struggle. In addition, the development of policies in this direction should be enforced, given the strong relationship that has been observed in many studies between climate change and changing fishery patterns.

From a Spanish perspective, hake landings are an important source of GHG emissions. Although this finding in itself should not be surprising, considering that it is the leading source of seafood intake in Spanish households, this first approximation provides visibility and actual CF values to the extent of its impact on society in terms of climate change. Consequently, future research should aim at improving the quality of the results provided in this case study by expanding the number of fishing fleets and hake species assessed, as well as by aiming to increase the sample size and assessment period. In addition, the complex postlanding pathways for hake species in Spain, including fresh, frozen, and processed distribution of hake products, should be further analyzed, including a cradle-to-cradle perspective to the hake sector.

Acknowledgements This chapter was developed thanks to funding from the Galician Government (Project reference: GRC 2010/37). The authors would like to thank Pescanova SA for the data disclosed for some of the fisheries examined.

References

Antelo M, Rodríguez D, Villasante S (2012) The Spanish fishing fleet and the economic value of southern stock of European hake fishery (*Merluccius merluccius*). Ocean Coast Manage 70:59–67

Armstrong CW, Sumaila UR, Erastus A, Orton Msiska O (2004) Benefits and costs of the Namibianisation policy. In: Sumaila UR, Boyer D, Skongen MD, Steinshamn SI (eds) Namibia's fisheries: ecological economic and social aspects. Eburon, Amsterdam, pp 203–214

Asche F, Guillén J (2012) The importance of fishing method, gear and origin: the Spanish hake market. Mar Policy 36:365–369

Avadí A, Fréon P (2013) Life cycle assessment of fisheries: a review for fisheries scientists and managers. Fish Res 143:21–38

Ayer NW, Tyedmers PH, Pelletier NL, Sonesson U, Scholz AJ (2007) Co-product allocation in life cycle assessments of seafood production systems: review of problems and strategies. Int J Life Cycle Assess 12:480–487

BSI (2008) PAS 2050:2008. Specification for the assessment of life cycle greenhouse gas emissions of goods and services. British Standards Institution

BSI (2011) PAS 2050: 2011. Specification for the assessment of life cycle greenhouse gas emissions of goods and services. British Standards Institution

BSI (2012) PAS 2050-2:2012. Assessment of life cycle greenhouse gas emissions. Supplementary requirements for the application of PAS 2050:2011 to seafood and other aquatic food products. ISBN 978-0-580-77964-0

Byrd-Bredbenner C, Lagiou P, Trichopolou AA (2000) A comparison of household food availability in 11 countries. J Hum Nutr Diet 13:197–204

Christensen V, Guenette S, Heymans JJ, Walters CJ, Watson R, Zeller D, Pauly D (2003) Hundred-year decline of North Atlantic predatory fishes. Fish Fish 4:1–24

Driscoll J, Tyedmers P (2010) Fuel use and greenhouse gas emission implications of fisheries management: the case of the new england atlantic herring fishery. Mar Policy 34:353–359

Druckman A, Jackson T (2009) The carbon footprint of UK households 1990–2004: a socio-economically disaggregated, quasi-multi-regional input–output model. Ecol Econ 68:2066–2077

European Commission (2013) European external action service. Treaties office database. Fisheries partnership agreements. http://ec.europa.eu/world/agreements/searchByActivity.do?parent=8520&xmlname=759&actName=Fisheries

FAOSTAT (2013) Statistics database. Food and Agriculture Organization of the United Nations—FAO. http://faostat.fao.org/site/291/default.aspx

Farmery A, Gardner C, Green BS, Jennings S (2013) Managing fisheries for environmental performance: the effects of marine resource decision-making on the footprint of seafood. J Clean Prod 64:368–376

García D, Prellezo R, Santurtun M, Arregi L (2011) Winners and losers of a technical change: A case study of long-term management of the Northern European Hake. Fish Res 110:98–110

García-Vázquez E, Machado-Schiaffino G, Campo D, Juanes F (2012) Species misidentification in mixed hake fisheries may lead to overexploitation and population bottlenecks. Fish Res 144:52–55

Girard S, Mariojouls, C (2008) What is the demand for farmed fish on European markets? In: IIFET 2008 Vietnam Proceedings

Guillén J, Franquesa R, Sole I (2004) Price relationships on the Spanish hake market. In: IIFET, 2004 Japan proceedings, p 13

Hertel TW, Golub AA, Jones AD, O'Hare M, Plevin RJ, Kammen DM (2010) Effects of US maize ethanol on global land use and greenhouse gas emissions: estimating market-mediated responses. Bioscience 60:223–231

Hornborg S, Nilsson P, Valentinsson D, Ziegler F (2012) Integrated environmental assessment of fisheries management: Swedish Nephrops trawl fisheries evaluated using a life cycle approach. Mar Policy 36:1193–1201

Hospido A, Tyedmers P (2005) Life cycle environmental impacts of Spanish tuna fisheries. Fish Res 76:174–186

ICES (2013) Report of the Working Group on the Assessment of Southern Shelf Stocks of Hake, Monk and Megrim (WGHMM). International Council for the Exploration of the Sea. 10–16 May 2013, ICES Headquarters, Copenhagen. ICES CM 2013/ACOM:11A. p 11

ILCD (2010) General guide for life cycle assessment—detailed guidance. European Commission, Joint Research Centre, Institute for Environment and Sustainability

ILCD (2011) Recommendations for life cycle impact assessment in the European context-based on existing environmental impact assessment models and factors. International reference life cycle data system. ISBN 978-92-79-17451-3

IPCC (2007) Contribution of Working Groups I, II and III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. IPCC, Geneva, Switzerland. pp 104

Iribarren D, Vázquez-Rowe I, Hospido A, Moreira MT, Feijoo G (2010a) Estimation of the carbon footprint of the Galician fishing activity (NW Spain). Sci Tot Environ 408:5284–5294

Iribarren D, Hospido A, Moreira MT, Feijoo G (2010b) Carbon footprint of canned mussels from a business-to-consumer approach. A starting point for mussel processors and policy makers. Environ Sci Policy 13:509–521

Iribarren D, Vázquez-Rowe I, Hospido A, Moreira MT, Feijoo G (2011) Updating the carbon footprint of the Galician fishing activity (NW Spain). Sci Total Environ 409:1609–1611

ISO (2006a) ISO 14040. Environmental management—life cycle assessment—principles and framework. International Organization for Standardization

ISO (2006b) ISO 14044. Environmental management—life cycle assessment—requirements and guidelines. International Organization for Standardization

ISO (2013) ISO 14067—Carbon footprint of products—requirements and guidelines for quantification and communication. http://www.iso.org/iso/iso_catalogue/catalogue_tc/catalogue_detail.htm?csnumber=59521

Klinger DH, Turnipseed M, Anderson JL, Asche F, Crowder LB, Guttormsen AG, Halpern BS, O'Connor MI, Sagarin R, Selkoe KA, Shester GG, Smith MD, Tyedmers P (2013) Moving beyond the fished or farmed dichotomy. Mar Policy 38:369–374

Laurent A, Olsen SI, Hauschild MZ (2012) Limitations of carbon footprint as indicator of environmental sustainability. Environ Sci Technol 46:4100–4108

Martín-Cerdeño VJ (2010) Consumo de pescado en España. Diferencias en función de las características del consumidor. MERCASA. Distribución y consumo. Septiembre–Octubre 2010 (in Spanish)

Parker R (2011) Measuring and characterizing the ecological footprint and life cycle environmental costs of Antartic krill (*Euphausia superba*) products. M.Sc. Thesis. Dalhousie University, Departement of Environmental Studies

Parker R, Vázquez-Rowe I, Tyedmers P (2014) Fuel performance and carbon footprint of the global purse seine tuna fleet. J Clean Prod, under review

Pelletier N, Audsley E, Brodt S, Garnett T, Henriksson P, Kendall A, Kramer KJ, Murphy D, Nemecek T, Troell M (2011) Energy intensity of agriculture and food systems. Annu Rev Environ Resour 36:223–246

Pelletier N, Tyedmers P (2011) An ecological economic critique of the use of market information in life cycle assessment research. J Ind Ecol 15:342–354

Piecyk M, McKinnon AC (2010) Forecasting the carbon footprint of road freight transport in 2020. Int J Prod Econ 128:31–42

Piquero Zarauz S, López E (2005) El consumo de pescado en España. Siglos XVIII-XX. Una primera aproximación. XI Congreso de Historia Agraria, Aguilar de Campoo, Spain. 15–18 July 2005 (in Spanish)

Punt A, Smith ADM (2001) The Gospel of maximum sustainable yield in fisheries management: birth, crucifixion and reincarnation. In: Reynolds JD, Mace GM, Redford KH, Robinsson HG (eds) Conservation of exploited species. Cambridge University Press, Cambridge, pp 41–66

Ramos S, Vázquez-Rowe I, Artetxe I, Moreira MT, Feijoo G, Zufía J (2011) Environmental assessment of the Atlantic mackerel (Scomber scombrus) season in the Basque Country. Increasing the timeline delimitation in fishery LCA studies. Int J Life Cycle Assess 16:599–610

Ramos S, Vázquez-Rowe I, Artetxe I, Moreira MT, Feijoo G, Zufía J (2014) Operational Efficiency and Environmental Impact Fluctuations of the Basque Trawling Fleet Using LCA+DEA Methodology. Turk J Fish Aquat Sc 14:77–90

Reap J, Roman F, Duncan S, Bras B (2008a) A survey of unresolved problems in life cycle assessment. Part 1: goal and scope and inventory analysis. Int J Life Cycle Assess 13:290–300

Reap J, Roman F, Duncan S, Bras B (2008b) A survey of unresolved problems in life cycle assessment. Part 2: impact assessment and interpretation. Int J Life Cycle Assess 13:290–300

Rogers SI, Clarke KR, Reynolds JD (1999) The taxonomic distinctness of coastal bottom-dwelling fish communities of the North-east Atlantic. J Anim Ecol 68:769–782

Searchinger T, Heimlich R, Houghton RA, Dong F, Elobeid A, Fabiosa J, Tokgoz S, Hayes D, Yu TH (2008) Use of US croplands for biofuels increases greenhouse gases through emissions from land use change. Science 319:1238–1240

Svanes E, Vold M, Hanssen OJ (2011a) Environmental assessment of cod (*Gadus morhua*) from autoline fisheries. Int J Life Cycle Assess 16:611–624

Svanes E, Vold M, Hanssen OJ (2011b) Effect of different allocation methods on LCA results of products from wild-caught fish and on the use of such results. Int J Life Cycle Assess 16:512–521

Thrane M (2004a) Energy consumption in the Danish fishery: identification of key factors. J Ind Ecol 8:223–239

Thrane M (2004b) Environmental impacts from Danish fish products. Hot spots and environmental policies. Doctoral Thesis. University of Aalborg, Aalborg, Denmark

Tyedmers P, Watson R, Pauly D (2005) Fueling global fishing fleets. Ambio 34:635–638

Vázquez-Rowe I, Iribarren D, Moreira MT, Feijoo G (2010) Combined application of life cycle assessment and data envelopment analysis as a methodological approach for the assessment of fisheries. Int J Life Cycle Assess 15:272–283

Vázquez-Rowe I, Moreira MT, Feijoo G (2011a) Life cycle assessment of fresh Hake fillets captured by the Galician fleet in the Northern Stock. Fish Res 110:128–135

Vázquez-Rowe I, Iribarren D, Hospido A, Moreira MT, Feijoo G (2011b) Computation of operational and environmental benchmarks within selected Galician fishing fleets. J Ind Ecol 15:776–795

Vázquez-Rowe I, Hospido A, Moreira MT, Feijoo G (2012a) Review: best practices in life cycle Assessment implementation in fisheries. Improving and broadening environmental assessment for seafood production systems. Trends Food Sci Technol 28:116–131

Vázquez-Rowe I, Moreira MT, Feijoo G (2012b) Environmental assessment of frozen common octopus (*Octopus vulgaris*) captured by Spanish fishing vessels in the Mauritanian EEZ. Mar Policy 36:180–188

Vázquez-Rowe I (2012) Fishing for solutions: environmental and operational assessment of selected Galician fisheries and their products. PhD Thesis. University of Santiago de Compostela

Vázquez-Rowe, Villanueva-Rey P, Moreira MT, Feijoo G (2013a) Protein energy return on investment ratio (EROI) for Spanish seafood products. AMBIO, doi = 10.1007/s13280-013-0426-2

Vázquez-Rowe I, Rege S, Marvuglia M, Thénie J, Haurie A, Benetto E (2013b) Application of three independent consequential LCA approaches to the agricultural sector in Luxembourg. Int J Life Cycle Assess 18:1593–1604

Vázquez-Rowe I, Villanueva-Rey P, Mallo J, De la Cerda JJ, Moreira MT, Feijoo G (2013c) Carbon footprint of a multi-ingredient seafood product from a business-to business perspective. J Clean Prod 44:200–210

Vázquez-Rowe I, Villanueva-Rey P, Moreira MT, Feijoo G (2013d) The role of consumer purchase and post-purchase decision-making in sustainable seafood consumption. A Spanish case study using carbon footprinting. Food Policy 41:94–102

Vázquez-Rowe I, Marvuglia A, Rege S, Benetto E (2014a) Applying consequential LCA to support energy policy: land use change effects of bioenergy production. Sci Total Environ 472:78–89

Vázquez-Rowe I, Villanueva-Rey P, Hospido A, Moreira MT, Feijoo G (2014b) Life Cycle Assessment of European pilchard (*Sardina pilchardus*) consumption. A case study for Galicia (NW Spain). Sci Total Environ 475:48–60

Verdegaia (2010) As emissions de efecto invernaddoiro en Galicia (1990–2008). http://www.footprintstandards.org (in Galician)

Villasante S (2009) Magnitud e implicaciones de la política pesquera comunitaria: aplicación de indicadores de sostenibilidad sobre el metabolismo de los ecosistemas marinos. PhD Thesis. University of Santiago de Compostela (in Spanish)

Weidema BP, Thrane M, Christensen P, Schmidt J, Løkke S (2008) Carbon footprint—a catalyst ofr life cycle assessment? J Ind Ecol 12:3–6

Welch AA, Lund E, Amiano P, Dorronsoro M, Brustad M, Kumle M, Rodríguez M, Lasheras C, Janzon L, Jansson J, Luben R, Spencer EA, Overvad K, Tjønneland A, Clavel-Chapelon F, Linsein J, Klipstein-Grobusch K, Benetou V, Zavitsanos X, Tumino R, Galasso R, Bueno-de-Mesquita HB, Ocké MC, Charrondière UR, Slimani N (2002) Variability of fish consumption within the 10 European countries participating in the European Investigation into Cancer and Nutrition (EPIC) study. Public Health Nutrition 5(6B):1273–1285

Ziegler F, Nilsson P, Mattsson B, Walther Y (2003) Life cycle assessment of frozen cod fillets including fishery-specific environmental impacts. Int J Life Cycle Assess 8:39–47

Ziegler F, Emanuelsson A, Eichelsheim JL, Flysjö A, Ndiaye V, Thrane M (2011) Extended life cycle assessment of Southern pink shrimp products originating in Senegalese artisanal and industrial fisheries for export to Europe. J Ind Ecol 15:527–538

Ziegler F, Winther U, Skontorp-Hognes E, Emanuelsson A, Sund V, Ellingsen H (2013) The carbon footprint of Norwegian seafood products. J Ind Ecol 17:103–116

A Life Cycle Assessment Application: The Carbon Footprint of Beef in Flanders (Belgium)

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Abstract Although several international carbon footprint (CF) calculation initiatives have been developed, studies that focus specifically on estimating the CF of beef are rather scarce. This chapter describes the application of a CF methodology based on the lifecycle assessment of greenhouse gas emissions for Flemish beef production using the Publicly Available Specification methodology (PAS2050; BSI 2011), which is currently the most developed, profound, and relevant method for the agricultural and horticultural sectors. Both primary and secondary data were used to model the meat system by means of a chain approach. The results, which are reported using the functional unit of 1 kg deboned meat, range from 22.2 to 25.4 kg CO_2 eq/kg of deboned beef meat. A sensitivity analysis on changes in herd and feed characteristics was conducted. Results were compared to other studies on the CF of beef in the EU and other livestock produce. Three major hotspots in the CF were revealed: rumen fermentation, the composition and production of feed, and manure production and usage, which contribute a lot to the overall CF. The CF is a good indicator of greenhouse gas emissions; however, it is not an indicator of the overall environmental impact of a product. This chapter helps to fill the void in CF literature that existed around beef products and to define a benchmark for the CF.

Keywords Beef · Carbon footprint · Greenhouse gases · LCA · Hotspots · Sustainability

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S. S. Muthu (ed.), *Assessment of Carbon Footprint in Different Industrial Sectors, Volume 2*, EcoProduction, DOI: 10.1007/978-981-4585-75-0_2,

1 Introduction

Meat forms a huge part of the human diet in many European countries (van Wezemael 2011). However, the livestock production that is needed to produce meat leads to substantial greenhouse gas (GHG) emissions, causing climate change effects (Johnson et al. 2007). Livestock production causes half of all GHG emissions related to the European diet (Kramer et al. 1999; European Commission 2009). Achieving sustainable development can therefore be established by limiting agricultural GHG emissions in order to reach a stabilization of GHG emissions (Dalgaard et al. 2011).

Achieving sustainable production hence proves the need for evaluating the current situation and assessing where the production system needs improvements (Eriksson et al. 2005). If one wants to identify where along the production chain improvements can be made, it is necessary to quantify all emissions during the lifecycle. Carbon footprinting is one of the methods able to calculate the climate change impact of livestock products (Espinoza-Orias et al. 2011). A carbon footprint (CF) quantifies the climate change impact of an activity, product, or service. Within the CF, all GHG emissions (carbon dioxide [CO_2], methane [CH_4], and nitrous oxide [N_2O]) are combined. It is a measure of the total amount of GHG emissions of a system or activity, considering all relevant sources, sinks, and storage within the spatial and temporal boundary of the population, system, or activity of interest. A CF is calculated as carbon dioxide equivalent using the relevant 100-year global warming potential (GWP100) (Wright et al. 2011).

Given the importance of beef in terms of world consumption and livestock production, a study was ordered by the Flemish Government to calculate the CF of Flemish beef production in order to benchmark with other countries. Moreover, beef is an interesting case to examine for the reason that beef is increasingly imported from Latin America to Europe; in addition, estimations for the CF of beef are not readily available, especially when compared with carbon footprint studies on milk (Blonk et al. 2008b; Muller-Lindenlauf et al. 2010; Sonesson et al. 2009; Thoma et al. 2010; Van Der Werf et al. 2009).

Most studies on CF are not clear in terms of methodology or standards for either the chosen system boundaries or system definition. Stakeholders with different backgrounds and interests might draw incorrect conclusions. Indeed, different approaches in methodology prevent fair comparisons of carbon footprints between products and sectors, for the reason that different calculations are used; hence, one compares apples and oranges. A carbon footprint is calculated by means of a life cycle assessment (LCA) (Finkbeiner 2009). The fact that each LCA has to deal with many different issues (e.g., allocation method, scope, system boundaries, data, inclusion of land use change; Finkbeiner 2009) makes it necessary that each of these aspects be described in a proper way. In our own study on CF methodology applied to livestock produce in Flanders (2011), a literature study was conducted on the state of the art in terms of existing LCA or CF studies on pig production; we found that important information was missing in several cases (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.

2 Background

The results of this chapter were obtained through a study conducted for the Flemish government in the Department of Agriculture and Fisheries. The purpose was to estimate the CF and furthermore identify hotspots in the life cycle of beef, pig meat, and milk production in Flanders. However, this article focuses solely on beef production. In Flanders, the environmental pressure from livestock production abounds, with a major impact on climate change by large emissions of GHGs.

Flemish farmers hold a total of 262,280 beef cattle per year. The beef cattle are distributed over 5,544 farms (Statistics Belgium 2010). Approximately 80 % of the farms specialize in beef production. The remaining farms produce a combination of crops and beef. Given the fact that specialized farms abound, we opted to focus on this type of farms for data collection and monitoring.

3 Methodology

3.1 Standards and Methods Used

A carbon footprint quantifies the total amount of GHG emissions for which a product, organization, or product is responsible. It is a measure of the contribution of persons, products, and organizations on the greenhouse gas effect. Figure 1 presents the different steps that occur when calculating a carbon footprint.

An LCA is used as starting point. LCA is a method to determine the total environmental impact of a product during the whole chain or lifecycle of the product. Carbon footprinting differs from LCA in one aspect: it focuses solely on quantifying GHG emissions causing climate change. Determining the carbon footprint is hence a choice to focus on one environmental indicator.

A product CF comprises all emissions related to each phase of the product's lifecycle, from cradle to grave. In practice, the boundaries of carbon footprint calculations are often shortened. The choice of the system boundaries depends upon the goal and application.

For this study, we made use of the Intergovernmental Panel on Climate Change (IPCC 2006a) guidelines in line with the National Inventory Report of Belgium (VMM et al. 2011). Although the IPCC (2006b) directive gives a description of the calculation of the total amount of GHG emissions, it does not include the allocation of GHG emissions to a particular product. In order to tackle this, a specific

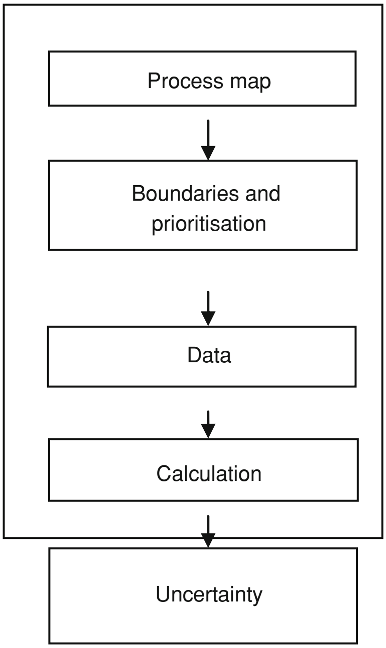

Fig. 1 The five necessary steps for calculating a carbon footprint (*Source* PAS 2050:2088; BSI)

methodology, such as the Publicly Available Specification (PAS) 2050, is needed (Espinoza-Orias et al. 2011). PAS2050 (BSI 2011) was chosen because it is one of the most profound methods available (among others; e.g. ISO 14067). In 2012, specific Product Category Rules (PCR) were developed according to the international Environmental Product Declaration (EPD) system (Environdec 2013) for mammal meat, including beef, in which slaughter activities, packaging processes, and storage are the core processes (Studio LCE 2012); hence, these were used as core activities in our 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.

3.2 Scope and System Boundaries

PAS2050 states that emission factors contributing <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 beef chain (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.

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 included
Animal	CH_4	IPCC method (Tier 2)
Manure storage and disposal	CH_4 and N_2O	IPCC method (Tier 2)
Manure application (not used for own feed mixtures)	CH_4 and N_2O	Allocation between animal (40 %) and vegetable production system (60 %) based on nitrogen uptake by plants
Energy and water consumption	CO_2, CH_4 and N_2O	Energy consumption (electricity, [red] diesel, gas); water consumption (tap and ground water)
Transport of goods	CO_2, CH_4 and N_2O	Assumptions made for the goods entering and leaving the farm
Processing materials	CO_2, refrigerant	Cleansing products, refrigerants

The study included the GHG emissions shown in Fig. 2. Production of materials, energy, and transport steps are included. The system boundary excludes production of capital goods, similar to most international studies.

3.3 Functional Units

Several functional units were defined upon agreement of the guiding committee of the project. This allowed better identification of the hot spots along the beef production chain. The functional units used were 1 kg of live beef meat, 1 kg of beef after slaughtering, and 1 kg of deboned beef meat.

3.4 Allocation Method

Another very important assumption describes the allocation of GHG emissions between the various byproducts emerging from a process. There are two major ways of allocation: physical and economic allocation. With regards to the project, a combination of both methods was applied, depending upon the chain stage. In terms of the slaughtering and deboning process, economic allocation was used: the economic value of the byproducts (bones, fat, skin, hide, heart, blood, etc.) represents the market prices multiplied by the mass fraction per incoming product (if it is a cost, then allocation share is zero). Manure contributes to the production of crops; therefore, physical allocation was used in order to allocate the GHGs from manure among crops and animal production. Overall, a combination of physical and economic allocation based upon several other references was used (Blonk et al. 2008a).

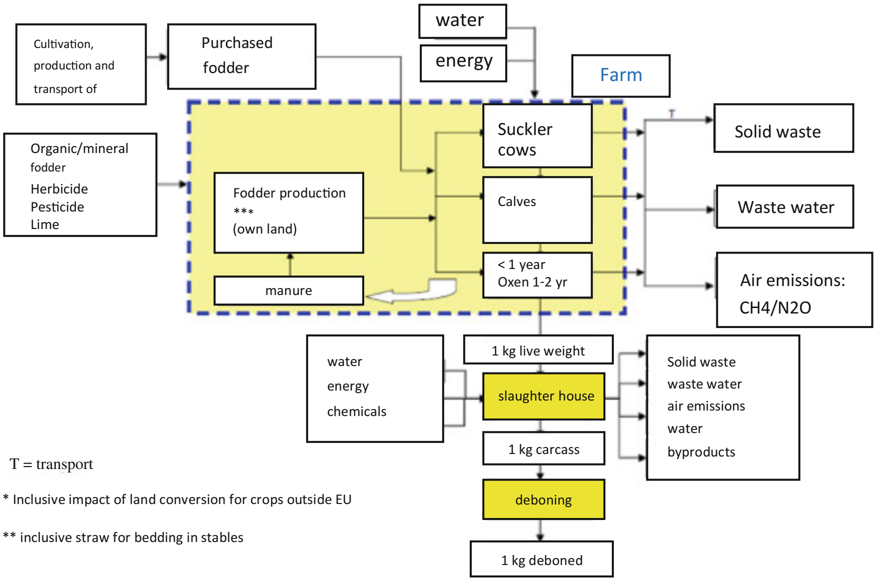

Fig. 2 The system boundaries. The boxes with *dashed lines* present a process; *solid lines* indicate a product flow. The *colored boxes* are the foreground system

3.5 Land Use and Land Use Change

According to the PAS2050 methodology, land use change (LUC) should be considered if land conversion took place in the last 20 years. This is not the case for agriculture in the EU; thus, LUC is zero. However, part of the feed is imported overseas; for those, FAO (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 (Blonk et al. 2008a; Ecoinvent 2011; 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, it was not included. International standards and guidelines for carbon footprinting also exclude it from the necessary calculations (ERM 2010).

4 Data Sources

PAS2050 has specific rules for using primary data over secondary data (BSI 2011). Primary data were complemented with secondary data and reports from umbrella organizations. Data were collected for 2009. However, certain data required more recent values, such as the feed compound composition, which changes daily.

Table 2 Yearly consumption of feed compound per beef cattle farm

Resources	Product (kg)	Yield[a] (kg product/ha)
Soy meal[b]	1,300	
Sugar beet pulp (dry)	910	
Sugar beet pulp (wet)	72.800	
Wet byproducts	27.430	
Milk powder	845	
Single forage	91	
Composite forage	69.680	
Grass silage and fresh grass (homegrown)	1.044.500	22.389[c]
Maize (homegrown)	359.800	48.670
Fodder beets (homegrown)	1.764	98.470

[a] Homegrown
[b] Origin of imported soy: 53 % Brazil, 11 % Argentina, 21 % United States, and 16 % Canada
[c] Weighted average meadow/temporary grassland

4.1 Raw Materials and Farm Level

The data on the representative conventional farm were collected through the Farmers Union dataset (Boerenbond 2011). This dataset was used to select the farms specializing in beef production. Flanders has 4,334 specialized beef cattle farms (NIR Belgium).

The database allowed identification of the average of the data to obtain a representative and existing farm. Outliers in the data were not used. Hence, the average represents a real farm with the following characteristics:

- 53 calves <12 months,
- 47 young cattle 1–2 years,
- 65 suckler cows.

Farms are confronted with the loss of animals. The mortality rate amounts to 1.25 % for mature animals and 11 % for calves. The replacement rate amounts to 34 %.

Data on manure production were collected from reports of the Flemish Centre for Manure processing and were linked to the number of animals on the farm (VLM 2011).

The fodder applied partly originates from the farm's own production and is partly purchased. Table 2 presents the overview per farm. The average composition of the feed concentrate was given by the Belgian Feed Compound Union (BEMEFA; personal communications, April–October 2011). The composition of the feed compound was given for October 4, 2010, which was randomly chosen during the course of the project. It varies daily according to the availability of components on the market. One can be sure that the feed used has an appropriate composition for the animals.

Emission factors were derived from BlonkMilieuadvies (Blonk et al. 2008a); Nielsen et al. (2010); and Ecoinvent (2011).

A suckler cow consumes the following raw feed components (homegrown) per year: 8.053 kg of grass, 7.697 kg of feed corn, and 636 kg of other raw feed components (mainly fodder beets). This is extended with purchased fodder consisting of 1.120 wet sugar beet pulp, 1.072 kg composite feed concentrate (see Table 2), 422 kg wet byproducts, 140 kg single forage, 20 kg soy meal, and 14 kg sugar beet pulp per suckler cow per year, as well as 13 kg milk powder.

The animals are kept in stables on a bed of homegrown straw. Suckler cows and female young cattle (1–2 years) stay outside for 24 h a day, during a period of 6 months/year. Female calves remain outside for 24 h a day, during a period of about 4 months. The male young cattle and the male calves stay inside. The animals produce 623 kg of manure per day. Approximately 32 % of this manure ends on the grassland during grazing, 60 % is preserved as stable, and 8 % as mixed manure.

The farmer possesses 48 ha of grass and cropland: 22.6 ha grassland, 10.3 ha of maize, 0.7 ha temporary grassland, 0.5 other roughage, and 13.9 ha of wheat. Fertilizer use consists of 650 kg of fertilizer per hectare of grassland (with 170 units of nitrogen) and 100 kg of starter fertilizer per hectare of maize (20 units of nitrogen and 2 units of phosphorus). The farm uses on average 2.65 kg of herbicides and 500 kg of lime, both per hectare.

The farm annually consumes 8.637 kWh of electricity and 8.054 L of oil fuel. In terms of water consumption, the farm consumes 773 m^3 of ground and 243 m^3 of tap water.

4.2 *Meat Processing*

The contacted slaughterhouses ($N = 4$) represented 33 % (weight) of processed beef in Flanders and hence were representative of the whole sector. Data were collected on meat weight and prices, byproducts and carcass, energy consumption, and transportation characteristics.

Missing data, such as the price of cuts and amount of waste/meat generated through slaughtering, were given by the Flemish Meat Federation (Febev).

5 Data Analysis

5.1 *Emissions from Fodder Production*

A distinction was made between homegrown and purchased fodder. The production of fodder also comprises the production and transportation of resources to sow, grow, and harvest the crops (seeds, fertilizers, pesticides, and diesel). The accompanying emissions are allocated to the crops. Land use during cultivation of

Table 3 Composition of feed concentrate for beef cattle. Allmash 16 is a commercial feed compound name

Resources	Beef cattle ALLMASH 16 (share in %)
Barley	12.5
Soy meal	5
Maize yellow from France	5.9
Maize gluten feed	22.5
Sugarbeet pulp	20
Linseed flakes	12.5
Rapeseed flakes	8.3

Source BEMEFA

the crops results in extra GHG emissions. Laughing gas is the most important greenhouse gas for land use.

For purchased fodder, crops are being transported to a processing plant. Emissions accompanying transport and processing are included. A 50/50 ratio for male and female for the young cattle and calves population is assumed.

5.1.1 Purchased Fodder

The resources of composed concentrate are processed to fodder consisting of different components. The BEMEFA was contacted to identify the composition. Databases were consulted on August 4, 2011; furthermore, feed specialists were consulted. Table 3 indicates the representative composition of approximately 80 % of feed concentrate for beef cattle.

The applied emission factors are derived from literature (Blonk Milieu Advies, University Wageningen). The available data were extended with other sources: the Ecoinvent database, LCA food database, and Carbon Trust. Table 4 presents the emission factors for the purchased fodder.

5.1.2 Homegrown Roughage

Farm land is applicable as grassland and moreover for the cultivation of fodder crops. The yield of the own crops is used as roughage. Table 5 presents the calculated emissions.

Energy and fuels used for machinery and transportation are included for the total energy consumption of the farm. They are not mentioned in Table 5. GHG emissions accompanying production and transportation of fertilizers, herbicides, insecticides, and fungicides are also included. Emission factors were calculated from the Eco-invent database. Emissions due to the application of these substances are mentioned in Table 6.

Table 4 Emissions accompanying purchased fodder per kilogram of product

Resources	kg CO_2 eq/kg product	Land use change (%)
Soy meal	3.06	71
Sugar beet pulp (dry)	0.11	
Sugar beet pulp (wet)	0.03	
Wet byproducts	0.03	
Milk powder	7.9	
Single forage	0.30	
Composite forage	0.42	19.8

Table 5 Emissions accompanying the cultivation of own crops

Resources	Area (ha)	kg CO_2 eq/year
Wheat (homegrown) for straw	13.9	17.659
Maize (homegrown)	10.3	32.150
Grass silage	23.3	23.9408
Fodder beets (homegrown)	0.5	4.989

Table 6 Emission factors: production, transportation of fertilizers, herbicides, and lime

Name	Value	Unit
Fertilizer (calcium ammonium nitrate)	8.81	kg CO_2 eq/kg N
Herbicide	10.730	kg CO_2 eq/kg
Lime (calcium carbonate)	0.02	kg CO_2 eq/kg

Laughing Gas Emissions

N_2O emissions due to crop cultivation are calculated according the IPCC 2006a, b, c method. Nitrogen sources applied to land are in this case fertilizers, natural fertilizers, and crop residue.

Direct laughing gas emissions are the result of denitrification. It is assumed that 1 % of all nitrogen applied to land converts to laughing gas (uncertainty interval 0.3–3 %). The IPCC value (1.25 %) was still applied in the national inventories until 2013. Current research in Flanders points out that 3.16 % of all nitrogen converts to N_2O. *Indirect* laughing gas emissions due to nitrogen leaching are calculated with data recorded in the National Inventory Report for greenhouse gases (NIR) of Belgium (2009). The amount of nitrogen leaching was determined with the Systems for the Evaluation of Nutrient Transport to Water model. In Flanders, 9 % of all applied nitrogen is leaching (NIR 2010, H6, p. 120). Of this, 0.75 % is finally converted to N_2O (IPCC 2006a, b, c). Indirect N_2O emissions due to nitrogen evaporation as ammonia (NH_3) and NO_x are calculated with the same data from the NIR Belgium (2009). The amount of evaporated nitrogen as NH_3 or NO_x, depends upon the nitrogen source:

Table 7 Emission factors electricity and gasoline oil

Name	Value	Unit	Source
Electricity	0.40	kg CO_2 eq/kWh	Energy covenant
Gasoline oil	2.66	kg CO_2 eq/kg	Energy covenant

(1) *Fertilizer* in Flanders: average NH_3 evaporation amounts to 3.3 % and the NO_x evaporation amounts to 1.5 % (NIR 2010).
(2) *Organic fertilizers* in Flanders: average nitrogen evaporation as NH_3 or NO_x amounts to 20 % (NIR 2009).

According to the IPCC calculation method, 1 % (0.2–5 %) of evaporated nitrogen (as NH_3 or NO_x) is converted to N_2O.
Lime Application
Lime is applied on land to increase the soil pH. This causes CO_2 emissions. For beef cattle farms, the total amount of lime applied per year is 1000 kg. The used emission factor is 0.48 kg CO_2 eq/kg lime (e.g. dolomite).

5.2 *Emissions from Cattle Breeding*

5.2.1 Energy Consumption Farm

The energy consumption is included as a whole and not allocated. In Table 7, emission factors are presented. Each suckler cow annually consumes about 1.512 kWh (10 % electricity and 90 % gasoline oil).

5.2.2 Animal Emissions: Rumen Fermentation

Emissions due to rumen fermentation are calculated based upon the IPCC guidelines (Tier 2 method). For calculating the necessary gross energy uptake (GE) per animal, the daily need, growth, and gestation is included. It is assumed that suckler cows produce a negligible amount of milk. The digestible energy (DE) is expressed as %GE. An adapted value is calculated based upon the fodder and the number of grazing days. It is calculated that approximately 169 MJ of GE is needed per suckler cow, 121 MJ for young cattle, and 82 MJ for calves. The digestible energy is calculated to be on average 74 %GE based upon the fodder. Table 8 represents the digestible energy per feed component. The time spent on grassland is included in order to determine an adapted digestible energy content of the animals' diet.

Approximately 6.5 % (weight basis) of the taken gross energy is converted to gas (methane) (IPCC 2006a). If the animals are fed more than 90 % with composite fodder, the above number can be lowered to 3 %. The taken gross energy is

Table 8 Digestible energy value for different types of fodder

Name	DE	Unit	Source
Wheat/barley	86	%GE	FAO
Maize/roughage	72	%GE	NIR Belgium
Soy meal	80	%GE	FAO
Beet pulp/citrus pulp	81	%GE	FAO
Wet byproducts	78	%GE	FAO
Composite fodder	80	%GE	NIR Belgium
Protein, vitamins	80	%GE	Proxy: composite fodder
Feed concentrate	80	%GE	Proxy: composite fodder
Composite young feed	80	%GE	Proxy: composite fodder
Fodder for young cattle	80	%GE	Proxy: composite fodder
Full milk	90	%GE	NIR Belgium
Grass silage	72	%GE	NIR Belgium
Fresh grass (grazing)	79	%GE	NIR Belgium

* Greenhouse gas emissions from the dairy sector: a Life Cycle Assessment 2010
Source FAO* and NIR Belgium

calculated based upon the fodder composition and the digestible energy content of each feed component.

5.3 Emissions Manure Storage and Usage

Manure production takes place on the meadow and in the barn. Suckler cows stay approximately 183 days/year on the meadow, female young cattle (between 1 and 2 years) also stay 183 days/year, and female calves (<1 year) stay approximately 122 days. Male young cattle and male calves are not put on the meadow. Manure produced in the stable is stored temporarily. It is assumed that 80 % of the manure production in the stable is being stored as stable manure. Manure disposal on grassland and manure storage are accompanied with methane and N_2O emissions. The calculations are explained below, based upon the IPCC 2006a, b, c guidelines (Tier 2).

5.3.1 Methane

Methane emissions related to manure production depend on the excreted volatile solids, the maximum methane production capacity of the manure, and the storage. The excreted volatile solids are calculated by using the IPCC (2006a) formula. Moreover, the IPCC 2006a, b, c reference value for the urine fraction (4 %) and dry matter content (8 %) were used (IPCC 2006b). Allocation of manure production between meadow and stable is presented in Table 9.

Table 9 Methane conversion factors and manure storage systems

Name	Stable manure (%)	Mixed manure (%)	Manure disposal on grassland (%)
Methane conversion factors	2	19	1
Manure suckler cows	40	10	50
Manure young cattle (1–2 years)	60	15	25
Manure calves (<1 year)	83		17

Source IPCC, NIR Belgium, Farmer's union

Table 10 N_{ex} per type of animal

Animal category	N_{ex} (kg/head.yr)
Calves (<1 year)	33
Young cattle (1–2 year)	58
Suckler cows	65

Source NIR Belgium/manure database

5.3.2 Laughing Gas

Through a combination of nitrification and denitrification, N_2O is released from stored manure or was disposed on land. The amount of produced laughing gas emissions depends upon the nitrogen excretion of the animals (N_{ex}). The excreted nitrogen per type of animal is taken from the NIR report of Belgium (Table 6.12).

The amount of N_2O from the total amount of nitrogen depends upon the manure storage. It is assumed that 0.5 % of total nitrogen is converted to N_2O during manure storage. For mixed manure stored underneath the slatted floor, it is assumed 0.1 % of the total nitrogen is converted to N_2O during storage. For manure disposed on the meadow by the animals, a 2 % conversion to N_2O is assumed (*direct emissions*) (Table 10).

Indirectly, there are N_2O emissions formed through volatilized NH_3 and NO_x. The amount of NH_3 and NO_x formed from the manure depends upon storage. Table 11 presents how much of the total nitrogen converts to NH_3 and NO_x. It is assumed that 1 % of indirect nitrogen losses converts to laughing gas (*indirect laughing gas emissions*).

5.3.3 Manure Usage for Crop Production

When manure is used on agricultural land for growing crops, emissions are allocated among crops and livestock. All produced manure is disposed of on the farm's own land. Accompanying emissions are described in Sect. 5.1.

Table 11 Nitrogen losses from manure as NH_3 or NO_x as a function of manure storage systems (IPCC 2006a, b, c)

Name	Nitrogen volatilization (NH_3/NO_x) (%)
Beef cattle—stable manure (fixed manure)	45
Beef cattle—mixed manure storage	40
Beef cattle—manure disposal on grassland	20

5.4 Emissions from Transport

Feed components are transported to the processing plant. Distances are limited within Europe. The soy component is transported overseas. Emissions related to both types of components are included in the applied emission factors and covered by the production of purchased fodder. For homegrown roughage, the necessary amount of fuel in the total energy consumption (Sect. 5.2.1) is included.

Feed is transported from the fodder processing plant to the farm (average distance of 30 km). The related emissions are covered within the farm data.

Cattle are transported from farm to slaughterhouse (average distance amounts 25 km).

5.5 Emissions from Meat Processing

Emissions originating from slaughtering are related to electricity consumption, fuel, cleansing products, water usage, and waste processing. Other transport methods are included as well. Emission factors are derived from Table 7 and the Ecoinvent (2011).

Furthermore, emissions are allocated to meat and other useful byproducts. Economic allocation is used at the slaughtering and deboning phase. The carcass yield is 67 % and the meat yield on carcass is 81 % for the Belgian White-Blue race.

6 Results

6.1 The CF of Beef

Results are presented in Fig. 3. In summary, 1 kg of deboned beef meat creates a *CF* of 22.2 kg CO_2 eq. Rumen fermentation, homegrown crops cultivation, and manure production and usage have the lion's share in the overall CF. The slaughtering process contributes 0.01 % of the total CF.

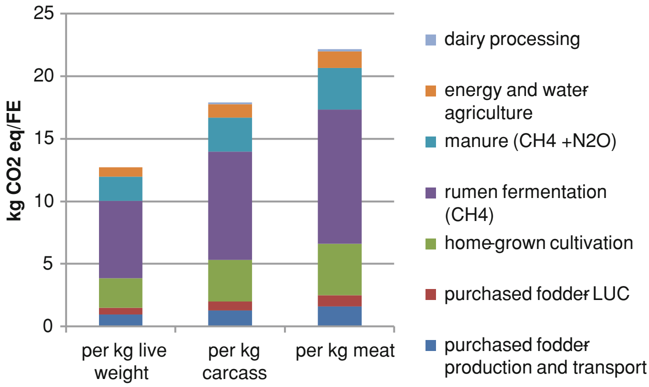

Fig. 3 The carbon footprint of 1 kg of beef in Flanders

The carbon footprint of live weight is 12.7 kg CO_2 equivalents. At the farm level, rumen fermentation represents 48.8 % of the carbon footprint, strongly determined by the feed uptake and digestibility. Feed consumption is estimated with available data; however, the reference values for digestibility are accompanied with a high uncertainty.

Fodder production is responsible for 29.5 % of the CF. Although only 13 % of the fodder is purchased, the impact is only 50 % compared to homegrown crop cultivation. The total impact of LUC contributes for 4 %. Manure storage and application on grassland contribute 15.3 % of the emissions (19 % is due to methane and 81 % due to N_2O emissions). Energy represents 5.9 % of this impact, electricity consumption represents 13.8 %, gasoline oil is 86 %, and water is 0.2 %.

At the slaughterhouse and deboning level, an extra 0.15 kg CO_2 eq/kg of carcass is added. The largest contribution (52.7 %) relates to waste management of the byproducts. Energy consumption contributes 37.5 %. Of this, 75 % is due to electricity consumption. The remaining 25 % is due to the combustion of fossil fuels. Furthermore, animal transport between the farm and slaughterhouse is 9.7 %. The production of process materials is negligible.

6.2 *CF Sensitivity*

The single outcome of CF calculations should be used with caution. A range of figures in which the CF is expected to be provides a more realistic insight (Flysjo et al. 2011b). Therefore, a sensitivity analysis[1] is conducted to define fluctuations.

[1] 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.

Table 12 Applied sensitivity analysis: impact of changes in herd and feed concentrate parameters on the CF

Parameter	Initial value	Min	Shift in CF	Max	Shift in CF
Mortality rate of animals <1 year	11 %	5 %	−0.6	15 %	+0.50
Moratlity rate of animals >1 year	1.25 %	0.5 %	−0.05	3 %	+0.1
Calving interval	365	–[a]	–	420	+2
Final weight bull/cow	680/690	660/670	+0.25	700/720	−0.3
Digestible energy content of fodder (%GE)	76 %	66 %	+1.9	86 %	−1.2

[a] The initial calving interval could not be lowered

6.2.1 Feed and Herd Characteristics

Table 12 presents trends in feed and herd characteristics and their possible impacts on the CF.

When the mortality rate for animals <1 year is decreased from 11 to 5 %, the CF decreases with 0.6 kg CO_2 eq/kg of deboned meat. Greater effects are identified for the rise in value for the GE from 76 to 86 %, leading to a CF fall of 1.2 kg CO_2 eq/kg of deboned meat and a decrease from 76 to 66 %GE, leading to CF rise of 1.9 kg CO_2 eq. Finally, a switch in the final weight also has an impact. Changing it from 680 to 700 decreases the CF with 0.3 kg CO_2 eq/kg of deboned meat, whereas a decrease in weight to 660 kg increases CF with 1.9 kg CO_2 eq.

6.2.2 Manure Storage/Disposal

An allocation between manure disposal on grassland (59 %) and in the barn is defined. For the latter, part of the manure is stored as barn manure (8 %) and the other as mixed manure (33 %). In total, three scenarios are considered. It is assumed that 100 % of the manure production is disposed on grassland (scenario 1), as barn manure (scenario 2), or stored as mixed manure (scenario 3) (Table 13).

The results are little influenced by these parameters. Storage as stable manure provides the least emissions. With extended time on grassland, animals consume more energy. The calculated gross energy is higher—hence, the rumen fermentation.

6.2.3 Influence of Allocation Method

Byproducts emerge in the slaughterhouse and deboning facility. Economic allocation was chosen because this method takes into account the value of the products. An alternative allocation method is based upon mass.

Table 13 Variation on parameters regarding manure storage

Parameter	Initial value	Scenario 1	Scenario 2	Scenario 3
Disposal grassland (%)	59	100	0	0
Stable manure (%)	8	0	100	0
Mixed manure (%)	33	0	0	100
Result (relative towards initial)	1	1.01	0.97	1.02

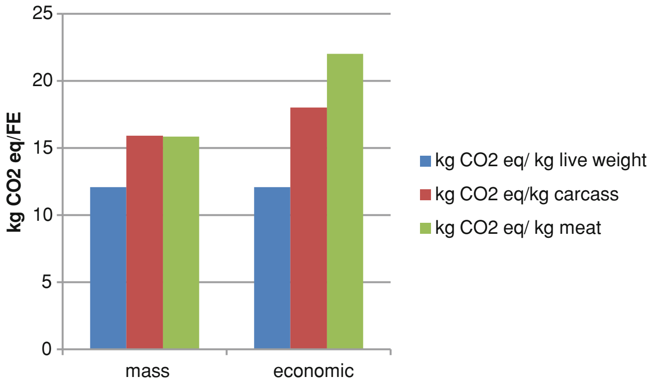

Fig. 4 Sensitivity of allocation method for beef meat

Figure 4 presents the impact of using different allocation methods on the overall CF.

The final calculated carbon footprint per kilogram of carcass and meat is significantly lower when one applies mass allocation. More emissions have to be allocated to other animal parts. It is therefore clear that there is no such thing as a single value for the CF. A range of values should always be given when the CF is reported, due to uncertainties and variations given the biological nature of the product.

6.2.4 CF Range

Based upon this sensitivity analysis, the overall estimated CF of beef production in Flanders falls between the range of 22.2 and 25.4 kg CO_2 eq/kg of deboned meat. For live weight, the carbon footprint ranges between 11.6 and 14.6 kg CO_2 eq/kg live weight; for carcass, the CF ranges between 16.3 and 20.5 kg CO_2 eq/kg carcass. Live weight obviously has the lowest carbon footprint because the deboning processes did not take place yet. The further down the beef production chain, the higher the carbon footprint.

Table 14 Comparison of the carbon footprint of Flemish beef with other international values

Study	Result (kg CO_2 eq)	Functional unit
Williams (2006)	16	1 kg carcass
Blonk et al. (2008a, b)	15.9	1 kg meat
Cederberg et al. (2009)	22.3	1 kg organic Swedish beef
Cederberg et al. (2009)	36.4	1 kg Japanese Kobe beef
Cederberg et al. (2009)	22	1 kg American beef
Own analysis	16.3–20.5	1 kg carcass
Own analysis	22.2–25.4	1 kg deboned meat

7 Discussion

7.1 Relative Importance

Comparing the CF of beef in Flanders with other international studies is not straightforward due to the different choices made. Table 14 makes a comparison.

Some authors report findings of a similar CF for beef, whereas others report a lower or higher CF. Williams (2006) reported a carbon footprint of 16 kg CO_2 eq/kg of carcass, which is a bit lower than the range being reported. Blonk et al. (2008a, b) studied the greenhouse gas emissions of meat (production). Their result is somewhat lower than our calculations. Cederberg et al. (2009) report somewhat higher CFs.

Next, the CF of beef can be compared with the CF for pigmeat and milk production. Within the same study, it is shown that the CF of pigmeat produced in Flanders lies between 3.1 and 6.4 CO_2 eq/kg of deboned pigmeat and of milk between 1.03 and 1.36 kg CO_2 eq/kg of milk consumed (1.5 % fat).

7.2 Mitigation Measures

Three huge hotspots in the production of beef meat were revealed: rumen fermentation, fodder production, manure production, and the usage of it. Some opportunities to reduce the CF of beef were defined.

In particular, the composition of feed has a very big impact on the overall CF. Within Europe, the use of soybean in feed concentrates has increased rapidly. However, the use of soy has a negative impact on the CF (negative LUC impact and transportation of feed components over long distances; Hortenhuber et al. 2011). Therefore, replacement with regional products can reduce the CF. When overseas products are to some extent indispensable, priority should be given to products produced in a sustainable way with a restricted impact on LUC. However, the composition of the feed depends more on availability, price, and the characteristics of the components. Price and availability are major economic factors influencing the final price of the feed and the possible usage by farmers. Hortenhuber et al.

(2011) clearly indicated that regional and local products are not always at people's disposal. Shifting production in Europe towards these alternatives might lead to LUC effects in Europe (Steinfeld et al. 2006). However, one cannot remove all carbon-negative components because this limits the economic sustainability of farming practices.

Therefore, parameters such as price, availability, and feed component characteristics need to be taken into account alongside the CF to ensure that meat production is not compromised in an effort to reduce the GHG emissions (Espinoza-Orias et al. 2011). This economic aspect is often neglected in other literature, as described by Verspecht et al. (2012). Manure production, storage, management, and usage is the second largest contributor to the overall CF. In Flanders, the most popular method of manure management involves separation of liquid and solid components of manure. The solid part gives rise to a similar quantity of nitrate emissions as the storage and use of untreated animal manure would do.

Both things exemplify the possible tradeoffs between dealing with GHG emissions and other aspects of sustainability, put in a larger perspective. Sustainability consists of three pillars: environmental protection, economic growth, and social equity. A mitigation measure only has a positive affect when all aspects lead to better or higher sustainability. Moreover, it is important to stress that the CF is a good indicator for GHG emissions as one environmental indicator, but it is not an indicator for environmental impact in general.

8 Conclusion

The CF of beef estimated in our study using the PAS 2050 methodology (BSI 2011) ranges from 22.2 to 25.4 kg CO_2 eq/kg of deboned beef meat. The main hotspots were found in rumen fermentation and fodder production, accounting for the greatest proportion of the total CF. Furthermore, manure management is another important hotspot in the production chain. These hotspots reveal where measures can be taken in order to decrease GHG emissions along the chain. Our study helps to fill the void in CF literature that existed around meat 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 beef in other regions, allowing better and fairer comparisons (Flysjo et al. 2011a) and hence assisting the definition of a benchmark for the CF. This in turn will stimulate the search for opportunities to reduce the CF 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; this study focused solely on the environmental impact of one indicator: climate change.

Acknowledgements This study was funded by the Flemish Administration, Department of Agriculture and Fisheries. The authors gratefully acknowledge this funding.

References

Blonk H, Kool A, Luske B (2008a) Environmental effects of Dutch consumption of protein-rich products, consequences of animal protein replacement anno 2008, (in Dutch). Blonk Milieuadvies, Gouda

Blonk H, Luske B, Dutilh C (2008b) Greenhouse gas emissions of meat—methodological issues and establishment of an information infrastructure. Blonk Milieuadvies, Gouda

Boerenbond (2011–2012) Corporate economic accounting (in Dutch [CD-ROM] (2009)). Leuven

British Standards Institute (BSI) (2011) PAS2050 specification for the assessment of the life cycle greenhouse gas emissions of goods and services. England

Campens V, Van Gijseghem D, Bas L, Van Vynckt I (2010) Climate and livestock (in Dutch), Department Agriculture and Fisheries, Division Monitoring and Study, Brussels.

Cederberg C, Sonesson U, Berglund M (2009) Greenhouse gas emissions in beef production. Report 4

Dalgaard R, Halberg N, Hermansen J (2007) Danish pork production: an environmental assessment. DJF Anim Sci 82:1–34

Dalgaard T, Olesen JE, Petersen SO, Petersen BM, Jorgensen U, Kristensen T, Hutchings NJ, Gyldenkaerne S, Hermansen JE (2011) Developments in greenhouse gas emissions and net energy use in Danish agriculture—how to achieve substantial CO(2) reductions? Environ Pollut 159(11):3193–3203

de Vries M, de Boer IJM (2010) Comparing environmental impacts for livestock products: a review of life cycle assessments. Livestock Sci 128:1–11

Ecoinvent (2011) Life cycle inventory (LCI) database [CD-ROM] (1998–2013). Swiss Centre for Life Cycle Inventories, St.-Gallen

Environdec (2013) The international environmental product declaration (EDP®) system—a communications tool for international markets. Stockholm

Eriksson IS, Elmquist H, Stern S, Nybrant T (2005) Environmental systems analysis of pig production—the impact of feed choice. Int J Life Cycle Assess 10(2):143–154

ERM (2010) Carbon footprint methodology applied to the Flemish livestock sector. ERM Issue, Brussels, p 184

Espinoza-Orias N, Stichnothe H, Azapagic A (2011) The carbon footprint of bread. Int J Life Cycle Assess 16(4):351–365

European Commission (2009) Adapting to climate change: the challenge for European agriculture and rural areas. Commission staff working document. Brussels

Eurostat (2013) Meat production and foreign trade (annual data). (Last update: 11 Oct 2012)

FAO (2010) Greenhouse gas emissions from the dairy sector, a life cycle assessment. Food and Agriculture Organization of the United Nations—Animal Production and Health Division. Rome, p 98

Finkbeiner M (2009) Carbon footprinting—opportunities and threats. Int J Life Cycle Assess 14(2):91–94

Flysjo A, Cederberg C, Henriksson M, Ledgard S (2011a) How does co-product handling affect the carbon footprint of milk? case study of milk production in New Zealand and Sweden. Int J Life Cycle Assess 16(5):420–430

Flysjo A, Henriksson M, Cederberg C, Ledgard S, Englund JE (2011b) The impact of various parameters on the carbon footprint of milk production in New Zealand and Sweden. Agric Syst 104(6):459–469

Hortenhuber SJ, Lindenthal T, Zollitsch W (2011) Reduction of greenhouse gas emissions from feed supply chains by utilizing regionally produced protein sources: the case of Austrian dairy production. J Sci Food Agric 91(6):1118–1127

IPCC (2006a) Intergovernmental Panel on Climate Change. Established by: United Nations Environmental Programme and World Meteorological Organization. Geneva. http://www.ipcc.ch/index.htm#.UOVJmrXLe_0. Accessed Jan–Aug 2011

IPCC (2006b) Chapter 10: emissions from livestock and manure management. In: 2006 IPCC guidelines for National Greenhouse Gas. Geneva. http://www.ipcc-nggip.iges.or.jp/public/2006gl/pdf/4_Volume4/V4_10_Ch10_Livestock.pdf. Accessed Jan–Aug 2011

IPCC (2006c) Chapter 11: emissions from managed soils, and CO2 emissions from lime and urea application. In: 2006 IPCC guidelines for National Greenhouse Gas. Geneva. http://www.ipcc-nggip.iges.or.jp/public/2006gl/pdf/4_Volume4/V4_11_Ch11_N2O&CO2.pdf. Accessed Jan–Aug 2011

IPCC (2007) Intergovernmental Panel on Climate Change. Established by: United Nations Environmental Programme and World Meteorological Organization. Geneva. http://www.ipcc.ch/index.htm#.UOVJmrXLe_0. Accessed July–Dec 2011

Johnson JMF, Franzluebbers AJ, Weyers SL, Reicosky DC (2007) Agricultural opportunities to mitigate greenhouse gas emissions. Environ Pollut 150(1):107–124

Kramer KJ, Moll HC, Nonhebel S (1999) Total greenhouse gas emissions related to the Dutch crop production system. Agric Ecosyst Environ 72(1):9–16

Leip A, Weiss F, Wassenaar T, Perez I, Fellmann,. Loudjani P, Tubiello F, Grandgirard D, Monni S, Biala K (2010) Evaluation of the livestock sector's contribution to the EU greenhouse gas emissions (GGELS)—final report. European Commission, Joint Research Centre

Muller-Lindenlauf M, Deittert C, Kopke U (2010) Assessment of environmental effects, animal welfare and milk quality among organic dairy farms. Livestock Sci 128(1–3):140–148

Nielsen P, Nielsen AM, Weidema BP, Frederiksen RH, Dalgaard R, Halberg N (2010) LCA food database. Faculty of Agricultural Sciences, Danish Institute for Fisheries Research, Denmark. http://www.lcafood.dk/. Accessed Jan–Aug 2012.

National Inventroy Report (NIR) Belgium (2009). Accessibale through website: http://unfccc.int/national_reports/annex_i_ghg_inventories

Sonesson U, Cederberg C, Berglund M (2009) Greenhouse gas emissions in milk production: decision support for climate certification. Klimatmarkning for mat, Sweden

Statistics Belgium (2010) Farm counting, (Landbouwtelling). Brussels

Steinfeld H, Gerber P, Wassenaar T, Castel V, Rosales Mand De Haan C (2006) Livestock's long shadow: environmental issues and options. FAO, Rome

Studio LCE (2012) UN CPC 2111. Product category rules for the assessment of the environmental performance of meat of mammals. Sept 11 2012. Contributed by: Siena University, COOP Italia, Assocarni

Thoma G, Popp J, Nutter D, Shonnard D, Ulrich R, Matlock M, Kim DS, Neiderman Z, Kemper N, East C, Adom F (2010) Regional analysis of greenhouse gas emissions from milk production practices in the United States. Seventh international conference on life cycle assessment in the agri-food Sector. Bari

Van Der Werf HMG, Kanyarushoki C, Corson MS (2009) An operational method for the evaluation of resource use and environmental impacts of dairy farms by life cycle assessment. J Environ Manage 90(11):3643–3652

van Wezemael L (2011) Consumer attitudes towards safety and health attributes of beef and beef technologies. Department of Agricultural Economics, Ghent University, Ghent

Verspecht A, Vandermeulen V, Ter Avest E, Van Huylenbroeck G (2012) Review of trade-offs and co-benefits from greenhouse gas mitigation measures in agricultural production. J Integr Environ Sci 9(1):147–157

VLM (2011) Manure bank dataset (in Dutch) [CD-ROM] (2009). Brussels

VMM, VITO, AWAC, IBGE-BIM, Federal Public Service of Health, Food chain safety and environment, IRCEL-CELINE, ECONOTEC (2011) Belgium's greenhouse gas inventory (1990–2008). National Inventory Report submitted under the United Nations Framework Convention on Climate Change and the Kyoto Protocol. Belgium. April 2010

Williams, AG, Audsley E, Sandars DL (2006) Determining the environmental burdens and resource use in the production of agricultural and horticultural commodities. Main Report. Defra Research Project IS0205. Cranfield University and Defra, Bedford. www.silsoe.cranfield.ac.uk, and www.defra.gov.uk

Wright L, Kemp S, Williams I (2011) 'Carbon footprinting': towards a universally accepted definition. Carbon Manage 2(1):61–72

Carbon Footprint and Energy Estimation of the Sugar Industry: An Indian Case Study

Varun and Manish Kumar Chauhan

Abstract The sugar industry plays a vital role in the world's economy. It also affects the environment directly and indirectly; hence, greenhouse gas (GHG) emission estimation and energy savings in the sugar industry are very important. To improve the efficiency of the plant and analyze the life cycle energy usage and emissions from the sugar industry, a complete Life Cycle Assessment (LCA) is needed. A major portion of sugar is produced from sugarcane. Sugar is produced as the main consumable product; molasses and bagasse are byproducts and filter cake is a waste product. These byproducts and waste products are used again for different purposes. The capacity of sugar plants in the present study is 12,000 tons of cane per day in a sugar mill, 60 MW in a cogeneration power plant, and 270 kL per day in a distillery. The LCA mainly focuses on primary energy usage and its externalities. In this chapter, energy usage and GHG emissions from the sugar industry were obtained through an economic input-output model. A comparative analysis of GHG emission has also been carried out using a process chain analysis approach.

Keywords LCA · Energy use · GHG emissions · Sugar industry · Economic input-output

Abbreviations

Descriptions

BOP	Balance of plant
Cd	Cadmium
CH_4	Methane

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S. S. Muthu (ed.), *Assessment of Carbon Footprint in Different Industrial Sectors, Volume 2*, EcoProduction, DOI: 10.1007/978-981-4585-75-0_3,

CMU	Carnegie Mellon University
CO	Carbon monoxide
CO_2	Carbon dioxide
COD	Chemical oxygen demand
CW	Circulating-water
EIO	Economic input-output
EOT	Electric overhead travelling crane
E&M	Electromechanical
FCS	Filtrate clarification system
GHG	Greenhouse gas
Gm	Gram
GWP	Global warming potential
HT	High tension
I-O	Input output
IPCC	International Panel on Climate Change
ISO	International standard organization
Kg	Kilogram
kJ	Kilojoules
klpd	Kiloliter per day
kWh	Kilowatt hour
LCA	Life cycle assessment
LCIA	Life cycle impact assessment
LT	Low tension
MCC	Motor control center
MJ	Megajoules
MS	Mild steel
Mt	Million tons
MW	Megawatts
NO_X	Nitrogen oxide
OC	Oliver Campbell
O&M	Operation and maintenance
PCA	Process chain analysis
PCC	Power control center
PPP	Purchase power parity
SO_2	Sulfur dioxide
T	Tons
TCD	Tons of cane per day
TJ	Terajoule
TSP	Total suspended particles
TSS	Total soluble salts
UASB	Upflow anaerobic sludge blanket
USDA	U.S. Department of Agriculture

VFD	Variable frequency drive
₹	Indian Rupees
$	US Dollar

Subscript

e	Electricity
eq	Equivalent

1 Introduction

Energy is an important input in the process of a nation's development. The need for energy security necessitates diversification of energy resources and the sources of their supply, as well as measures for conservation of energy. Future economic growth mainly depends on the long-term availability of energy that is affordable, accessible, and environmentally friendly.

Today, the world is facing major environmental problems, such as global warming, ozone layer depletion, and waste accumulation. Over the last few decades, the research indicates that the global climate is changing rapidly (IPCC 2001)—a change that will continue with time (Hulme et al. 2002). So, there is an urgent need to mitigate the undesirable problems arising from our modern way of life to save our environment and our planet.

India is blessed with the third largest coal supply in the world, but it has a lot of ash content and cannot be used indefinitely. The impact of the energy crisis is particularly felt in developing countries such as India, where an ever-increasing percentage of the national budget earmarked for development is diverted to the purchase of petroleum products. The oil embargo of 1973 triggered a worldwide search for alternative energy sources. So far, conventional sources of energy such as thermal, hydroelectric, and nuclear are the main sources of electricity generation.

Energy also produces a lot of waste and harmful gases that are emitted into the environment. These harmful gases contribute to greenhouse gas (GHG) emissions. Hence, the influence of the industrial sector on the environment cannot be ignored.

1.1 Energy Classification

Energy is one of the major inputs for the economic development of any country. In developing countries, the energy sector has a critical importance because of the huge investments that are required to meet ever-increasing energy needs. Energy can be classified based on the following criteria:

(a) Primary and secondary energy
(b) Commercial and noncommercial energy
(c) Renewable and nonrenewable energy

(a) Primary and secondary energy

Primary energy sources are those that are either found or stored in nature. Common primary energy sources are coal, oil, natural gas, and biomass (such as wood). Other primary energy sources include nuclear energy from radioactive substances, thermal energy stored in the earth's interior, and potential energy due to the earth's gravity (NEED 2011).

Secondary energy is a converted form of primary energy using an energy conversion process. It is a suitable form of energy that can also be used directly, such as coal, oil, or gas converted into steam and electricity or mineral oil into gasoline. Some energy sources also have nonenergy uses; for example, coal or natural gas can be used as feedstock in fertilizer plants.

(b) Commercial energy and noncommercial energy

The energy sources that are available in the market for a definite price are known as commercial energy. By far the most important forms of commercial energy are electricity, coal, and refined petroleum products. Commercial energy forms the basis of industrial, agricultural, transport, and commercial development in the modern world. In industrialized countries, commercialized fuels are predominant sources not only for economic production, but also for many household tasks of the general population (NEED 2011).

Energy sources that are not available in the commercial market for a price are classified as noncommercial energy. Noncommercial energy sources include fuels that are traditionally gathered rather than bought at a price, especially in rural households. These are also called traditional fuels. Noncommercial energy is often ignored in energy accounting (NEED 2011). Examples include firewood, cattle dung, agricultural waste, and solar energy for water heating, electricity generation, and drying grain, fish, and fruits; animal power for transport, threshing, lifting water for irrigation, and crushing sugarcane; and wind energy for lifting water and electricity generation.

(c) Renewable and Nonrenewable energy

Renewable energy is the energy obtained from sources that are essentially inexhaustible. Examples of renewable resources include wind power, solar power, geothermal energy, tidal power, and hydroelectric power. The most important feature of renewable energy is that it can be harnessed without the release of harmful pollutants. Nonrenewable energy includes conventional fossil fuels such as coal, oil, and gas, which are likely to deplete with time (NEED 2011).

1.2 Energy in Industry

With the growth of industrialization in India, demand for energy is increasing for manufacturing, commerce, and the transport sector. Even with regard to the primary sectors of the Indian economy—agriculture and allied sectors—demand for

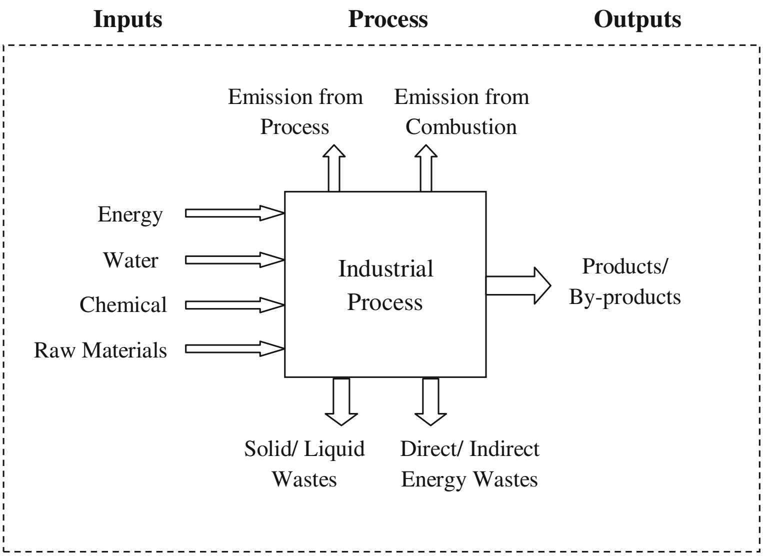


Fig. 1 Inputs and outputs in an industrial process (Bureau of Energy Efficiency 2011)

electricity and diesel has increased because of the increase of energy-intensive activities. The domestic energy demand for meeting fuel and lighting requirements has also increased during the past three decades. This is also due to the rapid increase in population and improvement in standards of living.

The industrial sector plays a vital role in the world economy. Industry accounts for more than one-third of all types of energy used in the world. Industries use a variety of highly energy-intensive processes, including steam, process heating, and motor-driven equipment such as air compressors, pumps, and fans. Industries have a lot of movable and high-power consumption parts that impact the environment. Thus, electricity and energy demand are very high in the industry market. Most of the energy that industry utilizes is supplied from a conventional electricity generation system (coal, oil, gas) (USDA 2011). Therefore, the reduction of electricity consumption is very essential to reduce environmental impacts.

Currently, energy crisis is a critical issue throughout the world. Generally, a lot of energy is consumed by industries, which also emit harmful pollutants in the environment. Day by day, energy resources are diminishing and GHG emissions are increasing in the world. Therefore, waste (air, water, etc.) utilization of industry has been a great concern. The usage of energy resources in industry leads to environmental damages by polluting the atmosphere. A few examples of air pollution are sulfur dioxide (SO_2), nitrous oxide (NO_X), and carbon monoxide (CO) emissions from boilers and furnaces and chlorofluorocarbon (CFC) emissions from refrigerants. Inputs, outputs, and emissions for a typical industrial process are shown in Fig. 1.

2 Sugar Industry Scenario

India is one of the leading producers of sugar in the world, producing approximately 20.8 million tons of sugar per year (ISMA 2013). Indian sugar industries play an important role in the growth of India's economy. India, Brazil, China, Thailand, and Pakistan are the top five sugar-producing countries (Foreign Agriculture Service 2012), accounting for nearly 40 % of the total worldwide. Sugar is being produced approximately in 115 countries in the world. Of these, 70 % of the countries produce sugar from sugarcane and 30 % by sugar beet, corn, cassava, etc. (Licht 2007; Contreras et al. 2009; Javalagi et al. 2010). Sugar industries are primarily based on sugarcane, however; in 2011–2012, approximately 1659 million metric tons of sugarcane was produced all over the world. Year by year, the demand for sugar is increasing in the market, so the production of sugarcane has been increased to fulfill the requirement.

Sugarcane is a form of chemical energy that is produced by conversion of solar energy. Sugarcane is a tall grass with large stems, growing mainly in tropical and subtropical countries (Renouf et al. 2010). In the past, sugar companies were producing only sugar, but now they are multitasking with the production of sugar, electricity, many types of fuels, many organic chemicals, a variety of papers, ethanol etc. These products are directly produced by sugar or its byproducts, so sugar industries are now called cane industries (Paturau 1989; ManoharRao 1997; Ramjeawon 2008).

An optimization of the system/industry can result in cost savings, reduced energy use, and less CO_2 emissions. Energy and environmental management tools (life cycle assessment, waste utilization, etc.) are very essential for improving the overall performance of the industries (Rajan 2001). There are various ways to improve the efficiency of sugar industry, such as increasing the calorific value of bagasse, reducing the process heat consumption in heating and evaporation of juice, reducing the power consumption of equipment, reducing mill bypass time, and automating processes (Ramjeawon 2004).

The amount of pollutants that is emitted by the industries into the environment is an important factor. Therefore, GHG emission analysis is necessary to determine the impact on the environment and humans. Reuse is better than recycling and recycling is better than single use (Baumann and Tillman 2004; Chauhan et al. 2011). Energy management is a method that saves valuable energy and also provides effective utilization of waste energy (Capehart et al. 1997). Waste energy utilization can help one to recognize the locations of high-intensity energy, which can also lead to improvement of the performance of industries. In sugar industries, energy savings in the form of steam and power are very necessary to face peak electricity demand in developing countries such as India, which are facing a severe shortage of electricity (Yarnal and Puranik 2009).

Bagasse and the molasses (byproducts) are used as input resources for the generation of the electricity in cogeneration plants and the production of ethanol in distilleries, respectively, as shown in Fig. 2. In the production of ethanol, the

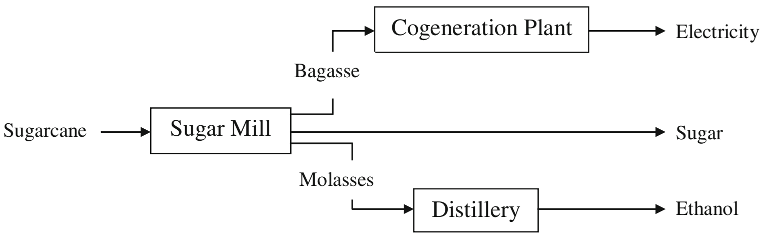


Fig. 2 Layout of the sugar industry (Dhampur Sugar Mills Limited Report 2011)

distillation process separates the ethanol and stillage or spent wash. Stillage is a waste product for a distillery, but it generates energy via advanced anaerobic digestion systems. An upflow anaerobic sludge blanket reactor is a type of anaerobic digestion system that is used to refine stillage into biogas (Nguyen and Gheewala 2008).

The sugar industry generally favors optimal utilization of waste produced. Sugar is mainly produced from sugarcane, which is mostly grown in tropical regions of the world. Bagasse and molasses are byproducts of the sugar industry, but they are used as a resource in other areas. Bagasse is used in cogeneration plants for generation of electricity and steam, which is used as an input resource in sugar mills, distilleries, cogeneration plants, and as a supply to grids for sale. During the off-season of the sugar industry, electricity is produced from other resources, such as coal, rice husk, and CH_4. In some countries, eucalyptus is used for electricity generation, so multifuel boilers may be used in cogeneration plants (Chauhan et al. 2011).

Molasses is also a byproduct that is used in distilleries for ethanol production. Distilleries produce ethanol as a product and stillage as a waste product. In an upflow anaerobic sludge blanket reactor, anaerobic digestion of stillage produces biogas as a resource and CH_4, which is used in cogeneration plants for electricity generation.

Filter cake/ash is a waste product of the sugar industry, which is used as fertilizer. The tops, leaves, and trash of sugarcane are also waste products of the sugar industry. These are used in mills to provide biomass feedstock. Sugar production requires some resources and produces a main product, by-products, and waste. Sugar production also emits some harmful gases and solid particles in the air and water, which directly affect the environment in terms of global warming, acidification, and eutrophication (Chauhan et al. 2011).

3 LCA

Life cycle assessment (LCA) is a technique for assessing the environmental aspects and potential impacts of a product. The concept of LCA was developed from the idea of a comprehensive environmental assessment of products, which began in Europe and the United States in the late 1960s and early 1970s (Boustead 1996). LCA studies should systematically and adequately address the environmental aspects of product systems, from raw material acquisition to its final disposal. The depth of detail and timeframe of an LCA study may vary to a large extent, depending on the defined goal and scope. In a full LCA, a certain procedure is followed that involves a number of steps, such as defining the goal and scope, drawing up inventory tables, and determining their impact assessment.

3.1 Types of LCAs

LCAs may be categorized into the following three types:

(a) Process LCA
(b) Input-output LCA
(c) Hybrid LCA

(a) Process LCA

Process LCA, also called process chain analysis (PCA), begins with the identification of one particular product as the object of study. This product may be either a good or a service. Then, the resources that are directly/indirectly required to produce the product are examined. When the list of such inputs is obtained, it is used to evaluate the total energy requirement and environmental emissions from this particular product. A process analysis requires extensive data on the production processes of the product that is selected for the study.

(b) Input-Output LCA

An alternative approach to process LCA (i.e. an LCA based on process modelling) is input-output (I-O) LCA. With I-O modelling, the product system that consists of supply chains is modelled using economic flow databases (input-output tables). These databases are collected and supplied by the statistical agencies of governments. They financially describe the amount that each industrial sector spends on the goods and services produced by other sectors. Emissions and associated impacts are then assigned to different sectors. I-O modelling provides greater comprehensiveness but also has certain limitations.

(c) Hybrid LCA

Hybrid LCA is a method in which both process LCA and input-output LCA have been combined. It was introduced in the early 1990s. Morriguchi et al. (1993) analyzed life cycle CO_2 emissions of an automobile by both process LCA and I-O analysis. The I-O model does not always guarantee a complete upstream system boundary, especially when the national economy, on which I-O table is based, relies on imports. Hondo et al. (1996) carried out process modelling for upstream processes that are not available in the Japanese I-O tables (e.g. coal mining) and used I-O analysis for the rest of the process.

3.2 LCA Methodology

LCA is a powerful tool for evaluating the possible impact of a product or system throughout its entire lifespan (birth to grave), from raw material acquisition, processing, manufacturing, use, and finally its disposal. LCA is a valuable tool to improve the environmental performance of the sugar industry in strategic decisions and to substantiate green energy claims (Ramjeawon 2004; Chauhan et al. 2011; ISO-1404 1997). It also acts as a tool to analyze the interactions between human activities and environment. It consists of four parts: definition of goal and scope, inventory analysis, impact assessment, and interpretation of results (Varun and Bhat 2008; Ometto et al. 2009). LCA is an important and comprehensive technique for analysis of the environmental impact of products/services. The principles of LCA are life cycle perspective, environmental focus, relative approach and functional unit, iterative approach, transparency, comprehensiveness, and priority of scientific approach.

In LCA analysis, assumptions, aim, scope, description of study area, methodologies, and output should be transparent (Varun and Prakash 2009; Varun and Bhat 2009); detailed methodology has been explained in ISO standards (ISO 1998, 2000a, b, 2006a, b). A methodological framework for the analysis of the environmental aspects of product life cycles consists of material and energy flow for the entire life cycle of a certain product. For this purpose, the product life cycle is divided into a number of processes; each process is described by the typical product input and output flow—that is, material input and output flow, water and air emissions, solid wastes, reusable materials, and so forth.

The increased awareness of the importance of energy in our society and the growing concern over future sources of energy have led to inquiries, such as how much energy is used in the production of goods and services. Primary energy is defined as the energy content of energy carriers that have not yet been subjected to any conversion. To enable aggregation of energy of different qualities, the different forms of energy need to be converted to the same energy form, which is called the primary energy equivalent. The conversion efficiency clearly differs between the different forms of used energy. An estimation of the energy use in the complete life cycle is the summation of the primary energy used in each phase of the system (construction, operation, and decommissioning).

Net energy has been defined as the amount of energy that remains for consumer use after the energy costs of finding, producing, upgrading, and delivering the energy have been paid (Huettner 1976). If a new technology consumes more energy than it produces, it has a negative net energy output, cannot provide any useful contribution to energy supplies, and should be dismissed as a net energy sink. Conversely, if a new energy technology can achieve a positive net energy output, then it should be adopted for use, even if the economic evaluation of its prospects is not found to be favorable in the case of energy scarcity (Mortimer 1991).

Table 1 Characterization factors for global warming potential (GWP) indicator calculation (Baumann and Tillman 2004; Intergovernmental Panel on Climate Change 2007)

Trace gas	GWP (kg-CO_{2eq}/kg)
CO_2	1
CH_4	25
N_2O	298
SF_6	22,800
CCl_4	1,400
CF_4	7,390
C_2F_6	12,200
C_3F_8	8,830
C_4F_{10}	8,860
$CBrF_3$	7,140
NF_3	17,200
1,1,1-trichloroethylene	110
CFC-11	4,750
CFC-12	10,900
CFC-13	14,400
CFC-113	6,130
CFC-114	10,000
HCFC-22	1,810
HCFC-123	77
HCFC-124	609
HFC-23	14,800
HFC-32	675
HFC-41	92
HFC-125	3,500
HFC-134	1,100
HFC-143	353

The combustion of coal, oil, and gas in thermal power plants emits mainly CO_2, SO_2, NO_x, CH_4, and airborne inorganic particulates, such as fly ash and suspended particulate matter (SPM). GHGs can be converted to their CO_2 equivalent (CO_{2eq}) using the characterization factors provided in Table 1 for a 100-year time horizon.

3.3 LCA of Sugar Industry

In the sugar industry, several harmful contents are emitted in the air and water. For 1 ton of sugar production at a Mauritius company, 0.002 kg CH_4, 1.7 kg total suspended particles, 1.21 kg SO_2, 1.26 kg NOx, 1.26 kg CO, and 160 kg CO_2 from fossil fuel use are emitted into air and 1.7 kg N_2, 19.1 kg chemical oxygen demand, and 13.1 kg total soluble salts are emitted into water (Ramjeawon 2004). In Nicaraguan sugar companies, when biomass is used in place of fuel oil, the emissions of CO_2 and SO_2 equivalent are 67 and 18 times lower (Broek et al. 2000). In the sugar industry in Mauritius, 0.27 tons of molasses and 591 kWh of electricity from bagasse are produced per 1 ton of sugar production (Ramjeawon

2004). In South Africa, 1 kg of sugar production produces byproducts such as 0.3 kg of molasses and 1.25 kg of fibrous residue (dry basis), known as bagasse (Botha and Blottnitz 2006). In Thailand, 1 ton of sugarcane produces 103.6 kg sugar and 45.2 kg molasses (Prasertsri 2007).

In Malaysia, electricity generation is mostly fossil-based, particularly natural gas and oil. Al-Amin et al. (2009) estimated the result of emissions from electricity production in the year 2020 on the basis of economic input-output (EIO) analysis. The emissions results are expected to be very high for CO_2 (800.52 m), SO_2 (3.84 m), and NOx (18.32 m) in comparison to the year 2000 in Malaysia. These types of energy I-O studies have been carried out by Hawdon and Casler (Hawdon and Pearson 1995; Casler and Wilbur 1984). In South Africa, 8.46 tons of sugarcane, 17,000 m^3 of water, 0.15 ha of land, and 71 kg of coal are required for 1 ton of raw sugar production. The 0.56 tons of filter cake, 0.38 tons of molasses, 2.4 tons of bagasse (with 50 % moisture content), and 368 kg of ash and slugs are also emitted for 1 ton of raw sugar production. Consumption of fossil energy is very high in the South African sugar industry compared with the industries in Mauritius and Brazil (Mashoko et al. 2010).

Bagasse and rice husk also produce CO_2 and ash as waste products of the industry; the use of these materials in cogeneration plants helps to increase the efficiency of the plant (Jafara et al. 2008). If the pollution content is high, the emitted byproducts are very harmful for human health, animals, crops, and the environment. Due to the high concentration of pollutants, greenhouse and global warming effects are produced (Tekin and Bayramoglu 2001). The production of surplus electricity from biomass, bagasse, and trash as a fuel is very important for the reduction of process steam demand and CO_2 emissions in order to prevent global warming (Ensinas et al. 2007).

In this chapter, GHG emissions and energy usage estimation are presented for the Indian sugar industry using EIO and PCA approaches. GHG emissions contain several harmful gases, but CO_2 is the major contributor. The carbon footprint is the sum of all emissions of greenhouse gases, such as CO_2 and CH_4, which were included by activity for a given time period. The carbon footprint is basically the amount of equivalent CO_2 released into the atmosphere as a result of the activities of an organization, system, event, or product. No study has been carried out for the Indian sugar industry to evaluate the carbon footprint and its energy usage. Hence, in the present study, carbon footprint and energy usage estimation were carried out for the Indian sugar industry.

4 Description of Study

In the present study, the Dhampur sugar industry was considered, which consists of three plants: a sugar mill of 12,000 TCD, a cogeneration power plant of 60 MW, and a distillery of 270 klpd (Dhampur Sugar Mills Limited Report 2011). Today, sugar industries do not produce only sugar but also several byproducts, which are

Table 2 Input resources, byproducts, and emissions for 1 ton of sugar production (Dhampur Sugar Mills Limited Report 2011)

(A) Input resource	Quantity
Sugarcane	10 T
Processing water	12.4 m^3
Steam	4.1667 T
Electricity consumption	316 kWh
(B) Byproduct	
Bagasse	3.333 T
Molasses	5.416 T
Filter cake	0.400 T
(C) Emission type	
Waste water	7.6 m^3
Hazardous waste	0.03 kg
Ashes and slugs	330 kg

used as input resources for other plants; waste products are also used as fertilizer in agriculture. For optimized waste utilization and maximum sugar production, the sugar industry is a combination of three plants—the sugar mill, cogeneration power plant, and distillery, as shown in Fig. 2.

4.1 Input Resources, Output Byproducts, and Emitted Waste

According to plant capacity and sugar industry data, the input resources for 1 ton of sugar production are given in Table 2. Some byproducts are also produced and some pollutants are emitted in soil, air, and water. The quantities of byproducts and pollutant emissions per ton of sugar production from sugar industry are given in Table 2.

4.2 Assumptions

Some assumptions and considerations are necessary for the study and analysis of any product or system. Generally, the sugar industry operates for 150 days per year, and the lifespan of plant is taken as 20 years. The production of sugar, ethanol, electricity, and filter cake is based on a plant's capacity or crushing of sugarcane per day. The lifespan is divided into three parts: electromechanical equipment, civil works, and operation and maintenance (O&M) works. O&M works are 6 % of electromechanical and 3 % of civil works (Varun and Bhat 2008).

4.3 Functional Units

Generally, the functional unit is the basis of normalized input-output data and also represents the final result taken. The functional unit is an important parameter when analyzing a system to determine a specific reference for a carbon footprint study. The functional units are different for each plant of the sugar industry. Three functional units are considered in the present study: 1 ton of sugar, 1 kWh of electricity, and 1 L of ethanol for the sugar mill, cogeneration power plant, and distillery, respectively. These functional units help to analyze and compare the results from other sugar industries easily.

4.4 Study Criteria and Boundaries

LCA is also known as a cradle-to-grave analysis, which includes the extraction of raw materials, processing, manufacturing, fabrication, distribution, use, and disposal of the product throughout its useful life. On the basis of these activities, the analysis is divided into different study criteria as cradle to gate, gate to gate, cradle to cradle, and cradle to grave. Cradle-to-gate analysis deals with a partial product life cycle from resource extraction to the factory gate before delivery to the consumer. Gate-to-gate analysis considers the procurement of resource materials to final delivery of a product to the customer. Cradle-to-cradle analysis is similar to cradle-to-grave analysis, but including the recycling of the product (Global Development Research Center 2011).

In the present study, the production of sugar, bagasse, molasses, and electricity generation have been considered for analysis using the EIO approach to estimate life cycle energy usage and GHG emissions. GHG emissions from the sugar mill have also been compared by using EIO and PCA approaches. For EIO-LCA, the components of the sugar industry and its plants that have been considered are shown in Fig. 3. This chapter uses a gate-to-gate analysis from sugarcane crushing to sugar production, from bagasse burning to electricity production, and from molasses processing to ethanol production.

In the sugar industry, sugar is produced through several processes, such as cutting, milling, filtration, raw juice heating, sulfiting, clarification, clear juice heating, evaporation, syrup sulfiting, solidification, grading, and packaging (Dhampur Sugar Mills Limited Report 2011; Ensinas et al. 2007). In a comparative analysis of EIO and PCA approaches, the sugar mill has been considered from sugarcane crushing to sugar packaging, as shown in Fig. 4.

5 Solution Methodology

For the life cycle assessment, there are two primary approaches: the PCA technique and the EIO technique. In this study, all processes were investigated using an EIO approach. With the help of input-output modeling, a product or system that consists of supply chains is modeled using economic flow databases (input-output tables). In an EIO-based LCA approach, two types of matrices are used, which

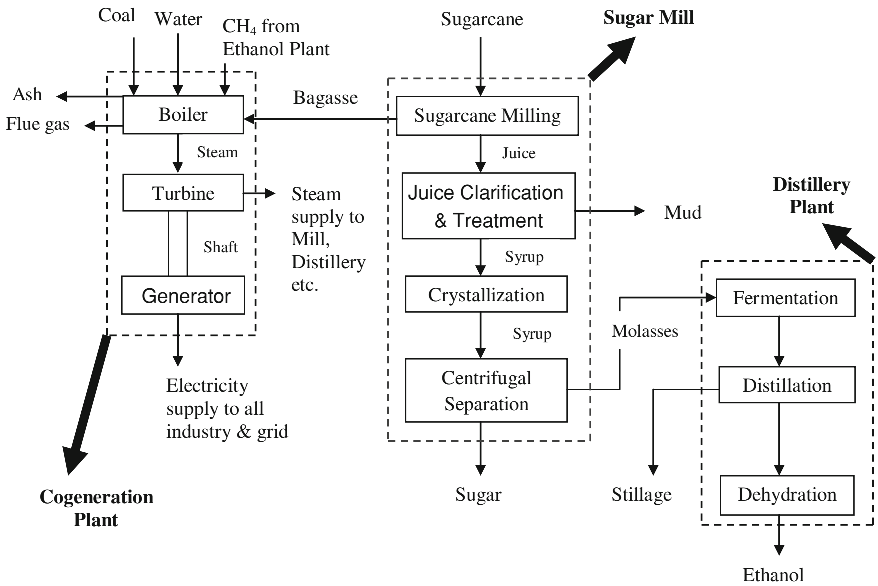


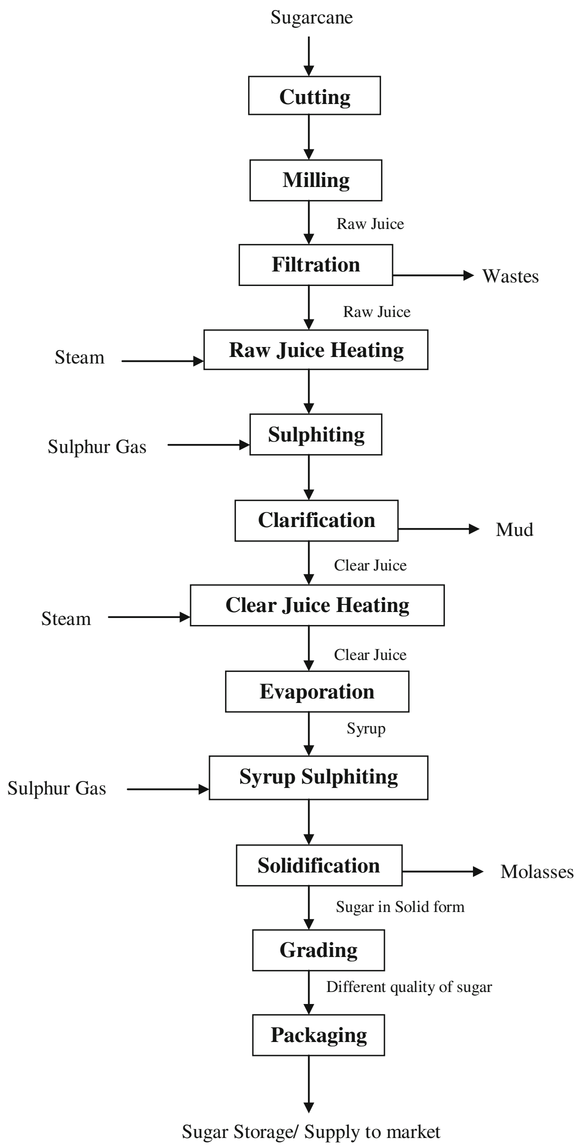
Fig. 3 Study criteria and schematic layout of the sugar industry (Chauhan et al. 2011)

consist of economic data and associated environmental coefficients (Bullard et al. 1978). These databases are collected by the statistical agencies of governments to explain the amount that each industrial sector spends on the goods and services that are produced by other sectors. EIO analysis gives some advantages, such as improved data accuracy, easier data handling, and excellent use of policy application (Chung et al. 2009).

Because no EIO-LCA model is available for India, the Carnegie Mellon EIO-LCA model (US Department of Commerce 2002; an industry benchmark) was used for this study after suitable modifications. The Carnegie Mellon University Green Design Institute developed this model based upon the US economy (CMU Green Design Institute 2008). The EIO-LCA model has several impact assessment factors (e.g. acidification potential, water usage); in the present study, it was used to determine the primary energy usage and GHG emissions of the manufacturing of major materials and equipment used in this particular industry.

The cost for different components that are used in the sugar industry were collected and inflated (using the inflation index) for 2004–2005, expressed in Indian Rupees (₹). The costs were converted into equivalent US dollars using the purchasing power parity (PPP) for that year (2004). The equivalent US dollar, as estimated using PPP, were adjusted for the year 2002 using the US inflation index. The cost corresponding to a component used in the sugar industry was converted into its equivalent US dollars for the year 2002 (Zhang et al. 2007; Varun and Bhat 2012). This equivalent cost was used in the EIO-LCA model.

Fig. 4 General layout of sugar mill processes (Chauhan MK 2011)



To use this model for Indian systems, the costs of industry equipment/materials were converted into equivalent US dollars ($) using PPP as follows:

$$\text{Equivalent cost in US \$ (2002)} = \frac{\text{Cost in Rs (2004)}}{\text{PPP in 2004}} \times \frac{\text{Inflation index for Year 2002 for US}}{\text{Inflation index for Year 2004 for US}} \quad (1)$$

Table 3 Inventory of energy usage and GHG emissions for electromechanical equipment in sugar mills

Components	Energy use (TJ)	GHG emissions (T-CO_{2eq})
Machinery and equipment for juice extraction plant with in-line shredder juice installation	1210.0	69308.914
Machinery and equipment for boiling house	391.0	24040.395
Machinery and equipment for centrifugal station	61.7	3909.966
Total	1662.7	97259.275

The PPP conversion factor is 9.2 for the year 2004. The inflation indexes for years 2002 and 2004 are 179.88 and 188.90, respectively. The total cost of the industry equipment/materials in US dollars is brought to the level of the year of the model (2002) by using the Consumer Price Index (inflation index). GHG emissions are normalized to an equivalent of CO_2 emissions based on International Panel on Climate Change (IPCC) 100-year global warming potential (GWP) (Baumann and Tillman 2004):

$$\text{GHG emissions} = \frac{\text{Total CO}_2 \text{ emissions throughout its life cycle } (\text{kg} - \text{CO}_{2\text{eq}})}{\text{Annual sugar production } (\text{kg/year}) \times \text{Life Time } (\text{years})}$$

6 Results Using the EIO Approach

6.1 Electromechanical Equipment

The components of electromechanical (EM) equipment are used in electrical and mechanical operations simultaneously. The total energy usage and GHG emissions for each type of equipment are obtained using the EIO-LCA model. The inventory of EM equipment used in the sugar industry, such as the sugar mill, cogeneration power plant, and distillery, are summarized in Tables 3, 4, 5 respectively. The inputs associated in all processes of sugar production, through the manufacturing of materials and equipment, were obtained from the sugar industry.

6.2 Civil Works

Civil works include all construction work in the sugar industry related to building establishment. The total energy usage and GHG emissions of sugar industry were obtained using the EIO-LCA model. The inventory of civil works used in the sugar industry with energy usage and GHG emissions are summarized in Table 6.

Table 4 Inventory of energy usage and GHG emissions for electromechanical equipment in cogeneration power plants

Components	Energy use (TJ)	GHG emissions (T-CO_{2eq})
Boiler	199.00	12791.305
Turbo generator	71.60	4490.564
Fuel, ash and dense phase ash handling system	22.00	1424.280
Variable frequency drive	5.54	341.101
Main cooling tower, circulating-water pumps and valves	13.80	835.517
Electric overhead travelling crane	2.37	154.221
Switchyard	4.97	295.742
Transformers	14.70	934.400
Low-tension panels: motor and power control centers	6.95	428.191
High-tension panels: motor and power control centers	6.81	420.026
Low-tension package, including lighting and earthing	7.00	415.491
Low-tension contract and bus duct	9.55	567.898
Cables	10.50	578.784
Distributed control system, balance of plant instrumentation	8.46	500.766
Piping and tanks	17.70	1197.484
Water treatment plant	8.31	505.302
Compressed air system	1.56	94.347
Ventilation system	1.90	118.841
Firefighting system	2.15	127.913
Total	414.87	26222.175

Table 5 Inventory of energy usage and GHG emissions for electromagnetic equipment in distilleries

Components	Energy use (TJ)	GHG emissions (T-CO_{2eq})
Fermentation equipment	70.10	4753.648
Multi-pressure vacuum distillation equipment	31.00	1850.657
Utility items (e.g. cooling towers, instrument air compressor, tubing, insulation)	7.59	459.943
Pump electrical instrumentation in storage section	5.59	333.844
Water treatment plant	4.27	259.455
Molasses bulk storage	44.40	3011.853
Storage section for alcohol along with receivers	12.70	863.640
Mild steel structure for fermentation, distillation, and alcohol storage section	31.90	2159.099
Biomethanation plant	58.00	3456.373
Total	265.55	17148.513

Table 6 Energy usage and GHG emissions for civil works in the sugar industry

Plant	Capacity	Energy use (TJ)	GHG emissions (T-CO_{2eq})
Sugar mill	12,000 TCD	69.3	4408.918
Cogeneration power plant	60 MW	65.0	4136.762
Distillery	270 klpd	94.0	598.742
Total		228.3	9144.422

Table 7 Energy usage and GHG emissions for operation and maintenance in the sugar industry

Plant	Capacity	Energy use (TJ)	GHG emissions (T-CO_{2eq})
Sugar mill	12,000 TCD	1010	65861.612
Cogeneration power plant	60 MW	488	31751.466
Distillery	270 klpd	350	22770.337
Total		1848	120383.415

6.3 Operation and Maintenance

O&M work is based on the annual maintenance cost and use of the machines, tools, etc. An estimation of the energy usage and GHG emissions from O&M in the sugar industry were obtained from the EIO-LCA model. Generally, O&M varies from 1 to 10 % of total cost, but a value of 6 % O&M was used in this chapter. Table 7 shows the annual energy usage and annual GHG emissions in operation and maintenance.

7 Results Using the PCA Approach

7.1 Primary Energy Usage

Every product and system requires primary energy, which is used in manufacturing and establishment. Primary energy usage varies according to a plant's capacity. The life cycle of sugar industry is divided into three parts: electromechanical, civil works, and O&M works. Therefore, energy usage had been evaluated for these life cycle steps in the sugar industry. Figure 5 shows the variation of primary energy use for civil, electromechanical, and O&M works share for the sugar mill, cogeneration power plant, and distillery.

For the sugar mill, cogeneration power plant, and distillery, the primary energy usage is 61, 43, and 38 % for electromechanical equipment; 2, 7, and 13 % for civil works; and 37, 50, and 49 % for O&M works, respectively. The total energy usages were 2742, 967.87, and 709.55 TJ for the sugar mill, cogeneration power plant, and distillery, respectively, as shown in Table 8.

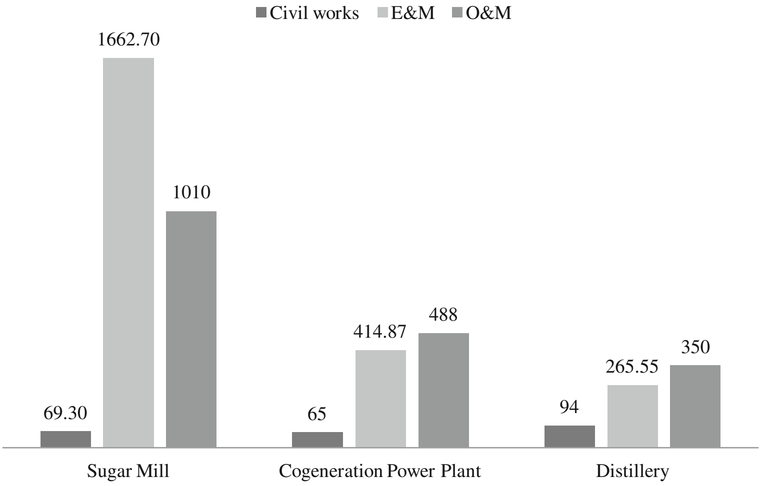


Fig. 5 Energy use in the sugar industry (values in TJ)

Table 8 Total life cycle energy usage and GHG emissions for the sugar industry

Components	Energy use (TJ)			GHG emissions (T-CO_{2eq})		
	Sugar mill	Cogeneration power plant	Distillery	Sugar mill	Cogeneration power plant	Distillery
Civil works	69.3	65.00	94.00	4408.918	4136.762	598.742
Electromechanical	1662.7	414.87	265.55	97259.276	26222.175	17148.513
O&M[a]	1010.0	488.00	350.00	65861.612	31751.466	22770.337
Total	2742.0	967.87	709.55	167529.806	62110.403	40517.592

[a] For a 20-year life span

The primary energy usage for the sugar industry 76 were 1.667 MJ/T of sugar, 228.811 kJ/kWh$_e$ and 161.687 kJ/L of ethanol for sugar mill, cogeneration power plant, and distillery, respectively, as shown in Table 9.

7.2 GHG Emissions

GHG emissions are directly related to global warming and are calculated in terms of equivalent CO_2 (CO_{2eq}). Every system emits GHG in manufacturing, establishment, uses, and processing. GHG emissions are a critical environmental issue. GHG emissions from each life cycle stage of the sugar industry have been evaluated using the EIO methodology. The variation of GHG emissions for civil, electromechanical, and O&M works for the sugar mill, cogeneration power plant, and distillery are shown in Fig. 6.

Table 9 Energy usage and GHG emissions of the sugar industry per unit of production

Plant	Capacity	Energy use	GHG emissions
Sugar mill	12,000 TCD	761.667 MJ/T of sugar	46.536 kg-CO_{2eq}/T of sugar
Cogeneration power plant	60 MW	228.811 KJ/kWh$_e$	14.683 gm-CO_{2eq}/kWh$_e$
Distillery	270 klpd	161.687 KJ/l of ethanol	9.233 gm-CO_{2eq}/liter of ethanol

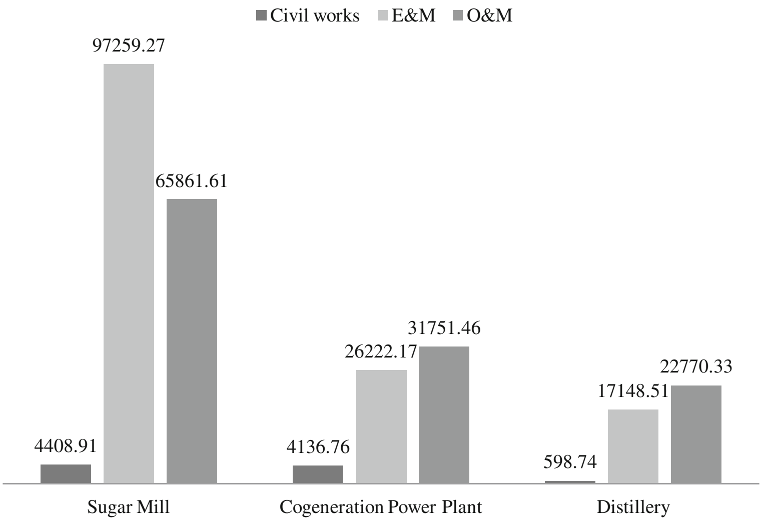


Fig. 6 Greenhouse gas emissions in the sugar industry (values in T-CO_{2eq}). E&M, electromechanical; O&M, operation and maintenance

GHG emissions from electromechanical equipment were 58, 42 and 42 %; from civil works were 3, 7 and 2 %; and from O&M works were 39, 51 and 56 % for the sugar mill, cogeneration plant, and distillery, respectively. The total GHG emissions were 167529.80, 62110.40, and 40517.59T-CO_{2eq} for the sugar mill, cogeneration power plant, and distillery, respectively, as shown in Table 8. The GHG emissions were 46.536 kg-CO_{2eq}/T of sugar, 14.683 gm-CO_{2eq}/kWh$_e$, and 9.233 gm-CO_{2eq}/L of ethanol for the sugar mill, cogeneration power plant, and distillery, respectively, as shown in Table 9.

8 Comparative Analysis of the Carbon Footprint of the Sugar Industry

The data collected from sugar industry has been used to estimate life cycle GHG emissions using two different LCA approach. The results obtained from both approaches were compared. GHG emissions mainly deal with the emission of

Table 10 GHG emissions (value in kg-CO_{2eq}) from sugar industry equipment by PCA

Equipment	Material	Embodied energy (MJ/kg)	Density (kg/m^3)	Quantity	Volume (m^3)	Weight (kg)	Total embodied energy (MJ)	GHG emissions (kg)	GHG emissions/ Ton of sugar
Roller mill 1–4	Steel	42	7,850	20	2.346	18415.543	15469056.26	1515967.514	0.421
Roller mill-5	Steel	42	7,850	7	3.218	25261.376	7426844.57	727830.768	0.202
Primary juice tank	Stainless steel	56.7	8,000	1	0.146	1171.469	66422.324	6509.388	0.001
Secondary juice tank	Stainless steel	56.7	8,000	1	0.146	1171.469	66422.324	6509.388	0.001
Mixed juice tank	Stainless Steel	56.7	8,000	1	0.284	2269.787	128696.928	12612.299	0.003
Juice heater 1–3	Stainless steel	56.7	8,000	3	2.013	16104.641	2739399.516	268461.153	0.074
Juice heater 4	Stainless steel	56.7	8,000	1	1.510	12080.368	684956.859	67125.772	0.019
Juice sulfiter	Stainless steel	56.7	8,000	2	0.296	2369.514	268702.906	26332.884	0.007
B.M.A. clarifier	Stainless Steel	56.7	8,000	1	9.907	79257.726	4493913.09	440403.483	0.122
Rapi clarifier	Stainless steel	56.7	8,000	1	10.776	86206.197	4887891.348	479013.352	0.133
Mud tank	Stainless steel	56.7	8,000	1	0.249	1990.835	112880.376	11062.277	0.003
Mud mixer	Stainless steel	56.7	8,000	1	0.272	2180.848	123654.0672	12118.098	0.003
OC vacuum filter 1–2	Stainless steel	56.7	8,000	2	2.0453	16362.264	1855480.694	181837.108	0.050
OC vacuum filter 3	Stainless steel	56.7	8,000	1	0.877	7017.131	397871.322	38991.389	0.010
OC vacuum filter 4	Stainless steel	56.7	8,000	1	1.391	11129.430	631038.710	61841.794	0.017
Buffer tank	Stainless steel	56.7	8,000	1	1.887	15098.573	856089.094	83896.731	0.023
FCS tank	Stainless steel	56.7	8,000	1	1.870	14964.251	848473.036	83150.357	0.0231
Clear juice tank	Stainless steel	56.7	8,000	1	0.585	4680.005	265356.268	26004.914	0.007
Clear juice heater	Stainless steel	56.7	8,000	1	1.510	12080.369	684956.859	67125.772	0.019
Evaporator K1	Stainless steel	56.7	8,000	1	9.431	75445.710	4277771.773	419221.633	0.116
Evaporator K2	Stainless steel	56.7	8,000	1	13.485	107878.691	6116721.784	599438.735	0.166
Evaporator K3	Stainless steel	56.7	8,000	1	14.962	119695.123	6786713.57	665097.930	0.185
Evaporator F1-F4	Stainless steel	56.7	8,000	4	9.431	75445.710	17111087.09	1676886.535	0.466
Evaporator F5	Stainless steel	56.7	8,000	1	8.205	65640.157	3721796.884	364736.095	0.101
Evaporator F6	Stainless steel	56.7	8,000	1	10.719	85753.983	4862250.886	476500.587	0.132

(continued)

Table 10 (continued)

Equipment	Material	Embodied energy (MJ/kg)	Density (kg/m^3)	Quantity	Volume (m^3)	Weight (kg)	Total embodied energy (MJ)	GHG emissions (kg)	GHG emissions/ Ton of sugar
Evaporator L1	Stainless steel	56.7	8,000	1	6.791	54325.815	3080273.693	301866.822	0.084
Evaporator L2	Stainless steel	56.7	8,000	1	2.681	21448.555	1216133.073	119181.041	0.033
Evaporator H	Stainless steel	56.7	8,000	1	3.742	29935.725	1697355.635	166340.852	0.046
Massecuite container A1, A2	Stainless steel	56.7	8,000	2	0.448	3581.953	406193.430	39806.956	0.011
Massecuite container A3	Stainless steel	56.7	8,000	1	0.399	3192.062	180989.908	17737.011	0.005
Massecuite container A4-A7	Stainless steel	56.7	8,000	4	0.482	3854.759	874259.268	85677.408	0.024
Massecuite container A8	Stainless steel	56.7	8,000	1	0.546	4372.198	247903.645	24294.557	0.007
Massecuite container A9	Stainless steel	56.7	8,000	1	0.606	4848.897	274932.490	26943.384	0.007
Massecuite container B1,B2	Stainless steel	56.7	8,000	2	0.448	3581.953	406193.430	39806.956	0.011
Massecuite container C1	Stainless steel	56.7	8,000	1	0.351	2810.026	159328.479	15614.191	0.004
Massecuite container C2	Stainless steel	56.7	8,000	1	0.482	3854.759	218564.8171	21419.352	0.006
Crystallizer (45T)	Stainless steel	56.7	8,000	15	0.351	2810.026	2389927.184	234212.864	0.065
Crystallizer (80T)	Stainless steel	56.7	8,000	3	0.482	3854.759	655694.451	64258.056	0.017
Centrifugal separator (1T)	Stainless steel	56.7	8,000	9	0.035	282.786	144305.712	14141.960	0.004
Centrifugal separator (1.5T)	Stainless steel	56.7	8,000	22	0.044	350.343	437017.612	42827.726	0.012
Centrifugal separator (2T)	Stainless steel	56.7	8,000	4	0.051	406.386	92168.351	9032.498	0.002
Bins	Stainless steel	56.7	8,000	4	0.450	3604.508	817502.341	80115.229	0.022
Grader G1	Stainless steel	56.7	8,000	2	0.348	2788.350	316198.953	30987.497	0.009
Grader G2	Stainless steel	56.7	8,000	1	0.450	3604.508	204375.585	20028.807	0.005
Flashing tower	Stainless steel	56.7	8,000	2	0.039	316.720	35916.035	3519.771	0.001
Hopper	Steel	42	7,850	1	0.3	2,355	98,910	9693.18	0.003
Crane carrier	Steel	42	7,850	1	3.316	26034.368	1093443.456	107157.459	0.030
Cutter	Cast iron	25	7,250	2	–	6,056	302,800	29674.4	0.0082
Fibrizer	Cast iron	25	7,250	1	–	9,720	243,000	23814	0.007
Gears	Cast iron	25	7,250	1	–	700,560	17514,000	1716,372	0.477
Pipelines	Cast iron	25	7,250	1	–	30550,870	763771,750	74849631.5	20.791

(continued)

Table 10 (continued)

Equipment	Material	Embodied energy (MJ/kg)	Density (kg/m^3)	Quantity	Volume (m^3)	Weight (kg)	Total embodied energy (MJ)	GHG emissions (kg)	GHG emissions/ Ton of sugar
Insulation	Glass wool	32.1	75.125	1	–	7,000	224,700	22020.6	0.006
Motor winding	Copper	40	8,920	50	–	95	190,000	18620	0.005
Motor casing	Cast iron	25	7,250	50	–	73	91,250	8942.5	0.002
Grand total							882269536.1	86462414.54	24.017

Table 11 GHG emissions for electromechanical equipment in the sugar industry by economic input-output analysis

Components	Cost in rupees 2004–2005 (10^6)	GHG emissions (T-CO_{2eq})	GHG emissions (kg-CO_{2eq}/T of sugar)
Machinery and equipment for juice extraction plant with in-line shredder juice installation	359.2	69308.914	19.252
Machinery and equipment for the boiling house	504.65	24040.395	6.678
Machinery and equipment for the centrifugal station	72.93	3909.966	1.086
Total	936.78	97259.275	27.016

CO_{2eq} contents into the environment. The machines and processing equipment used in the sugar industry emit different types and sizes of pollutants, and the insulation of the pipeline emits CO_2 contents into the environment. In this study, PCA and EIO techniques were used for the estimation of GHG emissions. The PCA technique is based on the embodied energy of materials used in the machinery of the sugar industry. With the help of embodied energy, CO_2 emissions from equipment and machinery were obtained for a 20-year life span. In the production of sugar, the total GHG emission from sugar industry equipment was determined to be 86462.41 ton-CO_{2eq} using the PCA technique, which is shown in Table 10 for the 20-year lifespan of the sugar industry.

By using the EIO technique, GHG emissions were obtained with the help of Carnegie Mellon EIO-LCA software. Total GHG emission from sugar industry equipment was determined to be 97259.275 ton-CO_{2eq} using the EIO technique, which is shown in Table 11 for the 20-year lifespan of the sugar industry. The results were 24.017 kg-CO_{2eq} per ton of sugar production by the PCA approach and 27.016 kg-CO_{2eq} per ton of sugar production by the EIO-LCA approach.

Every material, manufacturing process, electricity generation, and utilization of electricity emit GHGs. GHG emission in the sugar industry occurs through direct and indirect emissions. Direct emissions occur due to the processing of sugar production and indirect emissions occur due to the equipment and machinery that are used for sugar production. The machinery and equipment are manufactured using different materials by different processes, which also emit CO_2 into the environment.

9 Conclusion

In the Indian sugar industry, sugarcane is mostly used to produce sugar. The sugar industry also produces several byproducts, such as bagasse, molasses, and filter cake. The LCA of three plants of the sugar industry were examined in this chapter. The cost analysis was performed for primary energy usage and GHG emissions of the sugar industry and its plants. For the sugar mill, energy usage and GHG emissions for 1 ton of sugar production were 761.667 MJ and 46.536 kg-CO_{2eq}, respectively. For the cogeneration plant, energy usage and GHG emission for 1 kWh of electricity generation were 228.811 kJ and 14.683 gm-CO_{2eq}, respectively. For the distillery, energy usage and GHG emission were 161.687 kJ and 9.233 gm-CO_{2eq} per liter of ethanol, respectively. The energy usage and GHG emissions in sugar industry are 1227.61 MJ and 75.04 kg-CO_{2eq} per ton of sugar production using an EIO approach. The GHG emissions from a sugar mill were also estimated using the PCA approach to validate the results; it was found to be 86462.41 ton-CO_{2eq}. The values were also compared based upon functional units and were determined to be 24.017 kg-CO_{2eq} by PCA and 27.016 kg-CO_{2eq} by EIO for 1 ton of sugar production in a sugar mill. This GHG emission rate is very high and directly affects the environment. Therefore, a reduction of this rate is very essential to balance the environment.

Acknowledgements The authors thankfully acknowledge the authorities of Dhampur Sugar Mills Limited (DSML), Dhampur, Uttar Pradesh, for visiting the sugar industry and providing valuable data.

References

IPCC (2001) International Panel on Climate Change. Climate change: the scientific basis. Cambridge University Press, Cambridge

Hulme M, Jenkins G, Lu X, Turnpenny J, Mitchell T, Jones R, Lowe J, Murphy J, Hassell D, Boorman P, McDonald R, Hills S (2002) Climate change scenarios for the United Kingdom: The UKCIP02 scientific report. Tyndall Centre for Climate Change Research, University of East Anglia, Norwich

NEED (2011) National Energy Education Development. http://www.need.org/. Accessed 21 June 2011

USDA (2011) United States Department of Agriculture Report. http://www.usda.gov/. Accessed 21 June 2011

Bureau of Energy Efficiency (2011) National certification examination for energy managers and energy auditors. http://www.em-ea.org/. Accessed 7 May 2011

ISMA (2013) Indian Sugar Mill Association. http://www.indiansugar.com/. Accessed 7 July 2013

Licht FO (2007) International Sugar and Sweetener Report 139

Contreras AM, Rosa E, Perez M, Langenhove HV, Dewulf J (2009) Comparative life cycle assessment of four alternatives for using by-products of cane sugar production. J Clean Prod 17:772–779

Javalagi CM, Patil HR, Bhushi UM (2010) Statistical modeling of steam generation for cogeneration in Indian Sugar Industry: a case study. Cogenerat Distrib Generat J 25:18–34

Foreign Agriculture Service (2012) United States Department of Agriculture Report. http://www.usda.gov/. Accessed 21 June 2012

Renouf MA, Wegener MK, Pagan RJ (2010) Life cycle assessment of Australian sugarcane production with a focus on sugarcane growing. Int J Life Cycle Ass 15:927–937

Paturau JM (1989) By-products of the cane sugar industry. An introduction to their industrial utilization. Elsevier, Amsterdam

ManoharRao PJ (1997) Industrial utilization of sugar cane and its by-products. ISPCK, New Delhi

Ramjeawon T (2008) Life cycle assessment of electricity generation from bagasse in Mauritius. J Clean Prod 16:1727–1734
Rajan CG (2001) Optimization energy in industries. Tata McGraw-Hill Education Pvt Ltd, New Delhi
Ramjeawon T (2004) Life cycle assessment of cane-sugar on the Island of Mauritius. Int J Life Cycle Ass 9:254–260
Baumann H, Tillman AM (2004) The Hitch Hiker's guide to LCA. Student litteratur AB Lund, Sweden
Chauhan MK, Varun, Chaudhary S, Kumar S, Samrat S (2011) Life cycle assessment of Sugar Industry: a review. Renewable Sust Energy Rev 15:3445–3453
Capehart BL, Turner WC, Kennedy WJ (1997) Guide to energy management. The Fairmont Press Incorporation, Lilburn
Yarnal GS, Puranik VS (2009) Energy management in cogeneration system of Sugar Industry using system dynamics modeling. Cogenerat Distrib Generat J 24:7–22
Nguyen TLT, Gheewala SH (2008) Life cycle assessment of fuel ethanol from cane molasses in Thailand. Int J Life Cycle Ass 13:301–311
Dhampur Sugar Mills Limited Report (2011) www.dhampur.com. Accessed 7 April 2011
Boustead I (1996) LCA-how it came about: the beginning in the UK. Int J Life Cycle Ass 1(1):4–7
Moriguchi Y, Kondo Y, Shimizu H (1993) Analysing the life cycle impact of cars: the case of CO_2. Ind Environ 16:42–45
Hondo H, Nishimura K, Uchiyama Y (1996) Energy requirements and CO_2 emissions in the production of goods and services: application of an input-output table to life cycle analysis. CRIEPI report, Y95013, Central Research Institute of Electric Power Industry, Japan
ISO-14040 (1997) Environmental management—life cycle assessment—principles and framework. Int Org Stand, Geneva
Varun, Prakash R, Bhat IK (2008) Life cycle analysis of run-of river small hydro power plants in India. The Open Renew Energy J 1:11–16
Ometto AR, Hauschild MZ, Roma WNL (2009) Life cycle assessment of fuel ethanol from sugarcane in Brazil. Int J Life Cycle Ass 14:236–247
Varun, Bhat IK, Prakash R (2009) LCA of renewable energy for electricity generation systems-a review. Renewable Sust Energy Rev 13:1067–1073
Varun, Prakash R, Bhat IK (2009) Energy, economic and environmental impacts of renewable energy systems. Renewable Sust Energy Rev 13:2716–2721
ISO-14041 (1998) Environmental management—life cycle assessment—goal and scope definition and inventory analysis. Int Org Stand, Geneva
ISO-14042 (2000) Environmental management—life cycle assessment—life cycle impact assessment. Int Org Stand, Geneva
ISO-14043 (2000) Environmental management—life cycle assessment—life cycle interpretation. Int Org Stand, Geneva
ISO-14044 (2006) Environmental management—life cycle assessment—requirements and guidelines. Int Org Stand, Geneva
ISO-14040 (2006) Environmental management—life cycle assessment—principles and framework. Int Org Stand, Geneva
Huettner DA (1976) Net energy analysis: an economic assessment. Science 192(4235):101–104
Mortimer ND (1991) Energy analysis of renewable energy sources. Energy Policy 19(4):374–381
Intergovernmental Panel on Climate Change (2007) IPCC Fourth Assessment Report: Climate change 2007
Broek RVD, Burg TVD, Wijk AV, Turenburg W (2000) Electricity generation from eucalyptus and bagasse by sugar mills in Nicaragua. Biomass Bioenergy 19:311–335
Botha T, Blottnitz HV (2006) A comparison of the environmental benefits of bagasse-derived electricity and fuel ethanol on a life-cycle basis. Energy Policy 34:2654–2661
Prasertsri P (2007) Thailand sugar annual 2007, USDA foreign agriculture service—gain report—Global agriculture information network, gain report number: TH7048

Al-Amin AQ, Siwar C, Jaafar AH (2009) Energy use and environment impact of new alternative fuel mix in electricity generation in Malaysia. Open Renew Energy J 2:25–32

Hawdon D, Pearson P (1995) Input-output simulations of energy, environment, economy interactions in the UK. Energy Econ 17:73–86

Casler S, Wilbur S (1984) Enegry input-output analysis: a simple guide. Resour Energy 6:187–201

Mashoko L, Mbohwa C, Thomas VM (2010) LCA of the South African sugar industry. J Environ Plan Manag 53:793–807

Jafara AH, Al-Amin AQ, Siwar C (2008) Environmental impact of alternative fuel mix in electricity generation in Malaysia. Renew Energy 33:2229–2235

Tekin T, Bayramoglu M (2001) Exergy and structural analysis of raw juice production and steam-power units of a sugar production plant. Energy 26:287–297

Ensinas AV, Nebra SA, Lozano MA, Serra LM (2007) Analysis of process steam demand reduction and electricity generation in sugar and ethanol production from sugarcane. Energy Convers Manag 48:2978–2987

Global Development Research Center (2011) LCA definition. www.gdrc.org . Accessed 14 Feb 2011

Chauhan MK (2011) Life cycle assessment of Sugar Industry in India. Dissertation, National Institute of Technology Hamirpur

Bullard CW, Penner PS, Pilati DA (1978) Net energy analysis handbook of for combining process and input-output analysis. Resour Energy 1:267–313

Chung WS, Tohno S, Shim SY (2009) An estimation of energy and GHG emission intensity caused by energy consumption in Korea: an energy IO approach. Appl Energy 86:1902–1914

CMU Green Design Institute (2008) EIO-LCA model. http://www.eiolca.net/. Accessed 11 May 2011

Zhang Q, Karney B, Maclean HL, Feng J (2007) Life-cycle inventory of energy use and greenhouse gas emissions for two hydropower projects in China. J Infrastruct Syst ASCE 13(4):271–279

Varun, Prakash R, Bhat IK (2012) Life cycle greenhouse gas emissions estimation for small hydropower schemes in India. Energy 44(1):498–508

Carbon Footprint as a Single Indicator in Energy Systems: The Case of Biofuels and CO_2 Capture Technologies

Diego Iribarren and Javier Dufour

Abstract The energy sector is one of the main sources of greenhouse gas emissions, in both the transport and electricity subsectors. Taking into account the current context of the energy sector, relevant case studies concerning biofuels and CO_2 capture in power plants are defined and inventoried to evaluate their carbon footprints; the suitability of these carbon footprints as single indicators is then discussed. The methodological framework proposed in the Life Cycle Assessment standards is followed. The fuel systems evaluated involve second-generation biofuels from short-rotation poplar biomass: (i) synthetic fuels (gasoline and diesel) produced via biomass pyrolysis and bio-oil upgrading and (ii) hydrogen produced via biomass gasification and biosyngas processing. Four case studies of coal power plants with CO_2 capture technology are also evaluated, including post-combustion CO_2 recovery through chemical absorption, membrane separation, cryogenic fractionation, and pressure swing adsorption. Inventory data for the analysis are based on process simulation, robust databases, and scientific literature. The carbon footprints calculated show a promising life-cycle global warming performance of the energy products evaluated. However, conflicting results are found when evaluating other impact categories. Therefore, decisions and recommendations based solely on carbon footprints only capture a partial picture of the environmental performance, although different levels of risk are associated with the use of carbon footprints as single indicators, depending on the type of systems and products under evaluation. The use of multi-indicator approaches is recommended because the inclusion of additional impact categories leads to a more comprehensive evaluation of the environmental performance of energy product systems, thus facilitating a more sensible decision-making process oriented towards environmental sustainability.

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S. S. Muthu (ed.), *Assessment of Carbon Footprint in Different Industrial Sectors, Volume 2*, EcoProduction, DOI: 10.1007/978-981-4585-75-0_4,

Keywords Biofuel · Carbon dioxide capture · Carbon footprint · Energy · Life cycle assessment · Power

Abbreviations

ADP	Abiotic depletion potential
AP	Acidification potential
CCS	CO_2 capture and storage
CCU	CO_2 capture and utilization
CED	Cumulative non-renewable energy demand
CF	Carbon footprinting
CFB	Circulating fluidized bed
CO_2 eq	Carbon dioxide equivalent
DEA	Data envelopment analysis
EEA	European Environment Agency
EP	Eutrophication potential
FU	Functional unit
GCC	Gas and char combustor
GHG	Greenhouse gas
GWP	Global warming impact potential
IPCC	Intergovernmental Panel on Climate Change
ISO	International Organization for Standardization
LCA	Life cycle assessment
LCI	Life cycle inventory analysis
LCIA	Life cycle impact assessment
MEA	Monoethanolamine
ODP	Ozone layer depletion potential
PAS	Publicly available specification
POFP	Photochemical oxidant formation potential
PSA	Pressure swing adsorption
RED	Renewable energy directive
SMR	Steam methane reforming
TS	Technical specification
WGS	Water-gas shift

1 Introduction

The energy sector is one of the main sources of greenhouse gas (GHG) emissions. Both the transport and electricity subsectors are currently associated with high GHG emission rates. Moreover, the increasing energy demand worldwide could make this situation of environmental unsustainability even worse. These environmental concerns and the growing energy demand (and prices), as well as energy insecurity and social awareness of environmental issues (mainly climate change), have brought about the search for technological solutions that contribute to establishing a sustainable future energy sector (International Energy Agency 2012).

The attempts to provide the energy sector with sustainable energy systems involve not only conventional renewables (e.g., wind and solar power) but also a wide range of technological options that can be based on either novel processes or the modification of conventional process schemes (e.g., the use of the Fischer-Tropsch process to coproduce synthetic biofuels and electricity).

Biofuels are currently seen as the main option for substituting fossil fuels in the oil-dependent road transport subsector (European Commission 2006; Iribarren et al. 2012a). A wide range of biomass resources and technologies can be used to produce biofuels (Huber et al. 2006). Regarding resources, residual biomass could be a good option to yield sustainable biofuels (e.g., biodiesel production via esterification-transesterification of waste vegetable oils (Iribarren and Dufour 2012)), but it suffers from availability concerns when it comes to satisfying large fuel demands. Microalgae also have been studied as a possible feedstock for future bioenergy systems because of their high productivity and potentially high oil or carbohydrate content. However, significant efforts are still needed to overcome important barriers concerning immature cultivation and processing techniques for the use of microalgae to produce biodiesel and/or bioethanol (Mata et al. 2010; Iribarren et al. 2013a; Kohl et al. 2013). First-generation biofuels, based on food crops such as corn and sunflower, could fulfill the future biofuel demand, but at the expense of high land occupation. In fact, concerns regarding land use and competition between fuel and food have led the promotion of second-generation biofuels rather than first-generation ones. Lignocellulosic biomass from short-rotation plantations can be grown with low input requirements (including land needs) and could guarantee the supply of sustainable second-generation biofuels, therefore arising as a suitable feedstock for bioenergy conversion systems.

A variety of systems can be used to convert biomass into transportation fuels. Even though most of them produce biodiesel (e.g., systems based on oil transesterification) or bioethanol (e.g., via simultaneous saccharification and co-fermentation), other bioenergy systems (e.g., those based on the Fischer-Tropsch process using biosyngas or on the hydroprocessing of pyrolysis bio-oil) produce synthetic fuels (Iribarren et al. 2012a; Swain et al. 2011; Iribarren et al. 2013b). Furthermore, other conversion systems, such as those based on indirect biomass gasification, consider the production of hydrogen as an alternative biofuel (Spath et al. 2005; Susmozas et al. 2013).

Regarding the electricity sector, in addition to the use of conventional renewables and power generation from biomass, important efforts have been made to promote the implementation of CO_2 capture schemes in power plants (Mondal et al. 2012). CO_2 capture technologies are usually separated into pre-combustion, oxy-fuel combustion, and post-combustion technologies. Post-combustion methods include chemical absorption, which is the most developed technology.

Strategies based on CO_2 capture and storage (CCS; with or without enhanced resource recovery) or CO_2 capture and utilization (CCU) are especially interesting in power plants, as these facilities account for high CO_2 emissions (Iribarren et al. 2013c).

Environmental concerns regarding the energy sector are mainly focused on climate change. The promotion of CCS and the existing energy policies (e.g., the Renewable Energy Directive [RED] 2009/28/EC (European Union 2009)) clearly show the leading role of global warming when dealing with the environmental performance of the energy sector. Hence, a thorough and robust methodology for the quantification of greenhouse gas (GHG) emissions is needed. In this sense, the standardized Life Cycle Assessment (LCA) methodology (International Organization for Standardization 2006a, b) provides the basis for the calculation of carbon footprints (i.e., life-cycle GHG emissions). RED guidelines (European Union 2009) and current carbon footprinting (CF) specifications, such as PAS 2050:2011 (British Standards Institution 2011) and ISO/TS 14067:2013 (International Organization for Standardization 2013), follow this life-cycle approach, even though relevant differences exist among the different quantification schemes.

Although a large number of LCA studies on biofuels are available in scientific literature, they usually deal with the evaluation of individual case studies. These studies are often limited to the impact categories of global warming and cumulative energy demand, and they mostly evaluate first-generation biofuels (mainly biodiesel and bioethanol), even though the number of LCA studies on second-generation biofuels is increasing (Hoefnagels et al. 2010; Kendall and Yuan 2013). LCA studies on CO_2 capture in power plants are scarcer. Nevertheless, important efforts have already been made to compare CCS options in power plants, taking into account a life-cycle perspective and a wide range of environmental concerns (Iribarren et al. 2013c; Khoo and Tan 2006; Singh et al. 2011).

Although carbon footprints are valuable indicators of the performance of energy systems, their use as single indicators should be discussed because they could lead to a distorted image of the environmental performance of this type of systems. This chapter addresses this discussion through different case studies of biofuels and CO_2 capture in power plants. Relevant case studies are used to not only quantify the specific carbon footprints of relevant energy products, but also enable the formulation of general recommendations on the use of carbon footprints when it comes to evaluating the environmental performance of energy systems. In this sense, this chapter goes beyond common CF and LCA studies of energy systems because it is not restricted to a particular case study; instead, it attempts to provide (based on the discussion of quantitative results) general guidelines for the appropriate environmental evaluation of any energy system.

Figure 1 shows the roadmap for the chapter. Section 2 addresses the methodological framework of the study by defining its objectives, the life-cycle approach followed, and the specific case studies under evaluation regarding both biofuel systems and power generation systems with CO_2 capture, as well as data acquisition and methodological choices. After defining and inventorying the case studies in Sect. 2, Sect. 3 tackles the quantification of the carbon footprints of the

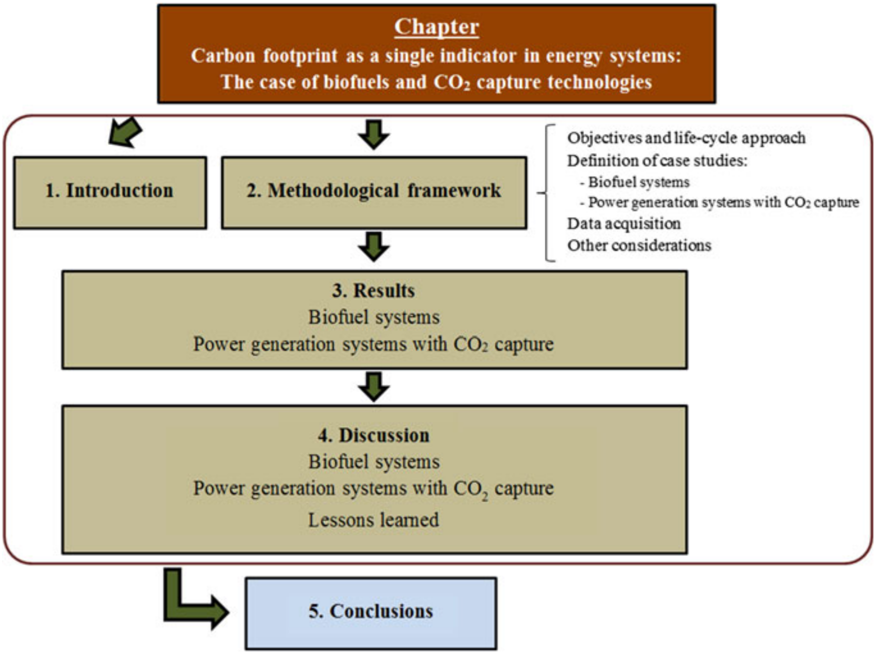

Fig. 1 Structure of the chapter

corresponding energy products, as well as the comparison of these carbon footprints with those of conventional equivalent products. Section 4 focuses on the discussion of the suitability of carbon footprints as single indicators when evaluating the environmental performance of energy systems. With this aim, Sect. 4 broadens the environmental scope of the case studies by evaluating additional impact categories, such as acidification and cumulative energy demand. Sections 4 and 5 use this specific discussion based on relevant case studies to draw more general conclusions and recommendations on the environmental evaluation of energy product systems.

2 Methodological Framework

2.1 Objectives and Life-Cycle Approach

The goal of this chapter is to show the potential effects of using carbon footprints as single indicators when evaluating energy systems. Specific case studies developed by the authors are used to illustrate these effects and identify the strong and weak points of CF.

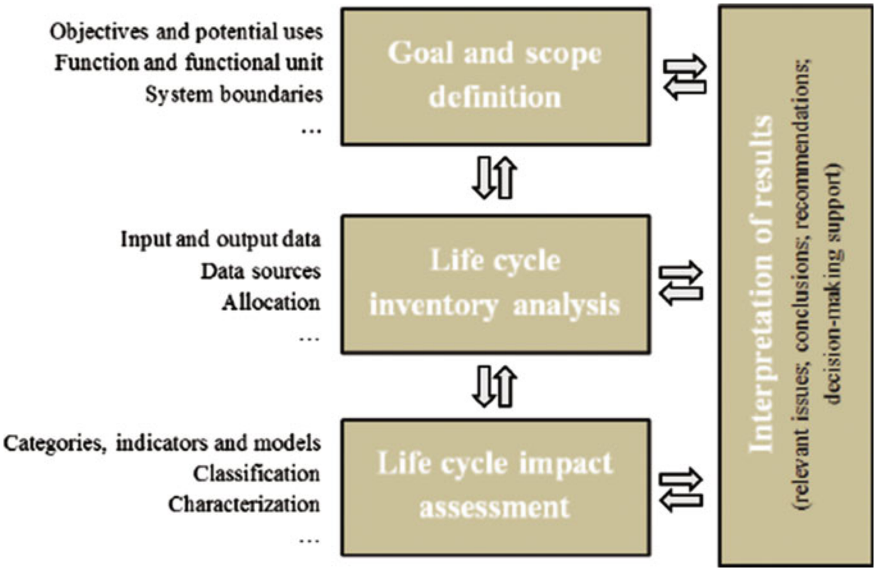

Fig. 2 LCA framework according to ISO14040:2006

The methodological framework proposed in the standardized LCA methodology is followed (International Organization for Standardization 2006a, b). As can be observed in Fig. 2, the study involves four interrelated stages: goal and scope definition, life cycle inventory analysis (LCI), life cycle impact assessment (LCIA), and interpretation.

The stage "Goal and scope definition" involves the definition of the objectives and potential uses of the study, as well as other key aspects such as the functional unit (FU), the system boundaries, assumptions, and restrictions. The LCI step requires data collection to carry out an inventory of the input and output data of the system under study. LCIA includes three mandatory steps: (i) selection of impact categories, indicators, and characterization models; (ii) classification (i.e., association of the inventory data with the selected impact categories); and (iii) characterization (i.e., calculation of the results of each category indicator through the conversion of the inventory elements to common units by using characterization factors, and aggregation of the converted results within the same impact category). Finally, in the interpretation stage, the results from the previous steps are summarized according to the goal and scope defined for the LCA study and discussed in order to identify relevant issues and provide conclusions, recommendations, and information for decision-making purposes (International Organization for Standardization 2006a, b).

2.2 Definition of Case Studies

According to the current context of the energy sector, a relevant set of case studies of biofuels and CO_2 capture in power plants is defined to evaluate the corresponding carbon footprints and discuss their suitability as single indicators.

2.2.1 Biofuel Systems

Two case studies of biofuel systems are considered. Both systems involve second-generation biofuels from poplar biomass. Poplar is selected as the biomass feedstock due to the current interest in short-rotation plantations with energy purposes. One of the selected systems deals with synthetic fuels (gasoline and diesel) obtained through biomass pyrolysis and bio-oil upgrading, whereas the other produces hydrogen via biomass gasification and biosyngas processing.

Synthetic Fuels from Pyrolysis Bio-Oil

The synthetic biofuel system (Fig. 3) includes cultivation and transportation of poplar biomass, bio-oil production through fast pyrolysis, and bio-oil upgrading to gasoline and diesel blendstocks. Additionally, the transportation of the synthetic fuels and their combustion in conventional engines are included (well-to-wheels approach). The FU for this case study is 1 t of fuel products, which corresponds to 602 kg of gasoline and 398 kg of diesel.

In the pyrolysis plant, poplar biomass (50 % moisture) is first pretreated in order to reduce its moisture content and particle size. The biomass delivered is dried to 7 % moisture in a direct-contact dryer using the hot exhaust gases coming from the gas and char combustor (GCC). Afterwards, it is ground in a crusher and passes through a sieve to guarantee a particle size below 3 mm. The pretreated biomass is converted into gas, char, and liquid fractions via fast pyrolysis in a circulating fluidized bed (CFB) reactor that operates at 520 °C and atmospheric pressure (residence time: 2.5 s) (Iribarren et al. 2012b). The heat required by the reactor is provided by the GCC.

The liquid fraction is usually called bio-oil. A two-stage hydrotreating process converts the bio-oil into a hydrocarbon mix. The bio-oil is stabilized in the first reactor under mild conditions (250 °C, 140 bar) and then deoxygenated to approximately 1.7 % oxygen content at more severe conditions (340 °C, 170 bar) in the second reactor (Iribarren et al. 2012b). The organic stream coming from the hydrotreating section is split up in the desired products using distillation columns and a hydrocracker.

The hydrogen required by the hydrotreating and hydrocracking reactors is produced in a steam reforming process that converts the light hydrocarbons contained in the off-gas streams from the hydrotreating and hydrocracking units into

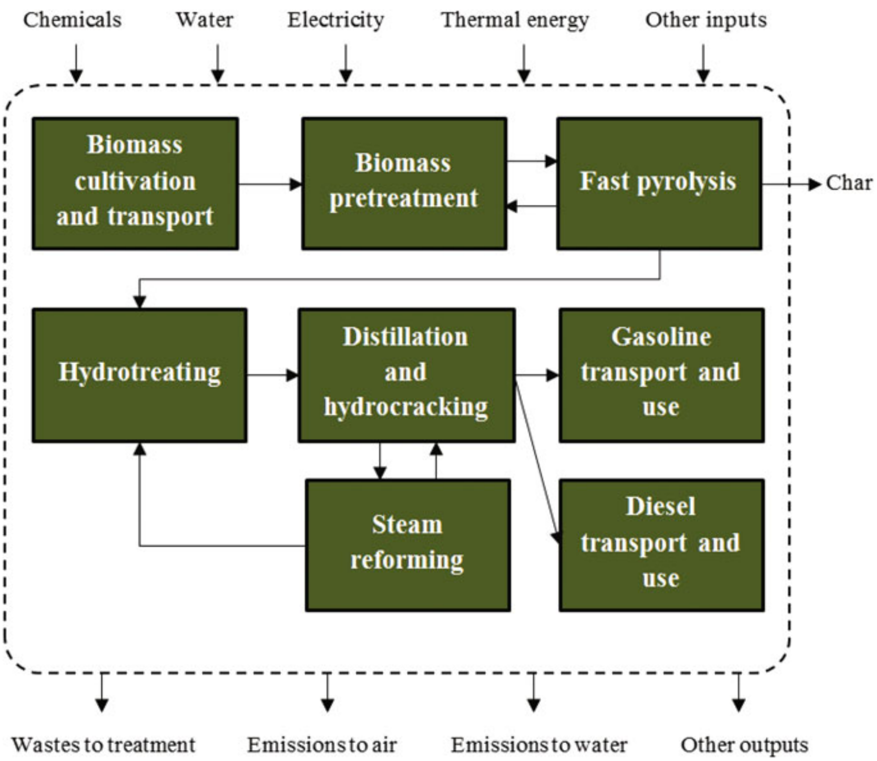

Fig. 3 Simplified diagram of the synthetic biofuel system

H_2 and CO. Additional natural gas is fed to the reactor in order to meet the hydrogen demand. After the steam reformer, a water–gas shift (WGS) reactor and a pressure swing adsorption (PSA) unit are used to finally obtain the desired hydrogen. The PSA off-gas and a fraction of the off-gas stream from the hydrocracker are fed to the off-gas combustor, which provides the heat required by the steam reforming reaction and the distillation columns (Iribarren et al. 2012b).

Hydrogen Via Indirect Biomass Gasification

The biohydrogen system (Fig. 4) includes poplar cultivation and transportation, biosyngas production through indirect gasification, syngas processing to hydrogen, and on-site power generation (cradle-to-gate approach). The FU for this case study is 1 kg of hydrogen produced (at plant; 99.9 vol% purity).

In the gasification plant, the poplar feedstock is milled and dried (from 50 to 12 % moisture content). The gasification process uses a low-pressure indirect gasifier consisting of two fluidized-bed reactors: a gasifier in which biomass reacts with steam at 870 °C and 1.6 bar producing raw syngas and char, and a combustor

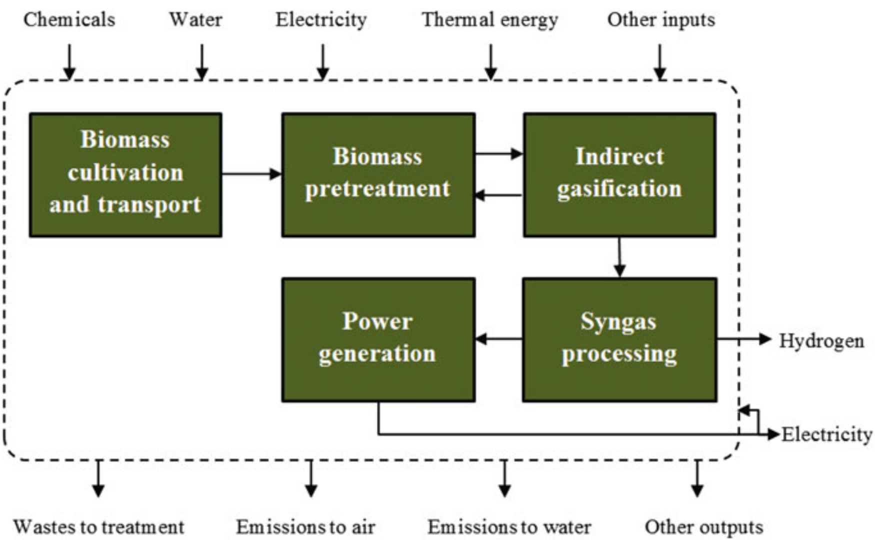

Fig. 4 Simplified diagram of the biohydrogen system

where the char fraction is burnt to provide the heat needed for the gasification process (Susmozas et al. 2013). The flue gas from the combustor is used to dry the poplar feedstock. The raw syngas undergoes a reforming process to convert tars and light hydrocarbons into CO and H_2.

The syngas stream is cooled and filtered in order to remove fine particles and condensed alkali compounds. Afterwards, the syngas is compressed and goes through a LO-CAT® process to remove sulfur compounds. The clean syngas undergoes a WGS process and, finally, hydrogen is separated from the rest of compounds in a PSA unit with 85 % efficiency (40 °C, 28 bar) (Susmozas et al. 2013).

The PSA off-gas is combusted to produce steam in a heat recovery steam generator. This steam is used on site to produce electricity in a steam cycle (30 MW). Part of the steam from the intermediate- and high-pressure sections of the turbine is used to satisfy the steam requirements of gasification and WGS (Susmozas et al. 2013).

2.2.2 Power Generation Systems with CO_2 Capture

Four alternative case studies of coal-fired power plants provided with post-combustion CO_2 capture technology are considered herein. As can be observed in Fig. 5, the four CO_2 capture systems evaluated involve the same steps, comprising the mining of the coal, through coal conditioning and power generation, to gas treatment and CO_2 capture (cradle-to-gate approach). Nevertheless, each specific

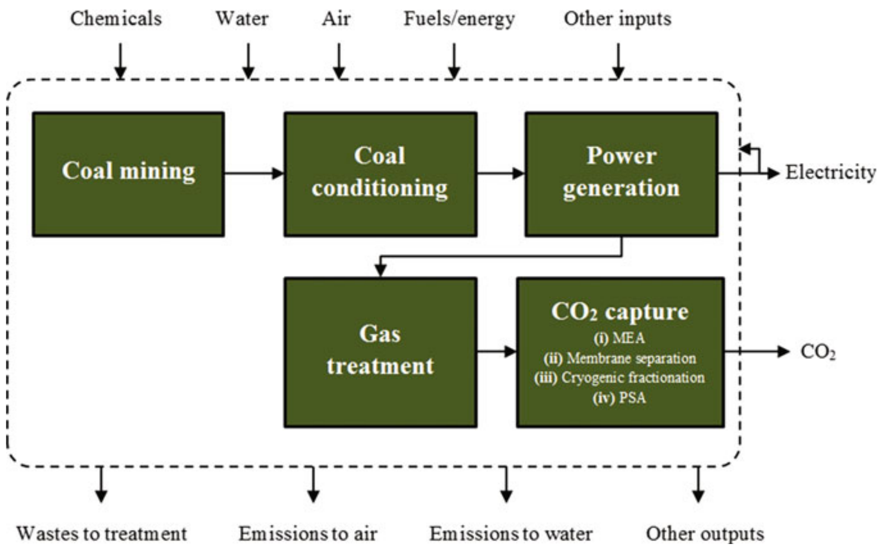

Fig. 5 General diagram of the CO_2 capture systems

post-combustion technology involves different material and energy flows. The FU for the four case studies is 1 kWh of net electricity (at plant).

The four CO_2 capture systems differ from each other in terms of the post-combustion technology selected (Iribarren et al. 2013c):

- Post-combustion CO_2 recovery via chemical absorption with monoethanolamine (MEA).
- Post-combustion CO_2 recovery via membrane separation.
- Post-combustion CO_2 recovery via cryogenic fractionation.
- Post-combustion CO_2 recovery via PSA.

It should be noted that these case studies stop at the generation of liquid CO_2, not including further steps such as CO_2 transport, storage, or beneficial use of carbon dioxide.

2.3 Data Acquisition

Key inventory data for the biofuel systems are derived from process simulation in Aspen Plus® (Aspen Technology 2013). Thus, the fast pyrolysis of poplar biomass and the subsequent bio-oil upgrading to synthetic fuels, as well as the indirect gasification of poplar biomass and the subsequent processing of the biosyngas to produce hydrogen, are simulated in Aspen Plus® in order to obtain LCI data. As an example, Fig. 6 shows the simulation diagram of the gasification plant, where

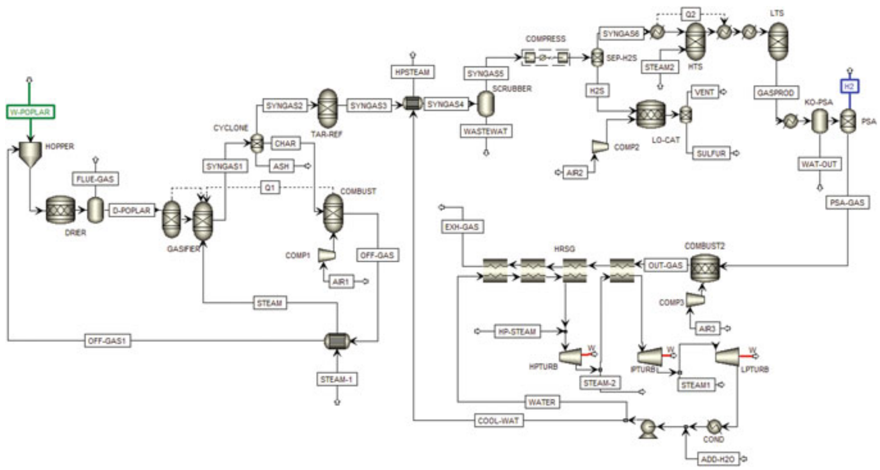

Fig. 6 Simulation diagram of the gasification plant for biohydrogen production

poplar biomass is pretreated and gasified to produce biosyngas, also including biosyngas processing to hydrogen and power generation (Susmozas et al. 2013).

Inventory data for the four CO_2 capture systems are based on scientific literature in the field of CCS (Iribarren et al. 2013c). Data for post-combustion CO_2 recovery through chemical absorption with MEA (Khoo and Tan 2006; Singh et al. 2011; Pehnt and Henkel 2009; Schreiber et al. 2009) are modified according to Khoo and Tan (2006) in order to include the alternative post-combustion technologies (i.e., membrane separation, cryogenic fractionation, and PSA).

Tables 1, 2 and 3 present a selection of key inventory data for each of the biofuel and CO_2 capture case studies. Further information on LCI data can be found elsewhere (Susmozas et al. 2013; Iribarren et al. 2013c; Iribarren et al. 2012b).

Data for poplar biomass are taken from specific literature (Gasol et al. 2009; Fan et al. 2011), whereas combustion emissions for the biosynfuel system are based on European Environment Agency (2009). Finally, data for background processes (e.g., waste management, transport, and production of chemicals and energy carriers) are retrieved from the ecoinvent® database (Frischknecht et al. 2007).

2.4 Other Considerations

Capital goods are not included in any case study. Economic allocation is used to distribute inventory data and environmental burdens when dealing with multi-functional systems (Curran 2007). In this respect, economic allocation is applied to

Table 1 Selection of inventory data for the biosynfuel system (functional unit: 1 t of synthetic fuel products)

Input	Units	Amount	Output	Units	Amount
Wet poplar biomass	t	6.77	Gasoline (to combustion)	kg	602.40
Poplar transport	t km	541.20	Diesel (to combustion)	kg	397.60
Natural gas	kg	223.73	Char (product)	kg	546.44
Electricity	MWh	1.20	CO_2 (direct emission at plant)	t	2.38
Biosynfuel transport	t km	200.00			

Table 2 Selection of inventory data for the biohydrogen system (functional unit: 1 kg of hydrogen)

Input	Units	Amount	Output	Units	Amount
Wet poplar biomass	kg	36.28	Hydrogen (product)	kg	1.00
Poplar transport	t km	2.90	Electricity (product)	kWh	2.07
			CO_2 (direct emission at plant)	kg	32.84

Table 3 Selection of inventory data for the CO_2 capture systems (functional unit: 1 kWh of net electricity)

Item	Units	Case MEA	Case membrane	Case cryogenics	Case PSA
Coal (input)	g	672.20	554.00	969.20	609.90
Net electricity (output)	kWh	1.00	1.00	1.00	1.00
Captured CO_2 (output)	kg	1.29	0.91	1.76	1.04
CO_2 (direct emission)	g	67.65	200.71	195.07	184.13

the biosynfuel system for both the pyrolysis section (bio-oil [allocation percentage: 89 %] and char [11 %]) and the bio-oil upgrading section (gasoline [63 %] and diesel [37 %]). Regarding the biohydrogen system, economic allocation is applied between the hydrogen (95 %) and electricity (5 %) products. In the CO_2 capture systems, the whole impact is attributed to the net electricity output (i.e., 0 % to the captured CO_2).

As a general concern in LCA and CF studies, different decisions on methodological choices such as boundary selection and allocation approach would lead to different results within each case study (Reap et al. 2008a). This fact, along with other factors such as data availability and quality, leads to uncertainty in the decision process (Reap et al. 2008b). Nevertheless, this chapter does not aim to report and compare accurate carbon footprints of the energy products evaluated, but rather to discuss the suitability of carbon footprints as single indicators when evaluating energy product systems. In Sect. 4, the discussion is based on broadening the environmental scope of the case studies (i.e., evaluating not only global warming, but also additional impact categories). Because all impact categories are evaluated for each individual case study based on the same system definition and

Table 4 Carbon footprints of the biofuel systems (results per functional unit)

Item	Units	Amount
Combusted synthetic biogasoline	kg CO_2 eq	84.93
Combusted synthetic biodiesel	kg CO_2 eq	164.59
Biohydrogen	kg CO_2 eq	0.39

the same inventory, uncertainty concerns are highly mitigated for the purposes of the study. Hence, no uncertainty analysis has been carried out for the case studies proposed.

3 Results

Specific LCA software (SimaPro 7) is used for the computational implementation of the inventories (Goedkoop et al. 2010). The global warming impact potential (GWP) of each case study is evaluated. Note that the GWP results are the carbon footprints of the energy systems assessed (expressed in terms of CO_2 eq). The calculation of these carbon footprints is carried out according to the characterization factors (100-year period) reported by the Intergovernmental Panel on Climate Change (Forster et al. 2007).

3.1 Biofuel Systems

Table 4 summarizes the carbon footprints of the biofuel products under study. Regarding synthetic biogasoline (combusted in a conventional passenger vehicle engine), the corresponding carbon footprint (84.93 kg CO_2 eq FU^{-1}) is due mainly to direct emissions arising from the fuel use phase, ahead of direct emissions from the energy conversion plant. When compared to conventional (fossil) gasoline (inventoried according to the ecoinvent® database (Dones et al. 2007)) also combusted in a common vehicle engine (European Environment Agency 2009), a GHG saving of 96 % is calculated. This high GHG saving clearly exceeds the 60 % GHG savings criterion stated in the RED for biofuels and bioliquids produced in installations in which production started on or after 1 January 2017 (European Union 2009).

Regarding synthetic biodiesel (combusted in a conventional passenger vehicle engine), a carbon footprint of 164.59 kg CO_2 eq FU^{-1} is calculated. As in the case of synthetic gasoline, direct emissions from the fuel use phase, followed by direct emissions from the energy conversion plant, are the main sources of GWP. In comparison with fossil diesel (inventoried according to the ecoinvent® database (Dones et al. 2007)) combusted in a common vehicle engine (European Environment Agency 2009), an 88 % GHG saving is estimated, also meeting the 60 % criterion of the RED.

The carbon footprint allocated to the hydrogen product within the biohydrogen system is 0.39 kg CO_2 eq FU^{-1}. Direct emissions to the air from the energy conversion plant account for the highest contribution to this carbon footprint. When compared to conventional hydrogen from steam methane reforming (SMR) as defined by Susmozas et al. (2013), a very high GHG saving (96 %) is estimated, which clearly shows that gasification-based biohydrogen is more suitable than conventional hydrogen in terms of global warming.

3.2 Power Generation Systems with CO_2 Capture

Figure 7 shows the carbon footprints of the electricity product from the four power generation systems equipped with CO_2 capture technology. The main sources of GWP identified in the four cases are the coal feedstock (leading contributor in the case study of chemical absorption with MEA) and direct emissions to the air arising from the coal power plant (leading contributor in the remaining case studies).

Furthermore, Fig. 8 compares the carbon footprint of the electricity produced in a conventional coal-fired power plant without CO_2 capture (as defined by Iribarren et al. (2013c)) with that of the electricity generated in the evaluated power plants provided with post-combustion CO_2 capture technology. As can be observed in Fig. 8, significant GHG savings (ranging from 57 to 75 %) are calculated for the different capture alternatives. Hence, from a life-cycle global warming perspective, CO_2 capture is found to be a suitable strategy to be implemented in power plants.

4 Discussion

This chapter does not focus on the quantitative results of the carbon footprints of biofuels and electricity, but it aims to discuss the suitability of these carbon footprints as single indicators of the environmental performance of energy systems.

Taking into account GWP as the sole criterion of environmental suitability, the results for biofuels in Sect. 3.1 show that they are an eco-friendly alternative to conventional fossil fuels. Similarly, Sect. 3.2 considers CO_2 capture as an appropriate option in power plants in order to generate environmentally friendly electricity.

Section 4 broadens the environmental scope of the study by evaluating a higher number of impact categories, thereby verifying the environmental appropriateness of biofuels and CO_2 capture. The CML method is used to evaluate the following impact potentials: abiotic depletion (ADP), ozone layer depletion (ODP), photochemical oxidant formation (POFP), acidification (AP), and eutrophication (EP)

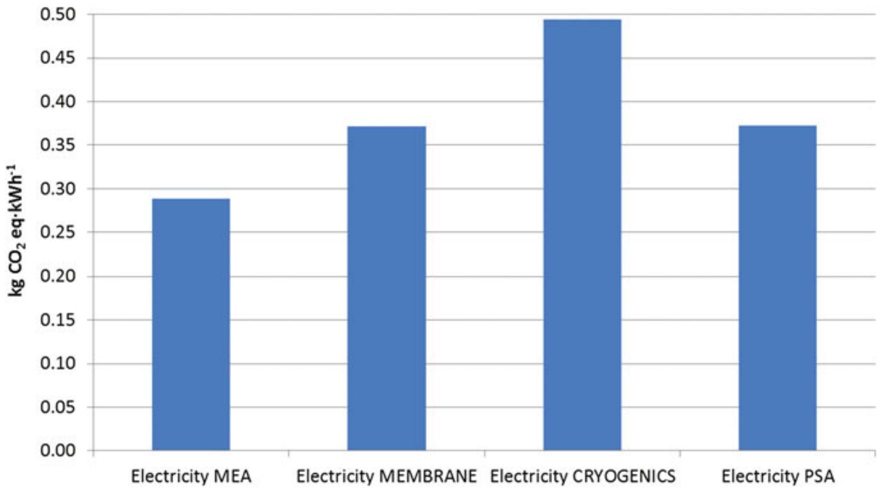

Fig. 7 Carbon footprints of the power generation systems with post-combustion CO_2 capture

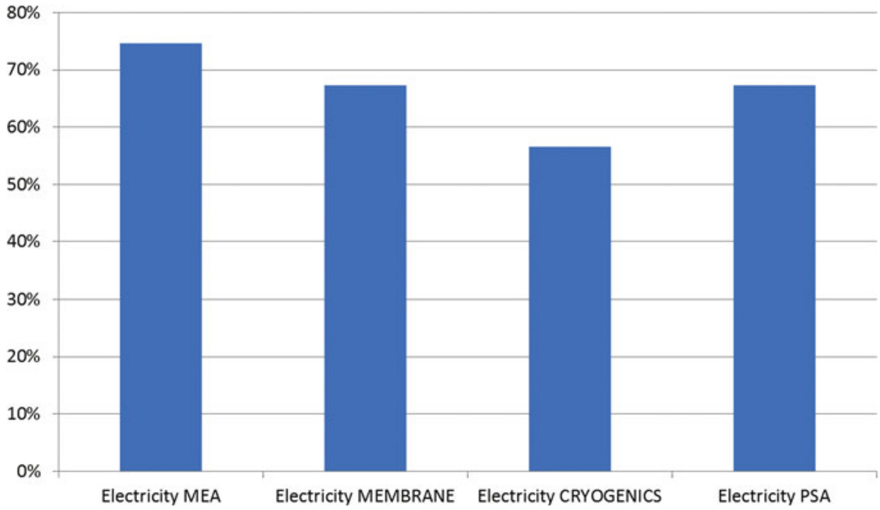

Fig. 8 Greenhouse gas savings linked to coal power plants with CO_2 capture relative to a conventional coal-fired power plant without CO_2 capture

(Guinée et al. 2001). The cumulative non-renewable (i.e., fossil and nuclear) energy demand (CED) is also quantified as an additional impact category (Verein Deutscher Ingenieure 2012). This wider set of common impact categories allows the identification of potential conflicts between GWP and other impacts when giving general recommendations on the substitution of conventional energy systems.

4.1 Biofuel Systems

Biohydrogen and synthetic biofuels result in a promising performance in terms of GWP (Sect. 3.1). However, the inclusion of additional impact categories could lead to a different picture of the life-cycle performance of these biofuels.

Figure 9 shows the comparison between synthetic biofuels and conventional fossil fuels when taking into account the extended set of impact categories. In the case of synthetic biogasoline (Fig. 9a), all categories (except for EP) identify synthetic biogasoline as a suitable alternative to fossil gasoline.

When compared to the use of GWP as a single indicator, the use of additional impact categories does not seem to influence significantly the recommendation in favor of synthetic biogasoline. Nevertheless, if special relevance is given to EP over the rest of categories, then this recommendation could be altered. The unfavorable EP result of synthetic biogasoline is linked to high electricity requirements and biomass cultivation (Iribarren et al. 2012a, b).

With respect to synthetic biodiesel (Fig. 9b), this biofuel is found to perform better than conventional fossil diesel in GWP as well as in four of the six additional impact categories under evaluation (i.e., CED, ADP, ODP, and POFP). Unless special attention has to be paid to AP and EP, the recommendation driven by GWP could be maintained. As seen in the case of synthetic biogasoline, the detrimental EP/AP performance of synthetic biodiesel is mainly due to electricity production and biomass cultivation (Iribarren et al. 2012a, b).

Figure 10 shows the comparison between biohydrogen and conventional SMR hydrogen for the extended set of impact categories. As can be observed in this figure, biohydrogen leads to important impact savings not only in GWP, but also in most of the additional impact categories (i.e., ADP, CED, ODP, and, to a lesser extent, POFP). Under these environmental categories, biohydrogen is recommended as an eco-friendly alternative to conventional hydrogen. However, if AP and EP are prioritized, then this recommendation could be wrong (as also seen in the case of synthetic biodiesel). The unfavorable AP/EP results of biohydrogen are closely linked to the need of fertilizers for biomass cultivation and to direct emissions to the air from the power generation section of the plant (Susmozas et al. 2013).

Overall, when evaluating biofuels, the recommendations driven by GWP seem not to be dramatically affected by the inclusion of additional impact categories. However, despite this generalization, the environmental suitability of biofuels is actually conditioned by the specific impact category in consideration. Although CED, ADP, and ODP usually show a behavior similar to that of GWP, other categories such as AP and EP are likely to lead to opposite recommendations. Furthermore, the consideration of a higher number of additional impact categories would result in a higher number of conflicts between the recommendations driven by GWP and those based on other impact categories.

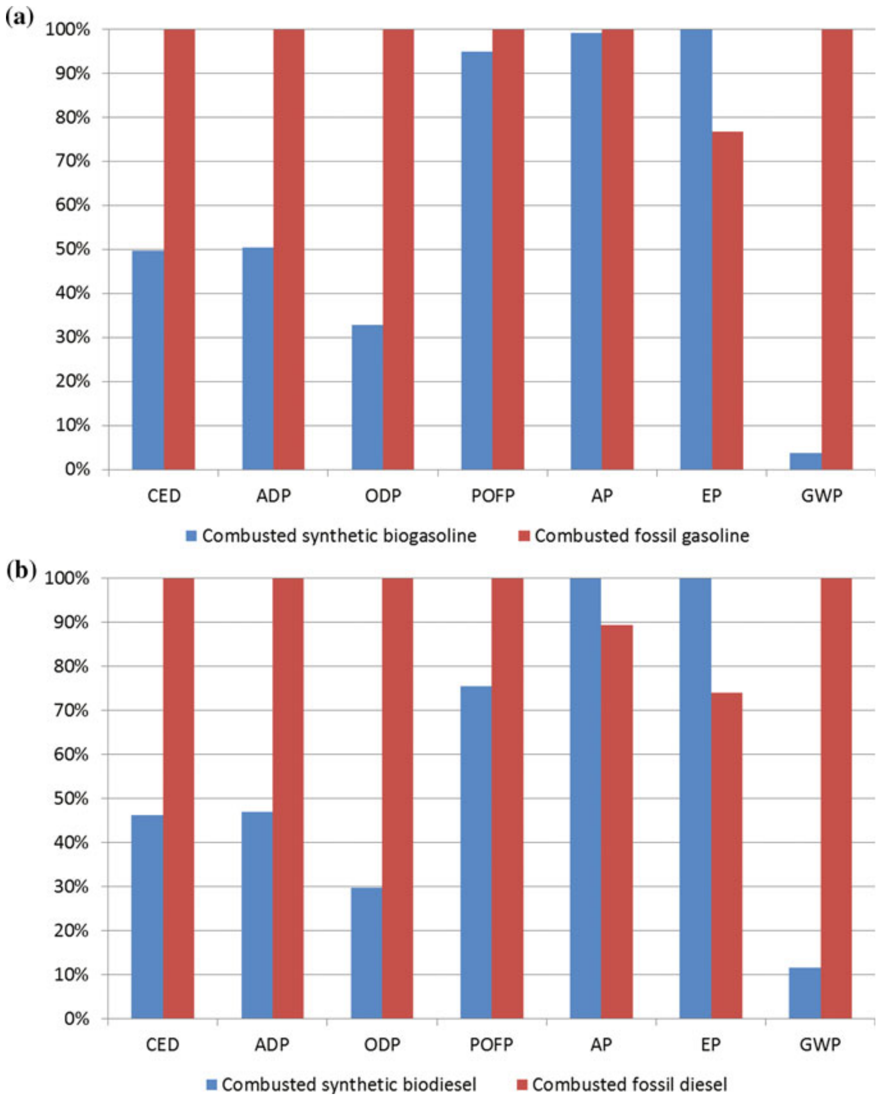

Fig. 9 Comparison of the environmental profile of synthetic biofuels and conventional fossil fuels: **a** gasoline, **b** diesel

4.2 Power Generation Systems with CO_2 Capture

When compared to conventional power plants from a life-cycle global warming perspective, the use of CO_2 capture technologies in power plants is found to be an appropriate strategy (Sect. 3.2). However, this suitability could be affected by the use of carbon footprints as single indicators. Figure 11 presents the comparison

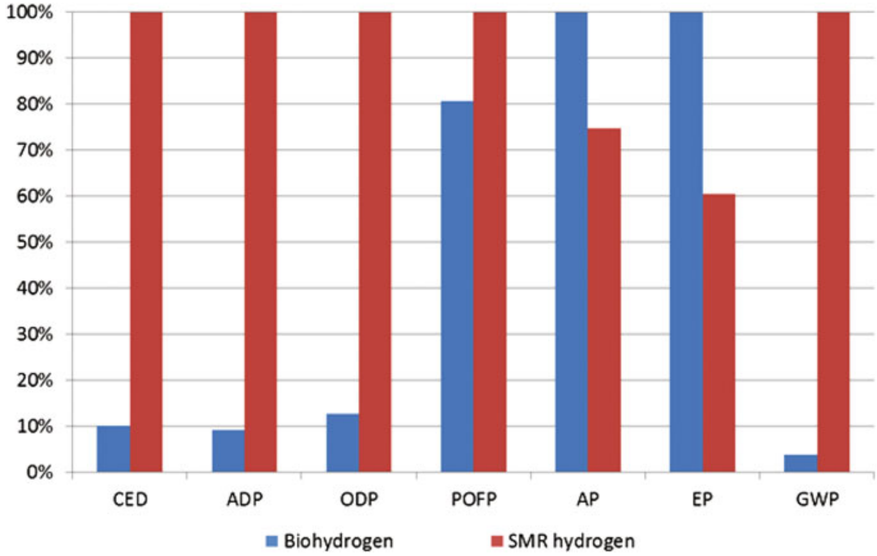

Fig. 10 Comparison of the environmental profile of biohydrogen from indirect biomass gasification and conventional hydrogen from steam methane reforming (SMR)

between conventional electricity (from a conventional coal-fired power plant without CO_2 capture (Iribarren et al. 2013c)) and the electricity generated in each of the four evaluated plants equipped with post-combustion CO_2 capture technology, taking into account the extended set of impact categories.

As can be observed in Fig. 11, most of the evaluated impact categories (namely, ODP, ADP, CED, and EP) show a worse performance of the electricity from coal power plants with CO_2 capture. Therefore, when evaluating power generation systems with CO_2 capture technology, the inclusion of additional impact categories in the assessment dramatically affects the identification of suitable energy systems.

Even though important GWP reductions are attained by implementing CO_2 capture strategies, the increased requirements of coal make the environmental benefits of these systems questionable, also affecting their thermodynamic performance (Iribarren et al. 2013c).

Overall, power generation with CO_2 capture faces concerns regarding its environmental and thermodynamic suitability. The use of carbon footprints as single indicators is very likely to lead to a misleading picture of the environmental performance of this type of systems, whose suitability highly depends on the impact categories considered.

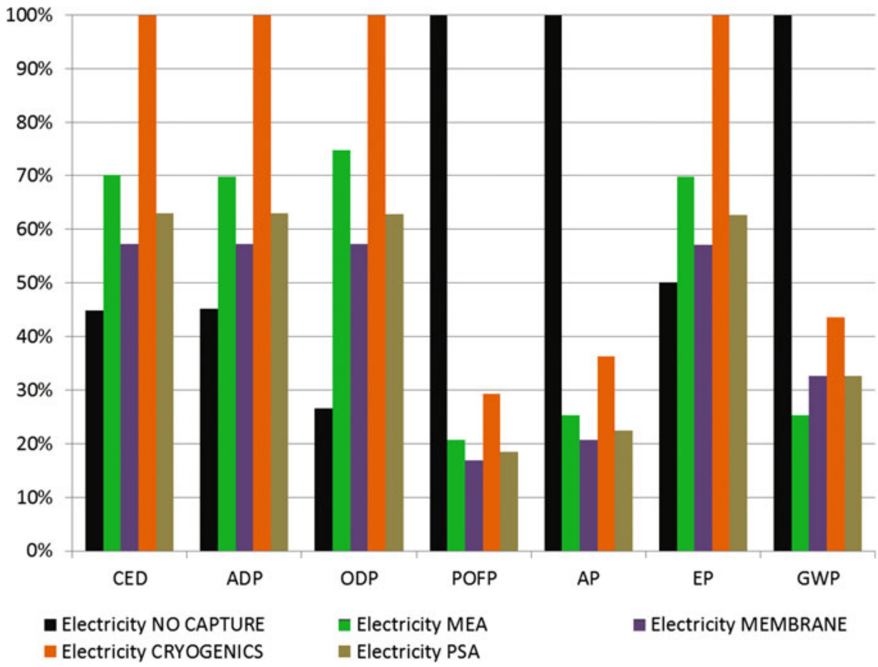

Fig. 11 Comparison of the environmental profile of electricity from coal power plants with CO_2 capture and electricity from a conventional coal-fired power plant

4.3 Lessons Learned

Biofuels and electricity from power plants with CO_2 capture are used in this chapter as relevant case studies of the energy sector. The individual study of their carbon footprints in Sects. 3.1 and 3.2 shows a promising life-cycle global warming performance of both types of energy products. Nevertheless, differences arise when it comes to expanding the scope of the study by including further impact categories (Sects. 4.1 and 4.2).

Decisions on the environmental suitability of a product always depend on the impact categories considered. Hence, decisions and recommendations based only on carbon footprints (i.e., only on the GWP results) will unavoidably capture a partial picture of the environmental performance of the evaluated product. Thus, the use of carbon footprints as single indicators is likely to result in a misleading interpretation of the environmental analysis.

However, different levels of risk seem to be associated with the use of carbon footprints as single indicators when assessing energy systems, depending on the type of systems and products under evaluation. In this respect, even though the use of a single indicator does not allow an unequivocal interpretation of the environmental performance of an energy product/system, the only use of carbon footprints to characterize biofuels can succeed in providing a general (simplified) picture of their performance. Nevertheless, analysts should be aware of the singularities of biomass-based systems, which probably affect certain categories such as AP and EP leading to opposite trends as compared to GWP. On the other hand, carbon footprints as single indicators of the environmental performance of electricity from power plants with CO_2 capture often distort the actual performance of the corresponding product systems, which usually involve energy-intensive technologies that seriously affect many categories such as CED and ADP.

Moreover, the correlation between GWP (carbon footprints) and other impact categories does not offer a general pattern for energy products (Laurent et al. 2012). In other words, despite the strong interactions that climate change shows with other global environmental issues, there is a weak correlation between carbon footprints and certain impact categories, such as toxicity-related categories or resource depletion (Laurent et al. 2012). Therefore, the use of a multi-indicator approach is generally safer, as also seen in CF studies of non-energy systems (Merrild 2009; Iribarren et al. 2010a).

When taking biofuels and electricity from power plants with CO_2 capture as representative case studies of the energy sector and trying to reduce ambiguity concerns, it is concluded that carbon footprints should not be the only criterion to assess energy product systems from a life-cycle environmental perspective. The inclusion of additional impact categories leads to a more comprehensive evaluation of the environmental performance of energy systems, thus facilitating a more sensible decision-making process oriented towards environmental sustainability (Iribarren and Dufour 2012; Iribarren et al. 2013c).

The recommendation on the use of multi-indicator approaches connects with the controversial discussion on the rough definition of CF as an LCA restricted to the GWP category. In this respect, taking into consideration ISO standards on LCA and admitting that CF is based on a life-cycle perspective, the terms CF and LCA should not be mixed in the same definition because LCA refers not only to a holistic approach but also to a comprehensive view of impacts (International Organization for Standardization 2006a, b).

Despite the appropriateness of multi-indicator evaluations, CF should not be trivialized. In fact, CF has succeeded in catalyzing life-cycle thinking, reaching policy makers, companies, and society (Iribarren et al. 2010a; Weidema et al. 2008). This success is closely linked to the interest in reporting environmental results (Finkbeiner 2009; Sinden 2009). The development of CF specifications such as PAS 2050:2011 (British Standards Institution 2011) has facilitated the systematic calculation of life-cycle GHG emissions, enhancing the communicability of carbon footprints.

Furthermore, although the carbon footprint of a product is a single indicator, CF involves a procedure that can be easily extended to evaluate impact categories other than GWP. This feature is due to the fact that the inventory data used in the CF study could be further used in LCA studies in order to get a more comprehensive understanding of the environmental performance of the evaluated product.

Finally, in addition to the possibility of performing an LCA using inventory data from the CF study, other methodological improvements could help to mitigate the concerns regarding the limited scope of CF in terms of evaluated impacts. For instance, when evaluating multiple similar entities, the Data Envelopment Analysis (DEA) methodology (Cooper et al. 2007) can be combined with either CF or LCA approaches, offering synergistic effects (Iribarren 2010; Vázquez-Rowe et al. 2010; Iribarren et al. 2010b; Vázquez-Rowe and Iribarren 2014). In particular, the combined use of CF and DEA moderates the reiterated limitation that CF cannot account for a comprehensive assessment of environmental impacts. This benefit of the combined CF and DEA approach is linked to the underlying nature of the method, which seeks GHG-emission benchmarking through the optimization of resource use (Vázquez-Rowe and Iribarren 2014). Because the optimization of resource use generally results in a better environmental performance in all impact categories (Schmidheiny 1992), the concern about the use of carbon footprints as single indicators is reduced.

5 Conclusions

The assessment of the life-cycle GHG emissions (i.e., carbon footprints) of different second-generation biofuels (synthetic fuels via biomass pyrolysis and hydrogen via biomass gasification) and electricity from coal power plants with alternative CO_2 capture technologies (chemical absorption, membrane separation, cryogenic fractionation, and pressure swing adsorption) was used to discuss the suitability of carbon footprints as single indicators when evaluating the environmental performance of energy product systems.

Although the carbon footprints calculated indicate a promising life-cycle performance of the energy products evaluated, opposite findings are seen when taking into account other impact categories. Therefore, carbon footprints as single indicators lead to a partial (and maybe misleading) picture of the environmental performance of energy products.

Although recommendations based solely on carbon footprints correspond with a partial picture of the environmental performance of the evaluated energy products, different levels of risk are associated with the use of carbon footprints as single indicators depending on the type of systems and products under study. For instance, carbon footprints can provide a general, simplified picture of the environmental performance of biofuels, whereas their use as single indicators for electricity from power plants with CO_2 capture usually distorts the actual environmental performance of the assessed product in a dramatic way.

Analysts are responsible for taking into consideration the singularities of each specific energy product system under evaluation because these singularities can seriously affect a wide range of impact categories, leading to trends opposite to GWP. For instance, the singularities of biomass-based systems affect certain categories, such as acidification and eutrophication, whereas energy-intensive technologies (e.g., CO_2 capture) affect categories such as abiotic depletion and cumulative energy demand.

The results for biofuels and electricity with CO_2 capture—as relevant case studies of the energy sector—show that carbon footprints should not be the only criterion for the environmental characterization of energy product systems from a life-cycle perspective. The use of multi-indicator approaches is considered to be more appropriate because it reduces ambiguity concerns.

Finally, even though the inclusion of additional impact categories facilitates more sensible decision- and policy-making processes oriented towards environmental sustainability, CF studies should continue to be undertaken. They not only address the globally relevant impact category of global warming, but also have proven to be an effective vehicle for the penetration of life-cycle thinking in companies, policies, and society. Furthermore, CF studies constitute a valuable source of inventory data that can be easily implemented in LCA studies to provide a more comprehensive environmental evaluation of energy systems.

Acknowledgements This research has been supported by the Regional Government of Madrid (S2009/ENE-1743) and the Spanish Ministry of Economy and Competitiveness (CTQ2011-28216-C02-02 and ENE2011-29643-C02-01). The authors would like to thank Jens F. Peters and Ana Susmozas for valuable scientific exchange.

References

Aspen Technology (2013) Aspen Plus®. http://www.aspentech.com/products/aspen-plus.aspx. Accessed 10 Oct 2013

British Standards Institution (2011) PAS 2050:2011 Specification for the assessment of the life cycle greenhouse gas emissions of goods and services. BSI, London

Cooper WW, Seiford LM, Tone K (2007) Data envelopment analysis: a comprehensive text with models, applications, references and DEA-solver software. Springer, New York

Curran MA (2007) Studying the effect on system preference by varying coproduct allocation in creating life-cycle inventory. Environ Sci Technol 41:7145–7151

Dones R, Bauer C, Bolliger R et al (2007) Life cycle inventories of energy systems: Results for current systems in Switzerland and other UCTE countries, ecoinvent report No. 5. Swiss Centre for Life Cycle Inventories, Dübendorf

European Commission (2006) Biofuels in the European Union—a vision for 2030 and beyond. European Communities, Luxembourg

European Environment Agency (2009) EMEP/EEA air pollutant emission inventory guidebook 2009. EEA, Copenhagen

European Union (2009) Directive 2009/28/EC of the European Parliament and of the Council of 23 April 2009 on the promotion of the use of energy from renewable sources and amending and subsequently repealing Directives 2001/77/EC and 2003/30/EC. Official Journal of the European Union, 5 June 2009

Fan J, Kalnes TN, Alward M et al (2011) Life cycle assessment of electricity generation using fast pyrolysis bio-oil. Renew Energ 36:632–641

Finkbeiner M (2009) Carbon footprinting—opportunities and threats. Int J Life Cycle Ass 14:91–94

Forster P, Ramaswamy V, Artaxo P et al (2007) Changes in atmospheric constituents and in radiative forcing. In: Solomon S, Qin D, Manning M et al (eds) Climate change 2007: the physical science basis—contribution of working group I to the fourth assessment report of the intergovernmental panel on climate change. Cambridge University Press, Cambridge, pp 129–234

Frischknecht R, Jungbluth N, Althaus HJ et al (2007) Overview and methodology, ecoinvent report No. 1. Swiss Centre for Life Cycle Inventories, Dübendorf

Gasol CM, Gabarrell X, Antón A et al (2009) LCA of poplar bioenergy system compared with *Brassica carinata* energy crop and natural gas in regional scenario. Biomass Bioenerg 33:119–129

Goedkoop M, Schryver A de, Oele M et al (2010) Introduction to LCA with SimaPro 7. PRé Consultants, Amersfoort

Guinée JB, Gorrée M, Heijungs R et al (2001) Life cycle assessment: an operational guide to the ISO standards. Centre of Environmental Science, Leiden

Hoefnagels R, Smeets E, Faaij A (2010) Greenhouse gas footprints of different biofuel production systems. Renew Sust Energ Rev 14:1661–1694

Huber GW, Iborra S, Corma A (2006) Synthesis of transportation fuels from biomass: chemistry, catalysts, and engineering. Chem Rev 106:4044–4098

International Energy Agency (2012) World Energy Outlook 2012. OECD/IEA, Paris

International Organization for Standardization (2006a) ISO 14040:2006 Environmental management—life cycle assessment—principles and framework. ISO, Geneva

International Organization for Standardization (2006b) ISO 14044:2006 Environmental management—life cycle assessment—requirements and guidelines. ISO, Geneva

International Organization for Standardization (2013) ISO/TS 14067:2013 Greenhouse gases—carbon footprint of products—requirements and guidelines for quantification and communication. ISO, Geneva

Iribarren D (2010) Life cycle assessment of mussel and turbot aquaculture: application and insights. University of Santiago de Compostela, Santiago de Compostela

Iribarren D, Hospido A, Moreira MT, Feijoo G (2010a) Carbon footprint of canned mussels from a business-to-consumer approach—a starting point for mussel processors and policy makers. Environ Sci Policy 13:509–521

Iribarren D, Vázquez-Rowe I, Moreira MT, Feijoo G (2010b) Further potentials in the joint implementation of life cycle assessment and data envelopment analysis. Sci Total Environ 408:5265–5272

Iribarren D, Dufour J (2012) Life cycle assessment of biodiesel production from free fatty acid-rich wastes. Renew Energ 38:155–162

Iribarren D, Peters JF, Dufour J (2012a) Life cycle assessment of transportation fuels from biomass pyrolysis. Fuel 97:812–821

Iribarren D, Peters JF, Petrakopoulou F, Dufour J (2012b) Well-to-wheels comparison of the environmental profile of pyrolysis-based biofuels. In: Krautkremer B, Ossenbrink H, Baxter D et al (eds) Proceedings of the 20th European biomass conference and exhibition. ETA-Florence Renewable Energies, Florence, pp 2195–2198

Iribarren D, Susmozas A, Sanz A (2013a) Contrasting the life-cycle performance of conventional and alternative diesel fuels. In: Silva C, Rivera A (eds) Diesel fuels: characteristics, performances and environmental impacts. Nova Science Publishers, New York, pp 153–167

Iribarren D, Susmozas A, Dufour J (2013b) Life-cycle assessment of Fischer–Tropsch products from biosyngas. Renew Energ 59:229–236

Iribarren D, Petrakopoulou F, Dufour J (2013c) Environmental and thermodynamic evaluation of CO_2 capture, transport and storage with and without enhanced resource recovery. Energy 50:477–485

Kendall A, Yuan J (2013) Comparing life cycle assessments of different biofuel options. Curr Opin Chem Biol 17:439–443

Khoo HH, Tan RBH (2006) Life cycle investigation of CO_2 recovery and sequestration. Environ Sci Technol 40:4016–4024

Kohl M, Iribarren D, Petrakopoulou F, Dufour J (2013) Life cycle assessment of bioethanol from microalgae. In: Krautkremer B, Ossenbrink H, Baxter D et al (eds) Proceedings of the 21st European biomass conference and exhibition. ETA-Florence Renewable Energies, Florence, pp 1818–1822

Laurent A, Olsen SI, Hauschild MZ (2012) Limitations of carbon footprint as indicator of environmental sustainability. Environ Sci Technol 46:4100–4108

Mata TM, Martins AA, Caetano NS (2010) Microalgae for biodiesel production and other applications: a review. Renew Sust Energ Rev 14:217–232

Merrild H (2009) Indicators for waste management: how representative is global warming as an indicator for environmental performance of waste management?. Technical University of Denmark, Kongens Lyngby

Mondal MK, Balsora HK, Varshney P (2012) Progress and trends in CO_2 capture/separation technologies: a review. Energy 46:431–441

Pehnt M, Henkel J (2009) Life cycle assessment of carbon dioxide capture and storage from lignite power plants. Int J Greenh Gas Con 3:49–66

Reap J, Roman F, Duncan S, Bras B (2008a) A survey of unresolved problems in life cycle assessment—part 1: goal and scope and inventory analysis. Int J Life Cycle Ass 13:290–300

Reap J, Roman F, Duncan S, Bras B (2008b) A survey of unresolved problems in life cycle assessment—part 2: impact assessment and interpretation. Int J Life Cycle Ass 13:374–388

Schmidheiny S (1992) Changing course: a global business perspective on development and the environment. MIT Press, Cambridge

Schreiber A, Zapp P, Kuckshinrichs W (2009) Environmental assessment of German electricity generation from coal-fired power plants with amine-based carbon capture. Int J Life Cycle Ass 14:547–559

Sinden G (2009) The contribution of PAS 2050 to the evolution of international greenhouse gas emission standards. Int J Life Cycle Ass 14:195–203

Singh B, Strømman AH, Hertwich EG (2011) Comparative life cycle environmental assessment of CCS technologies. Int J Greenh Gas Con 5:911–921

Spath P, Aden A, Eggeman T et al (2005) Biomass to hydrogen production detailed design and economics utilizing the Battelle Columbus Laboratory indirectly-heated gasifier. NREL, Golden

Susmozas A, Iribarren D, Dufour J (2013) Life-cycle performance of indirect biomass gasification as a green alternative to steam methane reforming for hydrogen production. Int J Hydrogen Energ 38:9961–9972

Swain PK, Das LM, Naik SN (2011) Biomass to liquid: a prospective challenge to research and development in 21st century. Renew Sust Energ Rev 15:4917–4933

Vázquez-Rowe I, Iribarren D (2014) Life-cycle benchmarking approaches oriented towards energy policy making: Launching the CFP+DEA method. J Clean Prod (in press)

Vázquez-Rowe I, Iribarren D, Moreira MT, Feijoo G (2010) Combined application of life cycle assessment and data envelopment analysis as a methodological approach for the assessment of fisheries. Int J Life Cycle Ass 15:272–283

Verein Deutscher Ingenieure (2012) VDI guideline 4600: cumulative energy demand (KEA) — terms, definitions, methods of calculation. VDI, Düsseldorf

Weidema BP, Thrane M, Christensen P et al (2008) Carbon footprint: a catalyst for LCA? J Ind Ecol 12:3–6

Reduction in the Carbon Footprint of Coal-Fired Thermal Power Plants by Promoting Compact Fluorescent Lamps and Light-Emitting Diodes in Households, Offices, and Commercial Centers

Sushant B. Wath and Deepanjan Majumdar

Abstract The electricity consumption of compact fluorescent lamps (CFLs) and light-emitting diode (LED) lamps is low, making them a useful tool for minimizing the rapidly increasing demand for electrical energy in India and elsewhere. This chapter aims to project the likely electrical energy conservation in a scenario of complete replacement of existing fluorescent tubes (FTs) by CFLs or LEDs at the Council of Scientific and Industrial Research (CSIR)-National Environmental Engineering Research Institute (NEERI), including the financial repercussions and indirect reduction in emissions of greenhouse gases (e.g. CO_2, N_2O, CH_4) and carbon footprint, as well as a few important air pollutants (e.g. SO_2, NO, black carbon, suspended particulate matter (SPM), mercury) in a coal-fired thermal power plant. The calculations show that the institute could save around 129,870 and 164,970 kW h of electricity per annum by replacing FTs with CFLs and LEDs, respectively, thereby saving approximately INR 1357,142 (US$21,935.37) and INR 1723,937 (US$27,863.85) in electricity costs per year for CFLs and LEDs, respectively. The use of CFLs and LEDs would be able to minimize approximately 47,127.14 and 59,863.94 kg of CO_2–C equivalent emissions over a 100-year time horizon, respectively. Moreover, reductions of approximately 961, 1,039, 10, 390, 19, and 0.55 kg of SO_2, NO, BC, SPM, PM_{10} and Hg emissions per year, respectively, could be achieved in electricity conservation by replacing FTs with CFLs at CSIR-NEERI. Reductions of approximately 1,221, 1,320, 13, 495, 25 and 0.7 kg of SO_2, NO, BC, SPM, PM_{10} and Hg emissions per year, respectively, could be achieved by replacing FTs with LEDs at CSIR-NEERI.

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S. S. Muthu (ed.), *Assessment of Carbon Footprint in Different Industrial Sectors, Volume 2*, EcoProduction, DOI: 10.1007/978-981-4585-75-0_5,

Keywords Climate change · Greenhouse gas · Global warming · India · Thermal power

1 Introduction

The carbon footprint (CF) or carbon profile is the total carbon dioxide (CO_2) plus other greenhouse gas (GHG; e.g. methane, nitrous oxide etc.) emissions associated with a product along its supply chain, sometimes calculated through the product's use and end-of-life recovery and disposal (European Platform on Life Cycle Assessment 2013). Electricity generation in power plants, heating with fossil fuels, transport operations, and various other industrial and agricultural processes are the main causes of these emissions.

A related measure, the global warming potential (GWP) of the GHGs, is used as an indicator to quantify carbon footprint. It is defined by the Intergovernmental Panel on Climate Change (IPCC 2005) as "the ratio of time integrated radiative forcing from a pulse emission of 1 kg of a substance, relative to that of 1 kg of carbon dioxide over a fixed horizon period." GWP is treated by the IPCC as "an indicator that reflects the relative effect of a greenhouse gas in terms of climate change considering a fixed time period, such as 100 years" (GWP100). Therefore, GWP is an index that attempts to integrate the overall climate impacts of a specific action (e.g. emissions of CH_4, NO_x, or aerosols). The overall contribution of these emissions to climate change can be expressed as one single indicator by adding together GWPs of different emissions. The duration of the perturbation is included by integrating radiative forcing over a time horizon (e.g. standard horizons for IPCC have been 20, 100, and 500 years). The time horizon thus includes the cumulative climate change and the decay of the perturbation (IPCC 2013).

Climate change is one of the key impact categories considered in Life Cycle Assessment (LCA), in which the IPCC factors for CO_2 equivalents are typically used. LCA covers the subset of carbon footprint data as an internationally standardized method (ISO 14040, ISO 14044; www.iso.org; accessed on 1.10.2013) for the evaluation of the environmental burdens and resources consumed along the life cycle of products—from the extraction of raw materials, the manufacture of goods, their use by final consumers or for the provision of a service, recycling, energy recovery, and ultimately disposal. Hence, a carbon footprint is a life cycle assessment with the analysis limited to emissions that have an effect on climate change. The available existing LCA databases are the only suitable background data sources for the footprint, which contains the life cycle profiles of the goods and services that anyone purchases, along with many of the underlying materials, energy sources, transportation, and other services (European Platform on Life Cycle Assessment 2013).

One of the indirect ways of reducing the carbon footprint of thermal power plants, which are a major contributor to GHG emissions, is by saving energy with efficient lighting. Proactive replacement of the fluorescent tube (FT) lights commonly used in houses and offices with compact fluorescent lights (CFLs) or light-emitting diodes (LEDs) will reduce direct carbon footprint of an office; it will also affect the economy in terms of savings in the electricity bill, as well as reduce the indirect carbon footprint of a thermal power plant that supplies power in the region. Furthermore, there is an additional advantage of reducing air pollutant emissions from thermal power generation by introducing CFLs/LEDs at the user level. Clearly, it is in the public's best interest to have an aggressive mix of strategies that includes mitigation, adaptation, and technological development to reduce serious harm to environment, human health, and the economy. Even small changes in policies or behaviors that save energy and reduce CO_2 emissions can have a profound effect on energy conservation, the national economy, and the global climate (Pearce and Hanlon 2007).

This chapter discusses the study and analysis of energy conservation options at the institutional level, with an objective to propose a roadmap for a reduction in the carbon footprint of coal-fired thermal power plants by promoting CFL and LED lights in households, offices, and commercial centers. Section 2 discusses the energy requirements, production scenarios, and dependence on fossil fuel for power generation in thermal power plants, the associated carbon footprint, and the need for its reduction. Section 3 discusses direct and indirect mechanisms and strategies that can be adopted for carbon footprint reduction. Carbon footprint reduction in natural gas–driven power plants is also discussed in detail. Section 4 discusses the indirect mechanism of carbon footprint reduction by replacement of existing FTs with either CFLs or LEDs, including a comparison between these three lighting options with their respective limitation and advantages. Section 5 discusses the methodology adopted for the case study undertaken at Council of Scientific and Industrial Research (CSIR)-National Environmental Engineering Research Institute (NEERI) involving replacement of 1,350 FTs with either CFLs or LEDs, in terms of energy and financial savings for the institute, along with the reduction in GHG emissions and other air pollutants (SO_2, NO, black carbon [BC], suspended particulate matter, and mercury) from the coal-fired thermal power plants. The detail results are discussed in Sect. 6 based on the methodology described in Sect. 5. The final recommendations for choosing the best options are discussed in the summary and conclusions in Sect. 7.

2 Energy Scenario, Associated Carbon Footprint, and the Need for its Reduction

2.1 Energy Scenario

In the summary for policymakers prepared after the 12th session of Working Group I in September 2013, IPCC reiterated that the atmospheric concentrations of greenhouse gases such as carbon dioxide, methane, and nitrous oxide have

Table 1 World electricity generation[a] from 1971–2010 by fuel in *terawatt hour* (TW h)

Source	1971 (%)	2010 (%)
Coal/Peat	38.3	40.6
Hydroelectric	21.0	16.0
Natural gas	12.1	22.2
Nuclear	3.3	12.9
Oil	24.7	4.6
Others[b]	0.6	3.7
Total electricity generation	**6,115 TW h**	**21,431 TW h**

[a] Excludes pumped storage
[b] Other includes geothermal, solar, wind, biofuels/waste, and heat
Source IEA (2012)

increased to unprecedented levels. In particular, CO_2 concentrations have increased by 40 % since pre-industrial times, primarily from fossil fuel (mainly coal/peat, oil, etc.) emissions and secondarily from net land use change emissions (IPCC 2013). The substantial increase in emissions of greenhouse gases occurred primarily due to a worldwide increase in demand for electricity, which was needed for rapid industrialization, urbanization, and mechanization in agriculture. The growth and development of a country is critically dependent on energy availability. Therefore, meeting the enhanced energy demand is very important and crucial for sustainable development.

In India, due to growing industrialization and population, energy demand has increased at a faster pace than its production and supply. Electricity demand in 2010–2011 was 861,591 million units against a supply of 788,355 million units, implying a shortage of 73,236 million units (8.5 %); in 2011–2012; demand was 933,741 million units against a supply of 837,374 units, showing a shortfall of 96,367 million units (10.3 %) (CEA 2012). Although the planning commission of India has given significant priority to the energy sector in 5-year plans and electricity generation has increased in recent years, per the Energy Statistics report (2013), India continued to have an overall energy deficit of 8.7 % and peak shortage of 9.0 % (ESR 2013). Along with the increase in demand in domestic and industrial sectors, transmission and distribution losses (which vary between 30 and 45 %) are also responsible for power shortages.

The power sector is the largest and fastest-growing carbon emitter globally. The use of coal for power generation is the biggest threat to climate and environmental sustainability. In recent years, coal use grew by 22 % worldwide (BP 2006), which has led to an increase in CO_2 emissions at a record rate of 3 % per year (IEA 2006). According to the International Energy Agency (IEA), CO_2 emissions from energy sources may increase up to 90 % by 2030 and coal will account for 43 % of global emissions unless policy interventions are made under the business-as-usual (BAU) scenario (http://www.sciencedaily.com/releases/2007/05/070504151722.htm, accessed on 6.10.2013). The use of various fuel sources for electricity generation in the world indicates a major reliance on fossil fuels such as coal, natural gas, and oil for the production of electricity (Table 1). The IPCC (2007a, b) warned that unmitigated climate change is likely to exceed the capacity of natural, managed, and human systems to adapt in the long run (Pearce and Harris 2007).

2.2 Associated Carbon Footprint and Need for its Reduction

Globally, thermal power plants are mostly run on coal, although some are based on natural gas and naptha. Coal/peat had the maximum share (40.6 %) of total electricity generation in the world during 2010 (Table 1); it is expected to continue to constitute a major share of electricity generation in the future. Coal is also the favorite fuel for electricity generation in large developing countries, such as China and India, due to its local availability and sustained high price of imported natural gas and oil. Coal is approximately 90 % of the total fuel mix for electricity generation. However, relatively lower calorific values, along with high ash content and inefficient combustion technologies, increase the emission of greenhouse gases and other pollutants from India's coal- and lignite-based thermal power plants (Mittal et al. 2012). In India, coal-fired power plants generate 75 % of all electricity and are the major source of air pollution in India's 20 largest cities. Since 1990, CO_2 emissions from coal use grew by 83 % in India (IEA 2006). Thermal power plants, using about 70 % of the total coal in India (Garg et al. 2002), are among the large point sources (LPS) that significantly contribute (47 % each for CO_2 and SO_2) to the total LPS emissions in India. Coal has the dominant role in electricity generation, which is approximately 54 % of the installed electricity generation capacity in India, per a report from the government of India (ESR 2013).

Combustion of coal at thermal power plants emits mainly carbon dioxide (CO_2), sulfur oxides (SO_x), nitrogen oxides (N_2O, NOx,), other trace gases and airborne inorganic particulates, such as fly ash and suspended particulate matter (SPM) (Raghuvanshi et al. 2006). The extent of CO_2 emissions from the combustion of coal depends on the quantity of coal consumed and average carbon (C) content of the coal. A small percent of unoxidized carbon remains largely as particulate matter. The high heating value of coal (or gross calorific value (GCV)) is related to its carbon content. Of the total carbon burnt, about 1 % escapes unoxidized (Marland and Rotty 1984). Based on the input parameters and ultimate analysis of the fuel used for power generation, the emission of CO_2 from thermal power plants can be computed. The input parameters are coal consumed per year, combustion system efficiency, and C content of the fuel. The combustion system efficiency has been considered equal to the average value (26 %) observed in pulverized systems. The total CO_2 emissions from the power sector can be obtained from the following equation:

$$Q_{CO_2} = C\,\rho\eta,$$

where Q_{CO_2} is the amount of carbon dioxide emitted in Mt; C is the carbon fraction of the fuel; ρ is the amount of fuel consumed in a particular year in MT (per year); and η is the combustion efficiency of the system.

Power plants also use small quantities (about 0.2–0.3 ml/unit of power) of diesel oil and furnace oil as supplementary fuels to boost the combustion and heat content, the consumption of which could range from 1 to 4 % of fuel (Raghuvanshi et al. 2006). This supplementary fuel is also instrumental in some CO_2 and other GHG emissions.

In India, it is estimated that CO_2 emissions may be expected to increase at an annual growth rate of 3 % between 2001 and 2025. This has been exacerbated by the low energy efficiency of coal-fired power stations in the country. It is well recognized that there will be continuous increases in CO_2 emissions in India and worldwide. The per capita CO_2 emission of India is about five times lower than the global per capita CO_2 emissions. Presently, the per capita CO_2 emissions of India have been reduced by about 3 times and the total CO_2 emissions have been reduced by about 2 times less than the global per capita emissions (Raghuvanshi et al. 2006).

Among the GHGs, CO_2 receives major attention even though its radiative forcing is much less than other greenhouse gases (e.g., CH_4, N_2O, chlorofluorocarbons) as CO_2 is emitted in much larger amounts into the atmosphere and also has substantial atmospheric lifetime. In fact, CO_2 was estimated to contribute approximately 60 % of the enhanced greenhouse gas effect (Houghton 1997). The International Energy Agency (IEA 2006) and IPCC's Third Assessment Report have put carbon dioxide on the top of the GHG list with GWP. CO_2 contributes approximately 70 % of the potential global warming caused by the emission of GHGs out of the various anthropogenic activities. However, there are large variations in CO_2 emissions per MW h of electricity generated by fossil fuels due to differences in generation efficiency, fuel selection, and plant age. However, there has been a steady decline in average emissions per MW h due to both a gradual switch from carbon-intensive fuels, such as coal, to low-carbon fuels, such as natural gas, as well as improvements in energy conversion efficiency (Morion et al. 2003).

The GHG emissions in the industrial sector from a thermal power plant are categorized by scope, as defined in the GHG protocol (Ahmad 2012):

- Scope 1: Direct emissions from sources that are owned or controlled by the company, including stationary, mobile, and fugitive emissions
- Scope 2: Indirect emissions, such as transmission/distribution losses, electricity used in the company's own buildings
- Scope 3: Other indirect emissions, such as staff commuting, raw material transportation, and waste disposal.

3 Mechanisms and Strategies for Carbon Footprint Reduction

Several strategies and mechanisms, which can be categorized as either direct or indirect, can be employed to reduce the carbon footprint of a thermal power plant.

3.1 Direct Mechanisms and Strategies

Direct mechanisms (e.g. increasing fuel efficiency, clean technology, renewable energy, alternative materials, green suppliers) can be applied directly to the sources contributing to GHGs; the application of such mechanisms directly reduces carbon footprint. The carbon footprint of thermal power plants could be reduced by a variety of direct mechanisms (Morrison 1989; Jaeger 1988):

3.1.1 Increasing the End-Use Efficiency of Fuel by Conservation and Improved Energy Conversion

Technologies for energy conservation and improved utilization include better insulation, cogeneration, and increased gas mileage. A reduction in overall energy consumption of up to 50 % is achievable with the available means. Traditionally, fossil fuel power stations have been designed around steam turbines to convert heat into electricity. Conversion efficiencies of new steam power stations can exceed 40 % of the lower heating value (LHV). New supercritical steam boiler, which are made with new materials, allow higher steam temperatures and pressures, thus enabling efficiencies of close to 50 %. In the long run, further improvements might be expected. Significant advancements have been made in combined cycle gas turbines, leading to increase in overall efficiency. The latest designs currently under construction can achieve efficiencies of more than 60 % LHV. All of these efficiency improvements result in a reduction of specific emissions per MW h. Hence, there is a potential for up to a 30 % reduction in CO_2 emissions by raising the overall efficiency from the 40 level to the 60 % level (ALSTOM 2000).

3.1.2 Replace Fossil Fuels with Renewable Energy Sources

Appropriate technologies are available for renewable energy sources, but the optimum system (e.g. wind, solar, or nuclear energy for electric power; solar thermal energy for domestic hot water) depends on geographic location and end use. Implementing this option would require a national energy plan with appropriate regulations and economic incentives.

3.1.3 Shift from Coal to Natural Gas

The combustion of natural gas produces approximately 40–50 % less CO_2 per unit energy delivered than the combustion of coal (Marland and Rotty 1983). Hence, switching fuel is at best a temporary amelioration and not a long-term solution. However, the easy availability and low cost of coal make it a preferable choice compared with the high price of imported natural gas. The carbon footprint scenario would drastically change in the case of a natural gas–powered thermal power plant because the emission factors of air pollutants for natural gas are starkly different from that of coal. At a natural gas power plant, the burning of natural gas produces NO_x and CO_2, but in lower quantities than burning coal or oil. Methane, a primary component of natural gas, can also be emitted when natural gas is not burnt completely, as well as the result of leaks and losses during transportation. Emissions of SO_2 and Hg compounds from burning natural gas are negligible. The US Environmental Protection Agency (EPA) estimates that average emissions rates in the United States from natural gas-fired generation are 1,135 lbs/MW h of CO_2, 0.1 lbs/MW h of SO_2 and 1.7 lbs/MW h of NO_x. Compared to the average air emissions from coal-fired generation, natural gas produces about half as much CO_2, less than a third as much NO_x, and one percent as much SO_x at the power plant (U.S.EPA eGRID 2000). But, in addition, the process of extraction, treatment, and transport of the natural gas to the power plant would generate additional emissions (USEPA 2013).

The emission factors of air pollutants are much less for natural gas than coal. However, the extent of carbon footprint reduction in a natural gas-powered thermal power plant by lighting energy conservation at the consumer level would be much less pronounced than at a coal-fired plant. Coal-fired power plants emit an estimated 1,747 MMT of CO_2-equivalent emissions, whereas natural gas–fired electricity generation accounted for approximately 373.1 MMT CO_2-equivalent emissions in the US (Worldwatch Institute 2011). The choice of GWP affects the relative life cycle GHG footprint of coal and gas; however, under all GWPs tested, the life cycle GHG footprint of gas is lower than coal. Natural gas–fired electricity is estimated to have 47 % lower life cycle GHG emissions than coal-fired electricity (Worldwatch Institute 2011).

3.1.4 Sequestration of CO_2

Liquid solvents, solid adsorbents, and separation processes could be used to remove CO_2 from flue gases. Although CO_2 emissions from coal-fired power plants could be reduced by up to 90 % by such mechanisms, power generation of those plants could be reduced by as much as 70 %. In addition to the extreme cost of these mechanisms, transportation and final disposal of the sequestered CO_2 pose unresolved problems.

3.2 Indirect Mechanisms and Strategies

Indirect mechanism or strategies (e.g. using efficient products; power saving in the consumer sector, such as households, offices, business centers, and commercial places) are not directly related to the sources. However, the application of these approaches would indirectly help in the reduction of carbon footprint by reducing the energy demand, which directly results in carbon footprint reduction.

4 Application of Indirect Mechanisms for Carbon Footprint Reduction at the Institutional Level

The whole issue of power conservation at the institutional level extends into the domain of environmental sustainability when the power saved here is reflected in the reduction of GHG emissions in a power plant, which would save coal and other fuels needed to produce the unnecessary electrical power. In its search for ecologically sustainable alternatives and recognition of climate change as a global issue, India stands committed to reduce its per capita greenhouse gas emissions below the levels of developed countries, even as it pursues its development objectives (BEE 2009). Because of the importance of energy conservation and its direct impact on the environment and natural resources, the Energy Conservation Act of 2001 was passed by the Indian parliament. The act provides a legal mandate to implement energy efficiency measures through the institutional mechanisms of the Bureau of Energy Efficiency in the Central Government and designated agencies in the states. Electrical energy savings at institute level are reflected in CO_2 emissions at power plants because it is the major gas emitted from fuel combustion using coal, the most important and widely used fuel in Indian power plants. The effects on other greenhouse gases, such as N_2O and CH_4, are less pronounced because their emissions are smaller. However, N_2O and CH_4 are more powerful greenhouse gases than CO_2, so even small reductions in their emissions may have equivalent or more pronounced effects on earth's radiative balance.

Electric lighting accounts for approximately 20 % of electricity consumption in India. The majority of these electric lights are incandescent tubes or bulbs, FTs, or general lighting systems, which all are of low electrical efficiency (BEE 2009). In recent years, more efficient CFLs have become popular. Another recent addition to lighting options is the LED.

Table 2 Comparison between fluorescent tubes (FTs), compact fluorescent lamps (CFLs), and light-emitting diodes (LEDs)

Description	FT	CFL	LED
Energy consumption	High	Comparatively very low	Comparatively very low
Electricity to light conversion	Less	High	High
Life (approximate) in hours	5,000	10,000	25,000[a]
Use of filament material	Yes	No	No
Use of mercury	Yes	Yes	No
Economy	Less	High	High
Required ballast for starting	Yes	No	No
Cost for same wattage for lifetime of light	High	Comparatively very low	Very high

[a] Although there are various types of LEDs available in the market (with reported lifetimes up to 60,000 h), a lifetime of 25,000 h was chosen here as a commonly reported value

4.1 Comparison of Fluorescent Tubes, Compact Fluorescent Lamps, and Light Emitting Diodes

CFLs are fluorescent lamps that fit into a standard incandescent light bulb socket. They consume less energy for the same lumen delivered than normal FTs or rods. Incandescent bulbs lose 95 % of the energy consumed as heat and give the rest as light, making them inefficient energy converters. In addition, they are fragile and short-lived, lasting only for 750–1,000 h of use (Freed 2007). On the other hand, a fluorescent lamp, tube, or rod is about 3 to 5 times more efficient than the standard incandescent lamp, can last longer, and is approximately 50 % more efficient (BEE 2009; Freed 2007; Devki 2006). FTs, CFLs, and LEDs are compared in Table 2 to highlight their respective advantages and limitations.

A CFL is known to convert a higher percentage of consumed electricity into light, consumes 2–2.5 times less energy for the same lumen output, and may last up to twice as long as the normal FT (BEE 2009). Nikola Tesla first introduced the fluorescent bulb at the World's Columbian Exposition in Chicago in 1893, but it was not commercially introduced until the 1980s in the United States (Freed 2007). The CFL is an energy-efficient alternative to incandescent bulbs and FTs (commonly known as bulb and tube lights, respectively, in India). The utility of CFLs has been popularly realized due to heightened awareness of electrical energy savings due to the economy. The working principle of a CFL is documented elsewhere (Freed 2007).

LEDs are the most promising of the new lighting technologies. The LED is a solid-state light source that emits white light (http://www.lightingafrica.org; accessed on 14.10.2013). LEDs also offer a number of other attributes that are highly desirable, including long service life, ruggedness, absence of mercury, low-voltage operation, compact/portable size, and a form factor that is well suited to directing light on a required task with very high optical efficiencies. LED light bulbs offer greater overall savings than FTs and even more than CFLs, using less energy and lasting 25,000 h. In terms of CO_2 emissions, LEDs emit only approximately 451 lbs/year, whereas CFLs emit approximately 1,051 lbs/year due to lower energy consumption.

Clean Development Mechanism released a new approved methodology (AMS-III-AR) for quantifying the carbon reductions of LED lighting systems in off-grid contexts (UNFCCC 2010); the underlying calculations were subsequently published by Mills and Jacobson (2011). Based on the minimum performance criteria specified in the new approved methodology, the savings could be approximately 0.16 tons of CO_2 per lamp (over a 2-year deemed service life). Therefore, LEDs may find increased market demand and applications worldwide. For example, 90 % of flashlights in parts of Kenya are based on LEDs (Johnstone et al. 2009). However, the efficiency of fuel-based lighting strategies, which are used in rural areas in many developing countries, can be as low as 0.04 lm/W—less than a thousandth of a modern LED light source.

4.2 Limitations of LEDs in Comparison with CFLs

LEDs, although more efficient than CFLs, have their own disadvantages. A greater initial investment is required for purchasing LEDs for same lumen than for CFLs and FTs. In addition, alternative fittings may be required; at present, they generally provide directional lighting. Also, most current commodity LED systems are low-price/low-quality products (Johnstone et al. 2009; Mink et al. 2010). Surveys of early adopters in Kenya showed that 87 % of LED flashlight buyers had problems within 6 months (Tracy et al. 2009). A market trial conducted in 2008 found that many of the lamps had failed by the time of a return visit 2 years later (Tracy et al. 2010a). However, with improvements, LED lights could become more viable at the consumer level, possibly emerging as the choice of future generations.

When replacing existing FTs, the use of well-established and proven lighting instruments, such as CFLs, is one way to ease energy demands because electrical energy can be conserved by the use of CFLs. The use of CFLs would imply that the same amount of electrical energy is saved for other uses or that some electricity production is avoided in a coal-fired thermal power plant, thus leading to lesser GHG and air pollutant emissions. A CFL is estimated to save 500–1,000 kg CO_2 and 4–8 kg of SO_2 emissions every year in the United States (Polsby 2003). Considering that several Indian coal types have high sulfur content and almost 75 % of Indian electricity generation comes from coal-fired power plants, the environmental benefits of using CFLs in India in terms of SO_2 emissions could be substantial (Kumar et al. 2003). Traditionally, the majority of government institutions and offices in India have used FTs because of their availability, popularity, and lower capital costs involved in their purchase. Furthermore, FTs are known for bright luminance and hence brighter interior lighting. CSIR-NEERI presently has approximately 1,350 FTs in the office and laboratories for lighting. Replacing the existing FTs in CSIR-NEERI with CFLs could be beneficial in terms of energy savings and the economy. From an environmental sustainability perspective, the use of CFLs could be very effective, leading to a minor to substantial reduction of emissions in coal-fired thermal power plants.

5 The Case Study at CSIR-NEERI

This case study was undertaken at NEERI, a research and development laboratory under the aegis of the CSIR, an autonomous research organization in India under Ministry of Science and Technology, Government of India. The institute is spread over 70 acres of land situated in Nagpur in the state of Maharashtra and takes its power supply from coal-based thermal power plants located in Maharashtra. The present study uses an indirect mechanism for calculating CF reduction, as well as estimates of the overall impact of a likely replacement of FTs by CFLs and LEDs in terms of electrical energy conservation at the organization level, the resultant reduction in power usage cost, and minimization in emissions of CO_2, other GHGs, radiatively active aerosol black carbon in a coal-fired thermal power plant. This chapter examines the impact of corrective energy-saving actions taken at the institutional level that would go on to create a substantial minimization of environmental costs in a thermal power plant by reducing its CF. This emission and energy inventory exercise can be replicated on smaller or larger scales in households, offices, commercial centers, and even in entire cities.

The emissions of CO_2, CH_4, and N_2O are also associated with many other operations in thermal power plants, such as kitchen/canteen-related combustions, transportation, and generator use. However, the calculations presented via this case study are related to the combustion of coal towards power production only. Furthermore, there may be emissions of other GHGs, such as SF_6 and chlorofluorocarbons, associated with some other select operations in a thermal power plant; however, because they are not the byproducts of coal combustion, they have not been considered for calculations of carbon footprint in this case study. Also, the possible reduction in the emissions of a few critical air pollutants along with the reductions in carbon footprint in a thermal power plant are also discussed.

5.1 Methodology

Emission inventory methodology was followed for GHG and air pollution reduction after estimating energy reduction in the institute by the likely replacement of existing FTs with CFLs/LEDs. Carbon footprint is related to GHG emission estimates only; therefore, it was calculated from the likely reduction in GHGs by the proposed replacement. The entire process of carbon footprint reduction calculation followed in this case study is presented in Fig. 1.

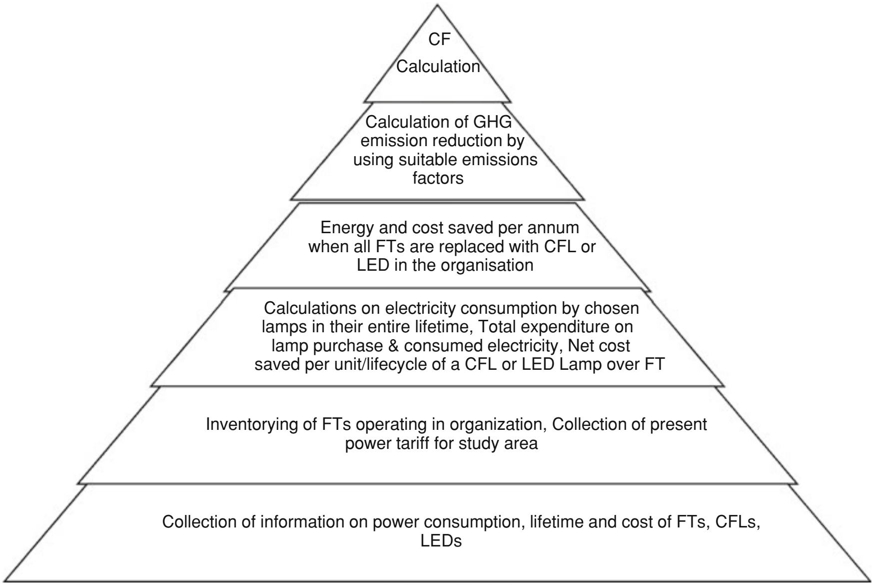

Fig. 1 Methodology followed for calculation of carbon footprint (CF) minimization in a thermal power plant by energy conservation at organization level

5.1.1 Electrical Energy Conservation and Economy

The study was undertaken in CSIR-NEERI to list and count the number of FTs being used. These lights have been installed in administrative offices, laboratories, scientists' cabins, the auditorium, bathrooms and toilets, the canteen, post office, bank, garages, vehicle stands, passages, verandas, workshops, and the guest house, which are all located within the institute. FTs installed in staff quarters and the colony were not included. The sockets with fluorescent rods were installed in recent years and were repaired or replaced when required. The necessary information regarding the total numbers of FT lights and frequency of replacement were collected from Electrical Division of Engineering Services Unit of the Institute. The electricity cost per unit of kW h energy consumption was collected from the approved tariff schedule of Maharashtra State Electricity Distribution Co. Ltd (2012). Electricity consumption and costs incurred for FTs and CFLs/LEDs based on their lifetime and the likely scenario of cost minimization per cycle of CFL lifetime was calculated (Table 3).

Table 4 estimates the annual cost savings in the institute in the likely scenario of complete replacement of FTs by CFLs/LEDs to estimate a direct impact on the annual cost savings of the institute. Two options exist for the replacement of FTs with CFLs/LEDs: (1) the replacement of the FTs with the CFLs/LEDs may be done at once completely or (2) replacement may be carried out in different phases. The second option assumes that all FTs are not withdrawn abruptly to introduce

Table 3 Economics of CFL/LED use compared with FTs based on lifetimes

Sr. no.	Particulars	Calculation
1.	Wattage for equal light from a FT[a], CFL, and LED (W)	A
2.	Life of FT/CFL/LED (h)	B
3.	Electricity consumption in 25,000 h (kW h)[b]	C = [A × B]/1000
4.	Total electricity cost incurred in lifetime at INR 10.45/kW h[c]	D = 10.45 × C
5.	Lamp purchase cost for 25,000 h (INR)[b]	E
6.	Total expenditure on purchase and electricity (INR)	F = D + E
7.	Net cost saved per unit life cycle of a lamp (INR)	F for FT—F for CFL/LED
8.	Cost saved per hour of CFL/LED operation (INR)	[(AW h of a FT—AW h of a CFL/LED)/1000] kW h × INR 10.45/kW h

[a] includes wattage of ballast in case of FT only
[b] 25,000 h is the highest lifetime amongst FT/CFL/LED; the calculation is based on the highest lifetime to put all three types on equal footing
[c] based on the tariff of Maharashtra State Electricity Distribution Co. Ltd (2012)

Table 4 Cost minimization per year by using CFLs/LEDs in place of FTs

Sr. no.	Particulars	Calculation
1.	Wattage for equal light (W)[a]	A
2.	Electricity consumption by a single light working for 2,600 h[b] per year at CSIR-NEERI (kW h)	B = [A × 2,600]/1000
3.	Energy saved per year when all FTs are replaced with CFLs/LEDs at CSIR-NEERI (kW h)	C = 1,350[c] × (B for FT—B for CFLs/LEDs)
4.	Total electricity cost saved by CFLs/LEDs per year at INR 10.45/unit[d] (INR)	D = 10.45 × C
5.	Total purchase cost of a lamp for an entire year (INR)	E = no. of units × price/unit
6.	Extra expenditure for CFLs/LEDs purchase (INR)	F = E for CFLs—E for FT
7.	Net monetary benefit for CFLs/LEDs use/year	D – F

[a] includes wattage of ballast
[b] working at 10 h a day for 260 working days a year (actual use may be greater)
[c] as per survey and data given by the engineering services electrical division of CSIR-NEERI
[d] based on the tariff of Maharashtra State Electricity Distribution Co. Ltd (2012)

CFLs/LEDs in their place; rather, FTs are slowly phased out by using CFLs/LEDs at the end of each FT's lifetime. Abrupt replacement would waste working FTs light hours and associated hardware (e.g. choke, ballast, starter), thus resulting in huge wastes of money. If these working items could be auctioned, part of the capital costs needed to replace the FTs with CFLs/LEDs could be recovered. Thus, the second option is recommended if FTs can be slowly replaced by CFLs within a year or two. Before adopting one of the options, it should be considered whether CFLs/LEDs come with a replacement warranty period of a minimum of 1 year for any malfunctioning; in that case, the lifespan of the CFLs/LEDs will be automatically increased by the duration of use for the older one.

To determine the better option in terms of economic and energy conservation, a payback period calculation has been used. In business and economics, a *payback period* refers to the period of time required for a return on an investment to "repay" the sum of the original investment. This aspect helps in better decision-making. The payback period was calculated as follows:

$$\text{Payback period} = \frac{\text{Total investment required for the replacement of all FTs with CFLs/LEDs}}{\text{Total electricity cost saved (INR) by CFLs/LEDs per year}} \tag{1}$$

5.1.2 Emission Factors

An emission factor is a representative value that relates to the quantity of a pollutant released into the atmosphere with an activity associated with the release of that pollutant. These factors are usually expressed as the weight of a pollutant divided by a unit weight, volume, distance, or duration of the activity emitting the pollutant. Such factors are used to estimate emissions from various sources of air pollution. These factors are averages of all authentic available data and are assumed to be representative of long-term averages of the source category. The emission factor is used to calculate the total emission from a source as an input for the emission inventory. The general U.S. EPA equation (http://www.epa.gov/air/aqmportal/management/emissions_inventory/emission_factor.htm; accessed on 6.10.2013) for emission estimation is $E = A \times EF \times (1 - ER/100)$, where E is emissions, A is the activity rate, EF is the emission factor, and ER is the overall emission reduction efficiency (%). Variations in the conditions at a given facility, such as the raw materials used, temperature of combustion, and emission controls, can significantly affect the emissions at an individual location. Whenever possible, the development of local emission factors is highly desirable.

For calculations of the reduction of emissions of GHGs and BC, emission factors for coal were resourced from scientific literature and suitable ones were chosen for the calculations. India-specific emission factors for the average Indian coal were used as shown in Table 5; various types of coals are used in Indian thermal power plants, with starkly different emission factors.

In this chapter, emission factors for CO_2, N_2O, and CH_4 (lbs/MBTU) were used (EIA 2001) for the general Indian coal type, which is considered to be sub-bituminous. The overall emission reduction of CO_2 is monumental and much more

Table 5 Emission factors of GHGs and BC for average Indian coal[a]

Agents with positive radiative forcing	Emission factor	Unit	References
CO_2	212.7	lbs $MBTU^{-1}$	Ohio Super Computer Centre (2008)[b]
CH_4	0.00141	lbs $MBTU^{-1}$	Energy Information Administration of US Department of Energy (2001)
N_2O	0.00326	lbs $MBTU^{-1}$	Energy Information Administration of US Department of Energy (2001)
BC	0.08	g kW h^{-1}	Energy Information Administration of US Department of Energy (2001)

[a] Average Indian coal is considered to be the coal used in the highest quantity in thermal power plants
[b] http://archive.osc.edu/research/archive/pcrm/emissions/summary.shtml; accessed 6.10.2013

pronounced than N_2O and CH_4. Evidently, the negative impact of CO_2 on the radiative budget of earth would also be much higher than the other two gases.

5.1.3 GHG Emission Reduction

The production of electricity in thermal power plants requires the combustion of fuel and generation of steam to drive turbines. The combustion of coal in power plants leads to emissions of large quantities of CO_2 along with small quantities of N_2O and CH_4. These gases are responsible for approximately 95 % of the energy-related emissions, whereas CO_2 from the energy sector represents approximately 80 % of the global anthropogenic GHG emissions (Roy et al. 2009). Indian thermal power plants connected to the national power grid are primarily run by the various Indian coal types, which are mostly of sub-bituminous in rank (Ohio Super Computer Centre 2008; Gupta 2009). The following formula was used to calculate the total coal energy that could be saved by complete replacement of FTs by CFLs/LEDs at CSIR-NEERI:

$$\begin{aligned}\text{k Cal coal energy saved/year} \\ &= \text{Average heat value of Indian coal (k Cal/kg)} \\ &\quad \times \text{ Specific coal consumption (kg/kW h)} \\ &\quad \times \text{ kW h power saved in CSIR - NEERI per year by using CFLs/LEDs}\end{aligned} \tag{2}$$

The overall performance of coal-based thermal power units were reported by Coal India Ltd.; their estimate reports the value of specific coal consumption (kg of coal/kW h power) as 0.705 for 2006–2007 (Coal India Ltd 2009), implying that 0.705 kg coal is burnt on an average in Indian power plants to generate a unit (kW h) of power. The Standing Committee on Energy (2001), part of the government of India, reported an average GCV of total coal dispatched by Coal India

Ltd to different sectors, including power, as 4,900 k Cal/kg. These average values were used in Eq. 2 to represent the condition in India. However, the heat value of actual coal used and specific coal consumption may vary marginally to appreciably in different power plants. After calculating k Cal coal energy saved per year by Eq. 2, it is converted to million British thermal units (MBTUs) of coal energy saved per year for the ease of application of the coal emission coefficients for CO_2, N_2O, and CH_4, which are reported in lbs gas/MBTU of coal by the Office of Integrated Analysis and Forecasting, Energy Information Administration in the U.S. Department of Energy (EIA 2001). By multiplying the respective gas emission coefficients (lbs/MBTU) with the MBTU coal power saved per year, reductions in individual gas emissions were calculated in kilograms per year. Subsequently, N_2O and CH_4 emission reductions (kg/year) were converted to CO_2–C equivalents by multiplying them with their respective GWPs for different time horizons reported by IPCC (2001). Calculations on Reduction of Air Pollutants in Power Plants.

Apart from greenhouse gases, coal-based thermal power plants also emit various air pollutants, among which SO_2 is a major one. The Ohio Super Computer Center (2008) studied the emissions from coal-fired thermal power plants in India; their calculations were used to generate an emission reduction scenario for electrical energy conservation at CSIR-NEERI. Based on the input parameters and the ultimate analysis of coal used for power generation, emissions of SO_2, NO, black carbon, and particulate matter from some of the prominent power plants in India were computed. Input parameters or operating conditions used were (i) actual air supplied, (ii) electric power generated per day, and (iii) coal used for unit power generation. Although thermal power plants sometimes use small quantities of diesel oil and furnace oil as supplementary fuels to boost the combustion and heat content, probable emissions from combustion of these supplementary fuels are not accounted for in the computations. For the estimation of emissions of the above-mentioned air pollutants from Indian thermal power plants, the available values of the ultimate analysis of coals used in the seven thermal power plants (Chandrapur, Dhanau, Singrauli, Dadri, Rihand, Kutch, and Nayveli) were used by the Ohio Super Computer Center (2008). Most thermal power plants in India use E- and F-grade coal only. The excess air used in the individual power plants, kilograms of coal used for unit (kW h) power generation, and per day power generation were used in these calculations.

SO_2 Emission Reduction

The Ohio Super Computer Center (2008) calculation indicated that the average SO_2 emission per unit of electricity generated from Indian thermal power plants was 7.4 g/kW h, although emissions varied between 4 and 31 g/kW h. Taking this average value, SO_2 emission minimization in a power plant by complete replacement of FTs by CFLs/LEDs at CSIR-NEERI was calculated as follows:

$$\begin{aligned} & SO_2\,\text{emission reduction (kg/year)} \\ & \quad = (\text{kW h electricity saved per year } \times 7.4\text{g/kW h})/1000 \end{aligned} \tag{3}$$

NO Emission Reduction

Emission calculations of another major air pollutant, NO, are based on the equilibrium reaction and an average gas temperature of 1,700 °K. However, in actuality, the gas temperature in the boiler varies from 900 to 2,500 °K and the reaction occurs in several phases. The Ohio Super Computer Center (2008) estimated that NO emissions for most power plants in India range from 6 to 10 g/kW h. Taking the mid-range value, NO emission minimization in a power plant by complete replacement of FTs by CFLs/LEDs at CSIR-NEERI is calculated as follows:

$$\begin{aligned} & \text{NO emission reduction (kg/year)} \\ & \quad = (\text{kW h electricity saved per year } \times \text{ 8 g/kW h})/1000 \end{aligned} \tag{4}$$

Black Carbon Emission Reduction

For soot or black carbon, the present model calculations show that emission factors were 0.08 g/kg of coal in Indian thermal power plants. This emission factor for coal is much lower than the average emission factors of 1.0, 0.325 and 0.2 g/kg proposed for underdeveloped, semi-developed, and developed countries, respectively, for industrial use (Mitra and Sharma 2002). However, the BC emission factor of 0.08 g/kg for coal obtained from the present calculations compares well with the emission factor of 0.075 g/kg proposed for the industrial use of hard coal (Bocola and Cirillo 1989; Williams 2001). The Nellore thermal power plant, with an estimated emission of 0.1 g/kW h, was found to be the largest emitter of soot. The other large-emitter thermal power plants included Faridabad, Harduaganj, Korba II and III, Kothagudem, Barauni, Muzaffarpur, and Talchar NTPC, where the soot emission ranged from 0.08 to 0.1 g/kW h. Taking the general emission factor for most Indian power plants to be 0.08 g/kW h (Ohio Super Computer Centre 2008), BC emission reduction by complete replacement of FTs by CFLs/LEDs at CSIR-NEERI was calculated as follows:

$$\begin{aligned} & \text{BC emission reduction (kg/year)} \\ & \quad = (\text{kW h electricity saved per year } \times \text{ 0.08/kW h})/1000 \end{aligned} \tag{5}$$

Suspended Particulate Matter Reduction

In terms of SPM, which is another significant air pollutant generated by coal-fired thermal power plants, the Ohio Supercomputer Center (2008) found that Chandrapur, Kothagudem, Nellore, Baauni, and Muzaffarpur thermal plants are amongst the largest emitters of SPM per unit electricity (i.e. 3–3.5 g SPM/kW h). Faridabad, Harduaganj, Obra, Panki, Paricha, Tanda, Korba II and III, Satpura, Ennore, Patratu, Calcutta, New Cossipore, Talchar NTPC, and IB Valley thermal power stations emit in the range of 2.5–3 g SPM/kW h. In the rest of the plants, SPM emissions are generally lower than 2.5 g/kW h. Taking the average emission factor of 3.0 g SPM/kW h within a range of 2.5–3.5 g SPM/kW h for Indian power plants, SPM emission reduction by complete replacement of FTs by CFLs/LEDs was calculated as follows:

$$\begin{aligned}&\text{SPM emission reduction (kg/year)}\\&\quad = (\text{kW h electricity saved per year } \times 3.0\ \text{g/kW h})/1000\end{aligned} \tag{6}$$

For PM_{10} emissions from coal-fired thermal power plants, several emission factors have been reported; these values vary by coal type, firing practice, and pollution control technology. Average U.S. coal steam-electric plants had a PM_{10} emission factor of 0.16 g/kW h in 1997, whereas the new coal steam-electric plants with the best available control technology was 0.15 g/kW h (USEPA 1996). In China, PM_{10} emission factors were reported to be 0.14, 0.15, and 0.15 g/kW h, respectively, for emissions controlled by only ESP, ESP and dry flue gas desulfurization (FGD), and ESP and wet FGD in some thermal power plants (USEPA 1996). Notably, all these emission factors are comparable and the modal value of these PM_{10} emission factors (i.e. 0.15 g/kW h) was chosen and used here for PM_{10} emission calculations. Using this emissions factor, the PM_{10} emission reduction was calculated using Eq. 6.

Mercury Emission Reduction

According to the U.S. EPA (2002), CFLs present an opportunity to prevent mercury (Hg) from entering air, where it mostly affects our health. The largest source of mercury in the atmosphere comes from burning fossil fuels such as coal. A CFL uses 75 % less energy than an incandescent light bulb and lasts at least six times longer. The EPA estimated that a power plant will emit 10 mg of Hg to produce the electricity to run an incandescent bulb, compared to only 2.4 mg of Hg to run a CFL for the same time. Figure 2 depicts Hg emissions affected by use of CFLs and incandescent lamps (USEPA 2002).

In view of the absence of any mercury in LED lights, the figure has been depicted only for incandescent bulbs and CFLs. Indian coal has been reported to have variable Hg content, ranging from 0.18 to 0.61 μg/g (average of 0.376 μg/g of coal) as per the Pollution Control Research Institute (2004) of Bharat Heavy Electricals, India. The Small Business Pollution Prevention Center (2006) of the Iowa Waste Reduction Center, USA, has reported an emission factor of 0.0000032 lbs/MBTU coal. This factor has been used to project Hg emission reduction per year by the same way, as calculated for greenhouse gases earlier.

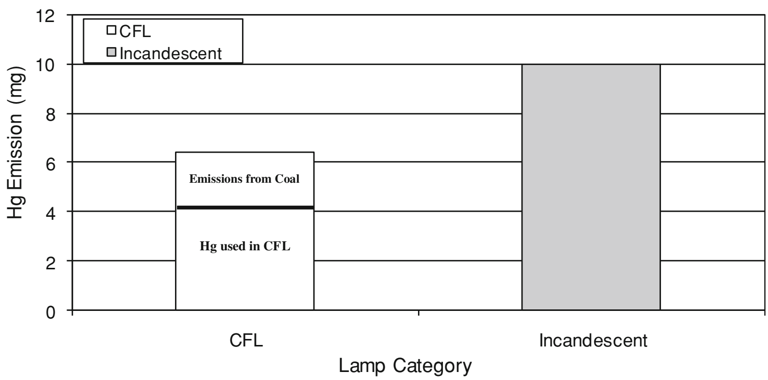

Fig. 2 Hg emissions affected by use of CFLs and incandescent lamps over a 5-year lifetime

5.1.4 Uncertainty and Assumptions

Uncertainties remain in all emission inventory calculations due to uncertainty in the emission factors that are used. The use of emission factors is not necessarily the best way to estimate emissions; however, given the practical difficulties in long and continuous emissions measurements of innumerable sources in many sectors, it is the only feasible option. Only the most suitable emission factors (amongst the available ones) have been used for calculations in this chapter. Further, the exact percentage of carbon in fuel directly affects CO_2 emissions; however, because coal composition can vary significantly for different sources and even for the same two batches from the same source, uncertainty is automatically incorporated in the estimates of actual CO_2 emissions from coal-fired systems or plants. The same is also applicable to SO_2, Hg, BC, and PM emissions because they depend on the sulfur, mercury, or carbon ash content of the fuels, which may vary appreciably (even from batch to batch). Moreover, some assumptions have been made for the calculations, as also indicated at the relevant places in the chapter:

1. An average GCV value of 4,900 k Cal/kg was used to represent average Indian coal, although the GCV of coal used in various power plants may also vary marginally but appreciably.
2. Emissions from the occasional combustion of diesel oil and furnace oil in power plants are not accounted for in the calculations.
3. The general Indian coal type is considered to be sub-bituminous; therefore, in particular, CO_2, N_2O, and CH_4 emission factors are used for this type of coal.

A unit rate power charge of INR 10.45 was considered for calculating the money saved annually. If other charges, such as demand charges, additional supply charges, energy charges, Time of Day (TOD) tariff Energy Charges (EC), fuel adjustment cost, which are also needed to pay for the consumed electricity, are considered, then the money saved annually would be substantially higher.

Table 6 Economics of CFL and LED usage *versus* FTs based on lifetime

Sr. no.	Particulars	Fluorescent Tube	CFL[e]	LED[e]
1.	Wattage for equal light (W)	55[a]	18	8
2.	Life (h)	5,000	10,000	25,000
3.	Electricity consumption in 25,000 h (kW h)	1,375	450	200
4.	Total electricity cost incurred in lifetime at INR. 10.45 /kW h[b]	14,369/- (US$232.34)	4,703/- (US$76.01)	2,090/- (US$33.78)
5.	Lamp purchase cost for 25,000 h (INR)	225/-[c] (US$3.64) (5 Lamps at INR 45 each)	375/- (US$6.06) (2.5 Lamps at INR 150 each)	895/- (US$14.47) (1 Lamp at INR 895 each)
6.	Total expenditure on purchase and electricity (INR)	14,594/- (US$235.88)	5,078/- (US$82.07)	2,985/- (US$48.25)
7.	Net cost saved per unit life cycle of a lamp (INR)	–	9,516/- (US$153.81)	11,609/- (US$187.63)
8.	Cost saved per hour of CFL/LED operation (INR)	–	0.39/- *For 1,350 FTs*: 522/- (US$8.44)	0.49/- *For 1,350 FTs*: 663/- (US$10.72)

[a] wattage includes 40 W for FT and 15 W for ballast
[b] based on the tariff of Maharashtra State Electricity Distribution Co. Ltd (2012)
[c] excluding the cost of ballast and starter
[e] CFLs and LEDs considered for equivalent FT lumens are from Bajaj Electrical Ltd

6 Results and Discussion

6.1 Electrical Energy Conservation and Cost Minimization

The electrical energy that could be saved if all FTs are replaced at the institute is quite substantial; this also reflects the amount of money saved. Taking a lamp's life cycle as the basis for calculating economics, Table 6 shows a benefit of INR 9,516/- (US$153.81) per unit for the CFL life cycle and INR 11,609/- (US$187.63) per unit for LED life cycle. Therefore, essentially, a CFL will not only run for approximately twice as long as an FT, but it will also lead to less expenditure or higher monetary gain in a single life cycle. The cost savings per hour of operation will be INR 522 (US$8.44) for CFL and INR 663 (US$10.72) for LED.

When calculating the annual economics (Table 7), it was observed that INR 1357,142/- (US$21,935.37) by CFL and 1723,937/- (US$27,863.85) by LED can be saved per year at the institute for just the power cost. The total electricity cost saved annually by LED is more than CFL by INR 366,795/- (US$5,928.47). However, taking into account the extra expenditure of INR 1005,750/- (US$16,255.86) required for the CFL versus LED, the net monetary benefit per year for CFL (INR 1215,392 (US$19,644.28)) is much more than LED (INR

Table 7 Cost and energy minimization per year through CFL/LED use in place of FT lights

Sr. no.	Particulars	Fluorescent tube lights	CFLs[e]	LEDs[e]
1.	Wattage for equal light (W)	55[a]	18	8
2.	Electricity consumption by a single light working for 2,600 h[b] per year at CSIR-NEERI (kW h)	143	46.8	20.8
3.	Energy saved per year when all 1,350[c] FTs are replaced with CFLs/LEDs at CSIR-NEERI (kW h)	–	129,870	164,970
4.	Total electricity cost saved (INR) by CFLs per year at INR10.45/kW h[d]	–	1357,142/- (US$21,935.37)	1723,937/- (US$27,863.85)
5.	Total purchase cost of lamps for an entire year (INR)	1,350[c] $\times$ 45 = 60,750/- (US$981.90)	1,350 $\times$ 150 = 202,500/- (US$3,272.99)	1,350 $\times$ 895 = 1208,250/- (US$19,528.85
6.	Extra expenditure for CFLs/LEDs purchase (INR)	–	141,750/- (US$2,291.09)	1147,500/- (US$18,546.95)
7.	Net monetary benefit for CFL/LED use per year	–	1215,392/- (US$19,644.28)	576,437/- (US$9,316.9)

[a] wattage includes 40 W for FTs and 15 W for ballasts
[b] working at 10 h a day for 260 working days a year
[c] based on the tariff of Maharashtra State Electricity Distribution Co. Ltd (2012)
[d] data given by the engineering services electrical division of CSIR-NEERI
[e] CFLs and LEDs as considered for equivalent FT lumens are from Bajaj Electrical Ltd
The exchange rate is US$1 = INR 61.87 (as of November 1, 2013). Values are rounded up to the next whole number because these are monetary values

Table 8 Carbon equivalent emission reduction per year in a power plant by electrical energy conservation through CFL use in CSIR-NEERI

Greenhouse gas	Emission coefficient (lbs gas/MBTU coal energy)	Emission reduction (kg/year)	CO_2 eq. emission reduction (kg/year)	CO_2–C eq. emission reduction (kg/year)
Time horizon—20 years				
CO_2 (1)[a]	212.7	171,991.85	171,991.85	46,906.87
N_2O (275)[a]	0.00326	2.64	726.00	197.71
CH_4 (62)[a]	0.00141	1.14	70.68	19.28
Total	–	–	**172,788.53**	**47,123.86**
Time horizon—100 years				
CO_2 (1)[a]	212.7	171,991.85	171,991.85	46,906.87
N_2O (296)[a]	0.00326	2.64	781.44	213.12
CH_4 (23)[a]	0.00141	1.14	26.22	7.15
Total	–	–	**172,799.51**	**47,127.14**
Time horizon—500 years				
CO_2 (1)[a]	212.7	171,991.85	171,991.85	46,906.87
N_2O (156)[a]	0.00326	2.64	411.84	112.32
CH_4 (7)[a]	0.00141	1.14	7.98	2.18
Total	–	–	**172,041.67**	**47,021.37**

[a] figures in parentheses are global warming potentials (GWPs) of the respective gases, relative to CO_2 (on wt. basis) (i.e. global warming contribution due to atmospheric emission of 1 kg of CH_4 or N_2O compared to emission of 1 kg of CO_2) (IPCC 2001)

576,437/- (US$9,316.90)). This makes the CFL option more economically beneficial than LED at the current time.

Considering the size of the CFLs in comparison to FTs, it is also imperative that lesser solid waste is generated when the CFLs complete their lifetimes and their use is discontinued.

The exchange rate is US$1 = INR 61.87 (as on November 1, 2013). Values are rounded up to the next whole number because these are monetary values.

From the payback period calculation from Eq. 1, it is obvious that the total investment required for replacement of all FTs will be recovered within 1 month in both cases. However, when comparing CFLs and LEDs, the payback period for CFLs is much faster than LEDs due to the higher investment price of LED lamps compared with CFLs. Thus, in terms of savings in the energy and electricity bills, replacement of FTs by CFLs is highly recommended.

6.2 Reduction of Air Pollutant Emissions in Power Plants

Apart from reducing the carbon footprint of a thermal power plant, low-power lighting also has the advantage of forcing a reduction in the emissions of air pollutants via a reduction in coal combustion in a thermal power plant. Any air pollutant that is emitted by coal combustion could be thus reduced, albeit to variable extents, depending on their emission factors.

Table 9 Carbon equivalent emission reductions per year in a power plant by electrical energy conservation through LED use in CSIR-NEERI

Greenhouse gas	Emission coefficient (lbs gas/MBTU coal energy)	Emission reduction (kg/year)	CO_2 eq. emission reduction (kg/year)	CO_2–C eq. emission reduction (kg/year)
Time Horizon—20 years				
CO_2 (1)[a]	212.7	218,476.14	218,476.14	59,584.40
N_2O (275)[a]	0.00326	3.35	921.25	251.25
CH_4 (62)[a]	0.00141	1.45	89.9	24.52
Total	–	–	**219,487.29**	**59,860.17**
Time horizon—100 years				
CO_2 (1)[a]	212.7	218,476.14	218,476.14	59,584.40
N_2O (296)[a]	0.00326	3.35	991.6	270.44
CH_4 (23)[a]	0.00141	1.45	33.35	9.10
Total	–	–	**219,501.09**	**59,863.94**
Time horizon—500 years				
CO_2 (1)[a]	212.7	218,476.14	218,476.14	59,584.40
N_2O (156)[a]	0.00326	3.35	522.6	142.53
CH_4 (7)[a]	0.00141	1.45	10.15	2.77
Total	–	–	**219,008.89**	**59,729.7**

[a] figures in parentheses are global warming potentials (GWPs) of the respective gases, relative to CO_2 (on wt. basis) (i.e. global warming contribution due to atmospheric emission of 1 kg of CH_4 or N_2O compared to emission of 1 kg of CO_2) (IPCC 2001)

Calculations based on the global warming potentials advocated by the IPCC (2001) shows the carbon equivalent emission reduction per year in a power plant by electrical energy conservation in CSIR-NEERI (Tables 8 and 9) through CFLs and LEDs, respectively.

Based on the selected emission factors, reductions of approximately 961.04, 1,038.96, 10.39, 389.61, 19.48, and 0.55 kg of SO_2, NO, BC, SPM, $PM_{10,}$ and Hg emissions per year, respectively, could be achieved through electricity conservation by replacing FTs with CFLs at CSIR-NEERI (Table 10). Similarly, for LEDs, reductions of approximately 1,220.78, 1,319.76, 13.20, 494.91, 24.75, and 0.7 kg of SO_2, NO, BC, SPM, $PM_{10,}$ and Hg emissions per year, respectively, could be achieved through electricity conservation by replacement of FTs at CSIR-NEERI.

In reality, the coal received at various power plants in India varies in quality and other factors, which may affect the emissions based on combustion efficiency, supplementary fuel use, etc. Therefore, the projections are extremely sensitive and prone to even short-term fluctuations. Moreover, the underlying assumption for generation in coal-fired power plant emission scenario is that no air pollution control system (APCS) is attached to the power plants, which is untrue. Most power plants are equipped with an advanced APCS, especially to control particulate emissions, such that the particulates (including black carbon) are reduced by 90 %, assuming the APCS is working at a good efficiency. As such, even if all these pollutants were formed on coal combustion, not all would escape through the

Table 10 Possible reduction in air pollution in a thermal power plant

Pollutant	Emission factors	Emission reduction (kg/year)	
		CFL	LED
SO_2	7.4 g/kW h	961.04	1,220.78
NO	8.0 g/kW h	1,038.96	1,319.76
BC	0.08 g/kW h	10.39	13.20
SPM	3.0 g/kW h	389.61	494.91
PM_{10}	0.15 g/kW h	19.48	24.75
Hg	0.0000032 lbs/MBTU coal	0.55	0.70

stacks in the same amounts, especially the particulates. Therefore, the current estimates are related to the reduction of air pollutant formation only in coal-fired thermal power plants, which may or may not enter into the atmosphere.

7 Summary and Conclusions

India is faced with the challenge of sustaining its rapid economic growth while dealing with the global threat of climate change. Climate change might alter the distribution and quality of India's natural resources and adversely affect the livelihood of its people. With an economy closely knitted to its natural resource base and climate-sensitive sectors, such as agriculture, water, and forestry, India may face threats in the future due to climate change. Effective emissions mitigation will require the entire country, regardless of energy demand and infrastructure, to use energy and manage natural resources in a sustainable manner. Because coal is the most abundant source available in India, it will continue to play a major role in future power generation. The issue of reducing greenhouse gases and air pollutants in a thermal power plant has great significance for climate change, along with issues such as cost and economy of resources. The cost of electricity from coal is expected to double by 2030 to US$40–55 per mega-watt hour (MW h). The additional cost of using carbon capture and storage for coal may raise the price to US$60–90 per MW h (IEA 2006). Some technology is available to limit CO_2 emissions, but it is extremely expensive. The options to limit the emission of CO_2 from electricity generation are to encourage reduction of the overall consumption of electricity through energy efficiency and conservation initiatives. Table 11 summarizes the benefits of replacing FTs with CFLs and LEDs.

Using efficient CFLs/LEDS in place of FTs can be one option for reducing energy consumption, but many other alternatives are available in the form of technologies as well as practices for saving energy consumption. Although this study was undertaken at the institute level (CSIR-NEERI), the benefits accrued from the replacement of FTs by CFLs/LEDs may be a good learning lesson for

Table 11 Summary of benefits accrued from the replacement of FTs by CFLs/LEDs at CSIR-NEERI

Items	Units	Savings (by CFL)	Savings (by LED)
Electricity consumption saved annually	kW h/year	129,870	164,970
Money saved annually	INR/year	1357,142/- (US$21,935.37)	1723,937/- (US$27,863.85)
GHG emissions saved (CO_2–C) annually from power plants	kg/year	47,123.86 (Time Horizon—20 years) 47,127.14 (Time Horizon—100 years) 47,021.37 (Time Horizon—500 years)	59,860.17 (Time Horizon—20 years) 59,863.94 (Time Horizon–100 years) 59,729.7 (Time Horizon—500 years)

other governmental and nongovernment institutes and establishments. Undertaking and implementing such practices would lead to energy conservation, energy economy, and environmental conservation.

However, the replacement of FTs with LEDs is a costly affair, mainly due to the high price of LEDs in the market. There is no economic advantage over FTs or CFLs. In due time, when LED technology becomes cheaper and market competition becomes more intense, the price is expected to come down. The option of LED usage may be considered only then. The carbon footprint reduction in power plants would be same as in case of CFLs for an equivalent wattage of 20 W. If the primary concern is price, CFLs would be a better choice because they consume almost the same amount of electricity; even though they have a shorter lifespan than LEDs (three CFLs are replaced to achieve the equivalent lifespan of a LED), the expenditure would still be much less than the purchase price of a LED.

On the advantageous side, LED lighting has the ability to distribute light evenly over a wide area. Therefore, if a specific minimum light level is needed over an entire parking structure, fewer LED fixtures could achieve this than fluorescent or another lighting technology. Furthermore, LED lighting technology is effective in cold temperatures, such as freezer storage areas; in sub-zero temperatures, fluorescent fixtures take a few minutes to warm up to full brightness. Because LEDs come up to full brightness instantly in sub-zero temperatures, they can be fitted with occupancy sensors that only turn them on when people are present. They can therefore cut energy consumption significantly by automatically turning off when no one is present (http://www.p-2.com/helpful-information/blog/370-is-led-the-most-efficient-lighting-technology/; accessed on 13.10.2013).

Acknowledgements The authors are grateful to the Head of Engineering Services (Electrical), CSIR-NEERI for providing requisite information for carrying out this study and also to the Head of Business Development Unit, Head, Air Pollution Division and Director, CSIR-NEERI for their continuous support and necessary guidance.

References

Ahmad RDR (2012) Carbon footprint initiative-development of a methodology on carbon inventory for TNB power plants. Stakeholder consultation workshop for national carbon disclosure programme, Malaysia, 12 July 2012

ALSTOM Power Technology (2000) Internal communication, Baden-Daettwil, Switzerland. www.alstom.com

Bocola W, Cirillo MC (1989) Air pollutants emissions by combustion processes in Italy. Atmos Environ 23:17–24

BP (2006) BP statistical review of world energy. http://www.bp.com

Bureau of Energy Efficiency (2009) Bachat lamp Yojana: CDM based CFL scheme. Ministry of Power, Government of India Report, www.bee-india.nic.in/bly/BLYconceptnoteandStatus_15Jan2009.doc

Central Electricity Authority (CEA) (2012) Load generation balance report 2011–2012. A report by Central Electricity Authority (CEA). Ministry of Power, Government of India, http://www.cea.nic.in/reports/yearly/lgbr_report.pdf

Coal India Ltd (CIL) (2009) http://www.coalindia.gov.in/customer2.htm. Accessed 31 Aug 2009

Devki Energy Consultancy Pvt Ltd (2006) Lighting. A report prepared for Bureau of Energy Efficiency and Indian Renewable Energy Development Agency, New Delhi, India

Energy Information Administration (EIA) (2001) Updated state-level greenhouse gas emission factors for electricity generation. Office of Integrated Analysis and Forecasting, Energy Information Administration, U.S. Department of Energy, USA

Energy Statistic Report (ESR) (2013) National statistical organisation ministry of statistics and programme implementation government of India http://mospi.nic.in/mospi_new/upload/Energy_Statistics_2013.pdf

European Platform on Life Cycle Assessment (2013) Carbon footprint—what it is and how to measure. European Commission Joint Research Centre. Institute for Environment and Sustainability. http://lct.jrc.ec.europa.eu/pdf-directory/Carbon-footprint.pdf. Accessed on 12 Oct 2013

Freed EC (2007) Ask the green architect: fluorescent bulbs. http://www.igreenbuild.com/cd_2855.aspx. Accessed 10 Oct 2013

Garg A, Kapse M, Shukla PR, Ghosh D (2002) Large point source (LPS) emissions from India: regional and sect oral analysis. Atmos Environ 36:213–224

Gupta P (2009) Baseline emission factor for coal based supercritical power plants. Central Electricity Authority, India www.cdmindia.com/admin/presentation/Praveen%20Gupta_305.ppt

Houghton J (1997) Global warming: the complete briefing. Cambridge University Press, UK

Intergovernmental Panel on Climate Change (IPCC) (2007a) Climate change: the physical science basis, summary for policymakers contribution of working group I to the fourth assessment report. IPCC Secretariat, Geneva, Switzerland

Intergovernmental Panel on Climate Change (IPCC) (2007b) Working group II contribution to the intergovernmental panel on climate change, fourth assessment report, climate change: climate change impacts, adaptation and vulnerability. IPCC Secretariat, Geneva, Switzerland

IPCC (2013) Climate change 2013, The physical science basis: summary for policymakers. Working group I contribution to the fifth assessment report of the Intergovernmental panel on climate change http://www.climate2013.org/images/uploads/WGI_AR5_SPM_brochure.pdf. Accessed 6 Oct 2013

International Energy Agency (IEA) (2012) Key world energy statistics 2012-report http://www.iea.org/publications/freepublications/publication/kwes.pdf. Accessed 6 Oct 2013

International Energy Agency (IEA) (2006). World energy outlook: focus on Asia. www.worldenergyoutlook.org. Accessed 10 Oct 2013

IPCC (2001) Climate change 2001: a scientific basis. In: Houghton JT, Ding Y, Griggs DJ, Noguer M, van de Linden PJ, Dai X, Johnson CA, Maskell K (eds) Intergovernmental panel on climate change. Cambridge University Press, Cambridge

IPCC (2005) Safeguarding the ozone layer and the global climate system: issues related to hydrofluorocarbons and perfluorocarbons. Intergovernmental panel on climate change. Cambridge University Press, UK

Jaeger J (1988) Developing policies for responding to climatic change. WMO/TD-No. 225, Beijer Institute, Stockholm, Sweden

Johnstone P, Tracy J, Jacobson A (2009) Pilot baseline study-report: market presence of off-grid lighting products in the towns of Kericho, Brooke, and Talek. Prepared for Lighting Africa

Kumar A, Jain SK, Bansal NK (2003) Disseminating energy-efficient technologies: a case study of compact fluorescent lamps (CFLs) in India. Energy Policy 31(3):259–272

Maharashtra State Electricity Distribution Co. Ltd (2012) Approved tariff schedule (with effect from August 1, 2012) www.Mahadiscom.In/Tariff/Tariff-Booklet-Aug-2012.Pdf. Accessed 15 Oct 2013

Marland G, Rotty R (1984). Carbon dioxide emission from fossil fuels: a procedure for estimation and results for 1950–1982. Tellus 36(b) 232–261

Marland G, Rotty RM (1983) Carbon dioxide emissions from fossil fuels: a procedure for estimation and results for 1950–1981, DOE/NBB- 0036. Oak Ridge Associated Universities, Oak Ridge

Mills E, Jacobson A (2011) From carbon to light: a new framework for estimating greenhouse gas emissions reductions from replacing fuel-based lighting with LED systems. Energ Effic 4:523–546

Mink T, Alstone P, Tracy J, Jacobson A (2010) Schatz Energy Research Center, Flashlights for sale in Kericho, Rift Valley Province, Kenya

Mitra AP, Sharma C (2002) Indian aerosols: present status. Chemosphere 49:1175–1190

Mittal ML, Sharma C, Singh R (2012) Estimates of emissions from coal fired thermal power plants in India. 20th emission inventory conference, August 13–16, 2012, Tampa, Florida, USA www.epa.gov/ttnchie1/conference/ei20/session5/mmittal.pdf. Accessed 15 Oct 2013

Morion J, Bozzuto C, Andrus H, Mccarthy M, Sundkvist SG, Griffin T (2003) Controlling fossil fuel power plant CO_2 emissions—near term and long range views. 2nd annual conference on carbon sequestration, Alexanadria, VA, USA, May 5–9

Morrison RE (1989) Global climate change, Congressional research service, Library of Congress, Washington, DC, Order Code IB 81005

Ohio Super Computer Centre (2008) A study on emissions of coal fired thermal power plant in India. http://www.osc.edu/research/archive/pcrm/emissions/coal.shtml

Pearce JM, Hanlon JT (2007) Energy conservation from systematic tire pressure regulation. Energy Policy 35:2673–2677

Pearce JM, Harris PJ (2007) Reducing greenhouse gas emissions by inducing energy conservation and distributed generation from elimination of electric utility customer charges. Energy Policy 35:6514–6525

Pollution Control Research Institute (PCRI) (2004) Assessment and development of environmental standards of heavy metals and trace elements emissions for coal based thermal power plants. Report no. PCI/001/2004. Pollution Control Research Institute, Bharat Heavy Electricals, Haridwar, India

Polsby E (2003) Marketplace: what to do when the lights go out. Home energy magazine online, November/December 1994 http://www.homeenergy.org/eehem/94/941115.html. Accessed 10 Oct 2013

Raghuvanshi SP, Chandra A, Raghav A (2006) Carbon dioxide emissions from coal based power generation in India. Energy Convers Manag 47:427–441

Roy J, Sarkar P, Biswas S, Choudhury A (2009) Predictive equations for CO_2 emission factors for coal combustion, their applicability in a thermal power plant and subsequent assessment of uncertainty in CO_2 estimation. Fuel 88:792–798

Small Business Pollution Prevention Centre (2006) Emissions for coal-fired power plants. Iowa Waste Reduction Centre, Iowa, USA

Standing Committee on Energy (SCE) Government of India, 2001 report http://164.100.24.208/ls/committeeR/Energy/18th/chapter6.htm. Accessed 11 Oct 2013

Tracy J, Alstone P, Jacobson A, Mills E (2010a) Market trial: off-grid LED lighting product market potential. Lumina project technical report no. 6. Lawrence Berkeley National Laboratory and Humboldt State University http://light.lbl.gov/pubs/tr/lumina-tr6-summary.html. Accessed 13 Oct 2013

Tracy J, Jacobson A, Mills E (2009) Research note #4; Quality and performance of LED flashlights in Kenya: common end user preferences and complaints. The Lumina Project

UNFCCC (2010) Approved methodologies for small-scale CDM project activities. AMS-III.AR: substituting fossil fuel based lighting with LED lighting systems–version http://cdm.unfccc.int/methodologies/DB/UMZGFR9COL8J0SRQXBVYR3DEM9F4TM. Accessed 12 Oct 2013

United State Environmental Protection Agency (USEPA) (2013) http://www.epa.gov/air/aqmportal/management/emissions_inventory/emission_factor.htm. Accessed 10 Oct 2013

USEPA (1996) Supplement B to compilation of air pollutant emission factors, vol I: stationary point and area sources, Technical report, Office of the Air Quality Planning and Standards, Research Triangle Park, NC

USEPA (2002) Fact sheet: mercury in compact fluorescent lamps (CFLs) http://www.gelighting.com/na/home_lighting/ask_us/downloads/MercuryInCFLs.pdf. Accessed 10 Oct 2013

Williams RH (2001) Toward zero emissions from coal in China. Energy Sustain Dev 5(4):39–65

Worldwatch Institute (2011) Comparing life cycle greenhouse gas emissions from natural gas & coal http://www.worldwatch.org/system/files/pdf/Natural_Gas_LCA_Update_082511.pdf. Accessed 10 Oct 2013

Assessment of Carbon Footprinting in the Wood Industry

Andreja Kutnar and Callum Hill

Abstract The management of natural resources is a subject that often arises when sustainable development is considered. Wood is a renewable, biological raw material used in numerous applications and is therefore growing in importance for sustainable development efforts. This chapter presents the applicability of carbon footprinting in the wood industry by comparing the carbon footprint of 14 primary wood products: air-dried and kiln-dried softwood and hardwood sawn timber, hard fiberboard, glued laminated timber for indoor and outdoor use, medium-density fiber board, oriented strand board, particleboard for indoor and outdoor use, plywood for indoor and outdoor use, and wood pellets. Furthermore, the use of timber products for the purposes of carbon storage and the effect of allocation methods on carbon footprinting are discussed. Additionally, the European policy strategies and actions directly impacting the forest products industry are discussed in relation to primary wood products. Also, wood as a building material and its placement in green building programs are considered.

Keywords Allocation · Carbon footprint · Carbon storage · Primary wood products · Sawn wood · Wood composites

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S. S. Muthu (ed.), *Assessment of Carbon Footprint in Different Industrial Sectors, Volume 2*, EcoProduction, DOI: 10.1007/978-981-4585-75-0_6,

1 Introduction

The European forest-based sector makes important contributions to Europe's sustainable, knowledge-based society by securing a renewable material supply and providing low-environmental-impact energy solutions. Wood use as a multifunctional material is expected to increase significantly in carbon-negative housing and furnishings, weight-efficient packaging and transportation, and heat and energy production, as well as being a raw material source for chemical production. Despite remarkable self-renewing capabilities, forests and their products cannot adequately provide enough raw materials for the growing global resource demands without significant improvements to resource utilization efficiency. The extension of material lifetimes by reasonable reuse and recycling loops has been identified as one of the most effective strategies for reducing pressure on resources. Furthermore, the European Union has set a goal of becoming a recycling society. The latest waste directive from 2008 (Directive 2008) contains an article requiring expanded reuse and recycling of materials, in addition to products. Amongst other things, it requires member countries to proceed with actions necessary to expand material and product recycling. To fulfill these requirements, simple recycling should be included in product design.

In the wood products sector, the waste hierarchy is presently underdeveloped and largely ignores the preferred option of maximizing the carbon storage potential of wooden materials. Reuse in solid form, with subsequent cycling of reclaimed wood in as many steps of material cascades as possible, is the best way to achieve the maximum carbon storage potential. Furthermore, the maintenance of natural resources is a subject that often appears when sustainable development is considered. In addition, as the world population increases and more nations develop economically, the strain on resources will continue to increase. As economic development and environmental pressures are linked, conserving both energy and resources has become paramount (Hill 2011).

In engineering, sustainable design is a design ideology that harbors the notion of sustainable human and societal development. However, every individual will approach the issue of sustainability in a different manner depending upon various factors, such as sustainability goals, background, awareness, and economic conditions. Resource sustainability can be defined as the development of opportunities for future generations to gain value from natural resources. One of the key aspects affecting efforts to become a sustainable society is construction. Sustainable construction principles are derived from ecological goals, which ideally produce buildings with no environmental impacts, a closed material loop, and full integration into the landscape after the service life of the structure is over. "Green buildings" represent the current efforts to achieve the sustainable construction ideal. According to the U.S. Environmental Protection Agency (EPA), Green Building is the "practice of creating structures and using processes that are environmentally responsible and resource-efficient throughout a building's life-cycle from siting to design, construction, operation, maintenance, renovation, and deconstruction." *Green building* is an ever-evolving, dynamic, and imprecise term; as technology evolves and new materials are developed, sustainability targets and the standards for what defines a green building also evolve. Furthermore, the role of life cycle assessment (LCA) in assessing the sustainability claims of green buildings and building materials is being introduced worldwide.

In this chapter, wood as building material, including the European Policy strategies and actions directly impacting the forest products industry, are discussed in relation to primary wood products. In total, 14 primary wood products are presented and their carbon footprints compared. Furthermore, the use of timber products for the purposes of carbon storage and allocation methods on carbon footprinting are discussed.

1.1 Wood as a Building Material

Wood is the most important renewable material resource. The utilization of wood in all aspects of human existence appears to be the most effective way to optimize the use of resources and to reduce the environmental impact associated with mankind's activities. Wood as a renewable biological raw material, used in numerous applications, is therefore gaining in importance.

Wood is the material of choice in many countries for residential and light commercial construction. In the United States, 90 % of the residential buildings are of wood-frame construction. Japan is not far behind. Wood use for construction, furniture, and other products aligns well with criteria for green building materials. Wood is a renewable resource, manufactured in nature using a large quantity of solar energy. Hence, no fossil fuels are required for the 'manufacturing' of wood. However, subsequently, processing of the wood will require an energy input that is often derived from fossil resources.

When waste wood is burned, it provides an independent source of energy. Energy from waste wood is converted solar energy (this is the embedded energy content), which has been stored in the wood since harvesting. Furthermore, the embodied energy associated with wood products is invariably lower when compared to other building materials, although this depends upon the number of subsequent processing steps for the wood product. For example, particleboard has a higher embodied energy than solid wood. At the end of the life of a wood product, it is possible to incinerate and use the embedded (i.e. trapped solar) energy, which is usually greater than the embodied energy. Consequently, when the carbon footprint of wood is calculated, the result is often a net benefit in that the atmospheric carbon stored is greater than that released to the atmosphere due to subsequent processing. Wood can be recycled, but not in the extensive manner of materials such as metals and glass. In most situations, the wood is downcycled to lower performance products. The production of wood is generally nonpolluting at all stages, although there have been instances in the past with polluted sites from chemical preservation processes (Buchanan 2006, 2010).

Another reason for building with wood is to increase the pool of carbon stored in wood and wood products. This is very important from a climate change standpoint. Green building programs often do not give proper credit to wood and its low embodied energy/carbon storage potential (Bowyer 2008). As a result, architects, builders, and contractors often overlook wood products. Within the green building sector, the wood industry must innovate and try to improve their market by creating a demand for new structural products.

Sustainability is increasingly becoming a key consideration of building practitioners, policy makers, and industry because the world has the aspiration of moving towards zero-energy construction. When buildings have net-zero energy consumption, the contribution of embodied energy and the associated greenhouse gas emissions become important. A zero-energy house can be built with different materials and construction methods that create different cumulative carbon footprints. Wood products can have a very low or negative carbon footprint. Therefore, the utilization of wood—the most important renewable material—in all aspects of human life appears to be the most effective way to optimize the use of resources and to reduce the environmental impact associated with mankind's activities.

Typically, the use of wood products results in lower greenhouse gas (GHG) emissions into the atmosphere than competing products and thus a lower overall environmental impact. However, to achieve sustainable development, certain criteria within a framework of economic, environmental, and social systems must be followed. It is important to note that only if wood is used effectively—through the whole value chain, from forest management and multiple use of forest resources through new wood and fiber-based materials, new processing technologies, and new end-use concepts, such as in the area of construction—can this lead to truly sustainable development. Therefore, research, development, and innovation related to "green" buildings should be informed through LCA analysis in all product stages, from primary processing, to use, through to disposal. Furthermore, research and development efforts should integrate knowledge and experience from various disciplines, engaging scientists from areas such as engineering, material science, forestry, environmental science, architecture, marketing, and business. These activities should be oriented towards new product development from renewable materials and utilization of the entire wood value chain, engineering solutions, and the cradle-to-cradle concept.

1.2 European Policy and Primary Wood Products

European policy is affecting and, indeed, directing current research, development, and marketing in the EU. Many policy strategies and actions directly affect the forest products industry. The main policies with direct impacts on the forest-based sector are the EU Sustainable Development Strategy (SDS, European Commission 2009), which was published in 2006 and reviewed in 2009; the EU Roadmap 2050 (European Commission 2011); and the recycling society directive (Directive 2008/

98/EC, European Parliament Council 2008). Additionally, with the support of the EU Commission, industry stakeholders created the Forest-based Sector Technology Platform (FTP). This group produced FTP Vision 2030 (Forest-based Sector Technology Platform 2013a, b), which is a strategy guide for the forest-based sector to help achieve the EU's goals of sustainable, inclusive growth.

1.2.1 Sustainable Development Strategy

The Sustainable Development Strategy (SDS) sets out a single, coherent strategy on how the EU will more effectively live up to its long-standing commitment to meet the challenges of sustainable development. It recognizes the need to gradually change the current unsustainable consumption and production patterns and move towards a more integrated approach to policy-making. It reaffirms the need for global solidarity and recognizes the importance of strengthening our work with partners outside the EU, including rapidly developing countries, which are expected to significantly impact global sustainable development. The overall intent of the SDS is to identify and develop actions to enable the EU to achieve continuous long-term improvement of quality of life. Specifically, the SDS calls for the creation of sustainable communities that are able to manage and use resources efficiently, tap the ecological and social innovation potential of the economy, and ultimately enjoy prosperity, environmental protection, and social cohesion.

1.2.2 Roadmap 2050

The Roadmap 2050 project mission is to provide a practical, independent, and objective analysis of pathways to achieve a low-carbon economy in Europe, which promotes energy security as well as the environmental and economic goals of the European Union. The Roadmap 2050 project is an initiative of the European Climate Foundation (ECF) and has been developed by a consortium of experts funded by the ECF. Roadmap 2050 breaks new ground by outlining plausible ways to achieve an 80 % reduction in greenhouse gas emissions from a broad European perspective, based on the best available facts elicited from industry members and academia; it was developed by a team of recognized experts rigorously applying established industry standards. Roadmap 2050 determines five priorities that must be established between 2010 and 2015 in order for Europe to progress towards implementation of an 80 % reduction target for greenhouse gas emissions by 2050:

(1) Energy efficiency (through aggressive energy-efficiency measures in buildings, industry, transport, power generation, agriculture, etc.)
(2) Low-carbon technology (development and deployment of offshore wind, biomass, electric vehicles, fuel cells, integrated heat pump and thermal storage systems, and networked high-voltage/direct-current technologies, including adoption of common standards, etc.)

(3) Advanced electricity grids and integrated market operation (i.e., an increase in regional integration and interconnection of electricity markets; effective transmission and distribution regulation, the development of regionally integrated approaches to planning and operation of grids and markets)
(4) Fuel shift in transport and buildings (fossil fuels are replaced in the building and transport sectors by decarbonized electricity and low CO_2 fuels, such as second-generation biofuels)
(5) Markets (a massive and sustained mobilization of investment into commercial low-carbon technologies)

1.2.3 European Recycling Society

The waste directive from 2008 (Directive 2008/98/EC) contains an article for the reuse and recycling of all consumer and industrial materials. Among other things, it requires member countries to proceed with the actions necessary to recycle materials as well as products. To fulfill these requirements, products should be developed with simple recycling as a product feature. In the wood products sector, the waste hierarchy is presently underdeveloped and largely ignores the EU's preferred option of maximizing the carbon storage potential of wooden materials by their reuse in solid form, with subsequent down-cycling of reclaimed wood in as many steps of a material cascade as possible (Leek 2010). At present in Europe, recovered wood volumes total approximately 55.4 million m^3. One third of this volume is burned for energy production, and one third is down-cycled and used for the production of particleboard, thus losing the favorable material properties of solid wood. The remaining (and largest) fraction of waste wood (20.4 million m^3) is not used at all at the moment in the EU27 and is landfilled (Leek 2010). However, this ignores the environmentally preferred option to maintain wood materials at a maximum quality level by reuse in solid form, therefore extending the carbon storage duration. This shortfall presents an opportunity for the forest-based sector to become a leader in achieving the European Commission's ambitious target of reduced CO_2 emissions with innovative production technologies, reduced energy consumption, increased wood product recycling, and the reuse and refining of side streams (e.g., manufacturing byproducts, such as sawdust as planer shavings).

1.2.4 Forest-based Sector Technology Platform

The FTP Vision 2030 supports the EU's Europe 2020 strategy for smart, sustainable and inclusive growth and identifies themes to address the 'grand societal challenges', as described by the European Commission, and drive towards the development of a bio-based society.

FTP Vision 2030 targets are grouped under four strategic themes that are essential for building a new forest-based sector in Europe by 2030. One of the themes, 'The forest-based sector in a bio-based society', is cross-cutting. The other three respond to a specific set of vision targets. These three strategic themes and specific vision targets are responsible management of forest resources, creating industrial leadership, and fulfilling consumer needs.

The European forest-based sector is directly affected by climate change, competition for wood resources, changing consumer demands, increasing competition, and the growing complexity of manufacturing processes. Traditional forest-based industries have used non-food renewable natural resources in a sustainable and responsible way; this growing and evolving sector now has great potential as a leader for a sustainable European bioeconomy in the future. The EU and the European forest-based sector can together contribute to achieving FTP Vision 2030 by implementing the revised Strategic Research and Innovation Agenda 2020 (SRA, Forest-based sector Technology Platform 2013a, b).

The SRA identifies strategic cross-sector alliances with other industries, investors, and public institutions as a vital role in the process. Open innovation concepts and methods that reach beyond the sector's usual technology providers, especially in the key area of enabling technologies (e.g., information and communication technologies, electronics, nanotechnology, sensor technologies and monitoring systems, advanced materials and manufacturing systems, industrial biotechnology) must be established to maintain the sector's competitive edge and accelerate development towards a bio-based society.

2 Primary Wood Products

Primary wood products are those produced directly from forest trees, including pulp, lumber, and wood composites. Wood composites are a family of materials that contain wood either in whole or fiber form as the basic constituent (Bodig and Jayne 1982). A binding adhesive of either natural or synthetic origin interconnects the wood or fiber elements. Composites are normally thought of as two-phase systems (i.e., particles interconnected by a binder); wood composites, however, are multiphase systems including moisture, voids, and additives. Furthermore, Berglund and Rowell (2005) defined a composite as two or more elements held together by a matrix. By this definition, what we call "solid wood" is also a composite. Solid wood is a three-dimensional composite composed of cellulose and hemicelluloses (with smaller amounts of inorganics and extractives), which are held together by a lignin matrix. The advantages of developing wood composites are to use smaller trees, to use waste wood from other processing, to remove defects, to create more uniform components, to develop composites that are stronger than the original solid wood, and to be able to make composites of different shapes.

Sawn softwood timber is most commonly used directly in structural applications or as a component of engineered products (e.g., glulams). Planed (also surfaced or dressed) timber has been machined to have a smooth, uniform surface and ensures proper sizing. Air-dried timber has been dried without mechanical aid, whereas kiln-dried timber has been dried with mechanical aid, often using co-generated electricity or natural gas as an energy source to provide heat and maintain regular air flow.

Conventional wood composites fall into five main categories based on the physical configuration of the wood: plywood, oriented strand board, particleboard, hardboard, and fiber board (Youngquist 1999). The performance of composites can be tailored to the end-use application of a product by optimally arranging the physical configuration of the wood, adjusting the density, varying the resin type and amount, and incorporating additives to increase water or fire resistance or to resist specific environmental conditions.

Because wood composites cover a wide field, it is hard to precisely define the term. Below, the description of various primary wood-based products, with accompanying carbon footprints presented in the following chapter, is summarized and simplified from Suchsland (2004) and descriptions given by Forest Products Laboratory (2010).

Hard fiberboard (also known as hardboard or high-density fiberboard [HDF]) is most often used for indoor, nonstructural applications, such as in furniture. This product is made by breaking wood (most often residues from other manufacturing processes) down to small fibers, then mixing the fibers with resin and wax to form mats that are compressed with pressure and heat. Hard fiberboard is very dense, typically more than 800 kg m^{-3}.

Glued laminated timbers are structural composite beams used to support large loads in building construction. Sawn timber, selected for stress-related mechanical properties, are glued and arranged in layers (with the high-grade timber in the outer layers and low-grade timber in the inner layers) with the grain direction parallel to the length of the timber. The size of the resulting glued laminated timbers may vary greatly, allowing the beams to be used as needed for a specific application. Glued laminated timbers for indoor use may use adhesives that are less resistant to the effects of the outdoor environment (e.g., relative humidity and temperature), while glued laminated timbers for outdoor use must use adhesives that are more resistant to changes in the outdoor environment.

Medium density fiberboard (MDF) is most often used for indoor, nonstructural applications, such as in furniture. This product is made by breaking wood (most often residues from other manufacturing processes) down to small fibers, then mixing the fibers with resin and wax to form mats that are compressed with pressure and heat. MDF density varies between 600 and 800 kg m^{-3}.

Oriented strand board (OSB) is a structural panel product most often used for roof, wall, and floor sheathing in construction. The product is made of usually made of three or more layers with strands in each layer oriented in alternating directions (i.e., parallel to the length of the panel or perpendicular to it). Water-resistant adhesives are used for OSB. The strands in the outer layer are oriented with the grain direction parallel to the length of the panel. The strands used are typically about three times longer than they are wide.

Particleboard is constructed by reducing wood product manufacturing residues (e.g., planer shavings, sawdust) and recycled wood products to small particles. Particle sizes often vary across the thickness of the board, with smaller particles in the outer layers and larger particles in the core layer. Particleboard is most commonly used for indoor uses, such as furniture, and has a density range of approximately 600–800 kg m^{-3}.

Plywood is made from thin layers of wood, which has been peeled from a log on a rotary lathe. These thin veneers are then combined in three or more (usually an odd number) of layers in alternating grain directions. The outer layers are aligned with the grain direction parallel to the length of the panel. Plywood for indoor applications may use an adhesive that is less water-resistant than plywood for outdoor use. In indoor applications, plywood is often used in furniture. Plywood for outdoor applications must use a water-resistant adhesive. Sheathing is the most common use of plywood in exterior applications.

Wood pellets are made by compressing wood residues from other manufacturing processes. Wood pellets are primarily used for industrial, commercial, and residential heating systems.

Wood-based composites have long been used as both decorative and structural components in the human environment. These materials extract the best properties of wood (and eliminate or minimize the defects) and combine them with other materials (adhesives, plastics, etc.) to create a wide variety of new products that meet market demands. In Europe, the most commonly produced wood based panels are particleboard and MDF. However, OSB, traditional plywood, insulation board, and hardboard are also important products. Other more recent products include laminated veneer lumber (LVL), light MDF (LDF), HDF, and cross-laminated timber (CLT). In the past years, technological innovations have advanced the field of wood-based panels. Most notably, hot pressing and the consequent viability of thermosetting resins have improved composites produced from particles and strands (particleboard, OSB), fibers (MDF, HDF) and veneers (plywood, LVL).

In spite of stronger regulations, the production of wood-based panels has recently experienced a dramatic, worldwide growth period. Europe and China each control more than 30 % of the worldwide capacity for wood-based panel production (Barbu and van Riet 2008). In Eastern Europe, new production is increasing, particularly in CIS and Turkey. In Western Europe, Germany is the main wood-based panel producer (25 %), followed by France and Poland (10 % each), then Italy and Spain (8 % each). Turkey has dramatically increased production and is now approaching Germany's capacity. Russia surpassed German production in 2011, but Germany may have latent capacity remaining from constricted production during the economic downturn (Forest-based Sector Technology Platform 2013a, b). Total European production was approximately 71 million m^3 in 2012, an increase of 14 % from 2002 (62 million m^3), but a decrease of 14 % from peak production in 2007 (81 million m^3) (Forest-based Sector Technology Platform 2013a, b). In Table 1, the European wood-based panel (excluding insulation boards), sawnwood, glulam, and wood pellets productions for 2012 are shown.

Table 1 European wood-based panel (WBP), sawnwood, glulam, and wood pellets production for 2012 (FAO 2013)

Product	Quantity (m^3)
Hardboard	4,408,653
MDF	11,852,683
Particleboard	45,243,727
Plywood[a]	3,204,944
OSB[a]	3,917,153
Total WBP	68,627,160
Sawn hardwood	13,533,427
Sawn softwood	126,751,739
Total sawnwood	140,285,166
Glulam[b]	4,800,000
Wood pellets[c]	9,262,990

[a] These numbers are from FAOStat, which combines plywood and OSB into one category. It was estimated OSB was 55 % of the total, and traditional plywood was the remaining 45 %

[b] Glulam estimate derived from graph 12.3.1 in the report for 2010: http://www.unece.org/fileadmin/DAM/timber/docs/tc-sessions/tc-65/md/presentations/19Dory.pdf

[c] Wood pellet quantity estimated from the report (2010 value): http://www.bioenergytrade.org/downloads/t40-global-wood-pellet-market-study_final.pdf (executive summary, Fig. 1.5, p. 8)

2.1 Environmental Impact of Primary Wood Products

With regard to greenhouse gas emissions, wood is a better alternative than other materials. Werner and Richter (2007) reviewed the results of approximately 20 years of international research on the environmental impact of the life cycle of wood products used in the building sector compared to functionally equivalent products from other materials. The study concluded that fossil fuel consumption, potential contributions to the greenhouse effect, and quantities of solid waste tend to be minor for wood products compared to competing products. Impregnated wood products tend to be more critical than comparative products with respect to toxicological effects and/or photo-generated smog depending on the type of preservative. Although composite wood products such as particle board or fiberboard make use of a larger share of the wood of a tree compared to products out of solid wood, there is a high consumption of fossil energy associated with the production of fibers and particles/chips as well as with the production of glues, resins, etc. Furthermore, wood is causing less emissions of SO_2 and generates less waste compared to the alternative materials (Petersen and Solberg 2005). However, treated wood, adhesively bonded wood, and coated wood might have toxicological impacts on human health and ecosystems.

Richter (2001) provided a comparison of environmental assessment data of different wood adhesives. The interventions increase from the polymerization adhesives to the polycondensation types. Within the polycondensation resins, the energy demand and emissions of substances increase with increasing percentage of aromatic compounds in the resin formulations. Limited LCA data have been published so far for resins based on renewable resources or components (e.g. tannins, lignins, proteins). A study of the use of a lignin-based phenolic adhesive in combination with a laccase initiating system has been conducted by Gonzalez-Garcia et al. (2011). This concluded that there was a significant impact associated with the enzyme production.

Incineration of wood products at the end of life provides various environmental benefits. The use of woody biomass as feedstock for biofuels production avoids the food versus fuel debate, which makes it more attractive from the environmental perspective (Wang 2005). However, Rivela et al. (2006a, b) applied a multicriterial approach in order to define the most adequate use of wood wastes. Based on environmental, economical, and social considerations, the study concluded that the use of forest residues in particleboard manufacture is more sustainable than their use as fuel. Cascading through several life cycles prior to incineration is a better option.

In a sensitivity analysis of an LCA of MDF manufacture, it was found that the final transport of product and the electricity generation profile had a significant influence upon the results (Rivela et al. 2007). A study of MDF production in a Brazilian context showed that the use of heavy fuel in the manufacturing process (including forestry operations) was the hotspot in all impact categories except ecotoxicity (Silva et al. 2013). Benetto et al. (2009) conducted an LCA of OSB production with emphasis on evaluating the environmental impact associated with a new wood drying process that had reduced emissions of volatile organic compounds. The study concluded that the environmental gains resulting from the new drying process were largely negated by changes required in the adhesive formulation. This shows the need to consider the whole process when considering the environmental impact of production and not focusing on making improvements of one part of the production. The combination of an OSB production plant with a biorefinery for the production of acetic acid and methanol has been studied from an LCA perspective recently (Earles et al. 2011). Significant reductions in human toxicity potential and freshwater ecotoxicity potential were recorded for the combined plant compared to a conventional OSB production process.

However, a renewable origin does not necessarily equate to environmental friendliness or sustainable use (Lindholm et al. 2010). Hall and Scrase (1998) provided a literature review concerning greenhouse gas and energy balances of bioenergy. The LCA study revealed that results may differ due to the type and management of raw materials, conversion technologies, end-use technologies, system boundaries, and reference energy systems with which the bioenergy chain is compared. A comprehensive sustainability assessment of biofuels is urgently needed to assess the economic, social, and environmental impacts of biofuel production and consumption (Halog 2009). Lindholm et al. (2010) modeled and calculated the environmental performance from an LCA prospective of different procurement chains of forest energy in Sweden. One of the conclusions of the study was that uncertainties and use of specific local factors for indirect effects (e.g., land-use change and nitrogen-based soil emissions) may give rise to wide ranges of final results.

Cherubini and Strømman (2011) performed a review of the recent bioenergy LCA literature. They concluded that most LCAs found a significant net reduction in greenhouse gas emissions and fossil energy consumption when bioenergy replaces fossil energy. Cherubini et al. (2009) explained the determination of energy balance and greenhouse gas emissions from bioenergy. The initial use of biomass for products followed by use for energy, known as cascading, can further enhance greenhouse gas savings, given what will be increasingly scarce resources of biomass. It has been shown that the environmental footprint associated with particleboard production can be reduced by using increasing amounts of recycled wood (Saravia-Cortez et al. 2013).

The number of LCA studies of wood-based composites is relatively limited, geographically distributed, and uses of a variety of databases and impact assessment protocols. A comparison between different production processes is not possible given the availability of information. Thus, a comparison of different production methods using common calculation rules is clearly required.

3 Carbon Footprint of Primary Wood Products

Following the common LCA methodology (ISO 14044, 2006), the scope and goal of the study was to compare the environmental impact of different primary wood products. The carbon footprint was chosen as indicator of environmental impact. Carbon footprinting summarizes the amount of GHG emissions caused by a particular activity or entity; it is also referred to as global warming potential (GWP). It is measured in tons (or kilograms) of carbon dioxide equivalent (CO_2e).

The comparison included 14 primary wood products: air-dried and kiln-dried softwood and hardwood sawn timber, hard fiberboard, glued laminated timber for indoor and outdoor use, medium-density fiber board, oriented strand board, particleboard for indoor and outdoor use, plywood for indoor and outdoor use, and wood pellets. The environmental impact of primary wood products was analyzed by the cradle-to-gate method, an assessment of a partial product life cycle that extends from manufacture ('cradle') to the factory gate (i.e., before it is transported to the consumer). Because the use phase and disposal phase of a product is highly dependent on the user and consequently the assumption of the product life cycle, the performance in use and life span are needed; the use phase and disposal phase of the product were omitted.

The environmental burdens associated with each primary wood product were considered from raw material acquisition through the manufacture/processing stages, accounting for the production and use of fuels, electricity, and heat, as well as the impact of transportation and distribution for all stages of the product supply chain. The functional unit for the calculation was 1 m^3. Data of energy inputs, raw materials, products, co-products, waste, and releases to air, water, and soil and the

Table 2 Life cycle inventory for carbon footprint calculations: 1 m^3 of sawn timber, hardwood, raw, kiln dried, $u = 10$ %, at plant/RER U (Ecoinvent 2.0)

	Quantity	Unit
Materials/fuels		
Electricity, medium voltage, production UCTE, at grid/UCTE U	33	kWh
Hardwood, allocation correction, 1/RER U	−0.136	m^3
Sawn timber, hardwood, raw, plant-debarked, $u = 70$ %, at plant/RER U	1.14	m^3
Technical wood drying, infrastructure/RER/I U	0.0000609	p
Wood chips, from industry, hardwood, burned in furnace 300 kW/CH U	1300	MJ
Emissions to air		
Heat, waste	119	MJ

Table 3 Life cycle inventory for carbon footprint calculations: 1 m^3 of sawn timber, hardwood, raw, air dried, $u = 20$ %, at plant/RER U (Ecoinvent 2.0)

	Quantity	Unit
Resources		
Occupation, industrial area, vegetation (land)	0.85	m^2a
Transformation, from unknown (land)	0.0085	m^2
Transformation, to industrial area, vegetation (land)	0.0085	m^2
Materials/fuels		
Hardwood, allocation correction, 1/RER U	−0.136	m^3
Sawn timber, hardwood, raw, plant-debarked, $u = 70$ %, at plant/RER U	1.14	m^3

upstream life cycle impacts of input materials were not specifically analyzed for this project. Instead, sound secondary life cycle data were sourced from the Ecoinvent database 2.0 (2010). In Tables 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14, 15, the life cycle inventory (LCI) of input/output data for the carbon footprint calculations for selected 14 primary wood products are given. The data collected were modeled in SimaPro (2009).

Carbon footprints were calculated with the methodology detailed in IPCC 2001 GWP 100a V1.02 (Climate Change 2001). IPCC 2007 contains the climate change factors of IPCC with a timeframe of 100 years. IPCC characterization accounts for the direct global warming potential of air emissions (excluding CH_4). They do not include indirect formation of dinitrogen monoxide from nitrogen emissions; do not account for radiative forcing due to emissions of NOx, water, sulfate, etc., in the lower stratosphere and upper troposphere; do not consider the range of indirect effects given by the IPCC; and do not include indirect effects of CO emissions. Embodied emissions do not include any offset for carbon stored in the timber materials.

In Table 16 and Fig. 1, the carbon footprints of selected primary wood products are presented. The products with the lowest carbon footprints are air-dried sawn timber, followed closely by kiln-dried sawn timber. This is unsurprising because these products are processed less than wood-based composites and require no adhesives. Wood-based composite production requires additional energy inputs to process raw materials, manufacturing byproducts, and recycled wood into the

Table 4 Life cycle inventory for carbon footprint calculations: 1 m^3 of sawn timber, softwood, raw, air dried, u = 20 %, at plant/RER U (Ecoinvent 2.0)

	Quantity	Unit
Resources		
Occupation, industrial area, vegetation (land)	0.749	m^2a
Transformation, from unknown (land)	0.00749	m^2
Transformation, to industrial area, vegetation (land)	0.00749	m^2
Materials/fuels		
Sawn timber, softwood, raw, forest-debarked, $u = 70$ %, at plant/RER U	1.1	m^3
Softwood, allocation correction, 1/RER U	−0.099	m^3

Table 5 Life cycle inventory for carbon footprint calculations: 1 m^3 of sawn timber, softwood, planed, air dried, at plant/RER U (Ecoinvent 2.0)

	Quantity	Unit
Materials/fuels		
Electricity, medium voltage, production UCTE, at grid/UCTE U	30.789	kWh
Planing mill/RER/I U	0.000000792	P
Sawn timber, softwood, raw, air dried, $u = 20$ %, at plant/RER U	1.1385	m^3
Softwood, allocation correction, 1/RER U	−0.138	m^3
Emissions to air		
Heat, waste	110.88	MJ

Table 6 Life cycle inventory for carbon footprint calculations: 1 m^3 of fiberboard hard, at plant/RER U (Ecoinvent 2.0)

	Quantity	Unit
Resources		
Water, cooling, unspecified natural origin/m^3 (in water)	0.18	m^3
Materials/fuels		
Aluminum sulfate, powder, at plant/RER U	0.9	Kg
Sodium hydroxide, 50 % in H2O, production mix, at plant/RER U	0.1	Kg
Paraffin, at plant/RER U	4.14	Kg
Electricity, medium voltage, production UCTE, at grid/UCTE U	408	kWh
Natural gas, burned in industrial furnace >100 kW/RER U	4140	MJ
Phenolic resin, at plant/RER U	9	Kg
Transport, lorry >16t, fleet average/RER U	99.5	Tkm
Transport, freight, rail/RER U	205	tkm
Industrial residue wood, mix, hardwood, $u = 40$ %, at plant/RER U	0.418	m^3
Industrial residue wood, mix, softwood, $u = 40$ %, at plant/RER U	1.25	m^3
Industrial wood, hardwood, under bark, $u = 80$ %, at forest road/RER U	0.16	m^3
Industrial wood, softwood, under bark, $u = 140$ %, at forest road/RER U	0.489	m^3
Wooden board manufacturing plant, organic bonded boards/RER/I U	3.33E-08	p
Emissions to air		
Heat, waste	1470	MJ
Waste to treatment		
Treatment, fiberboard production effluent, to wastewater treatment, class 3/CH U	0.799	m^3

Table 7 Life cycle inventory for carbon footprint calculations: 1 m^3 of glued laminated timber, indoor use, at plant/RER U (Ecoinvent 2.0)

	Quantity	Unit
Materials/fuels		
Diesel, burned in building machine/GLO U	33.6	MJ
Electricity, medium voltage, production UCTE, at grid/UCTE U	129	kWh
Heat, light fuel oil, at industrial furnace 1 MW/RER U	23	MJ
Sawn timber, softwood, raw, air dried, $u = 20$ %, at plant/RER U	1.37	m^3
Softwood, allocation correction, 1/RER U	−0.0522	m^3
Transport, freight, rail/RER U	81.2	tkm
Transport, lorry >16t, fleet average/RER U	38.2	tkm
Urea formaldehyde resin, at plant/RER U	12	kg
Wood chips, from industry, softwood, burned in furnace 300 kW/CH U	2680	MJ
Wood chips, softwood, from industry, $u = 40$ %, at plant/RER U	−0.84751	m^3
Wooden board manufacturing plant, organic bonded boards/RER/I U	3.33E-08	p
Emissions to air		
Formaldehyde	0.012	Kg
Heat, waste	463	MJ
Waste to treatment		
Disposal, polyurethane, 0.2 % water, to municipal incineration/CH U	0.974	Kg

Table 8 Life cycle inventory for carbon footprint calculations: 1 m^3 of glued laminated timber, outdoor use, at plant/RER U (Ecoinvent 2.0)

	Quantity	Unit
Materials/fuels		
Diesel, burned in building machine/GLO U	33.6	MJ
Electricity, medium voltage, production UCTE, at grid/UCTE U	129	kWh
Heat, light fuel oil, at industrial furnace 1 MW/RER U	23	MJ
Melamine formaldehyde resin, at plant/RER U	12	Kg
Sawn timber, softwood, raw, air dried, $u = 20$ %, at plant/RER U	1.37	m^3
Softwood, allocation correction, 1/RER U	−0.0553	m^3
Transport, freight, rail/RER U	81.2	tkm
Transport, lorry >16t, fleet average/RER U	38.2	tkm
Wood chips, from industry, softwood, burned in furnace 300 kW/CH U	2660	MJ
Wood chips, softwood, from industry, $u = 40$ %, at plant/RER U	−0.84056	m^3
Wooden board manufacturing plant, organic bonded boards/RER/I U	3.33E-08	p
Emissions to air		
Formaldehyde	0.012	Kg
Heat, waste	463	MJ

desired form, as well as adhesives and other additives to form the composite matrices, which considerably increases the carbon footprint of these wood products. The highest carbon footprint among the compared products was plywood for outdoor use, followed by hard fiberboard and plywood for indoor use.

In Figs. 2, 3, 4, 5, 6, 7, the emission contributions from different sources to the carbon footprints of 14 primary wood products are presented. The largest

Table 9 Life cycle inventory for carbon footprint calculations: 1 m^3 of medium-density fiberboard, at plant/RER U (Ecoinvent 2.0)

	Quantity	Unit
Resources		
Water, cooling, unspecified natural origin/m^3 (in water)	0.18	m^3
Materials/fuels		
Aluminum sulfate, powder, at plant/RER U	4.36	Kg
Paraffin, at plant/RER U	22.8	Kg
Electricity, medium voltage, production UCTE, at grid/UCTE U	355	kWh
Natural gas, burned in industrial furnace >100 kW/RER U	1670	MJ
Urea formaldehyde resin, at plant/RER U	49.6	kg
Transport, lorry >16t, fleet average/RER U	85.6	tkm
Transport, freight, rail/RER U	202	tkm
Wood chips, softwood, from industry, $u = 40$ %, at plant/RER U	−0.87564	m^3
Wood chips, from industry, softwood, burned in furnace 300 kW/CH U	2770	MJ
Industrial residue wood, mix, hardwood, $u = 40$ %, at plant/RER U	0.333	m^3
Industrial residue wood, mix, softwood, $u = 40$ %, at plant/RER U	0.998	m^3
Industrial wood, hardwood, under bark, $u = 80$ %, at forest road/RER U	0.127	m^3
Industrial wood, softwood, under bark, $u = 140$ %, at forest road/RER U	0.388	m^3
Wooden board manufacturing plant, organic bonded boards/RER/I U	3.33E-08	p
Emissions to air		
Formaldehyde	0.00927	kg
Heat, waste	1280	MJ

Table 10 Life cycle inventory for carbon footprint calculations: 1 m^3 of oriented strand board, at plant/RER U (Ecoinvent 2.0)

	Quantity	Unit
Materials/fuels		
Paraffin, at plant/RER U	5.3	kg
Diesel, burned in building machine/GLO U	15	MJ
Electricity, medium voltage, production UCTE, at grid/UCTE U	130	kWh
Natural gas, burned in industrial furnace >100 kW/RER U	203	MJ
Phenolic resin, at plant/RER U	44.7	kg
Transport, lorry >16t, fleet average/RER U	78.7	tkm
Transport, freight, rail/RER U	177	tkm
Wood chips, softwood, from industry, $u = 40$ %, at plant/RER U	−0.948	m3
Wood chips, from industry, softwood, burned in furnace 300 kW/CH U	3000	MJ
Industrial wood, softwood, under bark, $u = 140$ %, at forest road/RER U	1.19	m^3
Residual wood, softwood, under bark, air dried, $u = 20$ %, at forest road/RER U	0.115	m^3
Wooden board manufacturing plant, organic bonded boards/RER/I U	3.33E-08	p
Emissions to air		
Formaldehyde	0.00263	kg
Heat, waste	468	MJ

Table 11 Life cycle inventory for carbon footprint calculations: 1 m^3 of particle board, indoor use, at plant/RER U (Ecoinvent 2.0)

	Quantity	Unit
Resources		
Water, cooling, unspecified natural origin/m^3 (in water)	0.304	m^3
Materials/fuels		
Ammonia, liquid, at regional storehouse/RER U	0.64	kg
Hydrochloric acid, 30 % in H_2O, at plant/RER U	1.36	kg
Paraffin, at plant/RER U	11	kg
Electricity, medium voltage, production UCTE, at grid/UCTE U	104	kWh
Natural gas, burned in industrial furnace >100 kW/RER U	154	MJ
Heat, heavy fuel oil, at industrial furnace 1 MW/RER U	86	MJ
Heat, light fuel oil, at industrial furnace 1 MW/RER U	86	MJ
Urea formaldehyde resin, at plant/RER U	51	kg
Transport, lorry >16t, fleet average/RER U	63.3	tkm
Transport, freight, rail/RER U	152	tkm
Wood chips, softwood, from industry, $u = 40$ %, at plant/RER U	−0.34653	m^3
Wood chips, from industry, softwood, burned in furnace 300 kW/CH U	1100	MJ
Industrial residue wood, mix, hardwood, $u = 40$ %, at plant/RER U	0.217	m^3
Industrial residue wood, mix, softwood, $u = 40$ %, at plant/RER U	0.823	m^3
Industrial wood, hardwood, under bark, $u = 80$ %, at forest road/RER U	0.128	m^3
Industrial wood, softwood, under bark, $u = 140$ %, at forest road/RER U	0.215	m^3
Wooden board manufacturing plant, organic bonded boards/RER/I U	3.33E-08	p
Emissions to air		
Formaldehyde	0.003	kg
Heat, waste	375	MJ
Nonmethane volatile organic compounds, unspecified origin	0.166	kg
Particulates, <2.5 μm	0.0039	kg
Particulates, >10 μm	0.039	kg
Particulates, >2.5 μm, and <10 μm	0.0351	kg
Waste to treatment		
Treatment, particle board production effluent, to wastewater treatment, class 3/CH U	0.036	m^3

emissions source for both air-dried and kiln-dried sawn softwood timber is raw material processing, which includes harvesting (32.3 kg CO_2e), sawing (20.7 kg CO_2e), and the sawmill facility allocation (4 kg CO_2e). The increased raw material processing emissions for kiln-dried sawn softwood timber is due to the energy required for the drying process (18.7 kg CO_2e) (Figs. 2a, b). Manufacturing 1 m^3 of hardwood sawn timber results in a lower carbon footprint than softwood sawn timber. However, the raw material processing still accounts for the greatest contribution to the carbon footprint of air-dried hardwood sawn wood (Fig. 2c). As with softwood sawn timber, the kiln-drying process causes a significant increase in emissions (Fig. 2d).

In glued laminated timber, also known as glulam, emissions derive predominantly from timber harvest and initial lumber production of the softwood but also from the energy and adhesives required to bond the lumber (Fig. 3). Urea

Table 12 Life cycle inventory for carbon footprint calculations: 1 m^3 of particle board, outdoor use, at plant/RER U (Ecoinvent 2.0)

	Quantity	Unit
Resources		
Water, cooling, unspecified natural origin/m^3 (in water)	0.304	m^3
Materials/fuels		
Ammonia, liquid, at regional storehouse/RER U	0.64	Kg
Hydrochloric acid, 30 % in H_2O, at plant/RER U	1.36	Kg
Paraffin, at plant/RER U	11	Kg
Electricity, medium voltage, production UCTE, at grid/UCTE U	104	kWh
Natural gas, burned in industrial furnace >100 kW/RER U	154	MJ
Heavy fuel oil, burned in industrial furnace 1 MW, non-modulating/RER U	86	MJ
Light fuel oil, burned in industrial furnace 1 MW, non-modulating/RER U	86	MJ
Phenolic resin, at plant/RER U	51	Kg
Transport, lorry >16t, fleet average/RER U	63.3	Tkm
Transport, freight, rail/RER U	152	Tkm
Wood chips, softwood, from industry, u = 40 %, at plant/RER U	0.34653	m^3
Wood chips, from industry, softwood, burned in furnace 300 kW/CH U	1100	MJ
Industrial residue wood, mix, hardwood, u = 40 %, at plant/RER U	0.217	m^3
Industrial residue wood, mix, softwood, u = 40 %, at plant/RER U	0.823	m^3
Industrial wood, hardwood, under bark, u = 80 %, at forest road/RER U	0.128	m^3
Industrial wood, softwood, under bark, u = 140 %, at forest road/RER U	0.215	m^3
Wooden board manufacturing plant, organic bonded boards/RER/I U	3.33E-08	P
Emissions to air		
Formaldehyde	0.003	Kg
Heat, waste	375	MJ
Nonmethane volatile organic compounds, unspecified origin	0.166	Kg
Particulates, <2.5 μm	0.0039	Kg
Particulates, >10 μm	0.039	Kg
Particulates, >2.5 μm, and <10 μm	0.0351	Kg
Waste to treatment		
Treatment, particle board production effluent, to wastewater treatment, class 3/CH U	0.19	m^3

formaldehyde (UF) is the adhesive used for glued laminated timer for indoor use, which contributes 34.2 kg CO_2e (17 %) to the total carbon footprint of 1 m^3 of glued laminated timber (Fig. 3a). Melamine formaldehyde (MF) adhesive is used outdoor glued laminated timber. The MF adhesive has higher environmental impact then UF adhesive, which results in a higher carbon footprint of glued laminated timber for outdoor use (Fig. 3b). The MF adhesive contributes 55.2 kg CO_2e (24.8 %) to the carbon footprint of 1 m^3 of glued laminated timber for outdoor use.

For fiber composites (MDF and HDF), the extra energy required to convert the raw material to fibers, in addition to the energy required to apply pressure and heat to the products, is responsible for the bulk of the emissions from these products (Fig. 4a and b). However, the use of UF resin in MDF contributes significantly (28.5%) to the total carbon footprint of 1 m^3 of MDF board as well, despite

Table 13 Life cycle inventory for carbon footprint calculations: 1 m^3 of plywood, indoor use, at plant/RER U (Ecoinvent 2.0)

	Quantity	Unit
Resources		
Water, cooling, unspecified natural origin/m^3 (in water)	1.84	m^3
Materials/fuels		
Diesel, burned in building machine/GLO U	3.2	MJ
Electricity, medium voltage, production UCTE, at grid/UCTE U	306	kWh
Hardwood, allocation correction, 1/RER U	−1.32	m^3
Round wood, hardwood, under bark, $u = 70$ %, at forest road/RER U	2.7	m^3
Transport, freight, rail/RER U	348	tkm
Transport, lorry >16t, fleet average/RER U	157	tkm
Urea formaldehyde resin, at plant/RER U	83.2	kg
Wood chips, from industry, hardwood, burned in furnace 50 kW/CH U	8110	MJ
Wood chips, hardwood, from industry, $u = 40$ %, at plant/RER U	−1.9297	m^3
Wooden board manufacturing plant, organic bonded boards/RER/I U	3.33E-08	p
Emissions to air		
Formaldehyde	0.0832	kg
Heat, waste	1100	MJ
Waste to treatment		
Treatment, plywood production effluent, to wastewater treatment, class 3/CH U	1.84	m^3

Table 14 Life cycle inventory for carbon footprint calculations: 1 m^3 of plywood, outdoor use, at plant/RER U (Ecoinvent 2.0)

	Quantity	Unit
Resources		
Water, cooling, unspecified natural origin/m3 (in water)	1.84	m^3
Materials/fuels		
Diesel, burned in building machine/GLO U	3.2	MJ
Electricity, medium voltage, production UCTE, at grid/UCTE U	306	kWh
Hardwood, allocation correction, 1/RER U	−1.32	m^3
Melamine formaldehyde resin, at plant/RER U	83.2	Kg
Round wood, hardwood, under bark, $u = 70$ %, at forest road/RER U	2.7	m^3
Transport, freight, rail/RER U	348	Tkm
Transport, lorry >16t, fleet average/RER U	157	Tkm
Wood chips, from industry, hardwood, burned in furnace 50 kW/CH U	8110	MJ
Wood chips, hardwood, from industry, $u = 40$ %, at plant/RER U	−1.9297	m^3
Wooden board manufacturing plant, organic bonded boards/RER/I U	3.33E-08	P
Emissions to air		
Formaldehyde	0.0832	Kg
Heat, waste	1100	MJ
Waste to treatment		
Treatment, plywood production effluent, to wastewater treatment, class 3/CH U	1.84	m^3

Table 15 Life cycle inventory for carbon footprint calculations: 1 m^3 of wood pellets, $u = 10$ %, at storehouse/RER U (Ecoinvent 2.0)

	Quantity	Unit
Materials/fuels		
Electricity, medium voltage, production UCTE, at grid/UCTE U	164	kWh
Industrial residue wood, from planing, hard, air/kiln dried, $u = 10$ %, at plant/RER U	0.36	m^3
Industrial residue wood, from planing, softwood, kiln dried, $u = 10$ %, at plant/RER U	0.925	m^3
Transport, freight, rail/RER U	71.5	Tkm
Transport, lorry >16t, fleet average/RER U	35.8	Tkm
Wood pellet manufacturing, infrastructure/RER/I U	0.00000001	P
Emissions to air		
Heat, waste	591	MJ

Table 16 Carbon footprint of 1 m^3 of selected primary wood products from Ecoinvent 2.0 (2010)

Primary wood product	Carbon footprint (kg CO_2e)
Sawn timber, hardwood, raw, air dried, $u = 20$ %, at plant/RER U	57
Sawn timber, hardwood, raw, kiln dried, $u = 10$ %, at plant/RER U	79
Sawn timber, softwood, planed, air dried, at plant/RER U	85
Wood pellets, $u = 10$ %, at storehouse/RER U	103
Sawn timber, softwood, planed, kiln dried, at plant/RER U	104
Glued laminated timber, indoor use, at plant/RER U	204
Glued laminated timber, outdoor use, at plant/RER U	222
Particle board, indoor use, at plant/RER U	262
Oriented strand board, at plant/RER U	310
Particle board, outdoor use, at plant/RER U	329
Medium-density fiberboard, at plant/RER U	495
Plywood, indoor use, at plant/RER U	497
Fiberboard hard, at plant/RER U	581
Plywood, outdoor use, at plant/RER U	643

comprising only 10–20 % of the finished product. Paraffin, which is a hydrophobic agent that is present in small amounts (less than 1 %) in fiberboard, contributes 3.8 % of the total carbon footprint of 1 m^3 of MDF board. Compared to MDF, the carbon footprint of HDF board is higher due to higher energy consumption of the process (Fig. 4b).

In particle board and OSB, the main emission sources are adhesives (Fig. 5). Although the UF adhesive that is used in particle board for indoor applications only comprises approximately 6–9 % of the final product, it contributes 55.3 % to the total carbon footprint of 1 m^3 of particle board for indoor applications (Fig. 5a). Phenol formaldehyde (PF) adhesive is used for outdoor particleboard, which increases the share of carbon footprint attributed to the adhesive to 64.5 %

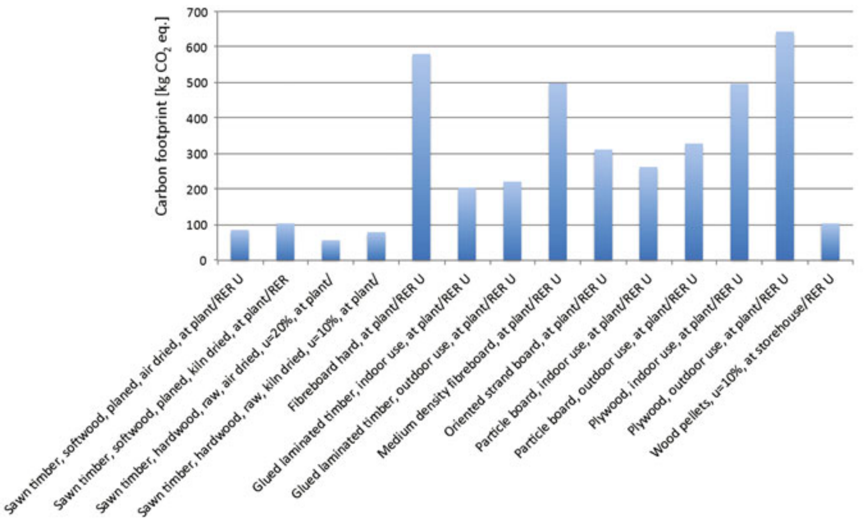

Fig. 1 Carbon footprint of selected primary wood products from Ecoinvent 2.0 (2010)

(Fig. 5b). PF adhesive is also used in OSB and accounts for 2–4 % of the product content, but contributes 59.6 % of the total carbon footprint (Fig. 5c). The marginally lower carbon footprint of OSB compared to particle board for outdoor applications is mainly a consequence of the lower adhesive content in OSB.

In plywood production, the main emission sources are the adhesives (Fig. 6). The UF adhesive in the plywood for indoor use contributes 47.7 % to total carbon footprint (Fig. 6a), whereas MF adhesive contributes 59.6 % to the total carbon footprint of plywood for outdoor use (Fig. 6b). The higher environmental impact of MF adhesive is the cause of the larger carbon footprint for outdoor plywood than for indoor plywood.

The main emission source during the production of wood pellets is the energy used during manufacturing, which includes compression (Fig. 7). Emissions are almost entirely from the energy demand during manufacturing because wood pellets are made mostly from manufacturing residues and contain no adhesives.

3.1 Carbon Storage

Trees capture atmospheric carbon dioxide via photosynthesis, and a proportion of this sequestered carbon is stored in the above-ground woody biomass. Wood is composed of three main biopolymers (cellulose, hemicellulose, and lignin). In a first approximation, the elementary composition can be assigned a stoichiometric ratio of CH_2O. This means that atmospheric carbon comprises a minimum of 40 % of the dry wood mass (increasing somewhat with increasing lignin content). Each

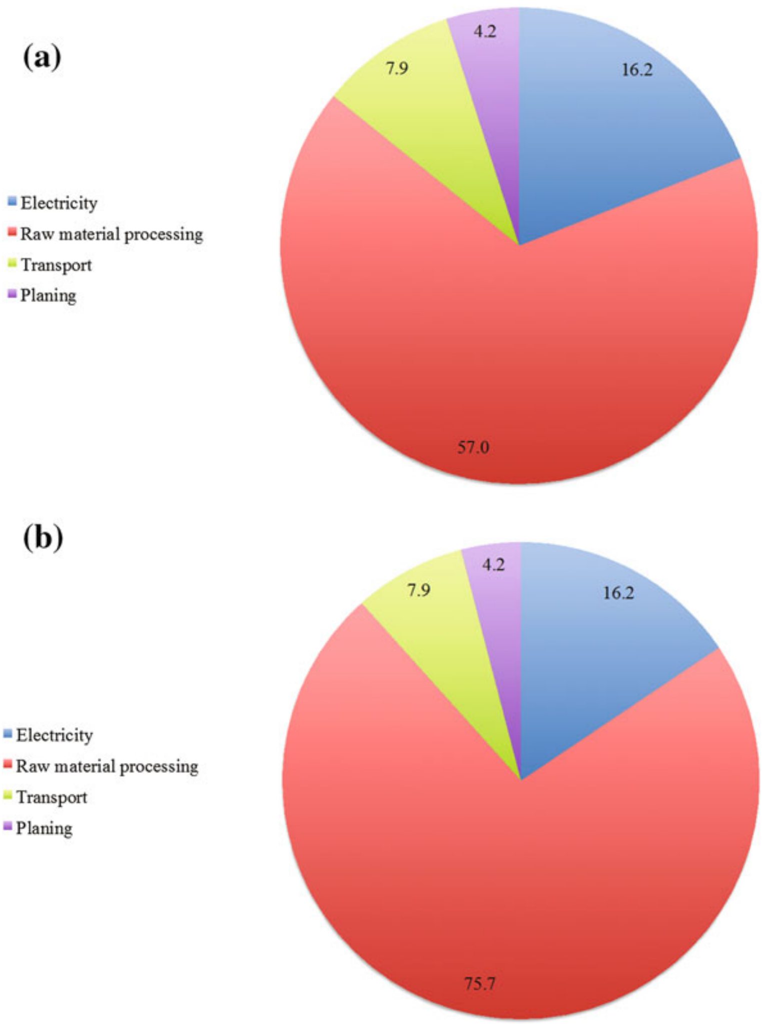

Fig. 2 Carbon footprint emission sources for 1 m^3 of sawn timber. **a** Air-dried softwood; **b** Kiln-dried softwood; **c** Air-dried hardwood; **d** Kiln-dried hardwood

ton of dry wood therefore equates to the removal of approximately 1.5 tons of atmospheric carbon dioxide (the ratio of the molecular weight of CO_2 compared to CH_2O: 44/30). The net benefit of this ability to store atmospheric carbon depends upon the length of time before the material is subsequently oxidized and the carbon released back to the atmosphere. In all situations where carbon flows and stocks are considered, it is essential that a distinction is made between biogenic and fossil carbon sources. Even with biogenic carbon, it is also important to differentiate between carbon that is held in long-term storage (such as old-growth forest) and that derived from newer managed or plantation forests.

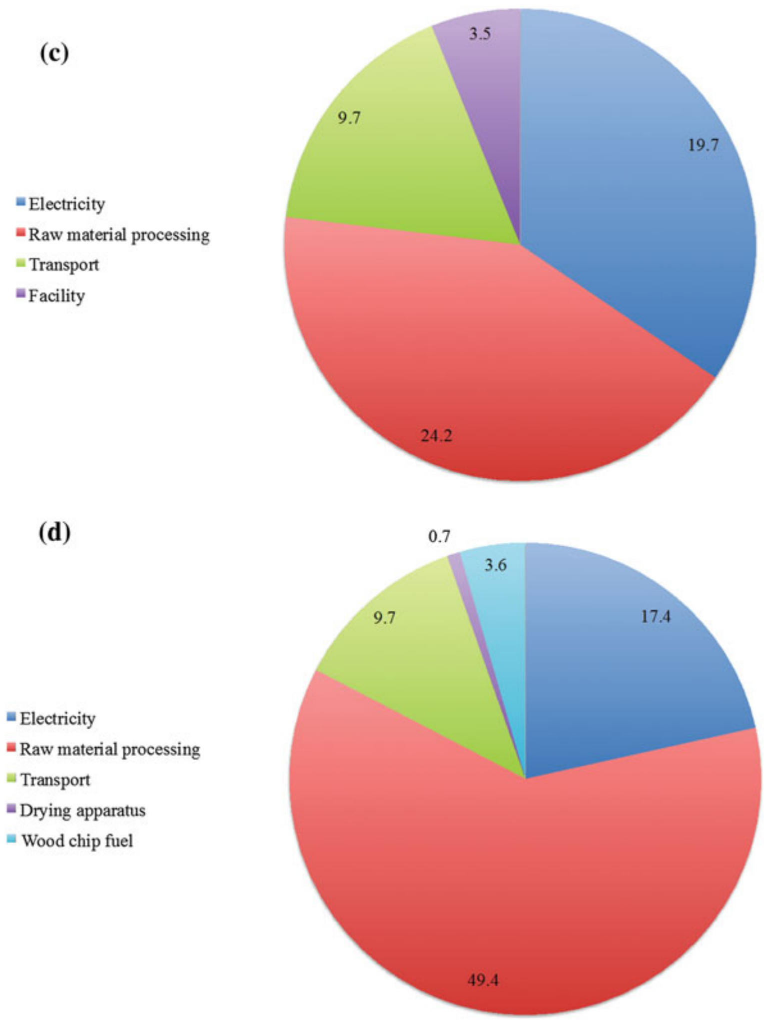

Fig. 2 continued

In Fig. 8, different scenarios for biogenic carbon storage and release are considered. In Fig. 8a, old-growth forest is burnt and the land is cleared for alternative use. The result is a release to the atmosphere of fossil carbon in the form of carbon dioxide (carbon stored in old-growth forest is treated the same as subterranean fossil carbon), which is shown as positive on the plot. This carbon content was previously held in long-term (historical) storage. Therefore, although technically this is biogenic carbon, it represents carbon that would have been in storage; prior to the industrial revolution, it was part of the natural biogenic cycle and can be considered equivalent to fossil carbon. The concentration of this 'fossil' carbon in

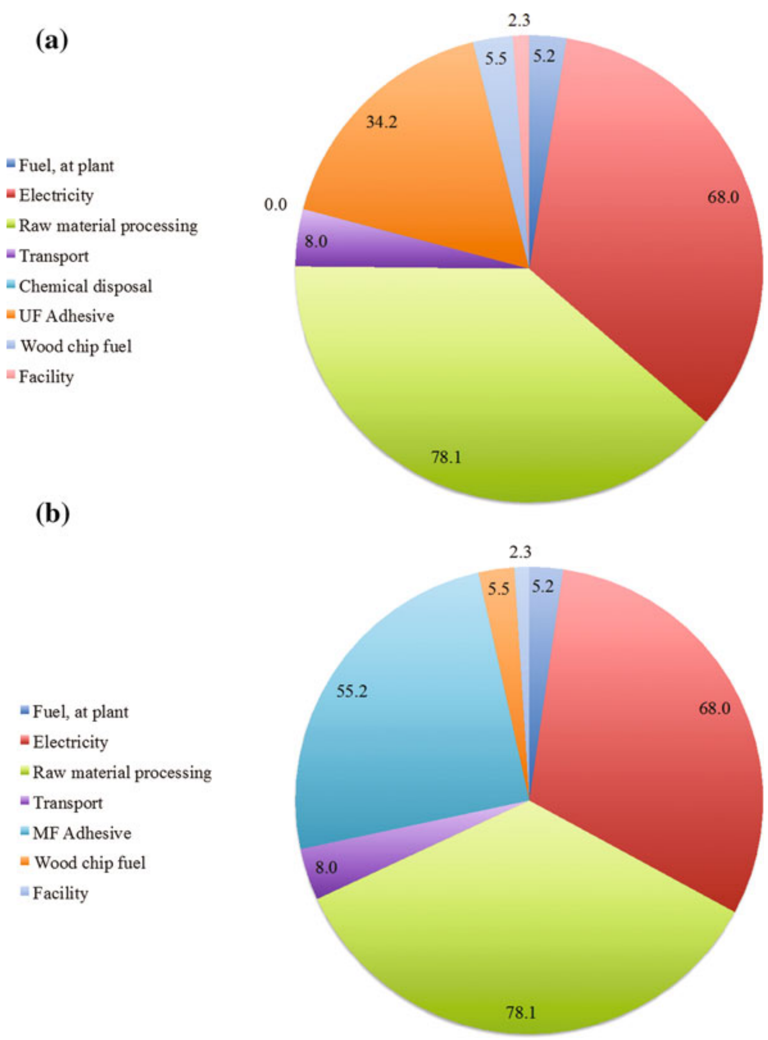

Fig. 3 Carbon footprint emission sources for 1 m^3 of glued laminated timber for indoor use (**a**) and glued laminated timber for outdoor use (**b**)

the atmosphere gradually decreases after the release (the Bern cycle) as it is removed by sequestration in oceanic and terrestrial sinks.

In Fig. 8b, a scenario is shown where a new forest plantation is established and the trees are allowed to grow for 50 years before harvesting and restocking. Carbon is removed from the atmosphere as the atmospheric carbon dioxide is photosynthetically bound in the biomass. The overall result is a benefit (shown as negative carbon) because atmospheric carbon dioxide has been sequestered. If the forest biomass is subsequently burnt with energy recovery after 50 years, then the

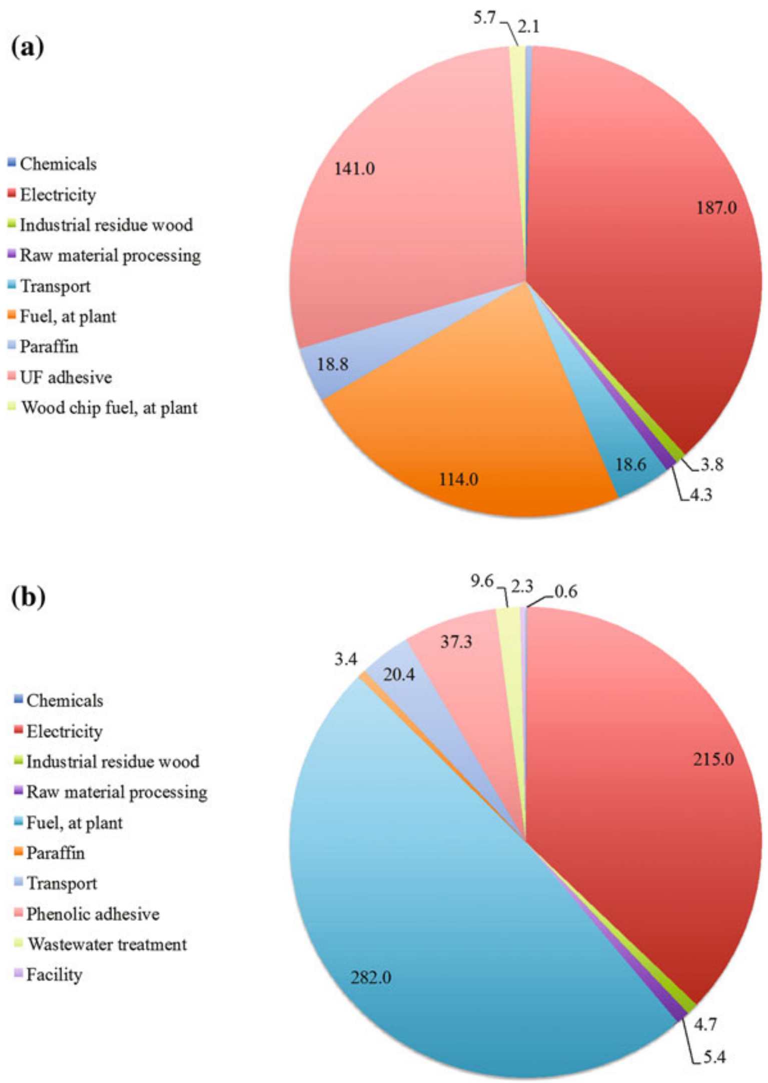

Fig. 4 Carbon footprint emission sources for 1 m^3 of medium-density fiberboard (**a**) and hard fiberboard (**b**)

above-ground biomass is oxidized and the accumulated atmospheric carbon is lost. The overall result is nonetheless still a benefit in terms of carbon sequestration. This is because there has been removal of atmospheric carbon dioxide during the 100-year period of consideration. When the aboveground biomass is subsequently burnt, this results in the return of atmospheric carbon dioxide. This only applies because new forest was created. However, the burning of virgin woody biomass cannot seriously be considered an effective mitigation strategy. Far better is one in

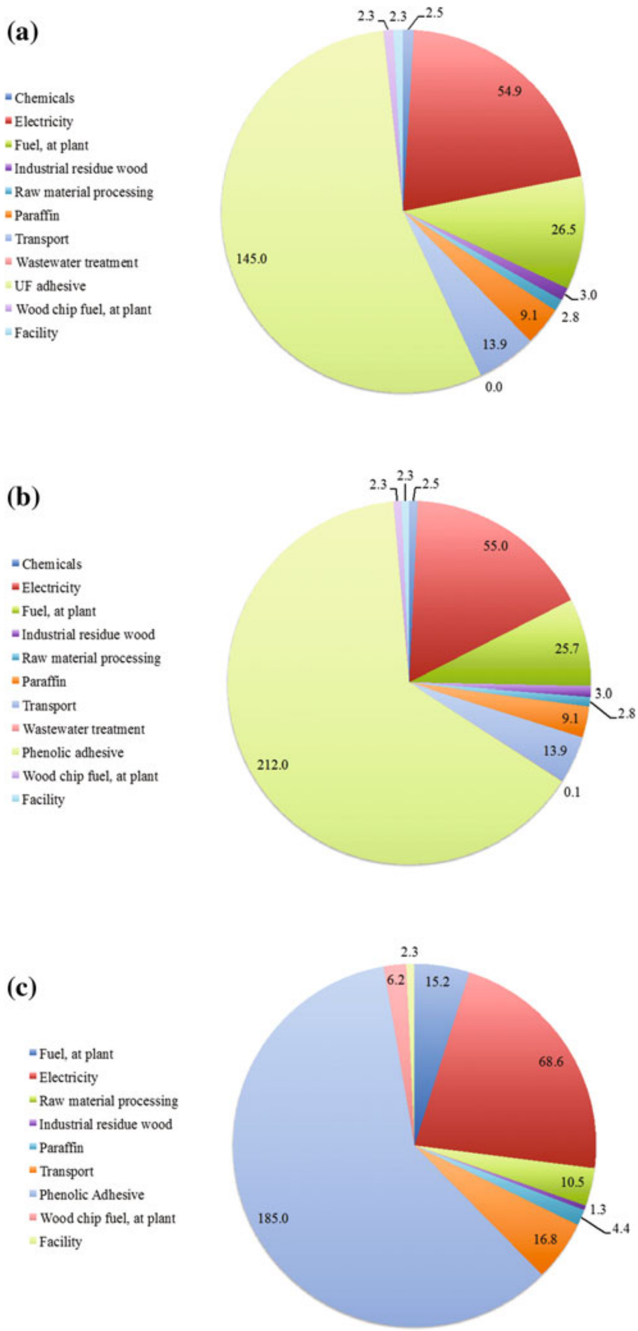

Fig. 5 Carbon footprint of emission sources and their contribution to the total carbon footprint of 1 m^3 of particle board for indoor use (**a**), particle board for outdoor use (**b**), and oriented strand board (**c**)

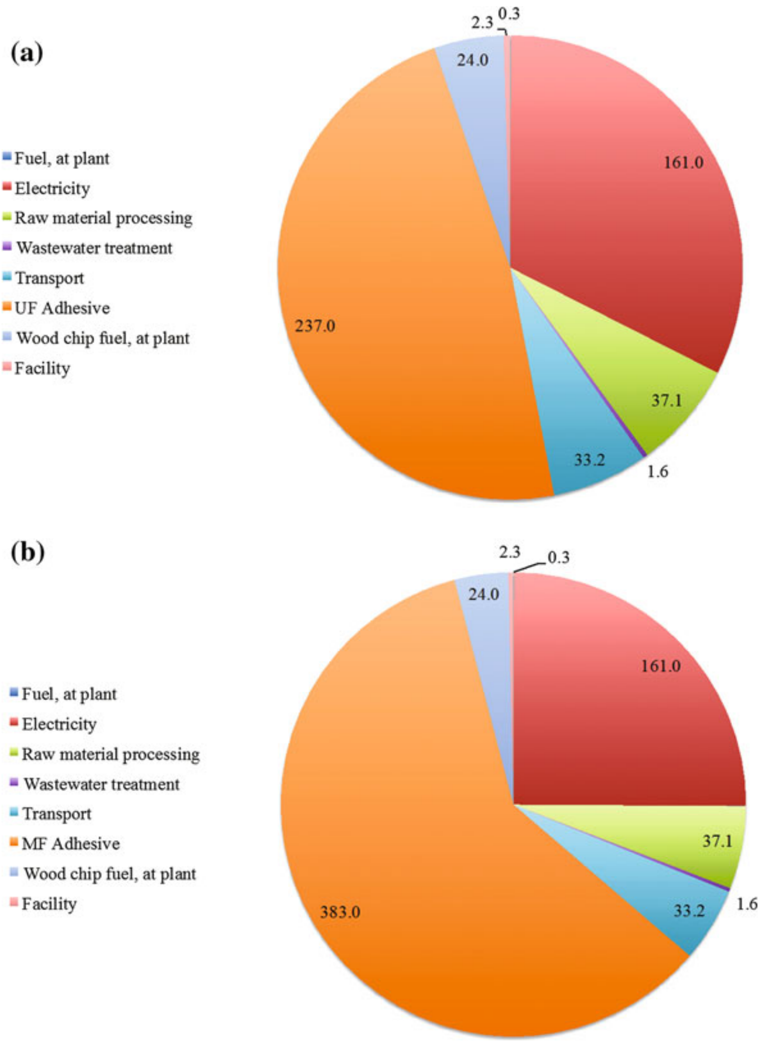

Fig. 6 Carbon footprint emission sources for 1 m^3 of plywood for indoor use (**a**) and plywood for outdoor use (**b**)

which the calorific value of the biomass is utilized and substituted for a fossil fuel alternative. The benefit then arises not only from the storage of atmospheric carbon in the growing biomass, but additionally from the avoided emission of the fossil carbon.

In Fig. 8c, the biogenic carbon embedded in the plantation forest is stored in timber products for 50 years, before it is used to generate energy. In this way, three benefits are realized. During the growth phase of the forest, carbon dioxide is sequestered due to the incremental growth of the trees. After harvesting, the carbon

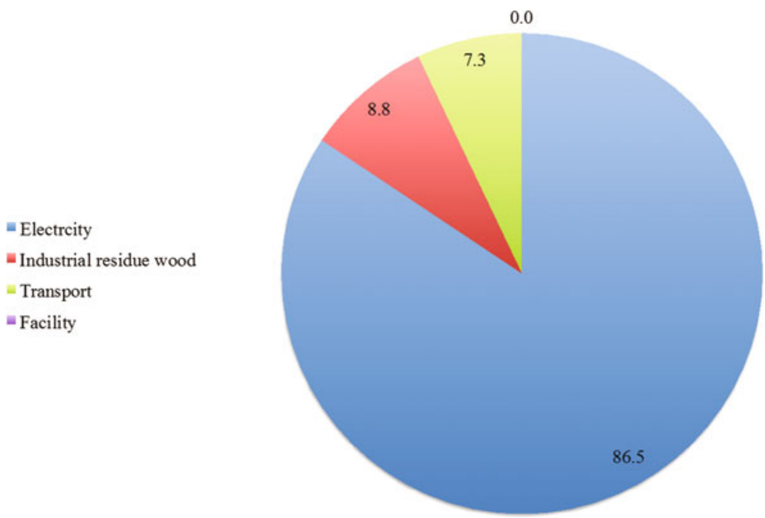

Fig. 7 Carbon footprint emission sources for 1 m^3 of wood pellets

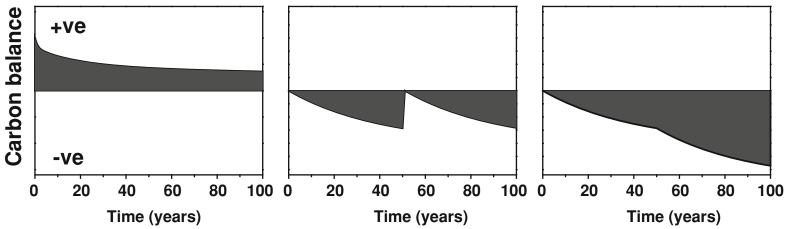

Fig. 8 Effect on carbon balance of burning old growth forest (**a**), burning plantation forest with a 50-year rotation (**b**), and using timber in long-life products (**c**)

continues to be stored in the timber products. It is only at the end of the life that this stored carbon is released into the atmosphere. Once again, if the wood is burnt with energy recovery, then there is also the benefit of the avoided emission of the fossil carbon. An even better option is to cascade the wood material down the product value chain through several life cycles before final incineration with energy recovery.

Although the storage of biogenic carbon clearly has benefits, it is necessary to consider an appropriate framework for reporting this. There has been some attempt to deal with the evaluation of biogenic carbon storage in long-life products in national standards. In the United Kingdom, this issue was dealt with in Publically Available Specification (PAS) 2050 (2011), which considers a 100-year assessment period following IPCC guidelines. Annex C of PAS 2050 (2011) describes the methodology to be used for calculating the storage of carbon in products. Two methods for calculating the weighted average of the effect of carbon storage in a product are given, although for a product with a life less than 2 years, no carbon storage benefit can be assigned. For products with a life of 2–25 years, a weighting factor is calculated, with a different weighting factor for other storage scenarios. This can only be applied to the storage of biogenic carbon, which is assigned a negative CO_2 value. However, this cannot be applied if the biogenic carbon is derived from old growth or native forests, where land use change has occurred. Emissions of biogenic carbon are not considered, because the origin of biogenic carbon is atmospheric carbon dioxide. Weighting factors are also applied for delayed release of GHGs.

In March 2011, the Construction Products Regulation (305/2011) was introduced, replacing the Construction Products Directive (89/106/EEC). The Construction Products Regulation states that if a European standard exists, it has to be used. In addition, it states that ‘for the assessment of the sustainable use of resources and of the impact of construction works on environment Environmental Product Declarations should be used when available.’ The Construction Products Regulation came into full force as of July 2013.

In order to develop a framework that allows for comparability of environmental performance between products, ISO 14025 (2009) was introduced. This describes the procedures required in order to produce Type III environmental declarations (EPD). This is based on the principle of developing product category rules (PCR), which specify how the information from an LCA is to be used to produce the EPD. A PCR will typically specify what the functional unit is to be for the product. Within the framework of ISO 14025, it is only necessary for the production phase (cradle to gate) of the lifecycle to be included in the EPD. It is also possible to include other lifecycle stages, such as the in-service stage and the end-of-life stage, but this is not compulsory.

ISO 14025 also gives guidance on the process of managing an EPD program. This requires program operators to set up a scheme for the publication of a PCR under the guidance of general program instructions. Until recently, PCRs have tended to be developed in an ad-hoc manner by different program operators, although there has been activity to harmonize the different rules. The situation now is one where European Standards are being introduced, which lay down the PCRs. For the construction sector core, the PCR is EN 15804 (2012). The standard that applies to sawn timber is the draft standard EN 16485 (2012), which at the time of writing has not yet been formally adopted. The draft standard allows for the reporting of sequestered carbon in timber products under the following conditions: ‘Consideration of the biogenic carbon-neutrality of wood is valid for wood from countries that have decided to account for Art. 3.4 of the Kyoto Protocol or which are operating under established sustainable forest management or certification schemes’. The methodologies for reporting sequestered carbon in timber products in EN 16485 (2012) are similar to those given in PAS 2050, in that different calculations are used for carbon stored in a product between 2 and 25 years and that stored in a product for 26–100 years. There is also a draft standard FprEN16449 (2013), which gives guidance on calculating the amount of sequestered carbon in timber.

Methodologies for accounting for the carbon stored in products are given in the International Reference Life Cycle Data (ILCD) Handbook, published by the European Commission Joint Research Centre (Institute for Environment and Sustainability), which also considers a 100-year assessment period. For carbon storage in products, the relevant sections are Sects. 7.4.3.6.4 and 7.4.3.7.3. It is recommended that fossil and biogenic carbon releases (e.g., CO_2 and CH_4) should be differentiated. Furthermore, all carbon emissions associated with land use changes and from biomass associated with virgin forests should be treated as fossil carbon. Emissions associated with plantation forests are to be inventoried as biogenic carbon. Uptake of atmospheric carbon dioxide is inventoried as 'resources from air'. A methodology is given for accounting for the removal and storage of atmospheric carbon dioxide. One of the issues discussed is that of carbon storage for a long period of time (e.g. 80 years) and how this then relates to the commonly used GWP100 parameter. GWP100 is a value given to the result of the emission of a pulse of a global warming gas in terms of its effect upon the environment for 100 years. Thus, if there is an emission of fossil-derived carbon dioxide into the atmosphere, its radiative forcing effect over a period of 100 years will gradually decrease as it is taken up by various natural sinks (the Bern cycle referred to earlier). For this reason, the parameter GWP100 is used (the global warming potential over a 100-year period).

In the case of carbon storage in a long-life material for 80 years, it would be incorrect to show the emission at end of life in terms of a GWP100 value because the total accounting time being considered is now 180 years. The ILCD methodology deals with this in the following way. The uptake of atmospheric carbon dioxide is inventoried as 'Carbon Dioxide–Resources from Air' and the emissions as 'Carbon Dioxide (biogenic)–Emissions to Air'. These two flows then cancel each other out. Meanwhile, the issue of the storage in the product is calculated by declaring a correction flow for delayed emission of the carbon dioxide and giving it a value of 0.01 times the CO_2 equivalent mass stored per year. The same method is used to calculate the storage of fossil carbon in a long-life product, except that there is no consideration given to the category 'Carbon Dioxide–Resources from Air.' Thus, there is a net effect of the release of the fossil derived CO_2 at the end of life, but the compensatory effect of the delayed emission of the fossil carbon is taken account of. With the introduction of Product Environmental Footprinting, it is likely that ILCD methodologies will be adopted.

4 Influence of Allocation Methods in Carbon Footprint Calculations of Wooden Products

When several products (or functions) from different product systems share the same unit process or group of unit processes, allocation may be required. Shared processes are often referred to as multifunction (or multifunctional) processes.

Allocation is needed in order to attribute the environmental load of the shared processes to the studied product and to each of the additional products delivered by the shared process. Allocation in general is defined in ISO 14040 (1997) as partitioning the input and/or output flows of a process to the product system under study. This means environmental aspects of the production process are apportioned to different co-products. Wherever possible, according ISO 14044 (2006), allocation should be avoided by either dividing the unit process or expanding the product system. If a process must be divided but data is not available, inputs and outputs of the verified system should be divided by its products or functions in such a way that separation shows basic physical relations among them. Where a physical relationship (i.e., mass, area or volume relationships) cannot be established or used as the basis for allocation, the inputs should be allocated between the products and the functions in a way that reflects other relationships between them, as defined in ISO 14041. For example, environmental input and output data might be allocated between co-products in proportion to the economic value of the products.

EN 15804 (2012) states that allocation should be based on physical properties (e.g., mass, volume) when the difference in revenue between co-products is low (of 1 % or less). In all other cases, allocation should be based on economic values. Furthermore, in EN 16485 (2012), allocation recommendations follow EN 15804 (2012), but different examples for the wood processing chain are given. According to EN 16485 (2012), allocations should respect the main purpose of the process studied and the purpose of the plant should be taken into account as well. Market prices from official statistics should be used for determination of revenues for assortments for which no company-specific prices are available. However, a discussion arises as impacts from allocation procedures differ between panels and sawmill industries. Concerning the different raw materials, processes, and co/by-products, a clear rule to harmonize the allocation procedures across all wood industry sectors should be determined in the future.

According to Jungmeier et al. (2002), it is generally agreed that environmental burdens should only be shared among products with a positive economic value—the products that are the intention of the process. Processes in the woodworking industry and manufacturing often produce multiple products. Those products can be either main products or by-products, and the environmental burden of the process should be distributed among these multiple products. As an example, the intended product of sawmills is sawn timber, but co-products with an assigned value, such as saw dust and wood chips, also accrue. The recommended procedure to account for the environmental impact of each of these products is to divide the unit process to be allocated into two or more subprocesses or to expand the product system to include additional functions related to the co-products. In some cases, it is not possible to use a wider approach and allocation within manufacturing processes has to be used. For instance, allocation would be required if an LCA focused on sawn timber production and it was necessary to determine the fraction of the environmental load associated with the sawmill that should be allocated to sawn timber versus to chips.

The treatment of allocation in LCA of wood-based products has been discussed for a long time and different solutions have been presented. It is generally accepted that different allocation procedures significantly influence the results of LCA of wood-based products.

Furthermore, wood is a renewable material that can be used for conventional wood products and energy production, among other uses. Consistent methodological procedures are needed in order to correctly address the entire product spectrum that wood products offer, multifunctional wood processing methods that generate large quantities of co-products (e.g. bark, wood chips), and reuse or recycling of paper and wood. Ten different processes in LCAs of wood-based products are identified where allocation questions can occur (Jungmeier et al. 2002): forestry, sawmill, wood industry, pulp and paper industry, particle board industry, recycling of paper, recycling of wood-based boards, recycling of waste wood, combined heat and power production, and landfill.

Mass and volume are usually used for physical allocation of wood-based products. Because moisture content varies in wood products and leads to enormous mass differences but negligible volume changes, volume should be considered instead of mass for allocation decisions. Different approaches to accounting for moisture content variances resulting from the inherent material properties of wood lead to deviating results. The moisture content of green wood is between 60 and 100 %, while most finished wood products show moisture contents between 7 and 20 %. Furthermore, co-products from the same process may have different moisture contents, which could directly affect the presumed physical relationships between them when allocation is based on mass and volume. On the other hand, the main problem of economic allocation is that, compared to mass or volume, prices are not stable and depend on and vary heavily with market conditions and fluctuations. Variations in the prices of sawn wood can be up to 10 % from year to year.

As a result of the COST Action E9 "Life cycle assessment of forestry and forest products," Jungmeier et al. (2002) provided the following recommendations for allocation in LCAs of wood-based products:

1. Energy and carbon content are characteristics of the wood and reflect the material and energy aspects of wood. A balance of the biological carbon and energy is necessary. Carbon uptake and the embodiment of energy as inherent material characteristics should always be allocated on a mass basis to avoid artefacts. The biogenic carbon neutrality does not necessarily indicate greenhouse gas neutrality, as carbon emissions can occur as methane or be derived from non-sustainable forestry.
2. Avoid allocation by an extension of system boundaries that combines material and energy aspects of wood. This means a combination of LCA of wood products and of energy from wood (bioenergy) with a functional unit for products and energy (e.g. 1 m^3 particle boards + 3 kWh energy).

3. Substitute energy from wood with conventional energy (e.g., energy from coal) in the LCA of wood products to get the functional unit of the wood product only (e.g. 1 m^3 particle boards), but identify the criteria for the substituted energy (e.g., kind and quality of energy, state of technology).
4. Substitution of wooden products with non-wooden products in an LCA of bioenergy is not advisable because the substitution criteria are too complex.
5. If avoiding allocation is not possible, the reasons should be documented.
6. If an allocation between different co-products is necessary for a certain process (e.g., sawmill), all upstream environmental effects also have to be allocated (e.g., upstream effects of sawmill can be transport and forestry).
7. Different allocation procedures must be analyzed and documented. In many cases, it seems necessary to make a sensitivity analysis of different allocation procedures for different environmental effects. It can also be useful to get the acceptance of the chosen allocation procedure by external experts.
8. For allocation in forestry, it is necessary to describe the main function of the forest from which the raw material is taken. In some cases, different types or functions of forests must be considered and described. The main function often indicates the allocation procedure.

Regarding the experiences from the examples, Jungmeier et al. (2002) identified the following most practical allocation for some specific processes: forestry—mass and volume; sawmill—mass and market price; wood industry—mass and market price.

In terms of the use of materials in the built environment and evaluating their environmental impact, we are still in a situation where there is huge variation in the way that LCA studies are performed. There has been action to make these studies more rigorous and prescriptive, with the introduction of EPDs and (within Europe) PCR for timber products, as well as for construction materials. Although the production of EPDs is presently voluntary, there will rapidly be a necessity to produce EPDs in order to meet the requirements of procurement. If we are to create carbon markets that are able to assign a monetary value to sequestered carbon stored in the built environment, it will become necessary to move towards a system where it is a legal requirement to have proper certification of the carbon footprint of products.

The formalization of procedures related to the chain of custody of forest products provides an opportunity for simultaneously incorporating LCA data. This represents an opportunity for the forest products sector that should be addressed. One of the problems with this sector is the diversity of sources, heterogeneity of material, and huge range of products that are produced. This is a much more complex situation than that faced by the concrete, steel, and polymer sectors. It is essential that the forest products industry adopts chain-of-custody systems that are integrated with LCA tools. The ability to track products through the value chain when they are used in buildings will be possible with the increasing adoption of building information modelling tools. It will be necessary to extend the chain of custody through first life and on to subsequent lives as the material is cascaded down the value chain, as well as at end of life when the sequestered carbon is finally returned to the atmosphere. This will allow for a really effective and accurate tool for informing LCA, policy makers, and the public. The forest products industry has considerable experience in chain-of-custody certification; this expertise should be harnessed in the future to use chain-of-custody procedures to ‘pull through’ environmental information. This information could be in the form of carbon certificates.

5 Conclusion

A cradle-to-gate analysis was used in this chapter to present the carbon footprint of 14 different primary wood products. The largest source of emissions for all sawn timber products is removing the timber from the forest, while for kiln-dried sawn timber the drying process follows closely behind. For fiber composites (MDF and HDF), the extra energy required to convert the raw material to fibers, in addition to the energy required to apply pressure and heat to the products, is responsible for the bulk of the emissions from these products. The adhesives used in particle board, plywood, and OSB are responsible for the largest fraction of emissions from these products. This is especially significant considering the low total volume they represent in the final products. Glulam emissions derive mostly from the harvest and initial production of the softwood, but also from the extra energy required to apply pressure and set the adhesives used. Wood pellets are made mostly from manufacturing residues; therefore, their emissions are derived almost entirely from the energy required during manufacturing, especially compression. Altering the system boundaries would yield different results. Furthermore, results would have been modified if the carbon footprint calculation accounted for carbon sequestration of wood, the use of recycled wood products, and other similar issues pertinent to LCA.

In Europe, carbon footprint is gaining immense importance and is expected to be mandated to accompany products and services. The environmental properties of wood and other construction materials are currently entering in building codes in construction. However, the limited availability of emissions data and its poor integration to real-life decision making within the construction sector have kept construction industries from using environmental arguments for material choices. Several studies have dealt with the LCA of forests and primary wood products (Richter 2001; Petersen and Solberg 2005; Puettmann and Wilson 2005; Rivela et al. 2006a, b; Werner and Richter 2007; Tucker et al. 2009; Cherubini et al. 2009; Lindholm et al. 2010; Oneil et al. 2010; Puettmann et al. 2010; Carre 2011; Cherubini and Strømman 2011).

However, there is still a lack of data. It is essential that research on timber processing and the resultant products place more emphasis on the interactive assessment of processes parameters, developed product properties, and environmental impact, including recycling and disposal options at the end of the service life towards upcycling after their service life based on the cradle-to-cradle concept. Intelligent concepts for reuse and recycling of valuable materials at the end of a single product life could reduce the amount of waste destined for landfills or down-cycling. With new and innovative production technologies, reduced overall energy consumption, increased recycling of wood products, and reuse and refining of side-streams, the sector can become a leader on the path to achieving the European Commission's ambitious target of 80 % reduction in CO_2 emissions by 2050. Also, other policy strategies and actions directly impact the forest products industry, such as the EU Sustainable Development Strategy (SDS, European Commission 2009) and the recycling society directive (Directive 2008/98/EC, European Parliament Council 2008). Furthermore, the standardization in the area of sustainability is currently under dynamic development.

Newly published standards for the sustainability of construction works (CEN TC 350 2012) open opportunities for EU-wide harmonization of calculations and reporting of environmental impacts of buildings. The most important standards are EN 15804 (2012) for construction product EPDs and EN 15978 (2011) for assessment of environmental performance. Many of the databases and tools mentioned above date from before the introduction of the CEN TC 350 standards. Furthermore, as the influence of green building programs continues to increase and the field matures, the primary green building programs will shift to the use of LCA as a means of using science and consistent methodology to inform green building decisions (Bowyer 2008) and move towards an integrated design process. It is vitally important to the industry that the PCRs used for the relevant EPDs allow for the reporting of sequestered atmospheric carbon in timber products.

The design of a building is a complex process involving a multitude of disciplines and expertise. Therefore, it is essential that a transparent and standardized approach to LCA is used to assess the ecological and environmental consequences of the materials, use phase of the buildings, and end of life. Unfortunately, the values can differ significantly between studies. The use of different input data, functional units, allocation methods, reference systems, and other assumptions complicates comparisons of the LCAs of green building studies. To be sustainable in a holistic way, an integrated design process should be adopted. Each system or discipline in a project has some effect on another system to varying degrees.

The goals of sustainable development to increase economic efficiency, protect and restore ecological systems, and improve human's well-being—or a combination of the three—are expected to lead to new concepts, products, and processes optimizing the multiple utilization/recycling of forest-based resources. The life cycle analysis and cradle-to-cradle concepts are also expected to be used as key tools in economic development, leading to new business opportunities through innovative products with properties optimized to the end-use requirements and sustainable use of resources.

Acknowledgment Andreja Kutnar would like to acknowledge the Slovenian Research Agency for financial support within the frame of the project Z4-5520.

References

Barbu M, van Riet C (2008) European panels market developments—current situation and trends. In: The proceeds of the SWST annual convention, Concepción, Chile, 2008. Madison 2008. Society of Wood Science and Technology

Benetto E, Becker M, Welfring J (2009) Life cycle assessment of oriented strand boards (OSB): from process innovation to ecodesign. Env Sci Technol 43(15):6003–6009

Berglund L, Rowell RM (2005) Handbook of wood chemistry and wood composites. CRC Press, Boca Raton, p 279–301

Bodig J, Jayne B (1982) Mechanics of wood and wood composites. Van Nostrand Reinhold Company. New York, p 712

Bowyer JL (2008) The green movement and the forest products industry. Forest Prod J 58(7/8):6–13

Buchanan AH (2006) Can timber buildings help reduce global CO_2 emissions? Proceedings, world conference on timber engineering, Portland

Buchanan AH (2010) Energy and CO_2 advantages of wood for sustainable buildings. Proceedings, world conference on timber engineering, Riva-del-Garda

Carre A (2011) A comparative life cycle assessment of alternative constructions of a typical Australian house design. Forest and Wood Products Australia Limited, p 121. www.fwpa.com.au

CEN/TC 350 (2012) Sustainability of construction works

Cherubini F, Strømman AH (2011) Life cycle assessment of bioenergy systems: state of the art and future challenges. Bioresour Technol 102:437–451

Cherubini F, Bird ND, Cowie A, Jungmeier G, Schlamadinger B, Gallasch S (2009) Energy- and greenhouse gas-based LCA of biofuel and bioenergy systems: key issues, ranges and recommendations. Resour Conserv Recycl 53:434–447

Climate Change (2001) IPCC third assessment report. The scientific basis. http://www.grida.no/climate/ipcc_tar/. Accessed 3 May 2010

Construction Products Regulation (305/2011) of the European Parliament and of the Council of 9 March 2011 laying down harmonised conditions for the marketing of construction products and repealing Council Directive 89/106/EEC

Earles J, Halo A, Shaler S (2011) Improving the environmental profile of wood panels via co-production of ethanol and acetic acid. Env Sci Technol 45(22):9743–9749

Ecoinvent 2.0 (2010) Swiss Centre for Life Cycle Inventories, Dübendorf

European Committee for Standardisation (CEN) (2012) EN 15804: sustainability of construction works—environmental product declarations—core rules for the product category of construction products

European Committee for Standardisation (CEN) (2011) EN 15978: sustainability of construction works—assessment of environmental performance of buildings—calculation method

European Committee for Standardisation (CEN) (2013) FprEN 16449: wood and wood-based products—calculation of the biogenic carbon content of wood and conversion to carbon dioxide Final draft 2013

European Committee for Standardisation (CEN) (2012). EN 16485: round and sawn timber—environmental product declarations—product category rules for wood and wood-based products for use in construction Draft 2012

European Commission (2009) Mainstreaming sustainable development into EU policies: 2009 review of the European Union Strategy for Sustainable Development. Communication. European Commission European Commission, Brussels. http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=CELEX:52009DC0400:EN:NOT

European Commission (2011) A roadmap for moving to a competitive low carbon economy in 2050. Communication. European Commission European Commission, Brussels. http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=CELEX:52011DC0112:EN:NOT

European Committee for Standardization (2011) 15942:2011: Sustainability of construction works—environmental product declarations—communication format business-to-business. Standard. European Committee for Standardization, Brussels

European Parliament, Council (2008) Directive 2008/98/EC of the European Parliament and of the Council of 19 November 2008 on waste and repealing certain Directives. Directive. European Parliament European Parliament, Brussels. http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=CELEX:32008L0098:EN:NOT

Food and Agriculture Organization of the United Nations (FAO) (2010) Global forest resource assessment 2010 main report. United Nations Food and Agriculture Organization, Rome

Food and Agriculture Organization of the United Nations (FAO) (2013) FAOSTAT database. http://faostat.fao.org/site/362/DesktopDefault.aspx?PageID=362. Accessed 4 Nov 2013

Forest-based Sector Technology Platform (2013a) Horizons—vision 2030 for the European forest-based sector. Research agenda. Filip De Jaeger, Gérant FTP Forest-based Sector Technology Platform, Brussels

Forest-based Sector Technology Platform (2013b) Strategic research and innovation agenda for 2020. Research agenda. Filip de Jaeger, Gérant FTP Forest-based Sector Technology Platform, Brussels

Forest Products Laboratory (2010) Wood handbook—wood as an engineering material. Forest Products Laboratory, Madison

Gonzalez-Garcia S, Feijoo G, Heathcote C, Kandelbauer A, Moreira M (2011) Environmental assessment of green hardboard production coupled with a laccase activating system. J Clean Prod 19(5):445–453

Hall DO, Scrase JI (1998) Will biomass be the environmentally friendly fuel of the future? Biomass Bioenergy 15(4/5):357–367

Halog A (2009) Models for evaluating energy, environmental and sustainability performance of biofuels value chain. Int J Glob Energy Energy Issues 32(1/2):87–101

Hill CAS (2011) An introduction to sustainable resource use. Taylor and Francis, London

International Organization for Standardization (ISO) (1997) 14040: environmental management—life cycle assessment—principles and framework. Standard. International Organization for Standardization, Geneva

International Organization for Standardization (ISO) (2006) 14044:2006: environmental management—life cycle assessment—requirements and guidelines. Standard. International Organization for Standardization, Geneva

International Organization for Standardization (ISO) (2009) 14025: environmental Labels and declarations—type III environmental declarations—principles and procedures. Standard. International Organization for Standardization, Geneva

Jungmeier G, Werner F, Jarnehammar A, Hohenthal C, Richter K (2002) Allocation in LCA of wood-based Products. Experiences of cost action E9. Part I. Methodology. Int J LCA 7(5):290–294

Leek N (2010) Post-consumer wood. In: EUwood—real potential for changes in growth and use of EU forests. Final Report. EUwood, Hamburg

Lindholm EL, Berg S, Hansson PA (2010) Energy efficiency and the environmental impact of harvesting stumps and logging residues. Eur J Forest Res 129:1223–1235

Oneil EE, Johnson LR, Lippke BR, McCarter JB, McDill ME, Roth PA, Finley JC (2010) Life-cycle impacts of inland Northwest and Northeast/North central forest resources. Wood Fiber Sci 42:29–51

PAS 2050 (2011) Specification for the assessment of the life cycle greenhouse gas emissions of goods and services

Petersen AK, Solberg B (2005) Environmental and economic impacts of substitution between wood products and alternative materials: a review of micro-level analyses from Norway and Sweden. Forest Policy Econ 7:249–259

Puettmann ME, Wilson JB (2005) Life-cycle analysis of wood products: cradle-to-gate LCI of residential wood building materials. Wood Fiber Sci 37:18–29

Puettmann ME, Bergman R, Hubbard S, Johnson L, Lippke B, Oneil E, Wagner FG (2010) Cradle-to-gate life-cycle inventory of US wood products production: corrim phase I and phase II products. Wood Fiber Sci 42:15–28

Richter K (2001) LCA—reuse/recycle. In: Johansson CJ, Pizzi T, van Leemput M (eds.) Wood adhesion and glued products, Report on the State of the Art of COST Action E13, p 161–180

Rivela B, Hospido A, Moreira T, Feijoo G (2006a) Life cycle inventory of particleboard: a case study in the wood sector. Int J LCA 11(2):106–113

Rivela B, Moreira M, Muñoz I, Rieradevall J, Feijoo G (2006b) Life cycle assessment of wood wastes: a case study of ephemeral architecture. Sci Total Env 357(1–3):1–11

Rivela B, Moreira T, Feijoo G (2007) Life cycle inventory of medium density fibreboard. Int J LCA 12(3):143–150

Saravia-Cortez A, Herva M, García-Diéguez C, Roca E (2013) Assessing environmental sustainability of particleboard production process by ecological footprint. J Cleaner Prod 52:301–308

Silva D, Lahr F, Garcia R, Freire F, Ometto A (2013) Life cycle assessment of medium density particleboard (MDP) produced in Brazil. Int J LCA 18(7):1404–1411

SimaPro, SimaPro Analyst Indefinite, Ecoinvent v2, Product Ecology Consultants, PEC, Netherlands, (2009). www.pre.nl/default.htm

Suchsland O (2004) The swelling and shrinking of wood: a practical technology primer. Forest Products Society, Madison

Tucker S, Syme M, Foliente G (2009) Life cycle assessment of forest and wood products in Australia. N Z Timber Des J 17(4):3–9

Wang M (2005) Energy and greenhouse gas emissions impacts of fuel ethanol. Center for Transportation Research Energy System Division, Argonne National Laboratory. NGCA Renewable fuels forum, the national Press club, 23 Aug 2005

Werner F, Richter K (2007) Wood building products in comparative LCA. A literature review. Int J LCA 12(7):470–479

Youngquist JA (1999) Wood handbook: wood as an engineering material. USDA Forest Service, Forest Products Laboratory, Madison. General technical report FPL; GTR-113: 10.1-10.31

Carbon Footprint of Recycled Products: A Case Study of Recycled Wood Waste in Singapore

Ruisheng Ng, Zhiquan Yeo, Hui Xian Tan and Bin Song

Abstract Recycling is a process that takes materials or products that are at the end of their lives and transforms them into either the same product or a secondary product. When a material is recycled, it is used in place of virgin inputs in the manufacturing process, rather than being disposed of and managed as waste. Therefore, recycling, especially the recycling of wood waste, is beneficial in delaying the release of greenhouse gas (GHG) emissions as well as leading to increased carbon storage in trees. According to Singapore Waste Statistics 2012, approximately 343,800 tons of wood waste and 247,800 tons of horticultural waste are generated annually. With limited land space and scarce natural resources, there is a huge incentive for Singapore to increase recycling rates. Furthermore, recycling leads to a reduction in carbon footprint and lower environmental impact. To quantify the potential environmental benefits of recycling wood waste, three approaches are introduced. However, there are several limitations associated with these approaches. To avoid under- and overestimating the avoided emissions due to the recycling of wood waste, a methodology for fair and reasonable assessment is introduced. A case study of a local wood waste recycling plant is presented to illustrate the proposed methodology. Results show that the recycled technical wood product has lower carbon footprint (12.8 kg CO_2e) than a virgin hardwood product (16.2 kg CO_2e). When the effects of avoided impact are taken into account, the carbon footprint of the technical wood product may have an even lower carbon footprint (-2.9 kg CO_2e), clearly illustrating the environmental benefits of recycling wood waste.

Keywords Carbon footprint · Recycling · Carbon storage · Wood waste · Avoided emissions

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S. S. Muthu (ed.), *Assessment of Carbon Footprint in Different Industrial Sectors, Volume 2*, EcoProduction, DOI: 10.1007/978-981-4585-75-0_7,

1 Overview of Recycling in Singapore

Recycling is a process that takes materials or products that are at end-of-life and transforms them into either the same product or a secondary product. When a material is recycled, it is used in place of virgin inputs in the manufacturing process, rather than being disposed of and managed as waste. Consequently, recycling provides greenhouse gas (GHG) emission reduction benefits in two ways, depending upon the material recycled: (1) it offsets a portion of "upstream" GHG emitted in raw material acquisition, manufacture, and transport of virgin inputs and materials, and (2) it increases the amount of carbon stored in forests when wood and paper products are recycled (U.S. Environmental Protection Agency 2012a). Therefore, recycling—especially recycling of wood waste—is beneficial in delaying the release of GHGs as well as leading to increase carbon storage in trees (U.S. Environmental Protection Agency 2010, 2012b; Ng et al. 2011, 2014).

Another motivating factor for companies to recycle is the external pressure to adopt recycling performance certification (Simpson 2012). There is increasing demand from big players in the industry to require their suppliers to adopt ISO 14001 certification (Lindsey 2000). ISO 14001 is an international standard for environmental management systems (ISO 2004). With the short- and long-term environmental and social benefits generated by recycling, recycling has become crucial to many industries.

1.1 Waste Management in Singapore

Singapore is a small island country with a total land area of 716 km^2. As of 2012, the population of Singapore is 5.31 million people (Department of Statistics Singapore 2013). With a high population density of 7,422 per km^2 and very limited natural resources, there is a huge incentive for Singapore to critically address its solid waste management issues. Singapore's output of solid waste has increased appreciably over the years, from 1,260 tons per day in 1970 to a high of 8,016 tons per day in 2012 (National Environmental Agency 2013a). Consequently, the traditional land-intensive method of landfilling waste is no longer viable. Conversely, recycling is a better alternative.

In Singapore, the National Environment Agency (NEA) plans, develops, and manages the country's waste management system. NEA is a public organization responsible for the licensing and regulation of solid waste collection and enforcement of illegal dumping in Singapore. From NEA's past reports, the total waste output has increased approximately 56 %, from 4,654,600 tons in 2000 to 7,269,500 tons in 2012 (National Environment Agency 2013b). If population growth is factored in (4.03 million in 2000 to 5.31 million in 2012), the actual increase in waste output per capita from 2000 to 2012 is approximately 18.5 %.

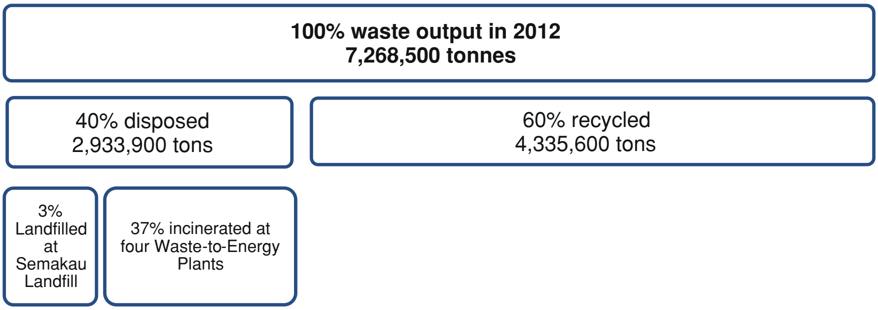

Fig. 1 Singapore Waste Statistics in 2012. *Source* National Environmental Agency (2013a)

Despite the increase in total waste output, the recycling rate has increased significantly from 40 % (1,857,300 tons) in 2000 to 60 % (4,335,600) in 2012 (Fig. 1). This data shows that although there is an increase in total waste output, the recycling rate in Singapore has increased steadily over the years.

In the 1960s, Singapore depended on several landfills around the island to manage the solid waste produced on the island. However, in the late 1970s, land limitation drove the government to seek alternative methods of solid waste disposal. The NEA later adopted waste-to-energy (WTE) incineration to reduce waste volume by 90 % before sending the waste to landfill. Today, there are research and development efforts to target on waste streams to further reduce the waste to landfill. In this chapter, the focus is on wood wastes only.

1.2 Wood Waste Management in Singapore

Due to the vision put forward by former Prime Minister Lee Kuan Yew to develop Singapore into a garden city, there are extensive rows of trees along every road and parks in every estate. The tree-growing efforts have contributed to substantial horticultural waste. The large amount of solid waste generated, together with abundance of wood and horticultural waste produced, have made wood waste management and wood waste recycling important subjects in Singapore.

Wood waste generally refers to waste from wooden pallets, crates, boxes, furniture, and wood planks used in construction. Horticultural waste includes tree trunks and branches, plant parts, and trimmings collected during the maintenance and pruning of trees and plants all over Singapore.

According to Singapore Waste Statistics 2012, approximately 343,800 tons of wood waste and 247,800 tons of horticultural waste are generated each year (National Environmental Agency 2013a). This represents 8.1 % of all waste generated in Singapore. Currently, 69 % of all wood waste and 44 % of all horticultural waste in Singapore are recycled (Table 1).

Table 1 Waste statistics and recycling rates in 2012

Waste type	Waste disposed (tons)	Total waste recycled (tons)	Total waste output (tons)	Recycling rate (%)
Wood/timber[a]	107,800	236,000	343,800	69
Horticultural[a] waste	139,800	108,000	247,800	44

[a] Includes 110,300 tons used as fuel in biomass power plants
Source National Environmental Agency (2013a)

Upon collection, used wooden pallets and crates are usually sent to recycling companies for repair and reconditioning. The pallets and crates are dismantled, then the wood parts are cut to size and fixed back to form new pallets and crates. Wood and horticultural waste are usually processed into wood chips for composting, cogeneration, or used to make new wood products (Zero Waste Singapore 2008).

Of the 343,800 tons of wood waste and 247,800 tons of horticultural waste generated per year in Singapore, 110,300 tons are used as fuel in biomass power plants. Also, there are 16 authorized wood waste and horticultural waste collectors in Singapore (National Environment Agency 2011). Some of these private companies have set up recycling facilities to recycle horticultural waste and wood waste. Horticultural waste is usually processed into compost, whereas wood waste is normally processed into charcoal and charcoal-related products.

1.3 Challenges in Wood Recycling

As of November 2013, the recycling rates for wood waste and horticultural waste stand at 69 and 44 %, respectively (National Environmental Agency 2013a). To further drive wood waste recycling efforts up in Singapore, there are several practical and technical challenges to be addressed.

1.3.1 Collection of Wood Waste

Although there are 16 wood and horticultural waste collectors in Singapore, most of these companies collect only wooden pallets, crates, cases, and planks. In addition, a few of these collectors do not provide collection services. Furthermore, since the waste wood collection fees of individual collectors are not disclosed, there is no uniform fee for wood waste collection in Singapore. This limits the recycling options for companies and gives rise to a situation where the fees vary across different collectors. To ensure that the market remains competitive, a new fee structure is necessary to enhance waste wood recycling efforts in Singapore.

1.3.2 Technology to Recycle Wood Waste

The various applications of recycled wood usually require the use of uncontaminated timber that has not been treated with chemicals. As such, the majority of wood waste that is being recycled mostly comes from wood packaging waste. It is a very labor- and time-intensive process to sort out the reusable wood pieces from the non-reusable ones. Also, the recycling process is highly energy intensive because the wood waste needs to go through shredding, metal separation, hammering, and drying processes. To facilitate the recycling process, there needs to be major technological shifts to improve the rate of timber recycling and increase the energy efficiency of the recycling process.

1.3.3 Performance of Recycled Wood Waste

In addition to the challenges to increase the rate of wood waste recycling, there is also the challenge to find demand for wood waste. This could be attributed to the social stigma associated with “used” wood. Used wood materials are often perceived to be of lower quality or inferior when compared to virgin wood. Furthermore, recycled wood products that are sold at higher prices cannot compete with virgin wood products for consumers in the market. However, recycled wood products that are sold at lower prices inevitably reinforce the idea that recycled wood products are of lower grades. In addition to the stigma, product designers generally find it more convenient and aesthetically more appealing to use standard-sized new wood planks for their products, especially furniture. The downside of recycled wood is that it normally comes in odd shapes and sizes. Consequently, in order for recycled wood to be more widely accepted and sought after, mindset shifts among society and technological advancement are necessary.

1.4 Opportunities in Wood Recycling

Despite the challenges in wood recycling, the process produces recycled wood that has many uses. With improvements in technology, wood recycling plants are becoming more efficient and cost-effective in their recycling process. The newer machineries can now manage larger quantities of wood waste simultaneously, while the removal of metal parts is becoming automated. The quality of the produced recycled wood is also improving. At present, recycled wood can be converted into different useful products of various grades—namely high-end, mid-range, and low-end.

High-end wood waste can be processed into “technical wood” (Ng et al. 2011). The technical wood can be used as a raw material for products such as tables, doors, flooring, and building materials. Wood waste, which is made of original hardwood fibers, can be processed to make molded pallets for transportation of goods (Ng et al. 2014). Recycled wood has the advantage of lower moisture content than virgin wood, which translates to stability and a reduction in thickness swelling. Low-end wood waste, on the other hand, can be used as fuel in biomass power plants (Khoo et al. 2008).

1.5 Carbon Footprint and Environmental Benefits

To assess the environmental impact of recycling, the carbon footprint (CFP) is chosen as the metric for measurement. The CFP measures the impact to the environment by human activities in terms of the amount of GHGs produced (Hertwich and Peters 2009). Typically, GHGs include CO_2, CH_4, N_2O, SF_6, HFCs, and PFCs. CFP is used because of its relevance in quantifying the effects of carbon storage and its global warming potential effect on climate change.

CFP assessments have been gaining traction in Singapore to quantify environmental benefits. For instance, CFP studies are carried out in the Singapore metal processing industry to assess the environmental performance and to identify strategies to improve manufacturing processes (Shi et al. 2011a, 2012; Ng et al. 2012). CFP studies are also carried out in the food industry to understand operational efficiency and look for opportunities to convert waste to useful products (Shi et al. 2011b). In the Singapore construction industries, CFP is also gaining interest for use as an indicator for quantifying environmental performance (Lu 2013; Shi et al. 2013).

There are also examples in Singapore recycling industries where CFP is used as an environmental performance indicator. For instance, CFP has been used as an indicator to quantify the environmental impact of a nickel recovery process (Yang et al. 2009). Rugrungruang et al. (2009) conducted a CFP assessment on a new biocomposite material, R3Plas^{TM}. The bio-composite is composed of recycled polypropylene (PP) and rice husk fiber and is used in rooftop plant boxes. The results show that the bio-composite can reduce a product's CFP by 33 % when the bio-composite is used to replace the virgin PP and by 8 % when substituting the recycled PP (Rugrungruang et al. 2009). Ng et al. (2014) have also assessed, quantified, and compared the CFP of recycled wood waste with virgin softwood in wooden pallets. The results demonstrate that recycled wood pallets have a lower CFP (3.547 kg CO_2e) than virgin softwood pallets (4.007 kg CO_2e); (Ng et al. 2014), which is an 11.4 % reduction. Chua et al. (2010) have also modelled the life cycle processes of manufacturing biodiesel by recycling waste cooking oil for transport in Singapore. The environmental impacts are compared against conventional ultra-low sulfur diesel. The biodiesel emits 0.006 kg CO_2e compared to ultra-low sulfur diesel of 1.08 kg CO_2e per kilometer travelled (Chua et al. 2010). This is about 99.4 % reduction. Hence, it is shown that recycling has many positive environmental impacts.

In this chapter, the focus is on the CFP study of recycled wood waste. There are a number of methodologies to quantify CFP. In particular, these methodologies discuss the use of avoided emissions as an additional benefit in recycling of wood waste (U.S. Environmental Protection Agency 2010; Eriksson et al. 2010; Miner 2010). However, there could be a possibility that avoided emissions may be overestimated because the scenarios where activities are avoided may be judgmental. Due to the limitation in existing methodologies, a new methodology termed "avoided impact" is proposed in this chapter. Its development and usage will be further discussed in the later sections.

This chapter consists of five sections. Following this introduction on recycling and wood waste management in Singapore, Sect. 2 of the chapter discusses the various methodologies used to assess carbon footprint in recycled wood products. Section 3 presents the proposed methodology in assessment of carbon footprint of recycled products from wood waste. Section 4 provides a case study on recycled doors to illustrate the possibilities of converting wood waste to useful wood products. Section 5 summarizes and concludes the chapter.

2 Methodologies to Assess Carbon Footprint in Recycled Wood Products

2.1 Considerations of Scenarios in Assessing Carbon Footprint

2.1.1 System Boundary

In all carbon footprint studies, the system boundary that is being scoped out by the practitioner delineates the processes to be included and excluded in the studies. Depending on the system boundary being set, the same product will yield vastly different results when different system boundaries are established. Therefore, care has to be taken when interpreting and comparing results for studies with different system boundaries, as direct comparison often cannot be made. Generally, for the purpose of comparing recycled products with non-recycled products, the two most common system boundaries being defined are *cradle-to-cradle* and *cradle-to-grave*.

2.1.2 Cradle-to-Cradle

In the cradle-to-cradle system boundary, the scope of the carbon footprint assessment covers the first lifecycle of a product until the beginning of the subsequent lifecycle. In the context of recycling, it encompasses the raw material extraction from Earth's natural resources; followed by product manufacture, use,

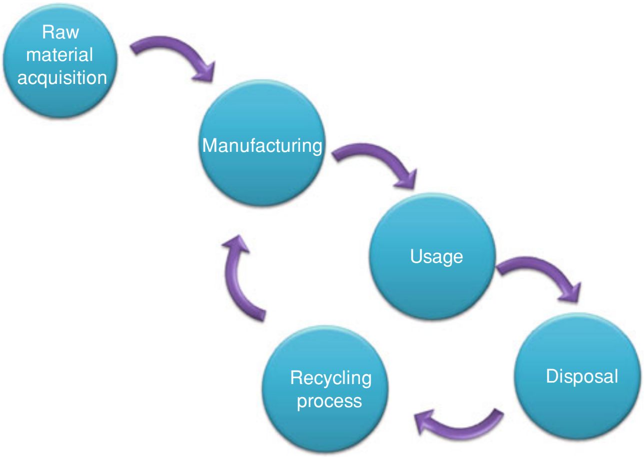

Fig. 2 Cradle-to-cradle, closed-loop recycling

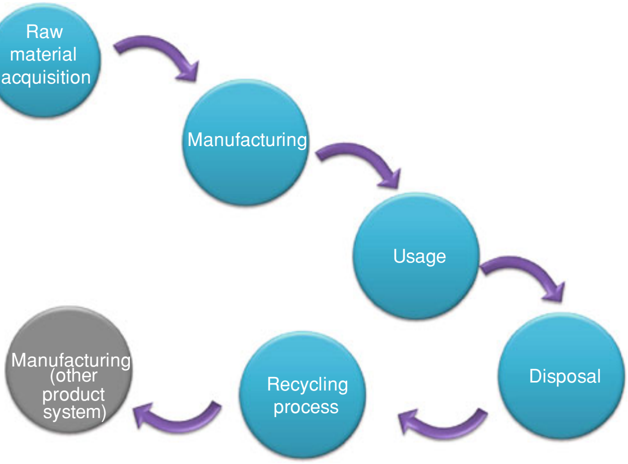

Fig. 3 Cradle-to-cradle, open-loop recycling

and post-use treatment of the materials; to a state ready for the use in a subsequent product lifecycle. If the subsequent lifecycle refers to the same product system, such recycling is termed *closed-loop recycling*, as shown in Fig. 2. However, if the subsequent lifecycle refers to another product system, such recycling is termed *open-loop recycling*, as shown in Fig. 3.

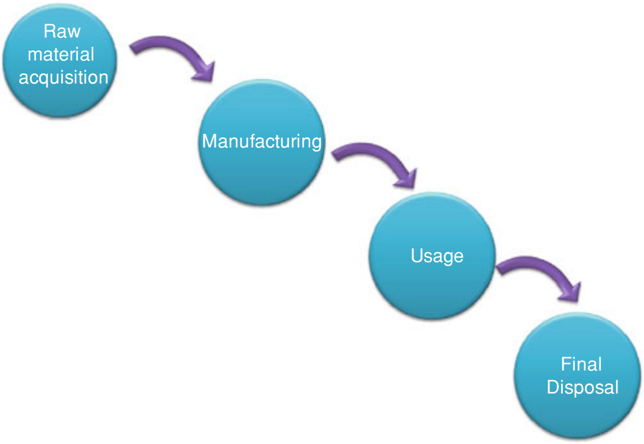

Fig. 4 Cradle-to-grave

2.1.3 Cradle-to-Grave

The cradle-to-grave system boundary typically describes the full lifecycle of a product. It begins from the raw material extraction from Earth's natural resources, followed by the manufacturing of the product, its usage, and its final disposal. This system boundary covers the typical phases of a product without recycling (Fig. 4).

2.1.4 Allocation Rules

Whenever processes are shared with other product systems, treatment to the process flows has to be employed to segregate the process flows into a portion belonging to the product system of the study and to the other portion belonging to other product systems not within the system boundary. This is known as allocation. According to the ISO/TS 14067, the allocation principles and procedures apply to not only processes with multiple outputs, but also to reuse and recycling situations (ISO 2013). The allocation procedure established in the ISO/TS 14067 takes reference from ISO 14044 (ISO 2006) and hence is consistent with the life cycle assessment (LCA) methodology.

The ISO standards on LCA clearly define a hierarchy for avoiding or solving allocation tasks (Ardente and Cellura 2011). The general procedure would start by first avoiding allocation altogether wherever possible. This can be done by dividing the unit process to be allocated into two or more sub-processes.

Following that, only those sub-processes that are related to the system boundary will be taken into account (Fig. 5). However, if the method of subdividing is not feasible, an alternative method to avoid allocation is recommended. This method is known as system expansion. In system expansion, the system boundary is

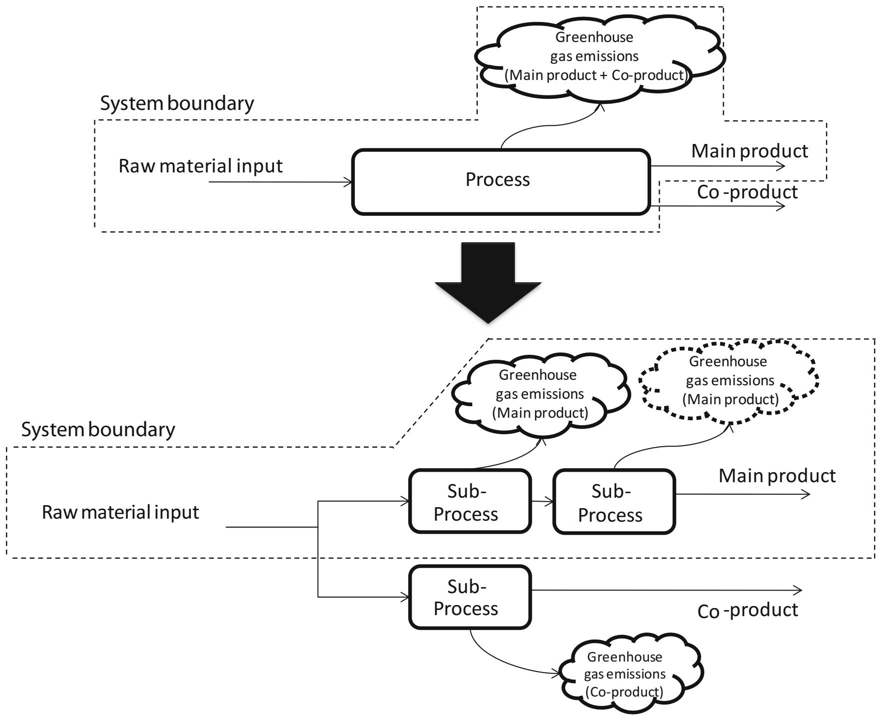

Fig. 5 Subdivision to avoid allocation

increased to include other related product systems that the unit process is supporting and, following that, subtracting the related environmental impact of the avoided emissions of the other product system that is not included the initial system boundary (Fig. 6).

Where allocation cannot be avoided, it is recommended that the process flows of the system be partitioned between its products or function in a way that reflects underlying physical relationships. This basis of emission allocation assumes that there are physical causalities that exist between emissions and the physical properties of the products and co-products (Ardente and Cellura 2011). In the case of wooden products, common options for allocation by physical relationship would be by the output mass or by the output volume. In the case of the former, allocation by mass would be subject to variations due to the fluctuating moisture content of the wooden products (Bergman 2008). This fluctuating moisture content of the wood would make the results inconsistent. Alternatively, allocation by volume can be selected as the basis of allocation. By using allocation by volume, the allocation will be less sensitive to the moisture content of the wood; hence, it is a more desirable basis for allocation to be performed.

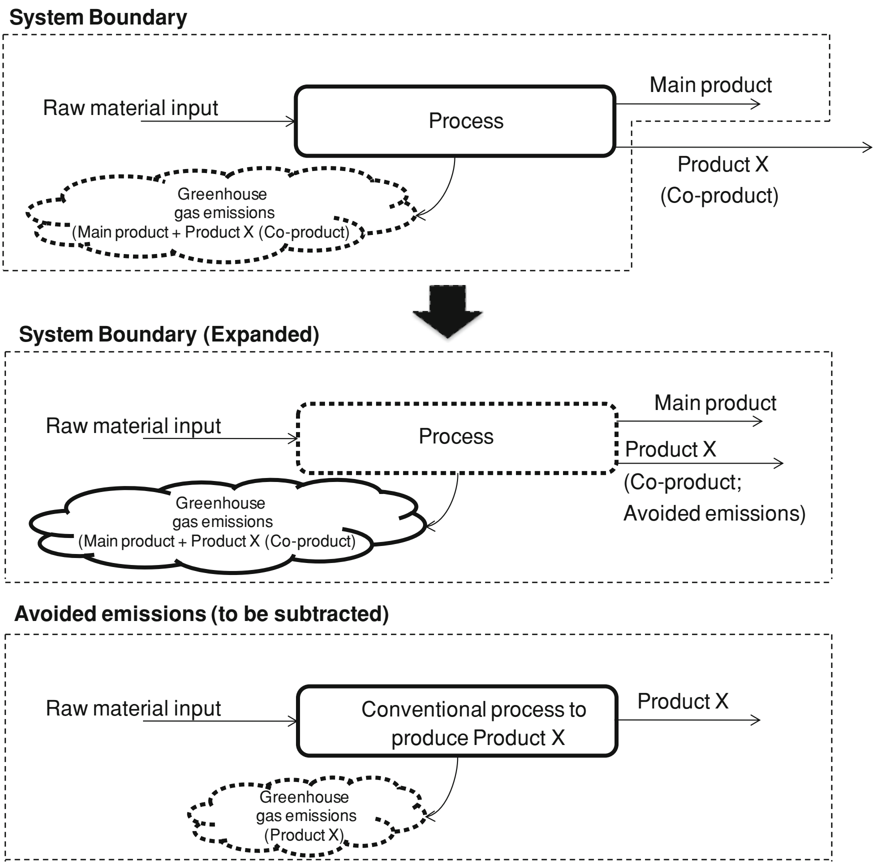

Fig. 6 System expansion to avoid allocation

The last step in the allocation procedure is to perform the allocation based on other nonphysical relationships that exist between the products, if allocation by physical relationship is not possible. In this case, the input–output data could be allocated based on the relative proportion of the economic values of the products. This allocation basis would allocate lesser emissions to low-value products, while those products that derive larger economic value would in turn be apportioned a larger share of emissions. It is argued that "the economic relationships reflect the socio-economic demands which cause the multiple-function systems to exist at all" (Azapagic and Clift 1999). However, the shortcoming of this allocation basis would be the ineffectiveness caused by large fluctuations in prices in products. This will bring about large inconsistencies in the results, which would be highly dependent on the market pricings of all the products concerned (Boustead et al. 1999).

2.1.5 Recycling Allocation

There are generally two types of recycling scenarios—open-loop recycling and closed-loop recycling. Open-loop recycling refers to the scenario whereby the post-recycled material does not get fed back to its own product system (i.e. it becomes a raw material for another product system). This could be due to the recycled material not being able to maintain its inherent properties as its virgin material state, thereby not being able to reuse for the same purpose. On the other hand, closed-loop recycling refers to the scenario whereby the post-recycled material is also the raw material within the same product system; hence, it forms a closed-loop material flow. This scenario will occur if the recycled material is able to maintain the same inherent properties as its virgin material input. Due to the two different scenarios that may exist in recycling, the methods for allocation also vary. The allocation of the carbon footprint typically involves the material acquisition and end-of-life stages.

In the case of open-loop recycling, the "cut-off method" (also referred to as the 100–0 method or the recycled content method) is used in allocation. To account for the material acquisition stage, all attributable processes due to the virgin and recycled material acquisition and preprocessing are allocated to the product. These include all upstream processes that are required to acquire both the virgin material and recycled material up to the point where they enter the production process. At the end-of-life stage, the treatment of the waste material output will be allocated to the product. The exclusions of this method are the processes to recover the material output at the end-of-life. The representation of the inclusions and exclusions of the open-loop recycling scenario is illustrated in Fig. 7.

For closed-loop recycling, the closed-loop method (also known as the 0–100 method, recyclability substitution approach, and the end-of-life approach) is used to allocate emissions to the product. In this method, it is assumed that the recycled material is able to displace the virgin material at the input. Therefore, the recycling rate of the material at the end-of-life is assumed to be equal to the virgin material displacement rate. With this assumption, the virgin material upstream emission is calculated with the post-displaced amount. For the end-of-life stage, all processes, including recycling of the material output, must be taken into account. This typically includes the collection of disposed materials, treatment of waste, material recovery, and preprocessing. Figure 8 illustrates the process inclusions of the closed-loop method for allocation. This method ensures that the recycling processes are included in the system boundary, as they are used to create the recycled material input back into the system. However, for this method, it is required to estimate the recycling rate of the system, which may be a challenge if the organization undertaking the CFP study does not control the product take-back process. In this case, general region-wide recycling statistical data might be used. However, this may not be reflective of the recycling rate of the actual product system under the study.

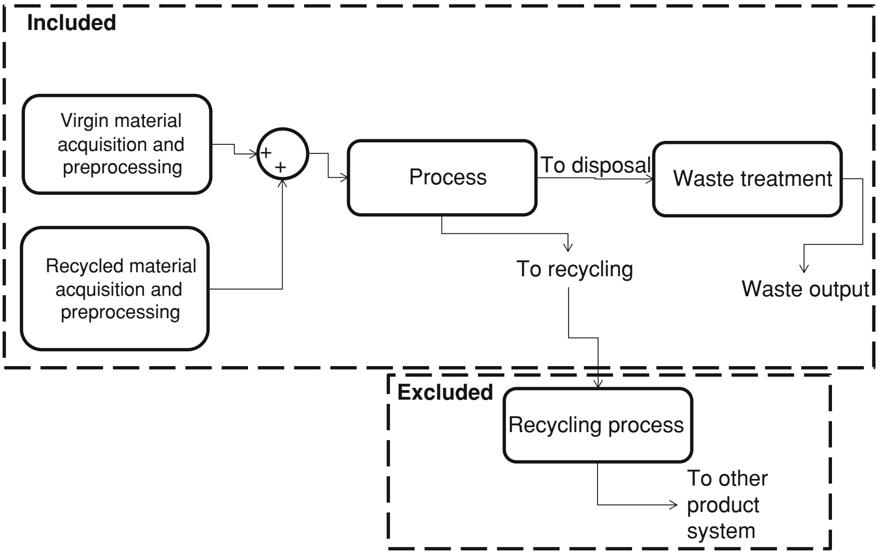

Fig. 7 Process flow diagram representation for the cut-off method

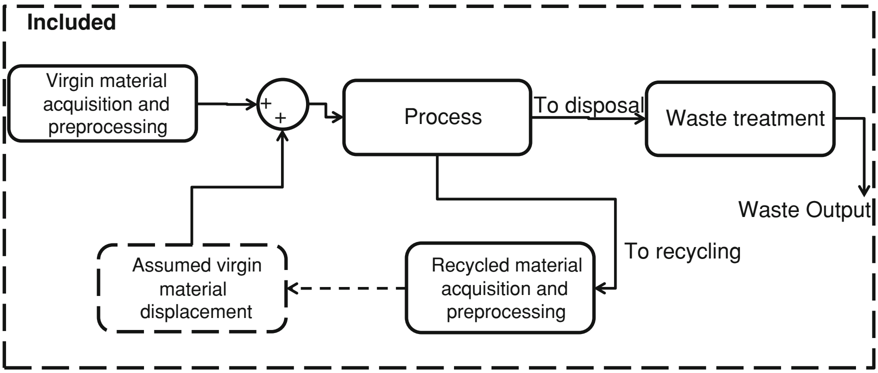

Fig. 8 Process flow diagram representation for the closed-loop method

2.2 Forest Carbon Sequestration in the Waste Reduction Model

The U.S. Environmental Protection Agency (EPA) developed a Waste Reduction Model (WARM) to capture the benefits of increased carbon storage as a consequence of recycling and source reduction of wood-related products (U.S. Environmental Protection Agency 2010). The carbon storage results from trees absorbing carbon dioxide during photosynthesis and converting it into cellulose and other materials, which are stored in trees as they grow.

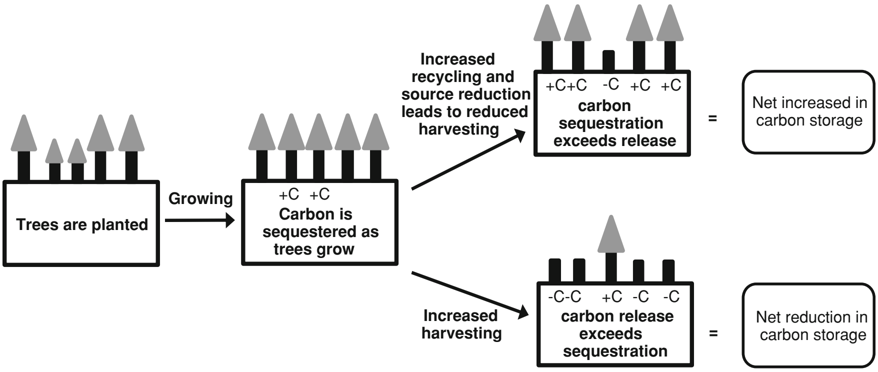

Fig. 9 Modeling of forest carbon sequestration in the Waste Reduction Model. *Source* U.S. Environmental Protection Agency (2010)

Assuming a business-as-usual scenario, the increase in recycling and/or source reduction will lead to lower demand in harvesting trees. As a result, the rate of tree growth is likely to exceed the rate of harvesting, leading to increased carbon storage in the forest. On the other hand, if the rate of harvesting exceeds the rate of tree growth, more carbon will be released. Figure 9 illustrates both scenarios when carbon sequestration exceeds release and when carbon release exceeds sequestration due to varying rates of harvesting.

2.3 Avoided Emissions Using System Expansion Approach

In a study by Eriksson et al. (2010), a methodology is developed based on the CEPI Carbon Footprint framework (CEPI 2007). The methodology includes a method to assess the net sequestration of biogenic CO_2 in the forest. In addition, the methodology also includes the avoided emissions quantification. The avoided emissions quantification involves the use of a system expansion approach. As described earlier in a system expansion approach, the system boundary is expanded to include other related product systems that the unit process is supporting. Subsequently, the related environmental impact of the avoided emissions of the other product system that is not included in the initial system boundary is subtracted. For instance, if the end-of-life product is incinerated, energy in the form of electricity or district heat can be recovered (Ekvall 1999). The recovered energy can be used to displace the emissions due to the combustion of other fuels. This displacement of emissions therefore results in avoided emissions.

- the methane from landfills that is avoided;
- the carbon storage in landfills that is avoided;
- the biomass energy that is precluded (i.e. recycled used products are not available for biomass energy);
- the difference in greenhouse gas intensity between virgin and recycled manufacturing;
- the differences in processing and transport requirements between virgin and recycled fiber;
- the impact of increased use of recovered fiber on forest carbon.

Fig. 10 Factors that affect the calculation of avoided emissions due to recycling. *Source* Miner (2010)

2.4 Avoided Emissions of the Global Forest Industry

In a report prepared by the Food and Agriculture Organization of the United Nations (Miner 2010), it was acknowledged that there were societal impacts that occurred outside the industry's value chain. These societal impacts may be avoided if recovery and recycling of materials and/or energy were to occur. The societal impacts that were avoided were generally categorized as "avoided emissions". However, quantifying these avoided emissions was extremely complex and uncertain, and it entails assumptions regarding uncertain scenarios.

In particular, it was highlighted in the report that the calculations involved in estimating the resulting value of avoided emissions were complex due to the need for consideration of several factors. To estimate avoided emissions due to recycling, several factors, such as those listed in Fig. 10, have to be considered:

However, it is acknowledged that there is high uncertainty in quantifying some of the avoided emissions (Miner 2010). Furthermore, if all six factors listed above are considered, it might over quantify the avoided emissions due to recycling. Therefore, there is a need for a fair and reasonable consideration in order not to over- or under quantify the effects of avoided emissions. Clearly, it is not illustrated in the description by Miner (2010).

3 Proposed Methodology in the Assessment of the Carbon Footprint of Recycled Products from Wood Waste

In the previous section, several methodologies were described to assess the CFP of the recycled products, specifically products with biogenic carbon. In particular, the methodologies discussed previously have estimated the avoided emissions based on scenarios that may not occur. These scenarios are based on the assumptions that emissions associated with activities will be avoided due to recycling. However, there could be a possibility that avoided emissions may be overestimated because the scenarios where activities are avoided may be judgmental.

Due to the limitation of judgmental call, Ng et al. proposed a methodology that is reasonable and conservative to quantify the avoided emissions (Ng et al. 2011, 2014). In that proposed methodology, Ng et al. also considered the avoided emissions as a result of recycling. However, to avoid overestimating the avoided emissions, the methodology only considers the amount of carbon stored in trees that would not have been harvested due to the effect of recycling. This avoidance of emissions is described by Ng et al. as "avoided impact" (Ng et al. 2011). Unlike the scenarios described elsewhere (U.S. Environmental Protection Agency 2010; Eriksson et al. 2010; Miner 2010), the proposed methodology did not include all activities and associated emissions that would have been avoided by recycling. To be more conservative, the avoided emissions are multiplied by a weighting factor (Eq. 5) that will always be equal or less than one. Consequently, only a fraction of the benefits due to the avoidance of emissions is reported.

3.1 Carbon Footprint Calculation

According to PAS 2050, two types of data—**activity data** and **emission factors**—are required for the calculation of CFP (British Standard International 2008). Activity data represent a quantitative measure of an activity. For instance, activity data may refer to the amount of material consumed, amount of energy required for processing, fuel consumed for transport goods, etc. The emission factor is a coefficient that quantifies the carbon emissions or removal of a gas per unit activity. Emission factors are usually derived based on the average of sampled measurements under a given set of operating conditions to develop a representative rate of emission for a given activity level. To quantify CFP, the activity data is multiplied with the associated emission factor. As described earlier, CFP typically refers to the six types of GHG emissions, which include CO_2, CH_4, N_2O, SF_6, HFCs, and PFCs. The unit of measurement is expressed in weight of carbon dioxide equivalents or CO_2e. All values are then aggregated to derive the total CFP. This is illustrated in Eq. 1.

$$\text{CFP} = \sum_{i}^{N} (\text{AD}_i \times \text{EF}_i) \tag{1}$$

where:

CFP is carbon footprint
AD_i is activity data for ith activity
EF_i is emission factor for ith activity
N is the total number of activity.

3.2 Carbon Storage Calculation

According to PAS 2050 (British Standard International 2008), the impact of carbon storage or uptake of atmospheric carbon shall be reflected as the weighted average time of storage during the 100-year assessment period, provided that this impact occurs over the product's lifecycle within the 100-year assessment period. Specifically, if a product has full carbon storage benefit occurring between 2 and 25 years after its formation, a weighting factor (Eq. 3) shall be applied to its carbon storage. Equations 2 and 3 show the carbon storage computation.

$$\text{CFP}_{\text{cs}} = \text{AD}_{\text{cs}} \times \text{WF}_{\text{cs}} \times (-\text{EF}_{\text{cs}}) \tag{2}$$

where:

CFP_{CS} is the carbon footprint due to carbon storage
AD_{CS} is the activity data of carbon storage
EF_{CS} is the emission factor of carbon storage
WF_{CS} is the weighting factor due to carbon storage, which is defined as:

$$\text{WF}_{\text{cs}} = \frac{0.76 \times t_o}{100} \tag{3}$$

where t_o is the number of years the full carbon storage benefit of a product exists following the formation of the product.

The constant of 0.76 is based on the rate of CO_2 removal from the atmosphere. This figure is derived based on the absorption rate of CO_2 in oceans and also terrestrial and aquatic biomass (Clift and Brandao 2008).

In other cases that are not covered in the previous scenario, a different weighting factor will be adopted. This weighting factor is illustrated in Eq. 4.

$$\text{WF} = \frac{\sum_{i=1}^{100} X_i}{100} \tag{4}$$

where:

i is each year in which carbon storage occurs

X is the proportion of total storage remaining in any year i

WF is the weighting factor for carbon storage.

3.3 Avoided Impact Calculation

To quantify the avoided impact, it follows a similar rationale as the carbon storage. This is because the avoided impact refers to the scenario where carbon is stored in non-harvested trees as a result of recycling. The assumption is that trees are like products that store carbon. Therefore, avoided impact calculation follows the same formula shown in Eq. 2, but the weighting factor is changed to Eq. 5. There is a need to emphasize that Eq. 5 is a heavier weighting factor than Eq. 3, which will result in greater benefit of carbon storage. The rationale for using a heavier weighting factor is that a tree that is still standing can store carbon and, at the same time, absorb carbon dioxide.

$$\mathrm{WF}_{\mathrm{AI}} = \frac{\sum_{i=1}^{100} X_i}{100} \tag{5}$$

where:

i is each year in which carbon storage occurs

X is the proportion of total storage remaining in any year i

$\mathrm{WF}_{\mathrm{AI}}$ is the weighting factor due to avoided impact.

Therefore, CFP due to avoided impact will be quantified according to Eq. 6.

$$\mathrm{CFP}_{\mathrm{AI}} = \mathrm{AD}_{\mathrm{AI}} \times \frac{\sum_{i=1}^{100} X_i}{100} \times (-\mathrm{EF}_{\mathrm{AI}}) \tag{6}$$

where:

$\mathrm{CFP}_{\mathrm{AI}}$ is the carbon footprint due to avoided impact

$\mathrm{AD}_{\mathrm{AI}}$ is the activity data due to avoided impact

$\mathrm{EF}_{\mathrm{AI}}$ is the emission factor due to avoided impact.

4 Case Study of a Door Made from Recycled Wood Waste

This section presents a case study that demonstrates the effects of adopting the proposed methodology for assessing the CFP of products made from recycled wood. In particular, this is a case study to compare the carbon footprint of a door made from two types of materials: typical virgin hardwood and technical wood.

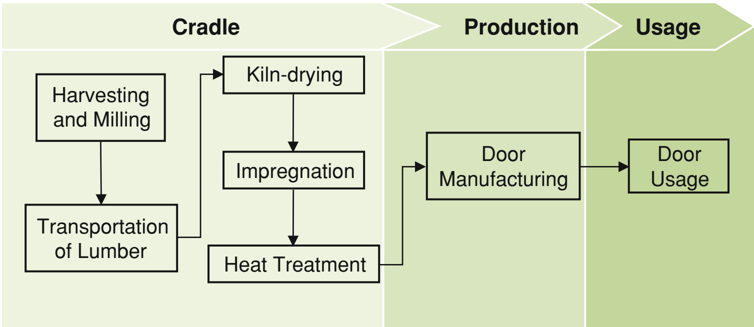

Fig. 11 System boundary for a door made from virgin hardwood. *Source* Ng et al. (2011)

Technical wood is a product created by recycling the wood waste. The scenarios and data are based on a Singapore wood waste recycling plant (LHT Holdings Ltd).

4.1 Goal Definition and Scope

4.1.1 Goal Definition

The goal of this study is to compare the CFP of recycled wood waste with virgin hardwood in the application of a wooden door. From this assessment, a baseline can be established and potential hotspots can be identified for continuous improvement.

4.1.2 Functional Unit

The functional unit selected for this study is one unit of door measuring 2,200 mm by 830 mm that has a product lifespan of 10 years. A baseline of 10 years is selected for comparative study. A sensitivity analysis will be provided towards the later section to analyze the effects of product lifespan to carbon footprint.

4.1.3 System Boundaries for Comparative Study of the Door

The system boundary determines the activities that should be included in the assessment. In a comparative study, both system boundaries should have the same scope. The scope of this comparative study covers life cycle stages, which include cradle, production, and usage. Figures 11 and 12 show the system boundaries. In both scenarios, the door knob, hinges, and surface coatings are excluded from the study.

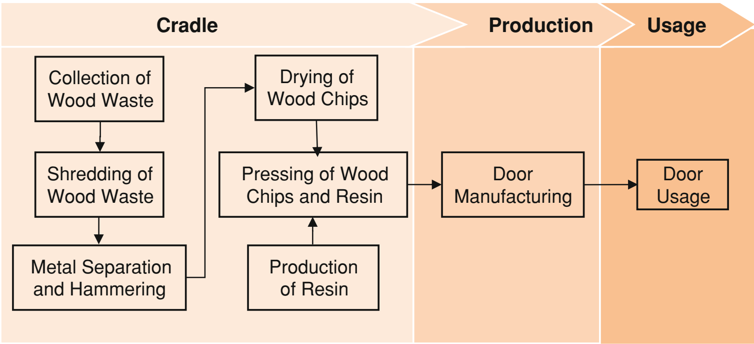

Fig. 12 System boundary for a door made from recycled technical wood. *Source* (Ng et al. 2011)

4.2 Results of the Comparative Assessment

The results of the comparative assessment are shown in the Life Cycle Inventory (LCI). The LCI is a list that includes the activities, activity data, emission factors, and respective CFP for each activity.

4.2.1 Life Cycle Inventory for Virgin Hardwood Timber

Typically, kapur is used to make virgin hardwood door. This type of hardwood is available in Pahang, Malaysia. Therefore, the virgin hardwood is harvested in Pahang, Malaysia. After harvesting, the hardwood is processed and milled before being transported to LHT in Singapore. On average, the density of kapur is 800 kg/m^3 at a moisture content of 12 % (Hopewell 2010). When the virgin hardwood lumber arrives at LHT, the average moisture content is approximately 40 %. The virgin hardwood lumber needs to kiln-dried to an average moisture content of 15 %. Table 2 lists the LCI to process 1 m^3 of virgin hardwood timber before door production. The CFP to process 1 m^3 of virgin hardwood timber is 131.2 kg CO_2e.

4.2.2 Life Cycle Inventory for a Virgin Hardwood Door

Based on the stated dimension of the door (2200 mm by 830 mm), the required amount of wood is derived as 0.026 m^3. To make the door fire retardant, an impregnation process involving a fire-retardant chemical is required. The main active ingredient of the fire retardant is boric acid. Hence, the emission factor is based on the emissions from the production of fire retardant. After the impregnation process, the timber is dried using a heat treatment process. Subsequently, the treated-hardwood timber is manufactured into a virgin hardwood door. Due to engineering scrap, a large amount of waste is generated from the door manufacturing. Therefore, approximately 50 % more virgin hardwood timber and fire

Table 2 Life Cycle Inventory per cubic meter of virgin hardwood timber

Activity	Activity data	Unit	Emission factor (kg CO_2e/unit of activity data)	CFP (kg CO_2e/ m^3 of timber)	Data sources	
					Activity data	Emission factor
Harvesting and milling	800	kg/m^3	0.087	69.924	Primary; Hopewell, (2010), Gan et al. (1999)	McCallum (2009), Ministry for the Environment New Zealand (2006), U.S. Department of Energy, Energy Information Administration (2009), Bergman (2008)
Transportation of lumber	440	ton-km	0.134	58.953	Primary; calculations	ELCD/PE-Gabi
Kiln-drying	4	kW h	0.576	2.304	Primary	National Environment Agency (2009)
Total				131.181		

Source Ng et al. (2011)

retardant are required. Similarly, 50 % more energy is required for impregnation and heat-treatment processes. Because the virgin hardwood door has a lifespan of 10 years, the benefit of carbon storage will be credited to the virgin hardwood door based on the computation method in Sect. 3.2. Table 3 lists the LCI to manufacture one unit of virgin hardwood door. The CFP of one functional unit of virgin hardwood door is 16.2 kg CO_2e.

4.2.3 Life Cycle Inventory for Technical Wood Timber

The technical wood timber is made from the collected wood waste in Singapore. The collected wood waste undergoes shredding, metal separation and hammering, and drying to produce dried wood chips. The amount of wood chips used will determine the density of the timber. The density of the technical wood door is approximately 840 kg/m^3 with a moisture content of 8 %. To make up this density, approximately 90 % by volume of the technical wood timber is made up of wood chips. The other 10 % will be the resin required to bond the wood chips together. The resin is a special mixture with 65 % of melamine-urea-formaldehyde (MUF) and 35 % of water by volume. The mixture of wood chips and the resin are then pressed by the mold to form the technical wood. To make the technical wood pest- and fungus-resistant, it has to undergo a high-temperature steaming process. Table 4 lists the LCI to process 1 m^3 of technical wood timber before door production. The CFP to process 1 m^3 of technical wood timber is 143.3 kg CO_2e.

4.2.4 Life Cycle Inventory for a Technical Wood Door

Unlike the virgin hardwood, technical wood has fire-, pest-, and fungus-resistant properties. Therefore, technical wood does not need to undergo impregnation and post-heat treatment processes. Another technical advantage over virgin hardwood is that technical wood can be pressed and molded into near-net shape blocks for door manufacturing. Consequently, there is a great reduction of material wastage and the engineering scrap rate is reduced to 3 %. This implies that the amount of technical wood timber required for the door is 0.027 m^3. Similar to the case of the virgin hardwood door, there is also a carbon storage benefit because the lifespan of the door is 10 years. In addition, there is an avoided impact credit for the technical wood door, as stated in Sect. 3.3. The avoided impact is due to the effects of recycling, which reduce the need to harvest trees. As a result, carbon can continue to be stored in non-harvested trees. The LCI to manufacture a technical wood door is shown in Table 5.

The net total CFP for one functional unit of technical wood door is −2.9 kg CO_2e. Care should be taken when interpreting negative emissions. In this case, the production of a technical wood door does not lead to negative emissions of 2.9 kg CO_2e. Instead, negative emissions refer to the potential avoidance of emissions by the technical wood door relative to the scenario of manufacturing a virgin

Table 3 Life Cycle Inventory per functional unit of virgin hardwood door

Activity	Activity data	Unit	Emission factor (kg CO_2e/unit of activity data)	CFP (kg CO_2e/m^3 of door)	Data sources	
					Activity data	Emission factor
Virgin hardwood	0.053	m^3	131.181	6.904	Primary	Table 2
Fire retardant	0.158	kg	0.048	0.008	Conrad Forest Products (2002), Arch Timber Protection (2008)	IDEMAT, Ecoinvent/SimaPro
Impregnation	0.532	kW h	0.576	0.306	Ecoinvent/SimaPro	National Environment Agency (2009)
Heat treatment	0.211	kW h	0.576	0.121	Conrad Forest Products (2002), Arch Timber Protection (2008)	National Environment Agency (2009)
Door manufacturing	20	kW h	0.576	11.519	Primary	National Environment Agency (2009)
Carbon storage	0.026	m^3	−99.527	−2.619	Primary	Nebel et al. (2009); calculations
Total				16.239		

Source Ng et al. (2011)

Table 4 Life Cycle Inventory per cubic meter of technical wood timber

Activity	Activity data	Unit	Emission factor (kg CO_2e/unit of activity data)	CFP (kg CO_2e/m^3 of timber)	Data sources	
					Activity data	Emission factor
Collection of wood waste	14.743	ton-km	0.134	1.976	Primary, calculations	ELCD/GaBi
Shredding, metal separation and hammering, drying, pressing, and steaming	240	kW h	0.274	65.798	Primary	Kannan et al. (2005); calculations
Production of resin	46.8	kg	1.614	75.521	Primary	National Renewable energy laboratory (2012)
Total				143.294		

Source Ng et al. (2011)

Table 5 Life cycle inventory per functional unit of technical wood door

Activity	Activity data	Unit	Emission factor (kg CO_2e/unit of activity data)	CFP (kg CO_2e/m^3 of door)	Data sources	
					Activity data	Emission factor
Technical Wood	0.027	m^3	143.294	3.888	Primary	Table 4
Door Manufacturing	20	kW h	0.576	11.519	Primary	National Environment Agency (2009)
Carbon Storage	0.026	m^3	−97.821	−2.574	Primary	Nebel et al. (2009); calculations
Avoided Impact	0.120	m^3	−130.957	−15.772	Calculations	Nebel et al. (2009); calculations
Total				−2.940		

Source Ng et al. (2011)

hardwood door. Without this relative scenario, the CFP of one functional unit of technical wood door will be 12.8 kg CO_2e. Nevertheless, the CFP of a technical wood door is still lower than a virgin hardwood door, by approximately 21 %. Figure 13 compares the CFP of a virgin hardwood door and a technical wood door, with and without considering the effects of avoided impact.

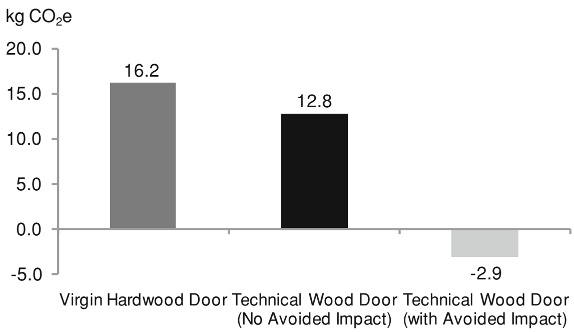

Fig. 13 Carbon footprint of a virgin hardwood door and a technical wood door, with and without considering the effects of avoided impact. *Source* Ng et al. (2011)

4.3 Discussion

The CFP of producing 1 m^3 of technical wood timber is 143.3 kg CO_2e, which is higher than the CFP of producing 1 m^3 of virgin hardwood timber (131.2 kg CO_2e). Even though technical wood timber is approximately 9.2 % higher than virgin hardwood timber, the cradle to end-of-use CFP of technical wood door (−2.9 kg CO_2e) is approximately 1.2 times lower than the CFP of virgin hardwood door (16.2 kg CO_2e). However, the CFP of a technical wood door has considered the effect of avoided impact, which results in negative emissions. Again, there is a need to emphasize that producing a technical wood door does not result in negative emissions. It is only in this comparative scenario that avoided impact has been taken into account. If the technical wood door is compared with other scenarios, such as the use of different types of virgin wood, the avoided impact will be different. This is because there will be other factors that have to be considered, such as the waste generated during milling and harvesting, engineering scrap during production, etc.

4.4 Contribution Analysis for Virgin Hardwood Door and Technical Wood Door

In this section, the activities that contribute to the CFP are analyzed to determine the hotspots. From Fig. 15, one of the main factors is avoided impact, which contributes 46.73 % of the CFP. As discussed previously, the negative emissions are due to the avoided emissions as a result of recycling. Specifically, the production of a technical wood door uses wood waste as a raw material, thus avoiding the harvesting of virgin hardwood. The volume of non-harvested virgin hardwood is approximately 0.12 m^3 of kapur. This volume of virgin hardwood if left standing in the forest will absorb carbon dioxide and store it as carbon for 10 years.

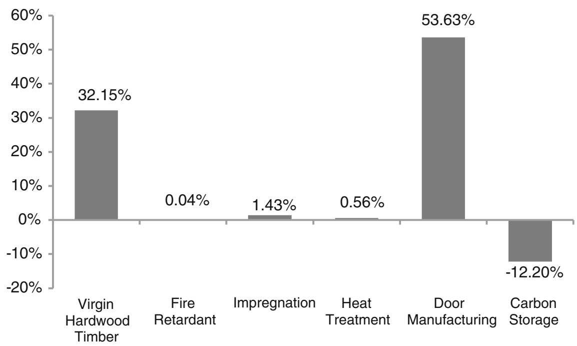

Fig. 14 Carbon footprint breakdown per functional unit of virgin hardwood door. *Source* Ng et al. (2011)

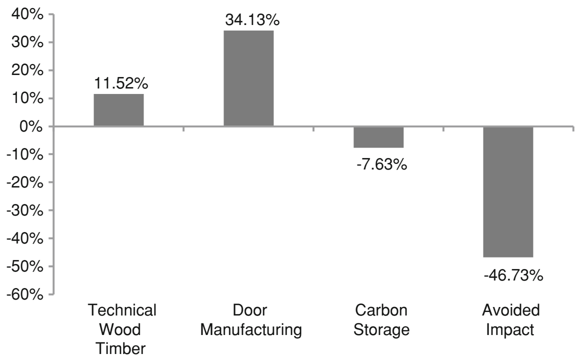

Fig. 15 Carbon footprint breakdown per functional unit of technical wood door. *Source* Ng et al. (2011)

There are two main reasons why greater amounts of virgin hardwood are used in the production of virgin hardwood doors. Firstly, the milling of harvested logs into planed timber generates a high amount of waste. Based on studies carried out by Bergman, only 43.7 % of harvested log is converted into planed timber (Bergman 2008). In order to produce 1 m^3 of planed timber, 2.3 m^3 of logs are required. Secondly, the door production also generates high amounts of waste. The engineering scrap is approximately 50 %. Therefore, it requires 50 % more virgin hardwood timber. In addition, there is a need to use 50 % more fire retardant, as well as energy for impregnation and heat-treatment processes. These additional requirements result in 32.15 % of the CFP for virgin hardwood doors (Fig. 14). On the other hand, technical wood timber contributes 11.52 % to the CFP of technical wood doors. Another significant hotspot in both cases is the door manufacturing. The energy consumption results in 53.63 and 34.13 % of the CFP for virgin hardwood doors and technical wood doors, respectively.

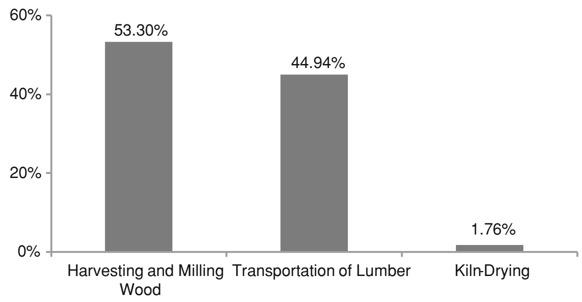

Fig. 16 Carbon footprint breakdown per cubic meter of virgin hardwood timber. *Source* Ng et al. (2011)

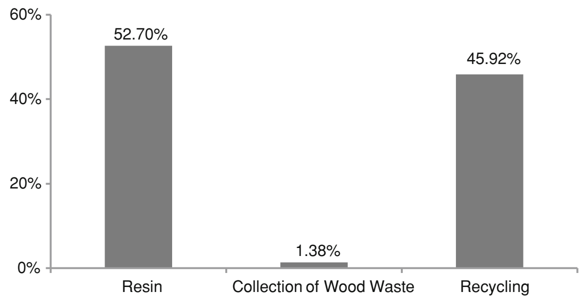

Fig. 17 Carbon footprint breakdown per cubic meter of technical wood timber. *Source* Ng et al. (2011)

4.5 Contribution Analysis for Virgin Hardwood Timber and Technical Wood Timber

Referring to Fig. 16, the most significant contribution is the harvesting and milling of virgin wood. Due to limited site-specific data, it is assumed that the activity data or operational data are similar to (McCallum 2009) and (Bergman 2008), with slight contextualization. The emission factors are also contextualized from the data in (Ministry for the Environment New Zealand 2006; U.S. Department of Energy, Energy Information Administration 2009; Malaysian Grid Emission Factor Calculation 2008). As explained in Sect. 4.4, a huge amount of waste is generated when the harvested logs are milled and planed into timber. This has resulted in a higher CFP.

Transportation of virgin hardwood lumber is another major contributor of CFP. Approximately 44.94 % of CFP per cubic meter of virgin hardwood timber is due to the transportation of virgin hardwood lumber from Pahang, Malaysia to LHT in Singapore. In the case of technical wood, transportation or collection of wood waste has a relatively insignificant contribution, accounting for less than 2 % (Fig. 17). It is demonstrated in this specific case that recycling of wood waste in Singapore reduces reliance on timber imports and long-haul transportation of timber, thereby contributing less to the CFP.

Table 6 Sensitivity analysis of door lifespans

	Door lifespan				
	10 years	15 years	20 years	25 years	30 years
Virgin hardwood door (kg CO_2e/door)	16.2	14.9	13.6	12.3	8.5
Technical wood door before adjustment (kg CO_2e/door)	12.8	11.5	10.3	8.9	5.2
Adjustment (Avoided impact)[a] (kg CO_2e/door)	−15.8	−23.6	−31.5	−39.4	−47.3
Technical wood door after adjustment (kg CO_2e/door)	−2.9	−12.1	−21.3	−30.5	−42.1

The values displayed above have been rounded up to one decimal place

[a] The avoided impact is due to the non-harvesting of 0.120 m^3 of kapur tree to produce virgin hardwood door for a baseline of 10 years. The avoided impact can only be attributed to the technical wood door as savings under scenarios set in this study. The longer the door lifetime of technical wood door, the greater the potential avoided impact

4.6 Sensitivity Analysis of Door Lifespans

In the earlier comparative study scenario, the base case of the door lifetime is 10 years. Because the emissions reduction due to carbon sequestration and avoided impact will increase as door lifespan increases, sensitivity analysis is carried out to forecast the change in emissions impact (Table 6).

As the door lifespan increases, the carbon footprint for both virgin hardwood doors and technical wood doors decreases. This is expected because the weighting factor (Eq. 3) increases as the door lifespan increases. Of note, when the door lifetime increases to beyond 25 years, the carbon footprint for both virgin hardwood doors and technical wood doors will decrease at an increasing rate (Fig. 18). Recall from Eq. 3 that the weighting factor (WF_{CS}) is applicable if the product can exist for between 2 and 25 years after the formation of the product. If the product (in this case, the door) can exist beyond 25 years, another weighting factor will apply. The weighting factor to be used is assumed to the same as the weighting factor for avoided impact (WF_{AI}) (Eq. 2). The rationale for assigning a heavier weight is to give more credit to a product that stores carbon for a longer lifetime (>25 years). Hence, the amount of the carbon footprint will decrease at an increasing rate should the lifespan of the door exceed 25 years.

4.7 Recommendations

To further reduce the CFP of the technical wood door, efforts may focus on the resin and recycling process. As shown in Fig. 17, resin and recycling contribute 52.7 and 45.92 % to the technical wood timber CFP, respectively.

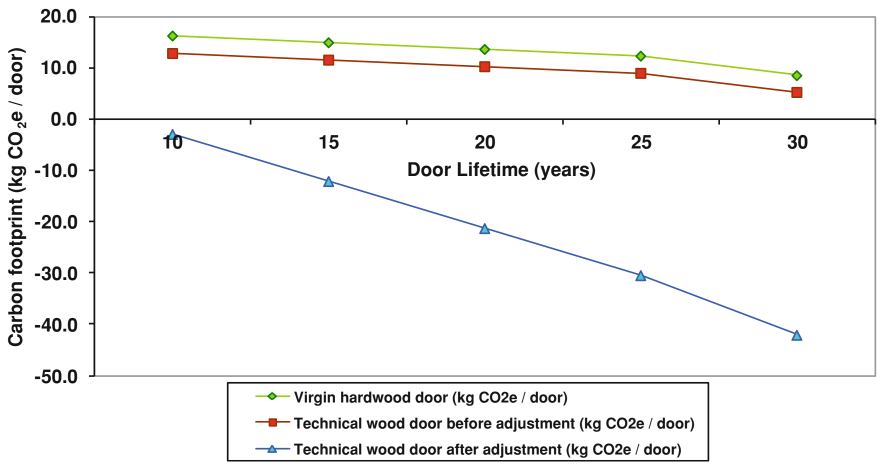

Fig. 18 Carbon footprint of the door at varying lifespans

Table 7 Resin alternatives and respective carbon footprint (CFP) reduction potential

Types of resin	CFP (kg CO_2e/m^3)	% change	CFP (kg CO_2e/door)	% change
Melamine-urea-formaldehyde (Baseline)	143.3	Base	12.8	Base
Melamine-urea-formaldehyde-1241	134.4	−6.20 %	12.6	−1.88 %
UF-1205	133.5	−6.83 %	12.6	−2.07 %
UF-1206	135.1	−5.69 %	12.6	−1.72 %

Source Ng et al. (2011)

Table 7 shows a list of resin alternatives that could possibly be used to replace MUF and achieve potential CFP reductions. The types of resin listed are Casco Products from Sweden. Their respective emission factors are from (Nilsson and Pålsson 2001). The CFP reduction potentials for the technical wood timber range from 5.69 to 6.83 %. The resin that shows the highest CFP reduction potential is resin UF-1205. It has a CFP reduction potential of 6.83 % per cubic meter of technical wood timber. The reduction of CFP in technical wood timber also led to a reduction of CFP in technical wood doors. Assuming resin UF-1205 is used for technical wood timber, the CFP of the technical wood door can be reduced by 2.07 %. However, there could be technical feasibility issues in terms of bonding strength if the resin were to be replaced. Therefore, a more in-depth feasibility assessment would have to be carried out before replacing the resin.

In the case of recycling, reducing energy consumption will lead to CFP reduction. A sensitivity analysis is carried out to assess the CFP reduction potential if the energy consumption were to be reduced. It can be observed from Table 8 that for every 5 % reduction in energy consumption during recycling process, there

Table 8 Sensitivity analysis of energy consumption in recycling and the respective carbon footprint (CFP) reduction potential

Energy reduction	CFP (kg CO_2e/m^3)	% change	CFP (kg CO_2e/door)	% change
Baseline	143.3	Base	12.8	Base
5 %	140.0	−2.30 %	12.7	−0.70 %
10 %	136.7	−4.59 %	12.7	−1.39 %
15 %	133.4	−6.89 %	12.6	−2.09 %
20 %	130.1	−9.18 %	12.5	−2.78 %

Source Ng et al. (2011)

are CFP reduction potentials of 2.3 % and 0.7 % in technical wood timber and technical wood doors, respectively.

4.8 Future Work

A case study was presented in this chapter to compare the CFP of a door made from virgin hardwood and recycled wood waste (technical wood). In particular, the concept of avoided impact has been demonstrated to assess the effect of recycling. Results show that a virgin hardwood door (16.2 kg CO_2e) has higher CFP than a technical wood door (12.8 kg CO_2e). The CFP of the technical wood door may even be lower (−2.9 kg CO_2e) if avoided impact is taken into account. Because recycling of wood waste avoids the need to harvest virgin wood, which delays the release of CFP, the avoided impact is thus credited to the technical wood door.

The contribution analysis highlights several significant factors that led to virgin hardwood doors having a higher CFP. Firstly, there is a high amount of waste generated (50 % waste) during door manufacturing. Secondly, there is high energy consumption during door manufacturing. Thirdly, there is low conversion rate of 43.7 % when harvested logs are milled and planned into timber. Fourthly, long-haul transportation is required for importing virgin hardwood lumber from Malaysia to Singapore.

Even though a technical wood door has a lower CFP than a virgin hardwood door, the contribution analysis has identified hotspots for reduction potentials. There are several significant factors that have resulted in a significant CFP: (1) high energy consumption during door manufacturing; (2) high energy consumption for recycling wood waste into technical wood; and (3) the use of resin for bonding the wood chips in technical wood.

In the case study presented herein, a baseline of 10-year lifespan was used to compare virgin hardwood doors and technical wood doors. Sensitivity analysis revealed that the carbon footprint will decrease for both the virgin hardwood door and the technical wood door if the lifespan increases. In particular, the carbon footprint will reduce at an increasing rate should the lifespan of the door exceed 25 years.

The scenario analysis of replacing resins shows that the CFP reduction potentials for the technical wood timber range from 5.69 to 6.83 %. In addition, assuming resin UF-1205 is used for technical wood timber, the CFP of the technical wood door can be reduced by 2.07 %. However, there could be technical challenges in terms of bonding strength if the resin were to be replaced. Therefore, a more in-depth feasibility assessment would have to be carried out before replacing the resin.

Sensitivity analysis shows that for every 5 % reduction in energy consumption during recycling process, there are CFP reduction potentials of 2.3 % and 0.7 % for technical wood timbers and technical wood doors, respectively. Hence, future works may improve the energy efficiency of the recycling process to reduce CFP.

5 Conclusion

Recycling is a process that takes materials or products that are at end-of-life and transforms them into either the same product or a secondary product. When a material is recycled, it is used in place of virgin inputs in the manufacturing process, rather than being disposed of and managed as waste. Therefore, recycling—especially recycling of wood waste—is beneficial in delaying the release of GHGs as well as leading to increased carbon storage in trees. With limited land space and scarce natural resources, there is a huge incentive for Singapore to increase recycling rates. Furthermore, recycling leads to reductions in carbon footprint and lower environmental impact.

To quantify the potential environmental benefits of recycling wood waste, three approaches were introduced. However, there are several limitations associated with these approaches. To avoid under- and over-estimating the avoided emissions due to recycling of wood waste, a methodology for fair and reasonable assessment was introduced. In particular, the term "avoided impact" was adopted to describe the scenario where carbon is stored in non-harvested trees as a result of recycling. The assumption is that trees are like products that store carbon.

A case study of a local wood waste recycling plant was presented to illustrate the proposed methodology. Results show that the recycled technical wood product has a lower carbon footprint (12.8 kg CO_2e) than the virgin hardwood product (16.2 kg CO_2e). When the effects of avoided impact are taken into account, the carbon footprint of the technical wood product may have an even lower carbon footprint (−2.9 kg CO_2e), clearly illustrating the environmental benefits of recycling wood waste.

Despite the environmental benefits, there is still room for improvement. A scenario analysis of replacing resins showed that the CFP of a technical wood door could be reduced by 2.07 %. Furthermore, a sensitivity analysis showed that for every 5 % reduction in energy consumption during the recycling process, there are CFP reduction potentials of 2.3 and 0.7 % in technical wood timber and technical wood doors, respectively. Through this case study, the environmental benefits have been highlighted. In particular, a product made from recycled wood waste has resulted in a lower CFP than a virgin wood product. It is hoped that the demonstration of quantifying the CFP of a recycled wood waste product will inspire and influence the public acceptance of recycled products to make a more sustainable living environment, in Singapore and the world.

References

Arch Timber Protection (2008) Fire retardant coatings for timber based products. Specifier's Guide, Arch Timber Protection

Ardente F, Cellura M (2011) Economic allocation in life cycle assessment, the state of the art and discussion of examples. J Ind Ecol 16(3):387–398

Azapagic A, Clift R (1999) Allocation of environmental burdens in multiple-function systems. J Clean Prod 7(2):101–119

Bergman RD (2008) Environmental impact of producing hardwood lumber using life-cycle inventory. Wood Fiber Sci 40(3):448–458

Boustead I, Dove WT, Halada K, Matsuno Y (1999) Primary metal industry ecoprofile calculations: a discussion of allocation methods. Opgeroepen op 1 Nov 2013, van http://www.boustead-consulting.co.uk/download/gifu99.pdf

British Standard International (2008) Specification for the assessment of the life cycle greenhouse gas emissions of goods and services. British Standard International

CEPI (2007) Framework for the development of carbon footprints for paper and board products (including separate appendices). Confederation of European Industries

Chua CB, Lee HM, Low JS (2010) Life cycle emissions and energy study of biodiesel derived from waste cooking oil and diesel in Singapore. Int J Life Cycle Assess 15(4):417–423

Clift R, Brandao M (2008) Carbon storage and timing of emissions—a note by Roland Clift and Miguel Brandao. Centre for Environmental Strategy, University of Surrey, Guildford

Conrad Forest Products (2002) Material safety data sheet: DRICON® Fire retardatnt treated wood and lumber. Conrad Forest Products

Department of Statistics Singapore (2013) Statistics. Opgeroepen op 12 Nov 2013, van http://www.singstat.gov.sg/statistics/latest_data.html#4

Ekvall T (1999) Key methodological issues for life cycle inventory analysis of paper recycling. J Clean Prod 7(4):281–294

Eriksson E, Karlsson P-E, Hallberg L, Jelse K (2010) Carbon footprint of cartons in Europe—carbon footprint methodology and biogenic carbon sequestration

Gan KS, Choo KT, Lim SC (1999) Timber notes—light hardwoods (Mersawa, Nyatoh, Pelajau, Penarahan, Perupok). Timber Technology Bulletin No 15

Hertwich E, Peters G (2009) Carbon footprint of nations: a global, trade-linked analysis. Environ Sci Technol 43(16):6414–6420

Hopewell G (2010) Information of kapur. Opgeroepen op 10 Nov 2010, van http://www.daff.qld.gov.au/26_5423.htm

ISO (2004) Environmental management systems—requirements with guidance for use. International Organization for Standardization, Geneva

ISO (2006) ISO 14044: Environmental management—life cycle assessment—requirements and guidelines. International Organization for Standardization, Geneva

ISO (2013) ISO/TS 14067: Greenhouse gases—carbon footprint of products—requirements and guidelines for quantification and communication. International Organization for Standardization, Geneva

Kannan R, Leong KC, Osman R, Ho HK, Tso CP (2005) Gas fired combined cycle plant in Singapore: energy use, GWP and cost—a life cycle approach. Energy Conversion and Management, pp 2145–2157
Khoo HH, Tan BH, Sagisaka M (2008) Utilisation of woody biomass in Singapore: technological options for carbonisation and economic comparison with incineration. Int J LCA 13:312–318
Lindsey T (2000) Key factors for promoting pollution prevention technology adoption. Pollut Prev Rev 10(1):1–12
Lu JP (2013) Design of durable concrete mix for sustainability. In: Proceedings of the SB 13 Singapore—realising sustainability in the tropics, Singapore
Malaysian Grid Emission Factor Calculation (2008) Malaysian grid emission factor calculation
McCallum D (2009) Report on carbon footprint project—Nelson forest
Miner R (2010) Impact of the global forest industry on atmospheric greenhouse gases. Food and Agriculture Organisation of the United Nations, Rome
Ministry for the Environment New Zealand (2006) Fuel combustion emission factors (transport fuels). Opgeroepen op 10 Nov 2009, van http://www.mfe.govt.nz/publications/climate/guidancegreenhouse-
National Environment Agency (2011) Collectors/traders for recyclables. Opgeroepen op 28 Sept 2013, van http://app2.nea.gov.sg/docs/default-source/energy-waste/waste-management/wood-horticultural-waste.pdf?sfvrsn=0
National Environment Agency (2009) Information on emission factors (For CDM projects in Singapore). Opgeroepen op 10 Nov 2010, van Information on Emission Factors: http://www.nccc.gov.sg/informationOnEmissionFactors.pdf
National Environment Agency (2013b) Waste management overview. Opgeroepen op 11 Nov 2013, van http://app2.nea.gov.sg/energy-waste/waste-management/overview
National Environmental Agency (2013a) Waste statistics and overall recycling. Opgeroepen op 12 Nov 2013, van http://app2.nea.gov.sg/energy-waste/waste-management/waste-statistics-and-overall-recycling
National Renewable Energy Laboratory (2012) U.S. life cycle inventory database. Opgeroepen op 9 Sept 2012, van, U.S. Life-Cycle Inventory database: http://www.nrel.gov/lci/
Nebel B, Alcorn A, Wittstock B (2009) Life cycle assessment: adopting and adapting overseas LCA data and methodologies for building materials in New Zealand. Ministry of Agriculture and Forestry
Ng R, Shi CW, Low JS, Lee HM, Song B (2011) Comparative carbon footprint assessment of door made from recycled wood waste versus virgin hardwood: case study of a Singapore wood waste recycling plant. In: Hesselbach J, Herrmann C (eds) Glocalized solutions for sustainability in manufacturing. Proceedings of the 18th CIRP international conference on life cycle engineering. Springer, Berlin, pp 629–634
Ng R, Shi CW, Tan HX, Song B (2014) Avoided impact quantification from recycling of wood waste in Singapore: an assessment of pallet made from technical wood versus virgin softwood. J Clean Prod 65:447–457
Ng R, Yeo Z, Shi CW, Rugrungruang F, Song B (2012) An algorithmic approach to streamlining product carbon footprint quantification: a case study on sheet metal parts. Int J Autom Technol 6(3):312–321
Nilsson B, Pålsson A-C (2001) SPINE@CPM. Chalmers University of Technology, Sweden
Rugrungruang F, Ng R, Teo T (2009) Carbon footprint assessment of R3PlasTM bio-composite material for eco-design of plant box. In: EcoDesign 2009—6th international symposium on environmentally conscious design and inverse manufacturing, Sapporo, Japan
Shi CW, Rugrungruang F, Yeo Z, Gwee HK, Ng R, Song B (2011a) Identifying carbon footprint reduction opportunities through energy measurements in sheet metal part manufacturing. In Hesselbach J, Herrmann C (eds) Glocalized solutions for sustainability in manufacturing. Springer, Berlin, pp 389–394
Shi CW, Rugrungruang F, Yeo Z, Song B (2011b) Carbon footprint analysis for energy improvement in flour milling production. In Hesselbach J, Herrmann C (eds) Glocalized Solutions for Sustainability in Manufacturing. Springer, Berlin, pp 246–251

Shi CW, Tan HX, Tjandra TB, Ng R, Rugrungruang F, Yeo Z, et al (2012) Effective energy management through energy monitoring: case study of sheet metal part manufacturing. In: Proceedings of international congress on informatics, environment, energy and applications, Singapore

Shi CW, Yeo Z, Ng R, Tan HX, Tjandra TB, Song B (2013) Carbon footprint assessment of a four storey industrial building constructed in Singapore. In: Proceedings of the SB 13 Singapore—realising sustainability in the tropics, Singapore

Simpson D (2012) Knowledge resources as a mediator of the relationship between recycling pressures and environmental performance. J Clean Prod 22:32–41

U.S. Department of Energy, Energy Information Administration (2009) Fuel emission factors from Appendix H of the instructions to form EIA-1605

U.S. Environmental Protection Agency (2012b) Forest carbon storage. U.S. Environmental Protection Agency, Washington, DC

U.S. Environmental Protection Agency (2010) Forest carbon storage in EPA's waste reduction model. U.S. Environmental Protection Agency, Washington, DC

U.S. Environmental Protection Agency (2012a) Recycling. U.S. Environmental Protection Agency, Washington, DC

Yang QZ, Ng RS, Qi GJ, Low HC (2009) Economic and environmental impact evaluation of nickel recovery. SIMTech Tech Rep 10(4):244–249

Zero Waste Singapore (2008, December 8) Wood and horticultural waste recycling. Opgeroepen op 15 Oct 2013, van http://www.zerowastesg.com/2008/12/08/wood-and-horticultural-waste-recycling/

Sector-Wise Assessment of Carbon Footprint Across Major Cities in India

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Abstract The concentration of greenhouse gases in the atmosphere has increased rapidly due to anthropogenic activities, resulting in a significant increase of the earth's temperature and causing global warming. These effects are quantified using an indicator such as global warming potential, expressed in units of carbon dioxide equivalent (CO_2eq), to indicate the carbon footprint of a region. Carbon footprint is thus a measure of the impact of human activities on the environment in terms of the amount of greenhouse gases produced. This chapter focuses on calculating the amount of three important greenhouses gases—carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O)—and thereby determining the carbon footprint of the major cities in India. National greenhouse gas inventories are used for the calculation of greenhouse gas emissions. Country-specific emission factors are used where all the emission factors are available. Default emission factors from Intergovernmental Panel on Climate Change guidelines are used when there are no country-specific emission factors. Emission of each greenhouse gas is estimated by multiplying fuel consumption by the corresponding emission factor. To calculate total emissions of a gas from all its source categories, emissions are summed over all source categories. The current study estimates greenhouse gas emissions (in terms of CO_2 equivalent) in major Indian cities and explores the linkages with the population and gross domestic product (GDP). Carbon dioxide equivalent emissions from Delhi, Greater Mumbai, Kolkata, Chennai, Greater Bangalore, Hyderabad, and Ahmedabad were found to be 38633.2, 22783.08, 14812.10,

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S. S. Muthu (ed.), *Assessment of Carbon Footprint in Different Industrial Sectors, Volume 2*, EcoProduction, DOI: 10.1007/978-981-4585-75-0_8,

22090.55, 19796.5, 13734.59, and 9124.45 Gg CO_2eq, respectively. The major sector-wise contributors to the total emissions in Delhi, Greater Mumbai, Kolkata, Chennai, Greater Bangalore, Hyderabad, and Ahmedabad are the transportation sector (32, 17.4, 13.3, 19.5, 43.5, 56.86 and 25 %, respectively), the domestic sector (30.26, 37.2, 42.78, 39, 21.6, 17.05 and 27.9 %, respectively), and the industrial sector (7.9, 7.9, 17.66, 20.25, 12.31, 11.38 and 22.41 %, respectively). Chennai emits 4.79 tons of CO_2 equivalent emissions per capita, the highest among all the cities, followed by Kolkata, which emits 3.29 tons of CO_2 equivalent emissions per capita. Chennai also emits the highest CO_2 equivalent emissions per GDP (2.55 tons CO_2 eq/lakh Rs.), followed by Greater Bangalore, which emits 2.18 tons CO_2 eq/lakh Rs.

Keywords Carbon footprint · Domestic sector · Global warming potential · Gross domestic product · India · Industries · Major cities · Transportation

1 Introduction

Greenhouse gases are the gaseous constituents of the atmosphere, both natural and anthropogenic, that absorb and emit radiation at specific wavelengths within the spectrum of thermal infrared radiation emitted by the Earth's surface, the atmosphere itself, and clouds (Intergovernmental Panel on Climate Change (IPCC) 2007a, b). The concentration of greenhouse gases (GHGs) in the atmosphere has increased rapidly due to anthropogenic activities, resulting in a significant increase in the temperature of the earth. The energy radiated from the sun is absorbed by these gases, making the lower part of the atmosphere warmer. This phenomenon is known as the natural greenhouse gas effect, whereas the enhanced greenhouse effect is an added effect caused by human activities. Increases in the concentration of these greenhouse gases result in global warming. The atmospheric concentrations of GHGs have increased due to increasing emissions in the industrialization era. Carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) are the major greenhouse gases. Among the GHGs, carbon dioxide is the major contributor to global warming, accounting for nearly 77 % of the global total CO_2 equivalent GHG emissions (IPCC 2007b).

In 1958, attempts were made towards high-accuracy measurements of atmospheric CO_2 concentration to document the changing composition of the atmosphere with time series data (Keeling 1961, 1998). The increasing abundance of two other major greenhouse gases, methane (CH_4) and nitrous oxide (N_2O), in the atmosphere have been reported (Steele 1996). Methane levels were found to increase at a rate of approximately 1 % per year in the 1980s (Graedel and McRae 1980; Fraser et al. 1981; Blake et al. 1982); however, during 1990s, its rate retarded to an average increase of 0.4 % per year (Dlugokencky et al. 1998). The increase in the concentration of N_2O is smaller, at approximately 0.25 % per year (Weiss 1981; Khalil and Rasmussen 1988). A second class of greenhouse gases—the synthetic

HFCs, PFCs, SF_6, CFCs, and halons—did not exist in the atmosphere before the twentieth century (Butler et al. 1999). CF_4, a PFC, is detected in ice cores and appears to have an extremely small natural source (Harnisch and Eisenhauer 1998).

The climate system is a complicated, inter-related system consisting of the atmosphere, land surface, snow and ice, oceans and other bodies of water and living things (Le Treut et al. 2007; Bouwman 1990; Bronson et al. 1997). Climate change is a serious threat to the global community. Rising global temperatures will affect the local climatic conditions and also melt the fresh water ice glaciers, causing the sea levels to rise. There is universal scientific understanding that the earth's climate is changed by GHG emissions generated by human activity (Anthony et al. 2006). Surface air temperature is the parameter generally taken into account for climate change. Extensive studies have been carried out to study the patterns of global and regional mean temperatures with respect to time (Hasselmann 1993; Schlesinger and Ramankutty 1994; North and Kim 1995).

The atmospheric concentrations of carbon dioxide equivalents with the possibility of increases in global temperatures beyond certain levels have been reported (Stern et al. 1996). The recent (globally averaged) warming by 0.5 °C is partly attributable to such anthropogenic emissions (Anthony et al. 2006). Changes in climate also result in extreme weather events, such as very high temperatures, droughts and storms, thermal stress, flooding, and infectious diseases. In the last 100 years, the mean annual surface air temperature has increased by 0.4–0.6 °C in India (Hingane et al. 1985; Kumar et al. 2000). This necessitates understanding the sources of global GHG emissions to implement appropriate mitigation measures.

Carbon Dioxide (CO_2) *Emissions*. CO_2 abundance was found to be significantly lower during the last ice age than over the last 10000 years of the Holocene per initial measurements (Delmas et al. 1980; Berner et al. 1980; Neftel 1982). CO_2 abundances ranged between 280 $\pm$ 20 ppm in the past 10000 years up to the year 1750 (Indermuhle 1999). There was an exponential increase of CO_2 abundance during the industrial era, to 367 ppm in 1999 (Neftel et al. 1985; Etheridge 1996; Houghton et al. 1992; IPCC 1996, 1998, 2000, 2001a, b) and to 379 ppm in 2005.

Methane (CH_4) *Emissions*. Anthropogenic activities such as fossil fuel production, enteric fermentation in livestock, manure management, cultivation of rice, biomass burning, and waste management release methane to the atmosphere to a significant extent. Estimates indicate that human-related activities release more than 50 % of global methane emissions (EPA 2010). Natural sources of methane include wetlands, permafrost, oceans, freshwater bodies, non-wetland soils, and other sources such as wildfires. Accelerating increases in methane and nitrous oxide concentrations were reported during the twentieth century (Machida 1995; Battle 1996). There was a constant abundance of 700 ppb until the nineteenth century. A steady increase brought methane abundances to 1745 ppb in 1998 (IPCC 2001b, 2003) and 1774 ppb in 2005 (IPCC 2006).

Nitrous Oxide (N_2O) Emissions. Nitrous oxide (N_2O) is produced by both natural sources and human-related activities. Agricultural soil management, animal manure management, sewage treatment, mobile and stationary combustion of fossil fuel, and nitric acid production are the major anthropogenic sources. Nitrous oxide is also produced naturally from a wide variety of biological sources in soil and water, particularly from microbial action (EPA 2010). From the measurements for N_2O, it is found that the relative increase during the industrial period is smaller than for other GHGs (15 %). The analysis showed a concentration of 314 ppb in 1998 (IPCC 2001b), rising to 319 ppb in 2005.

1.1 Carbon Emissions and Economic Growth

The transition to a very-low-carbon economy needs elementary changes in technology, regulatory frameworks, infrastructure, business practices, consumption patterns, and lifestyles (McKinnon and Piecyk 2010; Benjamin 2009). The emission of greenhouse gases into the atmosphere has caused concern about global warming, with efforts focusing on minimizing the emissions. Heavy industries are transferred to knowledge-based and service industries, which are relatively cleaner, as economic development continues (Shafik and Bandyopadhyaya 1992). At advanced levels of growth, there was a gradual decrease of environmental degradation because of increased environmental awareness and enforcement of environmental regulation (Stern et al. 1996). There is a need for a target that aids local and national governments in framing climate change policies and regulations.

Carbon dioxide emissions and energy consumption are closely correlated with the size of a country's economy (Cook 1971; Humphrey et al. 1984; Goldemberg 1995; Benjamin 2009). Carbon intensity is one of the most important indicators to help in measuring a country's CO_2 emission with respect to its economic growth. Carbon intensity refers to the ratio of carbon dioxide emissions per unit of economic activity, usually measured as GDP. It presents a clear understanding of the impact of the factors that are responsible for emissions and also helps policy makers to formulate future energy strategies and emission reduction policies (Ying et al. 2007). The analysis of changes in carbon intensity in developing countries helps to optimize fuel-mix and economic structure; meanwhile, it also provides detailed information on mitigation in the growth of energy consumption and related CO_2 emissions.

Carbon intensity value drops if there is a decrease in emissions or sharp rise in the economic growth of a country. Carbon dioxide emissions resulting from the consumption of energy in certain countries were compiled from published literature (International Energy Statistics, United States Energy Information Administration, EIA). Economic growth data were obtained from the World Bank (http://worldbank.org). GDP in domestic currencies were converted using official exchange rates from 2,000. Figure 1 illustrates the carbon intensity trend across major carbon players. India's overall carbon intensity of energy use has marginally decreased in recent years, despite coal's dominance. Strong growth in wind capacity and efficiency improvements in coal-based electricity production are some factors that are responsible for the decline of carbon intensity (Rao and Reddy 2007; Rao et al. 2009).

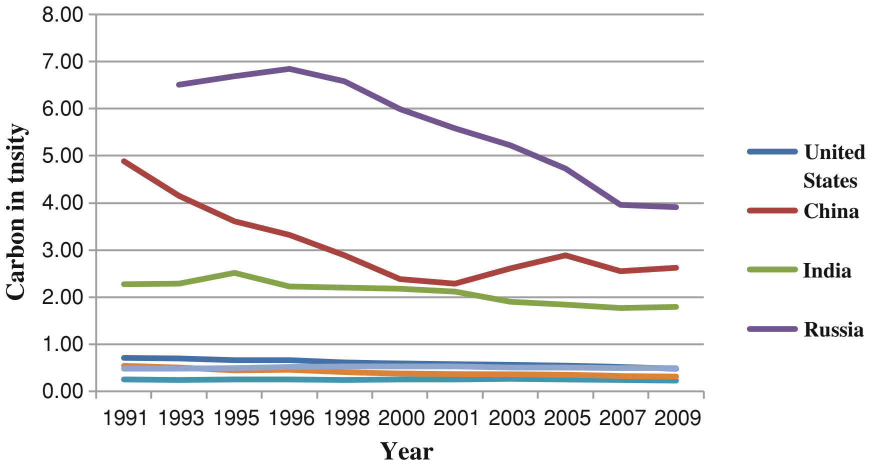

Fig. 1 Carbon intensity by country (kg CO_2/constant US $)

1.2 Carbon Footprint

Many organizations and governments are looking for strategies to reduce emissions from greenhouse gases from anthropogenic origins, which are responsible for global warming (Kennedy et al. 2009, 2010). The increasing interest in carbon footprint assessment has resulted from the growing public awareness of global warming. The global community now recognizes the need to reduce greenhouse gas emissions to mitigate climate change (Jessica 2008). Many global metropolitan cities and organizations are estimating their greenhouse gas emissions and developing strategies to reduce their emissions.

The *carbon footprint* is defined as a measure of the impact of human activities on the environment in terms of the amount of greenhouse gases produced. The total greenhouse gas emissions from various anthropogenic activities from a particular region are expressed in terms of carbon dioxide equivalent, which indicate the carbon footprint of that region (Andrew 2008). Carbon dioxide equivalent (CO_2e) is a unit for comparing the radiative forcing of a GHG (a measure of the influence of a climatic factor in changing the balance of energy radiation in the atmosphere) to that of carbon dioxide (ISO 14064-1, 2006a, b). It is the amount of carbon dioxide by weight that is emitted into the atmosphere and would produce the same estimated radiative forcing as a given weight of another radiatively active gas.

Carbon dioxide equivalents are calculated by multiplying the weight of the gas being measured by its respective global warming potential (GWP). It is a relative measure of how much heat a greenhouse gas traps in the atmosphere. It compares the amount of heat trapped by a certain mass of the gas in question to the amount of heat trapped by a similar mass of carbon dioxide. As defined by the IPCC, a GWP is an indicator that reflects the relative effect of a greenhouse gas in terms of climate change over a fixed time period, such as 100 years (GWP_{100}). GWP is expressed as a factor of carbon dioxide (whose GWP is standardized to 1). GWP depends on factors such as absorption of infrared radiation by a given species, spectral location of its absorbing wavelengths, and the atmospheric lifetime of the species (Matthew 1999). The GWP of major greenhouse gases over the next 20 years are 1 for CO_2, 25 for CH_4, and 298 for nitrous oxide (IPCC 2007a, b).

Need for Estimation of Carbon Footprint. Carbon footprint calculations have the potential to reduce the impact on climate change by increasing consumer awareness and fostering discussions about the environmental impacts of products. They offer valuable information for sustainable urban planning for policy makers and local municipalities (Bhatia 2008; Carbon Trust 2007a, b; Courchene and Allan 2008; Hammond 2007; Hoornweg et al. 2011; Laurence et al. 2011).

1.3 Carbon Footprint Studies in Cities

Emissions of GHG emissions at city levels with a detailed analysis of per capita GHG emissions for several large cities helps in evolving appropriate mitigation measures and resource efficiency (Hoornweg et al. 2011). Kennedy et al. (2009, 2010) developed a method for comparing emissions resulting from electricity consumption, heating and industrial fuel use, transportation, and waste sectors across 10 global cities. Similar studies by Sovacool and Brown (2009) provided a comparative account of carbon footprints in metropolitan areas, with suggestions for policymakers and planners regarding policy implications. The assessment of carbon footprint is being used for the management of climate change and to mitigate changes in climate at local levels. Studies on the carbon footprint of Norwegian municipalities were calculated to be related to the factors of size and wealth (Hogne et al. 2010).

1.4 Sector-Wise Assessment of GHG Emissions in India: Review

GHG Emissions in Electricity Generation Sector in India. GHG emissions from electricity use occur during the generation of electricity. Earlier studies have estimated the emission of gases due to power generation (Gurjar et al. 2004; Raghuvanshi et al. 2006; Chakraborty et al. 2007; Weisser 2007; Kennedy et al. 2009; Shobhakar 2009; Kennedy et al. 2010; Chun Ma et al. 2011; Qader 2009; POST 2006). India's reliance on fossil-fuel based electricity generation has aggravated the problem of high carbon dioxide (CO_2) emissions from combustion of fossil fuels, primarily coal, in the country's energy sector. Combustion of coal at thermal power plants emits mainly carbon dioxide (CO_2), sulfur oxides (SOx), nitrogen oxides (NOx), other trace gases, and airborne inorganic particulates, such as fly ash and suspended particulate matter. Inventory of carbon dioxide emissions from coal-based power generation in India were carried out for the present energy generation, with projections for next two decades (Raghuvanshi et al. 2006). A comprehensive emission inventory for megacity Delhi, India for the period 1990–2000 was developed, in which major CO_2 emissions were found from the power plants. Electricity generation, transport, domestic, industrial processes, agriculture emissions, and waste treatment were the major sectors for which the emission inventories were reported (Gurjar et al. 2004, 2010).

Measurements of CO_2 and other gases from coal-based thermal power plants in India have been reported. The emission rates of the GHGs were found to be dependent on factors such as the quality of coal mixture/oil, quantity used for per unit generation, age of the plant, and amount of excess air fed into the furnace (Chakraborty et al. 2007). A study of large point source emissions from India was carried out (Garg et al. 2001) for 1990 and 1995 using the IPCC (1996) method, indicating that CO_2 and SO_2 emissions were the major gases emitted from power plants.

GHG Emissions in Domestic and Commercial Sectors. Emissions from households and commercial establishments occur due to energy consumption for cooking, lighting, heating, and household appliances. Studies have been carried out using input-output analysis and aggregated household expenditure survey data to calculate the CO_2 emissions from energy consumption for different groups of households (BSI 2008; Murthy et al. 1997a, b; Pachauri and Spreng 2002; Pachauri 2004; Parikh et al. 1997; INCCA 2010; Garg et al. 2004, 2006, 2011). In 2007, at the national level, the residential sector emitted 137.84 million tons of CO_2 equivalents and the commercial sector emitted 1.67 million tons of CO_2 equivalent (INCCA 2010). A city-level emission inventory for key sectors found that the household sector was responsible for a major portion of emissions. Therefore, it is a target sector for emission reduction in both existing and new housing, in which energy efficiency is increased (Gupta et al. 2006; Reddy and Srinivas 2009).

GHG Emissions in the Transportation Sector. Emissions from the road transportation sector are directly related to gasoline and diesel consumption. Increases in emissions have been due to increases in the number of motor vehicles on the road and the distance these vehicles travel (Anil Singh et al. 2008). The traffic composition of six megacities of India (Delhi, Mumbai, Kolkata, Chennai, Bangalore, and Hyderabad) shows that there has been a significant shift from the share of slow-moving vehicles to fast-moving vehicles and public transport to private transport (Jalihal et al. 2005; Jalihal and Reddy 2006). Various studies have been carried out in India with regard to the emissions resulting from the transportation sector (Bhattacharya and Mitra 1998; Ramanathan 1975; Ramanathan and Parikh 1999; MiEF 2004). The trends of energy consumption and consequent emissions of greenhouse gases such as CO_2, CH_4, and N_2O and ozone precursor gases such as CO, NO_x, and NMVOC in the road transport sector in India for the period from 1980 to 2000 have been studied. Efforts are being made to apportion the fuels, both diesel and gasoline, across different categories of vehicles operating on the Indian roads (Anil Singh et al. 2008; Ramachandra and Shwetmala 2009) and determine the major sources of air pollutants in urban areas (Gurjar et al. 2004; Das et al. 2004; Gurjar et al. 2010).

Emissions from vehicles have been estimated using various model calculations (Goyal and Ramakrishna 1998). Studies have calculated emissions on the basis of activity data, vehicle kilometers travelled, vehicle category, and subcategories (Ramanathan and Parikh 1999; CPCB 2007; Mittal and Sharma 2003; ALGAS 1998; ADB 2006; Baidya and Borken Kleefeld 2009). Emission factors for Indian vehicles have been developed by the Automotive Research Association of India in co-ordination with MoEF, CPCB and State Pollution Control Boards (ARAI 2007). Inventory estimates for the emissions of greenhouse gases and other pollutants and effects of vehicular emission on urban air quality and human health have been studied in major urbanized cities in India (Sharma et al. 1995; Sharma and Pundir 2008; Gurjar et al. 2004; Ghose et al. 2004; Ravindra et al. 2006; Jalihal and Reddy 2006).

GHG Emissions in the Industrial Sector. Industry is a major source of global GHG emissions. The industrial sector is responsible for approximately one-third of global carbon dioxide emissions through energy use (William 1996). In India, emission estimates from large point sources, such as thermal power, steel industry, cement plants, chemical production and other industries, have been carried out by various researchers (Mitra 1992; Mitra et al. 1999a, b; Garg et al. 2001, 2004; Mitra and Bhattacharya 2002; Gurjar et al. 2004; Garg et al. 2006). CO_2 emissions from iron and steel, cement, fertilizer, and other industries such as lime production, ferroalloy production, and aluminum production have been estimated (Garg et al. 2006, 2011).

Six industries in India have been identified as energy-intensive industries: aluminum, cement, fertilizer, iron and steel, glass, and paper manufacturing. The cement sector holds a considerable share within these energy-intensive industries (Schumacher and Sathaye 1999; Bernstein et al. 2007). At the country level, trends of GHG emissions from industrial processes indicated 24,510 CO_2 equivalent emissions in the year 1990, 102,710 CO_2 equivalent emissions in 1994, 168,378 CO_2 equivalent emissions in 2000 and 189,987.86 CO_2 equivalent emissions in 2007 (Sharma et al. 2009, 2011; Kumar 2003). Under the aegis of INCCA, a national-level GHG inventory for CO_2, CH_4, and N_2O inventory was published in 2010 for the base year 2007, which showed from industrial processes and product use (Sharma et al. 2011).

GHG Emissions in Agriculture Sector. Agricultural activities contribute directly to emissions of GHGs through a variety of processes. The major agricultural sources of GHGs are methane emissions from irrigated rice production, nitrous oxide emissions from the use of nitrogenous fertilizers, and the release of carbon dioxide from energy sources used to pump groundwater for irrigation (Nelson et al. 2009). Where there is open burning associated with agricultural practices, a number of greenhouse gases are emitted from combustion. All burning of biomass produces substantial CO_2 emissions. In India, the crop waste generated in the fields is used as feed for cattle and domestic biofuel; the remainder is burnt in the field

(Reddy et al. 2002). Rice paddy soils contain organic substrates, nutrients, and water; therefore, they are an increasing source of methane resulting from the anaerobic decomposition of carbonaceous substances (Alexander 1961). The anaerobic bacterial processes in the irrigated rice cultivated fields are considered to be among the largest sources of methane emission (Sass and Fisher 1998); the annual global contribution of methane is estimated to be $\sim$190 Tgy^{-1} (Koyama 1963; Yanyu et al. 2006).

Studies on CH_4 emission from Indian rice fields have been carried out by different researchers to study the effects of soil type, season, water regime, organic and inorganic amendments, and cultivars (Parashar et al. 1991; Mitra 1992; Parashar et al. 1993, 1994; Adhya et al. 1994; Sinha 1995; Mitra et al. 1999a, b; Chakraborty et al. 2000, 2007; Jain et al. 2000; Khosa et al. 2010). Average methane flux varied significantly with different cultivars, ranging between 0.65 and 1.12 mg m^{-2} h^{-1} (Mitra et al. 1999a, b). CH_4 emissions from Indian rice paddies, therefore, is estimated to be 3.6 $\pm$ 1.4 Tg y^{-1}, which is lower than the value of 4.2 (1.3 to 5.1) Tg y^{-1} obtained using the IPCC 1996 default emission factors (Gupta et al. 2009). India emitted 3.3 million tons of CH_4 in 2007 from 43.62 million ha cultivated (Gupta 2005; MoA 2008; INCCA 2010). The application of nitrogen fertilizer in upland irrigated rice has led to increased N_2O emissions (Kumar et al. 2000; Majumdar et al. 2000; Ghosh et al. 2003; Garg et al. 2004, 2006). Total seasonal N_2O emission from different treatments ranged from 0.037 to 0.186 kg ha^{-1} (Ghosh et al. 2003; Aggarwal et al. 2003; Bhatia et al. 2008; Bhatia 2008; INCCA 2010).

GHG Emissions in the Livestock Sector. There are two major sources of methane emission from livestock: enteric fermentation resulting from digestive process of ruminants and animal waste management (IPCC 2006; Bandyopadhyay et al. 1996). Animal husbandry accounts for 18 % of GHG emissions that cause global warming (Naqvi and Sejian 2011). Methane emission from enteric fermentation from Indian livestock ranged from 7.26 to 10.4 MT/year (Garg and Shukla 2002). In India, more than 90 % of the total methane emission from enteric fermentation is contributed by large ruminants (cattle and buffalo), with the rest from small ruminants and other animals (Swamy and Bhattacharya 2006). The production and emission of CH_4 and N_2O from manure depends on digestibility and composition of feed, species of animals and their physiology, manure management practices, and meteorological conditions such as sunlight, temperature, precipitation, wind, etc. (Gaur et al. 1984; Yamulki et al. 1999).

In India, studies have been carried out in which the emission inventories for enteric fermentation and manure management were done at the national level (Garg et al. 2001; Naqvi and Sejian 2011; Gurjar et al. 2004, 2009; Garg et al. 2011). The total emission of methane from Indian livestock was estimated to be 10.08 MT, considering different categories of ruminants and type of feed resources available in different zones of the country (Singhal et al. 2005). CH_4 and N_2O country-specific emission factors for bovines were found to be lower than IPCC (1996) default values. Inventory estimates were found to be approximately 698 $\pm$ 27 Gg CH_4 from all manure management systems and 2.3 $\pm$ 0.46 tons of N_2O from solid storage of manure for the year 2000 (Gupta et al. 2009). Using the emission factors provided in the report (NATCOM 2004), it is estimated that the Indian livestock emitted 9.65 million tons in 2007. Buffalo are the single largest emitter of CH_4, constituting 60 % of the total CH_4 emission from this category, simply because of their large numbers compared to any other livestock species and also because of the large CH_4 emission factor with respect to others (INCCA 2010). By using the IPCC guidelines, the total CH_4 emitted from enteric fermentation in livestock was found to be 10.09 million tons; emissions from manure management were estimated to be approximately 0.115 million tons of CH_4 and 0.07 thousand tons of N_2O (INCCA 2010).

GHG Emissions Inventory in the Waste Sector. The main GHG emitted from waste management is CH_4. It is produced and released into the atmosphere as a byproduct of the anaerobic decomposition of solid waste, whereby methanogenic bacteria break down organic matter in the waste. Similarly, wastewater becomes a source of CH_4 when treated or disposed anaerobically. It can also be a source of N_2O emissions due to the protein content in domestically generated wastewater (INCCA 2010; Hogne et al. 2010; Marlies et al. 2009). Industrial wastewater with significant carbon loading that is treated under intended or unintended anaerobic conditions will produce CH_4 (IPCC 2006).

Waste landfills are considered to be the largest source of anthropogenic emissions. Methane emissions from landfills are estimated to account for 3–19 % of the anthropogenic sources in the world (IPCC 1996). Landfill gas, primarily a mix of CO_2 and CH_4, is emitted as a result of the restricted availability of oxygen during the decomposition of the organic fraction of waste in landfills (Talyan et al. 2007). Methane emissions have been estimated for specific particular landfill sites and regions in India (Kumar et al. 2000, 2004, 2009; Gurjar et al. 2004; Ramachandra and Bachamanda 2007; Subhasish et al. 2009; Rawat and Ramanathan 2011).

CH_4 emission estimates were found to be approximately 0.12 Gg in Chennai from municipal solid waste management for the year 2000, which is lower than the IPCC (1996) values.

Municipal solid waste (MSW) management in major cities in India has been assessed; parameters such as waste quantity generated, waste generation rate, physical composition, and characterization of MSW in each of the cities are carried out (Kumar et al. 2009). Solid waste generated in Indian cities increased from 6 Tg in 1947 to 48 Tg in 1997 (Pachauri and Sridharan 1998), with a per capita increase of 1–1.33 % per year (Rao and Shantaram 2003). Per INCCA (2010), 604.51 Gg of CH_4 was emitted from solid waste disposal sites in India.

Methane is generated from domestic and industrial wastewater. The main factor in determining the extent of CH_4 production is the amount of degradable organic fraction in the wastewater (Fadel and Massoud 2001), which is commonly expressed in terms of biochemical oxygen demand (BOD) or chemical oxygen demand (COD). The disposal and treatment of industrial waste and MSW are not a prominent source of methane emissions in India, except in large urban centers. In India, methane emissions from domestic/commercial and industrial wastewater were found to be 861 and 1050 Gg, respectively, for the year 2007. Approximately

15.81 Gg of nitrous oxide is emitted from the domestic/commercial wastewater sector (Garg et al. 2001; Sharma et al. 2011).

A sector-wise review highlights the fragmented efforts of assessing the carbon footprint in India. There are no comprehensive efforts to assess the carbon footprint among all sectors in rapidly urbanizing cities, which is vital for evolving appropriate city-specific mitigation measures. The objectives of this chapter are to quantify and analyze sector-wise greenhouse gas emissions in terms of carbon dioxide equivalent (CO_2 eq) across major cities in India.

Section 2 presents methods for quantifying the carbon footprint for electricity, domestic, industry, transportation, agriculture, livestock, and waste sectors; it also provides a brief account of cities chosen for the current study. Section 3 provides a detailed account of sector-wise carbon footprints for major cities in India, with a synthesis of intercity variations. This is followed by conclusions and gaps in the current study in Sects. 4 and 5, respectively. Annex 1 provides the sector-wise carbon footprints for major cities in India.

2 Method

2.1 Study Area

Carbon footprint has been assessed for eight major metropolitan cities (populations of >4 million per 2011 census) in India: Delhi, Greater Mumbai, Kolkata, Chennai, Greater Bangalore, Hyderabad, and Ahmedabad. Except for Ahmedabad, all of these cities are class X (formerly class A1) cities as per the classification of Ministry of Finance (HRA 2008). Table 1 lists the location, population, and GDP for all chosen cities. Geographic locations of the cities are depicted in Fig. 2.

Delhi. Delhi is the capital of India with long history, covering an area of 1483 km^2 with a population of 16,127,687 (in 2009). This city borders Uttar Pradesh state to the east and Haryana on the north, west, and south. In 2009, Delhi had a GDP of Rs. 219,360 crores at constant prices, which primarily relies on the integral sectors such as power, telecommunications, health services, construction, and real estate (SOE 2010).

Greater Mumbai (*Bombay*). Greater Mumbai, the capital of Maharashtra, is one of the major port cities located at the Coast of Arabic Sea in the west coast in India. The Greater Mumbai region consists of the Mumbai city district and Mumbai suburban district. It covers a total area of 603.4 km^2, with a population of 12,376,805 (in 2009). It is also the commercial and entertainment capital of India, generating a GDP of Rs. 274,280 crores at constant prices and contributing to 5 % of India's GDP (MoUD 2009; MMRDA 2008).

Kolkata (*Calcutta*). Kolkata, the capital of West Bengal, is located on the east bank of the Hooghly River. The Municipal Corporation of Kolkata covers an area of 187 km^2, with a population of 4,503,787 (in 2009). The GDP of Kolkata in the

Table 1 Locations, populations, and GDPs of major metropolitan cities in India

Cities	Latitude and longitude[a]	Population (2009)[b]	GDP (constant prices, crores) for 2009[c]
Delhi	28°25’ N and 76°50’ E	16,127,687	219360.35
Greater Mumbai	18.9° N and 72.8° E	12,376,805	274280.15
Kolkata	22°34’ N and 88°24’ E	4,503,787	136549.41
Chennai	13°04’ N and 80°17’ E	4,611,564	86706.92
Greater Bangalore	12°59’ N and 77°37’ E	8,881,631	90736.07
Hyderabad	17°28’ N and 78°27’ E	6,007,259	76254.10
Ahmedabad	23.02° N and 72.35° E	5,080,596	64457.80

Sources [a] Balachandran et al. (2000); Srivastava et al. (2007); Gupta et al. (2006, 2007, 2009); Ramachandra and Kumar (2010); Latha et al. (2003); Bhaskar et al. (2010)
[b] Population for the year 2009 was estimated using the 2001 and 2011 Census of India
[c] Pricewaterhouse Coopers study 2009

year 2009 was estimated to be Rs. 136,549 crores at constant prices, resulting in the city being a major commercial and financial hub in Eastern and Northeastern India.

Chennai (*Madras*). Chennai, the capital of the state of Tamil Nadu, is located on the Coromandel Coast of the Bay of Bengal. It had a population of 4,611,564 in the year 2009, with an area of 174 km^2, which is expanded to 426 km^2 by the city corporation in the year 2011. The economy of the city mainly depends on sectors such as automobile, software services, health care industries, and hardware manufacturing, resulting in an estimated GDP of Rs. 86,706 crores at constant prices during the year 2009 (Loganathan et al. 2011).

Greater Bangalore. Greater Bangalore is the principal administrative, cultural, commercial, and knowledge capital of the state Karnataka. It covers an area of 741 km^2 and had an estimated population of 8,881,631 in 2009. During the year 2009, Bangalore’s economy of Rs. 90,736 crores at constant prices made it one of the major economic centers in India. The city’s economy depends on information technology, manufacturing industries, biotechnology, and aerospace and aviation industries (Ramachandra et al. 2012).

Hyderabad. Hyderabad, the capital of Andhra Pradesh, is located at the north part of the Deccan plateau, with a population of 6,007,259. The municipal Corporation of Hyderabad covers an area of 179 km^2, whereas Greater Hyderabad is spread over an area of 650 km^2. The city’s economic sector depends on traditional manufacturing, knowledge, and tourism, resulting in a GDP of Rs. 76,254 crores at constant prices in the year 2009.

Ahmedabad. Ahmedabad, an industrial city, is situated on the banks of Sabarmati River in north-central Gujarat. It covers an area of 205 km^2, with a population of 5,080,596 in the year 2009. Ahmedabad is the second largest industrial center in western India after Mumbai. Automobiles, textiles, pharmaceuticals, and real estate are the major sectors contributing to the economy, which was Rs. 64,457 crores at constant prices in the year 2009.

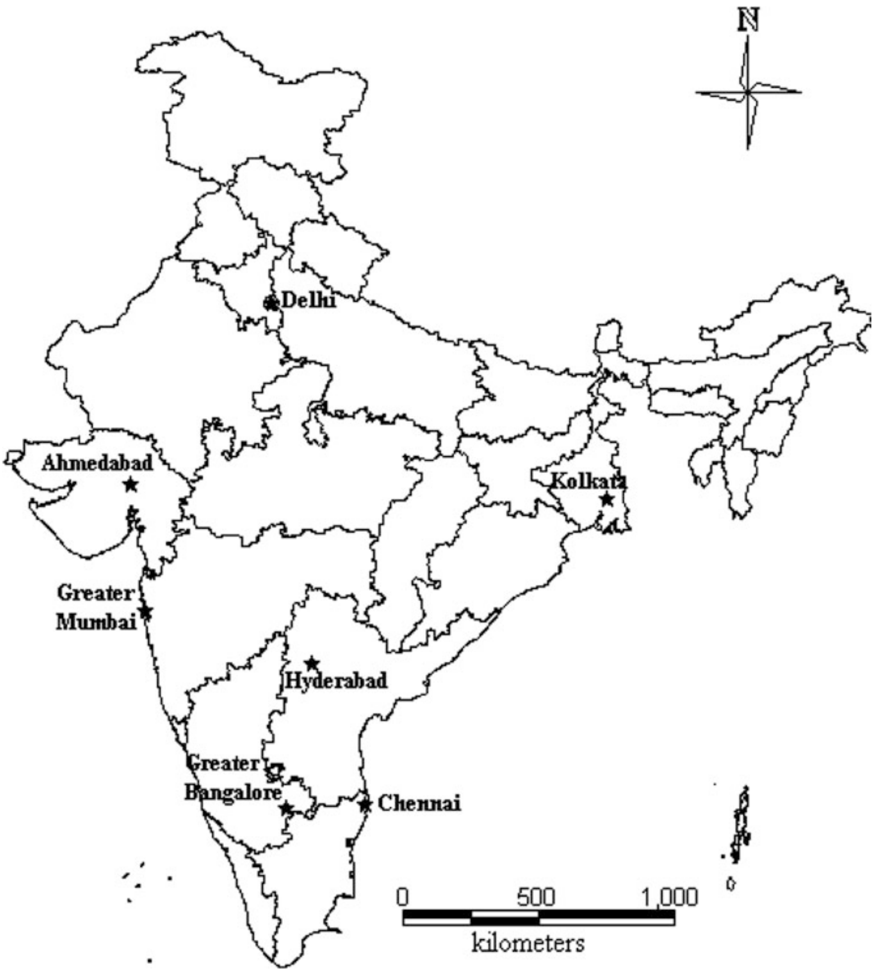

Fig. 2 Study area, indicating the major cities in India. *Source* Energy and Wetlands Research Group, Centre For Ecological Sciences, Indian Institute of Science

2.2 Quantification of Greenhouse Gases

The major three greenhouse gases quantified are carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O). The non-CO_2 gases are converted to units of carbon dioxide equivalent (CO_2e) using their respective GWPs. The total units of CO_2e then represent a sum total of the GWP of all three major greenhouse gases. The categories considered for GHG emission inventory are the following:

(i) Energy: electricity consumption, fugitive emissions
(ii) Domestic or household sector
(iii) Transportation

(iv) Industrial sector
(v) Agriculture-related activities
(vi) Livestock management
(vii) Waste sector.

National GHG inventories compiled from various sources were used for the calculation of GHG emissions. Country-specific emission factors were compiled from the published literature. In the absence of country-specific emission factors, the default emission factors from the IPCC were used. The emission of each GHG was estimated by multiplying fuel consumption by the corresponding emission factor. The total emissions of a gas from all its source categories (Ramachandra and Shwetmala 2012; Pandey et al. 2011; Global Footprint Network 2007) are summed as given in Eq. 1:

$$\text{Emissions}_{\text{Gas}} = \sum\nolimits_{\text{Category}} A \times \text{EF} \tag{1}$$

where

$\text{Emissions}_{\text{Gas}}$	emissions of a given gas from all its source categories
A	amount of an individual source category utilized that generates emissions of the gas under consideration
EF	emission factor of a given gas type by type of source category

GHG Emissions from Electricity Consumption. The combustion of fossil fuels in thermal power plants during electricity generation results in the emission of GHG into the atmosphere. CO_2, oxides of sulfur (SOx), nitrogen oxides (NOx), other trace gases, and airborne inorganic particulates, such as fly ash and suspended particulate matter, are the most important constituents emitted from the burning of fossil fuels from thermal power plants (Raghuvanshi et al. 2006; Ramachandra and Shwetmala 2012; TEDDY 2006, 2011). The emissions were computed based on consumption in the following categories: domestic, commercial, industrial, and others (which includes consumption in railways, street lights, municipal water supply, sewage treatment, etc.) based on the amount of electricity consumed by these sectors. The total GHG emissions were calculated on the basis of fuel consumption required for the generation of electricity using Eq. 2:

$$\begin{aligned}\text{Emissions (t)} = {} & \text{Fuel consumption (kt)} \times \text{Net calorific value of fuel (TJ/kt)} \\ & \times \text{Emission factor (t/TJ)}\end{aligned} \tag{2}$$

Electricity is generated from various sources (coal, hydro, nuclear, gas, etc.). The proportion of electricity generated from each source for each study region was compiled from secondary sources (state electricity board, central electrical authority, etc.). The quantity of respective fuel is computed with the knowledge of the relative share of fuel and the quantity of fuel required for generating one unit of

Table 2 Net calorific values and CO_2, CH_4, and N_2O emission factors for different fuel

Fuel	NCV (TJ/kt)	CO_2 EF (t/TJ)[a, b]	CH_4 EF (t/TJ)[b]	N_2O EF (t/TJ)[b]
Coal	19.63	95.81	0.001	0.0015
Natural gas	48	56.1	0.001	0.0001
Naphtha	44.5	73.3	0.003	0.0006
Diesel oil	43.33	74.1	0.003	0.0006
Natural gas	48.632	64.2	0.003	0.0006
Low-sulfur heavy stock	40.19	73.3	0.003	0.0006
Residual fuel oil	40.4	77.4	0.003	0.0006
Low-sulfur fuel Oil	41	73.3	0.003	0.0006
Heavy fuel oil	40.2	73.3	0.003	0.0006

Note NCV Net Calorific Value, *EF* Emission factor
Sources [a] Indian Network for Climate Change Assessment (INCCA 2010)
[b] 2006 IPCC Guidelines for National Greenhouse Gas Inventories (2006), TEDDY (2006, 2011)

electricity (e.g., 0.7 kg coal is required for the generation of 1 unit [KWh] of electricity). The data related to electricity consumption in different cities was taken from the respective electricity boards providing electricity to that city. Table 2 lists the emission factors and the net calorific values (NCVs) of respective fuels. The total emissions obtained from the amount of fuel consumed is then distributed into major sectors, such as domestic, commercial, industrial, and others, based on the amount of electricity consumed in that sector during the inventory year 2009–2010. Apart from the fuel consumption on the basis of electricity consumption, the fuel consumption and the emissions resulting thereby were also determined for the auxiliary consumption in the power plants located within the city boundaries and the transmission loss resulting from these power plants.

Fugitive Emissions. Fugitive emissions are the intentional or unintentional release of GHGs during the extraction, production, processing, or transportation of fossil fuels. Exploration for oil and gas, crude oil production, processing, venting, flaring, leakages, evaporation, and accidental releases from oil and gas industry are sources of CH_4 emission (INCCA 2010; Ramachandra and Shwetmala 2012). Refinery throughput is the total amount of raw materials processed by a refinery or other plant in a given period. In the present study, the emissions from refinery crude throughput are calculated from the refineries present within city boundaries, per Eq. 3:

$$\begin{aligned} \text{Emissions (Gg)} = \; & \text{Refinery crude throughput (Million tons)} \\ & \times \text{ Emission factor (Gg/Million tons)} \end{aligned} \tag{3}$$

The methane emission factor for refinery throughput is 6.75904×10^{-5} Gg/million tons (IPCC 2000, 2006).

GHG Emissions from the Domestic Sector. The large demand for energy consumption in the domestic sector is predominantly due to activities such as cooking,

Table 3 Net calorific values (NCV) and CO_2, CH_4 and N_2O emission factors for domestic fuels used in the study

Fuel	NCV (TJ/kt)	CO_2 EF (t/TJ)[a, b]	CH_4 EF (t/TJ)[b]	N_2O EF (t/TJ)[b]
Liquefied petroleum gas	47.3	63.1	0.005	0.0001
Piped natural gas	48	56.1	0.005	0.0001
Kerosene	43.8	71.9	0.01	0.0006

Note EF emission factor
Source [a] Indian Network for Climate Change Assessment (2010)
[b] 2006 IPCC Guidelines for National Greenhouse Gas Inventories (2006)

lighting, heating, and household appliances. Per the Census of India (2001), in urban areas, the most commonly used fuel is liquefied petroleum gas (LPG; 47.96 %), followed by firewood (22.74 %) and kerosene (19.16 %). Electricity consumption is another major source of energy utilization in urban households. The pollution caused by domestic fuel use is a major source of emissions in cities, which causes indoor air pollution that contributes to overall pollution. The type of fuels used in households also affects air pollution.

The emissions resulting from electricity consumption in the domestic sector are attributed to this sector. GHG emissions from fuel consumption in the domestic sector can be calculated by Eq. 4 (Ramachandra and Shwetmala 2012).

$$\begin{aligned}\text{Emissions (t)} = {} & \text{Fuel consumption (kt)} \times \text{Net calorific value of fuel (TJ/kt)} \\ & \times \text{Emission factor (t/TJ)}\end{aligned} \tag{4}$$

Table 3 lists the NCVs and emission factors for domestic fuels.

GHG Emissions from the Transportation Sector. The transportation sector is one of the dominant anthropogenic sources of GHG emissions (Mitra and Sharma 2002). The urban population predominantly depends on road transportation; therefore, there is an increase in sales of vehicles in urban areas every year. The type of transport and fuel, apart from type of combustion engine, emission mitigation techniques, maintenance procedures, and age of the vehicle, are the major factors on which road transportation emissions depend (Ramachandra and Shwetmala 2009, 2012). Emissions are estimated from either the fuel consumed (fuel sold data) or the distance travelled by vehicles. A bottom-up approach was implemented based on the number of registered vehicles, annual vehicle kilometers travelled, and corresponding emission factors for the estimation of gases from the road transportation sector (Gurjar et al. 2004; Ramachandra and Shwetmala 2009). In national-level studies, the fuel consumption approach has been used to calculate the emissions from road transport (Sikdar and Singh 2009; INCCA 2010).

A bottom-up approach was used in this study. Emissions were calculated using the data available on number of vehicles, distance travelled in a year, and the

Table 4 CO_2, CH_4, and N_2O emission factors (EFs) for different types of vehicles

Type of Vehicle	CO_2 EF (g/km)[a]	CH_4 EF (g/km)[b]	N_2O EF (g/km)[b]
Motorcycles, scooters, and mopeds	27.79	0.18	0.002
Cars and jeeps	164.22	0.17	0.005
Taxis	164.22	0.01	0.01
Buses	567.03	0.09	0.03
Light motor vehicles (passengers)	64.16	0.18	0.002
Light motor vehicles (goods)	273.46	0.09	0.03
Trucks and lorries	799.95	0.09	0.03
Tractors and trailers	515.2	0.09	0.03

Sources [a] Emission factor development for Indian Vehicles, ARAI (2007)
[b] *EEA* European Environmental Agency (2009), Gurjar et al. 2004

respective emission factor for different vehicles. Emissions from road transportation were calculated per Eq. 5:

$$E_i = \sum \left(\mathrm{Veh}_j \times D_j\right) \times E_{i,j,\mathrm{km}} \tag{5}$$

where

E_i Emission of the compound (i)
Veh_j Number of vehicles per type (j)
D_j Distance travelled in a year per different vehicle type (j)
$E_{i,j,\mathrm{km}}$ Emission of compound (i) for vehicle type (j) per driven kilometer

Emission factors are listed in Table 4.

In this study, the number of registered vehicles in inventory year 2009 was taken from the Motor Transport Statistics of the respective states and also from the Road Transport Year Book (2007–2009) when the city-level data were not available from the local transport authorities. The Supreme Court passed an order in July 1998 to convert all public transport vehicles to compressed natural gas (CNG) mode in Delhi, which marked the beginning of CNG vehicles in India (Sandhya Wakdikar 2002; Chelani and Sukumar 2007). Emissions from the number of vehicles using CNG as a fuel were also calculated in the major cities where CNG was introduced to mitigate the emissions resulting from transportation. The vehicle kilometer travelled per year values were taken from the Central Pollution Control Board of India (CPCB 2007; Chelani and Sukumar 2007). The annual average mileage values of different vehicles used are given in Table 5.

GHG emissions for the major cities in India were calculated considering the fuel consumption for navigation in the major ports of Mumbai, Kolkata, and Chennai. The 2006 IPCC guidelines provide a method to calculate emissions from navigation (IPCC 2006). Using the ship type in the ports and gross registered tonnage (GRT), the total fuel consumed is calculated, from which the emissions are calculated. The type of ships and GRT data are available from Basis Ports

Table 5 Vehicle kilometers travelled (VKT)

Types of vehicles	VKT
Motorcycles, scooters, and mopeds	10,000
Cars and jeeps	15,000
Taxis	30,000
Buses	60,000
Light motor vehicles (passengers)	40,000
Light motor vehicles (goods)	40,000
Trucks and lorries	30,000
Tractors and trailers	5,000

Source Transport fuel quality for the year 2005 from CPCB (2007)

Statistics of India (2009–10). Equation 6 was used to compute the emissions using the fuel consumption in different ship types using GRT and the ship type data as given below,

$$\begin{aligned}\text{Emissions (t)} = {} & \text{Fuel consumption (kt)} \times \text{Net calorific value of fuel (TJ/kt)} \\ & \times \text{Emission factor (t/TJ)}\end{aligned} \tag{6}$$

Container $= 8.0552 + (0.00235 \times \text{GRT})$
Break Bulk (General Cargo) $= 9.8197 + (0.00413 \times \text{GRT})$
Dry Bulk $= 20.186 + (0.00049 \times \text{GRT})$
Liquid Bulk $= 14.685 + (0.00079 \times \text{GRT})$

High-speed diesel (HSD), light diesel oil (LDO), and fuel oil are the major fuels used for shipping in India (Ramachandra and Shwetmala 2009). The average of NCV values and emission factors are used to calculate the emissions for fuel consumption. CO_2 emission factors for fuel oil and HSD/LDO are taken as 77.4 and 74.1 t/TJ, respectively. CH_4 and N_2O emission factors are taken as 0.007 and 0.002 t/TJ, respectively, for navigation (IPCC 2006). At the country level, the emissions from shipping were calculated using the fuel consumption data (NATCOM 2004; Garg et al. 2006; Ramachandra and Shwetmala 2009; MCI 2008).

GHG Emissions from the Industry Sector. GHG emissions are produced from a wide variety of industrial activities. Industrial processes that chemically or physically alter materials are the major emission sources. The blast furnace in the iron and steel industry, manufacturing of ammonia and other chemical products from fossil fuels used as chemical feedstock, and the cement industry are the major industrial processes that release a considerable amount of CO_2 (IPCC 2006). There are no data available for the calculation of emissions from small- and medium-scale industries, which number in the thousands in major cities. In this study, the emissions were calculated from the major polluting industrial processes in the industries that are located within city boundaries. In cities such as Mumbai, the

Table 6 Values used to calculate GHG emissions from the fertilizer industry

Parameter	FR (GJ/tone NH_3 produced)	CCF (kg C/GJ)	COF (fraction)
Value	37.5	15.30	1

Source 2006 IPCC Guidelines for National Greenhouse Gas Inventories (2006)

presence of large petrochemical plants, fertilizer plants, and power plants leads to emissions (Kulkarni et al. 2000).

The GHGs estimated for the type of industries located within city boundaries based on the availability of the data are discussed below. Ammonia (NH_3) is a major industrial chemical and the most important nitrogenous material produced. Ammonia gas is directly used as a fertilizer, in paper pulping, and in the manufacture of chemicals (IPCC 2006). Ammonia production data were obtained from the fertilizer industry (RCF 2010; MFL 2010); emission factors and other parameters (Table 6) were obtained from (IPCC 2006) guidelines. Emissions from ammonia production were calculated by Eq. 7.

$$E_{CO_2} = \text{AP} \times \text{FR} \times \text{CCF} \times \text{COF} \times 44/12 - R_{CO2} \tag{7}$$

where

E_{CO_2}	emissions of CO_2 (kg)
AP	ammonia production (tons)
FR	fuel requirement per unit of output (GJ/ton ammonia produced)
CCF	carbon content factor of the fuel (kg C/GJ)
COF	carbon oxidation factor of the fuel (fraction)
R_{CO2}	CO_2 recovered for downstream use (urea production) in kilograms

The glass industry can be divided into four major groups: containers, flat (window) glass, fiberglass, and specialty glass. Limestone ($CaCO_3$), dolomite Ca, $Mg(CO_3)_2$, and soda ash (Na_2CO_3) are the major glass raw materials that are responsible for the emission of CO_2 during the melting process (IPCC 2006). Equation 8 is used when there are no data available on glass manufactured by process or the carbonate used in the manufacturing of glass.

$$CO_2 \text{ emissions} = \text{Mg} \times \text{EF} \times (1 - \text{CR}) \tag{8}$$

where

CO_2 emissions	emissions of CO_2 from glass production (tons)
Mg	mass of the glass produced (tons)
EF	default emission factor for the manufacturing of glass (tons CO_2/tons glass)
CR	cullet ratio for the process (fraction)

Table 7 gives the values of the different parameters that are used to calculate GHG emissions from the glass industry. In the present study, fuel consumption data from major industries within the city boundaries are used to calculate the

Table 7 Values used to calculate GHG emissions from the glass industry

Parameter	Emission factor (tons CO_2/tons glass)	Cullet ratio
Value	0.2	0.5

Source 2006 IPCC Guidelines for National Greenhouse Gas Inventories (2006)

emissions where all data are available (annual reports of Vitrum Glass 2010; G-P (I) Ltd. 2010; EMAMI 2010; Kesoram Ind. Ltd 2010; TN Petro Products Limited 2010; Khoday India Limited 2010). The fuel consumption by industry for the year 2009–10 was obtained from the companies' annual reports, from which emissions were calculated by accounting for fuel utilization.

GHG Emissions from Agriculture-Related Activities. Agriculture-related activities, such as paddy cultivation, agricultural soils, and the burning of crop residue, are considered in the quantification of GHG emissions. Flooded rice fields are one source of methane emissions. During the paddy growing season, methane is produced from the anaerobic decomposition of organic material in flooded rice fields, which escapes to the atmosphere through rice plants by the mechanism of diffusive transport (IPCC 1997). Oxygen supply is seized from the atmosphere to the soil due to the flooding of rice fields, which leads to anaerobic fermentation of organic matter in the soil, resulting in the production of methane (Ferry 1992).

There are three processes of methane release into the atmosphere from paddy fields. The major phenomenon is CH_4 transport through rice plants (Seiler et al. 1984; Schutz et al. 1989). This accounts for more than 90 % of total CH_4 emissions. Methane loss as bubbles (ebullition) from paddy soils is also a common and significant mechanism. The least important process is the diffusion loss of CH_4 across the water surface (IPCC 1997). The emission of methane from rice fields depends on various factors, such as the amendment of organic and inorganic fertilizers, characteristics of the rice varieties, water management, and the soil environment (Mitra et al. 1999a, b). CH_4 emissions from rice cultivation have been estimated by multiplying the seasonal emission factors by the annual harvested areas. The total annual emissions are equal to the sum of emissions from each subunit of harvested area, which was calculated using Eq. 9 (IPCC 2000):

$$\mathrm{CH}_{4\mathrm{Rice}} = \sum\nolimits_{i,j,k} \left(\mathrm{EF}_{i,j,k} \times A_{i,j,k} \times 10^{-6}\right) \tag{9}$$

where

$CH_{4\ \mathrm{Rice}}$ annual methane emissions from rice cultivation (Gg CH_4 year^{-1})

$\mathrm{EF}_{i,j,k}$ seasonal integrated emission factor for i, j, and k conditions (kg CH_4 ha^{-1})

$\mathrm{A}_{i,j,k}$ annual harvested area of rice for i, j, and k conditions (ha year^{-1})

i, j, and k represent different ecosystems, water regimes, types and amounts of organic amendments, and other conditions under which CH_4 emissions from rice may vary

It is advisable to calculate the total emissions as a sum of the emissions over a number of conditions. For studies at city levels, Eq. 10, from the revised (IPCC 1996) guidelines, was used (IPCC 1997).

$$\mathrm{Fc} = \mathrm{EF} \times A \times 10^{-9} \tag{10}$$

where

Fc estimated annual emission of methane from a particular rice water regime and for a given organic amendment (Gg/year)

EF methane emission factor integrated over the integrated cropping season (g/m^2)

A annual harvested area cultivated under conditions defined above. It is given by the cultivated area times the number of cropping seasons per year (m^2/year)

This method was used because the area of paddy fields based on the type of ecosystem (irrigated, rain fed, deep water, and upland) is not available at the city level. A seasonally integrated emission factor of 10 g/m^2 was used, as obtained from the revised 1996 IPCC guidelines (IPCC 1997).

Agricultural soils contribute to the emission of two major GHGs: methane and nitrous oxide. N_2O is produced naturally in soils through the processes of nitrification and denitrification. Nitrification is the aerobic microbial oxidation of ammonium to nitrate and denitrification is the process of anaerobic microbial reduction of nitrate to nitrogen gas (N_2). Nitrous oxide is a gaseous intermediate in the reaction sequence of denitrification and a byproduct of nitrification that leaks from microbial cells into the soil and ultimately into the atmosphere. This method therefore estimates N_2O emissions using human-induced net nitrogen (N) additions to soils (e.g., synthetic or organic fertilizers, deposited manure, crop residues, sewage sludge) or of mineralization of N in soil organic matter following drainage/management of organic soils or cultivation/land-use change on mineral soils (IPCC 2006; Granli and Bockman 1994).

The emissions of N_2O resulting from anthropogenic N inputs or N mineralization occur through both a direct pathway (i.e., directly from the soils to which the N is added/released) and through two indirect pathways: (i) following volatilization of NH_3 and NOx from managed soils and from fossil fuel combustion and biomass burning, and the subsequent redeposition of these gases and their products NH_4^+ and NO_3^- to soils and waters; and (ii) after leaching and runoff of N, mainly as NO_3^-, from managed soils. Total N_2O emissions are given by the following equation:

$$N_2O \text{ emissions} = N_2O_{\mathrm{Direct}} \text{ emissions} + N_2O_{\mathrm{Indirect}} \text{ emissions} \tag{11}$$

Direct N_2O emissions. The sources included for the estimation of direct N_2O emissions are synthetic N fertilizers, organic N applied as fertilizer, urine and dung N deposited on pasture, range and paddock by grazing animals, N in crop residues, N mineralization associated with loss of soil organic matter resulting from change of land use or management of mineral soils, and drainage/management of organic soils:

$$N_2O_{Direct}-N = N_2O-N_{NInput} + N_2O-N_{OS} + N_2O-N_{PRP} \quad (12)$$

where

N_2O_{Direct}–N annual direct N_2O–N emissions from managed soils (kg N_2O–N year^{-1})

N_2O–N $_{N\ Input}$ annual direct N_2O–N emissions from N inputs to managed soils (kg N_2O–N year^{-1})

N_2O–N $_{OS}$ annual direct N_2O–N emissions from managed organic soils (kg N_2O–N year^{-1})

N_2O–N $_{PRP}$ annual direct N_2O–N emissions from urine and dung inputs to grazed soils (kg N_2O–N year^{-1})

$$N_2O-N_{NInput} = \left[\left[(F_{SN} + F_{ON} + F_{CR} + F_{SOM}) \times \ EF_1\right] + \left[(F_{SN} + F_{ON} + F_{CR} + F_{SOM})_{FR} \times EF_{1FR}\right]\right] \quad (13)$$

where

F_{SN} annual amount of synthetic fertilizer N applied to soils (kg N year^{-1})

F_{ON} annual amount of animal manure, compost, sewage sludge, and other organic N additions applied to soils (kg N year^{-1})

F_{CR} annual amount of N in crop residues (above-ground and below-ground), including N-fixing crops and from forage/pasture renewal, returned to soils (kg N year^{-1})

F_{SOM} annual amount of N in mineral soils that is mineralized, in association with loss of soil C from soil organic matter as a result of changes to land use or management (kg N year^{-1})

EF_1 emission factor for N_2O emissions from N inputs (kg N_2O–N (kg N input)$^{-1}$)

EF_{1FR} emission factor for N_2O emissions from N inputs to flooded rice (kg N_2O–N (kg N input)$^{-1}$)

$$N_2O-N_{OS} = \left[\left(F_{OS,CG,Temp} \times \ EF_{2CG,Temp}\right) + \left(F_{OS,CG,Trop} \times \ EF_{2CG,Trop}\right) + \left(F_{OS,F,Temp,NR} \times \ EF_{2F,Temp,NR}\right) + \left(F_{OS,F,Temp,NP} \times \ EF_{2F,Temp,NP}\right) + \left(F_{OS,F,Trop} \times EF_{2F,Trop}\right)\right] \quad (14)$$

where

EF_2 emission factor for N_2O emissions from drained/managed organic soils, kg N_2O–N ha^{-1} year^{-1}

The subscripts CG, *F*, Temp, Trop, NR, and NP refer to cropland and grassland, forest land, temperate, tropical, nutrient rich, and nutrient poor, respectively.

$$N_2O-N_{PRP} = \left[\left(F_{PRP,CPP} \times \ EF_{3PRP,CPP}\right) + \left(F_{PRP,SO} \times \ EF_{3PRP,SO}\right)\right] \quad (15)$$

where

F_{PRP} annual amount of urine and dung N deposited by grazing animals on pasture, range, and paddock, kg N year^{-1}

EF_{3PRP} emission factor for N_2O emissions from urine and dung N deposited on pasture, range, and paddock by grazing animals, kg N_2O–N (kg N input)$^{-1}$

The subscripts CPP and SO refer to cattle/poultry/pigs and sheep/other animals, respectively.

$$F_{ON} = F_{AM} + F_{SEW} + F_{COMP} + F_{OOA} \tag{16}$$

$$F_{AM} = N_{MMSAvb} \times [1 - (\mathrm{Frac}_{FEED} + \mathrm{Frac}_{FUEL} + \mathrm{Frac}_{CNST})] \tag{17}$$

$$F_{PRP} = \sum_{T} \left[N_{(T)} \times N_{ex(T)} \times \mathrm{MS}_{(T,PRP)} \right] \tag{18}$$

where

F_{ON} total annual organic N fertilizer applied to soils other than by grazing animals (kg N year^{-1})

F_{AM} annual amount of animal manure N applied to soils (kg N year^{-1})

F_{SEW} annual amount of total sewage N that is applied to soils (kg N year^{-1})

F_{COMP} annual amount of total compost N applied to soils (kg N year^{-1})

$N_{MMS\ Avb}$ amount of managed manure N available for soil application, feed, fuel, or construction (kg N year^{-1})

Frac_{FEED} fraction of managed manure used for feed

Frac_{FUEL} fraction of managed manure used for fuel

Frac_{CNST} fraction of managed manure used for construction

$N_{(T)}$ number of head of livestock species/category T in the country

$N_{ex(T)}$ annual average N excretion per head of species/category T (kg N animal^{-1} year^{-1})

$\mathrm{MS}_{(T,\ PRP)}$ fraction of total annual N excretion for each livestock species/category T that is deposited on pasture, range, and paddock

Organic soils contain more than 12–18 % organic carbon. Indian soils are generally deficient of organic carbon (<1 %). Only some soils in Kerala and the northeast hill regions contain higher organic carbon (5 %). Therefore, the area under organic soil has been taken as nil (Bhatia et al. 2004).

Indirect N_2O emissions. Sources considered for estimation of indirect N_2O emissions include synthetic N fertilizers, organic N applied as fertilizer, urine and dung N deposited on pasture, range and paddock by grazing animals, N in crop residues, and N mineralization associated with loss of soil organic matter resulting from change of land use or management of mineral soil. The N_2O emissions from atmospheric deposition of N volatilized from managed soils were estimated by Eq. 19.

$$N_2O_{(ATD)}-N = [(F_{SN} \times \mathrm{Frac}_{GASF}) + ((F_{ON} + F_{PRP}) \times \mathrm{Frac}_{GASM})] \times EF_4 \quad (19)$$

where,

$N2O_{(ATD)}$–N	annual amount of N_2O–N produced from atmospheric deposition of N volatilized from managed soils (kg N_2O–N year^{-1})
F_{SN}	annual amount of synthetic fertilizer N applied to soils (kg N year^{-1})
Frac $_{GASF}$	fraction of synthetic fertilizer N that volatilizes as NH_3 and NOx (kg N volatilized (kg of N applied)$^{-1}$)
F_{ON}	annual amount of managed animal manure, compost, sewage sludge, and other organic N additions applied to soils (kg N year^{-1})
F_{PRP}	annual amount of urine and dung N deposited by grazing animals on pasture, range, and paddock (kg N year^{-1})
Frac $_{GASM}$	fraction of applied organic N fertilizer materials (F_{ON}) and of urine and dung N deposited by grazing animals (F_{PRP}) that volatilizes as NH_3 and NOx (kg N volatilized [kg of N applied or deposited]$^{-1}$)
EF_4	emission factor for N_2O emissions from atmospheric deposition of N on soils and water surfaces (kg N–N_2O [kg NH_3–N + NOx–N volatilized]$^{-1}$)

N_2O emissions from leaching and runoff in regions where leaching and runoff occurs were estimated using Eq. 20:

$$N_2O_{(L)}-N = (F_{SN} + F_{ON} + F_{PRP} + F_{CR} + F_{SOM}) \times \mathrm{Frac}_{LEACH-(H)} \times EF_5 \quad (20)$$

where

$N_2O_{(L)}$–N	annual amount of N_2O–N produced from leaching and runoff of N additions to managed soils in regions where leaching/runoff occurs (kg N_2O–N year^{-1})
F_{SN}	annual amount of synthetic fertilizer N applied to soils in regions where leaching/runoff occurs (kg N year^{-1})
F_{ON}	annual amount of managed animal manure, compost, sewage sludge, and other organic N additions applied to soils in regions where leaching/runoff occurs (kg N year^{-1})
F_{PRP}	annual amount of urine and dung N deposited by grazing animals in regions where leaching/runoff occurs (kg N year^{-1})
F_{CR}	amount of N in crop residues (above- and below-ground), including N-fixing crops and from forage/pasture renewal, returned to soils annually in regions where leaching/runoff occurs (kg N year^{-1})

F_{SOM}	annual amount of N mineralized in mineral soils associated with loss of soil C from soil organic matter as a result of changes to land use or management in regions where leaching/runoff occurs (kg N year^{-1})
Frac $_{LEACH\text{-}(H)}$	fraction of all N added to/mineralized in managed soils in regions where leaching/runoff occurs that is lost through leaching and runoff (kg N [kg of N additions]$^{-1}$)
EF_5	emission factor for N_2O emissions from N leaching and runoff (kg N_2O–N [kg N leached and runoff]$^{-1}$)

Conversion of $N_2O_{(ATD)}$–N and $N_2O_{(L)}$–N emissions to N_2O emissions was calculated using Eq. 21:

$$N_2O_{(ATD)/(L)} = N_2O_{(ATD)/(L)}-N \times 44/28 \tag{21}$$

Large quantities of agricultural waste are produced from the farming systems in the form of crop residue. The burning of crop residues is not a net source of CO_2 because the carbon released to the atmosphere during burning is reabsorbed during the next growing season (IPCC 1997). However, it is a significant net source of CH_4, CO, NOx, and N_2O. In this study, the emissions are calculated for two GHGs—CH_4 and N_2O. Non–CO_2 emissions from crop residue burning were calculated using Eq. 22:

$$\text{EBCR} = \sum \text{crops}\ (A \times B \times C \times D \times E \times F) \tag{22}$$

where

EBCR	Emissions from residue burning
A	Crop production
B	Residue-to-crop ratio
C	Dry matter fraction
D	Fraction burnt
E	Fraction actually oxidized
F	Emission factor

GHG Emissions from the Livestock Sector. Major activities resulting in the emission of greenhouse gases from animal husbandry are enteric fermentation and manure management. Enteric fermentation is a digestive process by which carbohydrates are broken down by the activity of micro-organisms into simple molecules for absorption into the blood stream. Factors such as the type of digestive tract, age and weight of the animal, and quality and quantity of feed consumed affects the amount of CH_4 released. Ruminant livestock (cattle, sheep) are the major sources of CH_4, whereas moderate amounts are released from nonruminant livestock (pigs, horses). CH_4 emissions from enteric fermentation were calculated using Eq. 23:

$$\text{Emissions} = \text{EF}_{(T)} \times N_{(T)} \times 10^{-6} \tag{23}$$

where

Emissions	methane emissions from enteric fermentation (Gg CH_4 year^{-1})
$\text{EF}_{(T)}$	emission factor for the defined livestock population (kg CH_4 head^{-1} year^{-1})
$N_{(T)}$	the number of heads of livestock species/category T
T	species/category of livestock

To estimate the total emissions from enteric fermentation, the emissions from different categories and subcategories were summed together.

Methane emissions from manure management were calculated using Eq. 24:

$$\text{Emissions} = \text{EF}_{(T)} \times N_{(T)} \times 10^{-6} \tag{24}$$

where

Emissions	methane emissions from manure management (Gg CH_4 year^{-1})
$\text{EF}_{(T)}$	emission factor for the defined livestock population (kg CH_4 head^{-1} year^{-1})
$N_{(T)}$	the number of head of livestock species/category T
T	species/category of livestock

Nitrous oxide emissions from manure management were calculated by Eq. 25:

$$\text{Emissions} = \text{EF}_{(T)} \times N_{(T)} \times \text{ N−excretion } \times 10^{-6} \tag{25}$$

where

Emissions	nitrous oxide emissions from manure management (Gg CH_4 year^{-1})
$\text{EF}_{(T)}$	emission factor for the defined livestock population (kg N head^{-1} year^{-1})
$N_{(T)}$	the number of heads of livestock species/category T
T	species/category of livestock
N–excretion	nitrogen excretion value for the livestock (kg head^{-1} year^{-1})

CH_4 and N_2O emission factors used in this study are shown in Table 8. N_2O emissions from manure management for livestock species of dairy cattle, nondairy cattle, young cattle, and buffaloes were taken as 60, 40, 25, and 46.5 kg/head/yr, respectively.

Table 8 Methane emission factors (EFs) used to calculate emissions from livestock management

Livestock	EF for enteric fermentation ($kg\ CH_4$/head/year)[a]	EF for manure management ($kg\ CH_4$/head/year)[a]
Dairy cattle	46	3.6
Nondairy cattle	25	2.7
Young cattle	25	1.8
Buffaloes	55	4
Sheep	5	0.3
Goats	5	0.2
Pigs	1	4
Horses and ponies	18	1.6

Source [a] Revised 1996 IPCC Guidelines for National Greenhouse Gas Inventories (IPCC 1997), Gurjar et al. (2004)

GHG Emissions from the Waste Sector. Methane (CH_4) is the major greenhouse gas emitted from the waste sector. Three major categories are considered in this study: municipal solid waste disposal, domestic waste water and industrial waste water. Considerable amounts of CH_4 are produced from the treatment and disposal of municipal solid waste. CH_4 produced at solid waste disposal sites (SWDS) contributes approximately 3–4 % to the annual global anthropogenic greenhouse gas emissions (IPCC 2001a, b). The IPCC method for estimating CH_4 emissions from SWDS is based on the first-order decay method, which assumes that CH_4 and CO_2 are formed when the degradable organic component in waste decays slowly throughout a few decades. No method is provided for N_2O emissions from SWDS because they are not significant. Emissions of CH_4 from waste deposited in a disposal site are highest in the first few years after deposition; then, the bacteria responsible for decay consume the degradable carbon in the waste, causing emissions to decrease (IPCC 2006). CH_4 emissions from SWDS were calculated by Eq. 26:

$$\text{Emissions } CH_4 = \left[\left[\text{MSW} \times \text{MCF} \times \text{DOC} \times \text{DOC}_f \times F \times 16/12\right] - R\right] \times (1 - \text{OF}) \tag{26}$$

where

MSW	mass of waste deposited (Gg/year)
MCF	methane correction factor for aerobic decomposition in the year of deposition (fraction)
DOC	degradable organic carbon in the year of deposition (Gg C/Gg waste)
DOC_f	fraction of degradable organic carbon that decomposes (fraction)
F	fraction of CH_4 in generated landfill gas (fraction)
R	methane recovery (Gg/year)
16/12	molecular weight ratio CH_4/C (ratio)
OF	oxidation factor (fraction)

The methane (CH_4) correction factor (MCF) accounts for the fact that unmanaged SWDS produce less CH_4 from a given amount of waste than anaerobic managed SWDS. An MCF of 0.4 was used in this study for unmanaged and shallow landfills (IPCC 2006). A degradable organic carbon value of 0.11 was obtained from NEERI (2005). The fraction of degradable organic carbon that decomposes (DOC $_f$) was taken as 0.5 (IPCC 2006) the fraction of CH_4 (F) in generated landfill gas was taken as 0.5 (IPCC 2006). It was considered that the there is no CH_4 recovery in the disposal sites in the major cities, and the oxidation factor was taken as zero for unmanaged and uncategorized solid waste disposal systems.

When treated or disposed anaerobically, wastewater can be a source of CH_4 and also N_2O emissions. Domestic, commercial, and industrial sectors are the sources of wastewater. The wastewater generated may be treated onsite or in a centralized plant, or disposed untreated to nearby bodies of water. Wastewater in closed underground sewers is not believed to be a significant source of CH_4. The wastewater in open sewers will be subjected to heating from the sun and the sewer conditions may be stagnant, causing anaerobic conditions to emit CH_4 (Nicholas 2006). There is a variation in the degree of wastewater treatment in most developing countries. Domestic wastewater is treated in centralized plants, septic systems, or may be disposed of in unmanaged lagoons or waterways, via open or closed sewers. Though the major industrial facilities may have comprehensive onsite treatment, in a few cases industrial wastewater is discharged directly into the water bodies (IPCC 2006).

The extent of CH_4 production depends primarily on the quantity of degradable organic material in the wastewater, the temperature, and the type of treatment system. More CH_4 is yielded from wastewater with higher COD or BOD concentrations when compared to wastewater with lower COD or BOD concentrations. An increase in temperature will also increase the rate of CH_4 production. N_2O is associated with the degradation of nitrogen components (urea, nitrate, and protein) in the wastewater. Domestic wastewater mainly includes human sewage mixed with other household wastewater, from sources such as effluent from shower drains, sink drains, and washing machines (IPCC 2006). Equation 27 was used to estimate CH_4 emissions from domestic wastewater:

$$\mathrm{CH_4emissions} = \left[\sum_{i,j}(U_i \times T_{i,j} \times \mathrm{EF}_j)\right](\mathrm{TOW} - S) - R \tag{27}$$

where

CH_4 Emissions	CH_4 emissions in inventory year (kg CH_4/year)
TOW	total organics in wastewater in inventory year (kg BOD/year)
S	organic component removed as sludge in inventory year (kg BOD/year)

U_i	fraction of population in income group i in inventory year
$T_{i,j}$	degree of utilization of treatment/discharge pathway or system, j, for each income group fraction i in inventory year
i	income group: rural, urban high income, and urban low income
j	each treatment/discharge pathway or system
EF_j	emission factor (kg CH_4/kg BOD)
R	amount of CH_4 recovered in inventory year (kg CH_4/year)

The emission factor (EF_j) was calculated using Eq. 28:

$$EF_j = Bo \times MCF_j \tag{28}$$

where

EF_j	emission factor (kg CH_4/kg BOD)
j	each treatment/discharge pathway or system
Bo	maximum CH_4 producing capacity (kg CH_4/kg BOD)
MCF_j	methane correction factor (fraction)

The total amount of organically degradable material in the wastewater (TOW) is a function of human population and BOD generation per person. It is expressed in terms of biochemical oxygen demand (kg BOD/year), as given by Eq. 29:

$$TOW = P \times BOD \times 0.001 \times I \times 365 \tag{29}$$

where

TOW	total organics in wastewater in inventory year (kg BOD/year)
P	country population in inventory year (person)
BOD	country-specific per capita BOD in inventory year (g/person/day)
0.001	conversion from grams BOD to kg BOD
I	correction factor for additional industrial BOD discharged into sewers (the collected default is 1.25 and uncollected default is 1.00)

N_2O emissions can occur as both direct and indirect emissions. Direct emissions are from the treatment plants, whereas indirect emissions are from wastewater after disposal of effluent into waterways, lakes, or the sea. Direct emissions of N_2O may be generated during both nitrification and denitrification of the nitrogen present (IPCC 2006). Equation 30 was used to estimate N_2O emissions from wastewater effluent:

$$N_2O \text{ emissions} = N_{\text{effluent}} \times EF_{\text{effluent}} \times 44/28 \tag{30}$$

where

N_2O emissions	N_2O emissions in inventory year (kg N_2O/year)
N_{effluent}	nitrogen in the effluent discharged to aquatic environments (kg N/year)
EF_{effluent}	emission factor for N_2O emissions from discharged to wastewater (kg N_2O–N/kg N)
44/28	conversion of kg N_2O–N into kg N_2O

EF_{effluent} of 0.005 kg N_2O–N/kg N is used in this study (default value: IPCC 2006).

Total nitrogen in the effluent was calculated by Eq. 31:

$$N\ \text{effluent} = (P \times \text{Protein} \times F_{\text{NPR}} \times F_{\text{NON-CON}} \times F_{\text{IND-COM}}) - N_{\text{sludge}} \quad (31)$$

where

N effluent	total annual amount of nitrogen in the wastewater effluent (kg N/year)
P	human population
Protein	annual per capita protein consumption (kg/person/year)
F_{NPR}	fraction of nitrogen in protein (kg N/kg protein)
$F_{\text{NON-CON}}$	factor for nonconsumed protein added to the wastewater
$F_{\text{IND-COM}}$	factor for industrial and commercial co-discharged protein into the sewer system
N sludge	nitrogen removed with sludge (kg N/year)

Per capita protein consumption (Pr) value is taken as 21.462 (Nutritional Intake in India 2009–2010). The fraction of nitrogen in protein (F_{NPR}), fraction of non-consumption protein ($F_{\text{NON-CON}}$), and fraction of industrial and commercial co-discharged protein ($F_{\text{IND-COM}}$) values were taken as 0.16, 1.4 (fraction), and 1.25 (fraction) kg N/kg protein, respectively (IPCC 2006).

Industrial wastewater may be treated onsite by the industries or can be discharged into domestic sewer systems. The emissions are included in domestic wastewater emissions if released into the sewer system. Methane is produced only from industrial wastewater with significant carbon loading that is treated under intended or unintended anaerobic conditions (IPCC 2006). Major industrial wastewater sources having high CH_4 production potential are pulp and paper manufacture, meat and poultry industry, alcohol, beer and starch production, organic chemical production, and food and drink processing industries. In this study, industrial wastewater emissions were calculated based on the data availability from the industries located within the city limits. The method for estimation of CH_4 emissions from onsite industrial wastewater treatment is given in Eq. 32:

$$CH_4 \text{ emissions } = \sum_i (\text{TOW}_i - S_i)\ \text{EF}_i - R_i \tag{32}$$

where

CH_4 Emissions	CH_4 emissions in inventory year (kg CH_4/year)
TOW_i	total organically degradable material in wastewater from industry i in inventory year (kg COD/year)
i	industrial sector
S_i	organic component removed as sludge in inventory year (kg COD/year)
EF_i	emission factor for industry i, kg CH_4/kg COD for treatment/discharge pathway or system(s) used in inventory year

If more than one treatment practice is used in an industry, then a weighted average is taken for this factor:
R_i amount of CH_4 recovered in inventory year, kg CH_4/year

The emission factor (EF_j) for each treatment/discharge pathway or system was calculated using Eq. 33:

$$\text{EF}_j = \text{Bo} \times \text{MCF}_j \tag{33}$$

where

EF_j	emission factor for each treatment/discharge pathway or system (kg CH_4/kg COD)
j	each treatment/discharge pathway or system
Bo	maximum CH_4 producing capacity (kg CH_4/kg COD)
MCF_j	methane correction factor (fraction)

The TOW is a function of industrial output (product) P (tons/year), wastewater generation W (m^3/ton of product), and degradable organics concentration in the wastewater COD (kg COD/m^3):

$$\text{TOW} = P \times \text{BOD} \times 0.001 \times I \times 365 \tag{34}$$

where

TOW	total organically degradable material in wastewater for industry 'i' (kg COD/year)
i	industrial sector
P_i	total industrial product for industrial sector i (t/year)
W_i	wastewater generated (m^3/t_{product})
COD_i	chemical oxygen demand (kg COD/m^3)

3 Results and Discussion

3.1 GHG Emissions from the Energy Sector

The major energy-related emissions considered under this sector are emissions from electricity consumption and fugitive emissions. Emissions resulting from consumption of fossil fuels and electricity in domestic and industrial sections are represented independently under their respective sectors.

Electricity Consumption. The major sectors for which greenhouse gases are assessed under electricity consumption are consumption in domestic sector, commercial sector, industrial sector, and others (public lighting, advertisement hoardings, railways, public water works and sewerage systems, irrigation, and agriculture). Emissions resulting from electricity consumption in the domestic and industrial sectors are attributed to the respective sector, along with the emissions from fuel consumption and industrial processes. GHG emissions from electricity consumption in the commercial sector and other sectors are represented in isolation for comparative analysis among the cities. Emissions resulting from auxiliary power consumption in plants located within the city boundaries and from the supply loss were also calculated in this study.

Figure 3 illustrates the emissions resulting from electricity consumption in commercial and other sectors, along with auxiliary consumption in power plants and supply losses. During the year 2009–10, the commercial sector in Delhi consumed 5339.63 MU of electricity, resulting in the release of 5428.55 Gg of CO_2 equivalent emissions. The emissions hold a share of 29.66 % of emissions when compared with emissions from commercial sector in other cities. Electricity consumption in other subcategories (which includes Delhi International Airport Limited, Delhi Jal Board, Delhi Metro Rail Corporation, public lighting, railway traction, agriculture and mushroom cultivation, and worship/hospital) consumed 2064.73 MU, resulting in the emission of 2099.11 Gg of CO_2 equivalents, which is responsible for 36.51 % of total emissions when compared with other cities. Auxiliary fuel consumption and supply losses resulted in 857.69 Gg of CO_2 equivalent emissions, accounting for 27.07 % of total emissions from this sector. CO_2 equivalent emissions from commercial, others, and auxiliary consumption and supply losses along with their shares are summarized for all cities in Table 9.

Fugitive Emissions. The intentional or unintentional release of greenhouse gases that occurs during the extraction, production, processing or transportation of fossil fuels is known as fugitive emissions (IPCC 2006). In the present study, fugitive emissions occurring from refinery crude throughput activity were estimated for Greater Mumbai city. The CH_4 emissions were found to be 0.0013 Gg for the year 2009–10, which gives a converted value of 0.033 Gg of CO_2 equivalents.

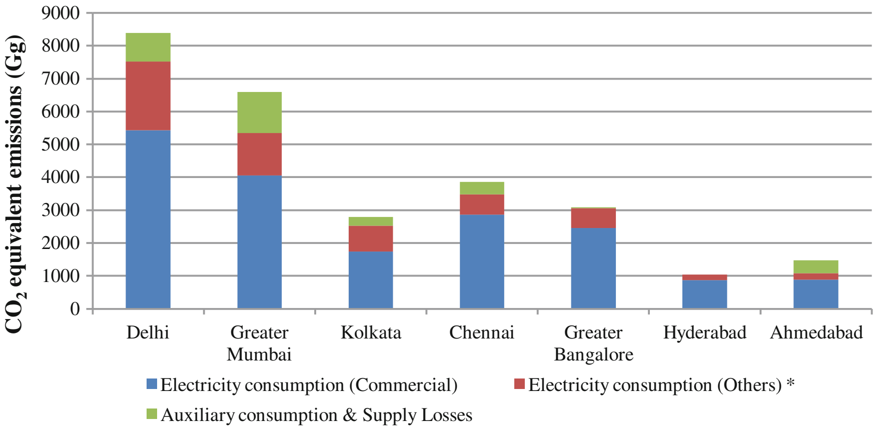

Fig. 3 Carbon dioxide equivalent emissions (CO_2 eq) from electricity consumption

Table 9 CO_2 equivalent emissions from electricity consumption by city

Cities	Commercial sector		Others[a]		Auxiliary consumption and supply losses	
	Gg	%	Gg	%	Gg	%
Delhi	5428.55	29.66	2099.11	36.51	857.69	27.07
Greater Mumbai	4049.85	22.13	1291.49	22.46	1247.54	39.38
Kolkata	1746.34	9.54	777.46	13.52	269.43	8.50
Chennai	2859.07	15.62	624.18	10.86	375.61	11.86
Greater Bangalore	2456.80	13.43	603.46	10.50	24.85	0.78
Hyderabad	870.4	4.76	165.74	2.88	_	_
Ahmedabad	888.73	4.86	188.09	3.27	392.85	12.40

Note [a] Others include electricity consumption in street lights, advertisements, public water works and sewer systems, irrigation and agriculture, pumping systems, religious/worship, and crematoriums and burial grounds

3.2 GHG Emissions from the Domestic Sector

The domestic sector contributes a considerable amount of emissions in city-level studies. The major sources include electricity consumption for lighting and other household appliances and consumption of fuel for cooking. In the present study, GHGs emitting from electricity consumption in domestic sector and fuel consumption were calculated. The major fuels used in this study are LPG, piped natural gas (PNG), and kerosene, based on the availability of data.

Figure 4 shows the total GHG emissions converted in terms of CO_2 equivalent from the domestic sector in major cities. In Delhi during the base year 2009, 11690.43 Gg of CO_2 equivalents were emitted from the domestic sector, which is the highest among all the cities, accounting for 26.4 % of the total emissions when compared with the other six cities (Fig. 4). Electricity consumption accounted for

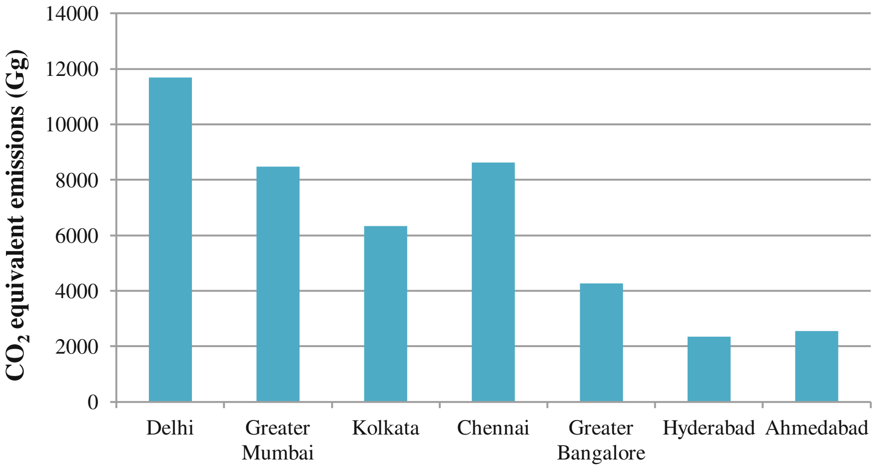

Fig. 4 Carbon dioxide equivalent emissions (CO_2 eq) from the domestic sector

9237.73 Gg of emissions out of the total domestic emissions. Earlier estimate show an emission of 5.35 million tons (5350 Gg) of CO_2 emissions from the domestic sector in Delhi during the year 2007–08 (Dhamija 2010). Greater Mumbai, which covers both Mumbai city and the suburban district, emits 8474.32 Gg of CO_2 equivalents from the domestic sector, which shares 19.14 % of the total emissions. The domestic sector in Kolkata results in 6337.11 Gg of CO_2 equivalents (14.31 % of total emissions).

Chennai ranks second in the list with 8617.29 Gg of CO_2 equivalents, contributing to approximately 19.5 % of total emissions. Greater Bangalore accounts for 4273.81 Gg of emissions from the domestic sector, which is 9.65 % of total emissions from the domestic sector. Hyderabad and Ahmedabad are responsible for 2341.81 Gg of CO_2 equivalent and 2544.03 Gg of CO_2 equivalent, respectively. These two cities together share 11 % of the total domestic emissions.

3.3 GHG Emissions from the Transportation Sector

In the major cities, the transportation sector is one of the major anthropogenic contributors of greenhouse gases (Mittal and Sharma 2003). Emissions resulting from the vehicles registered within the city boundaries and also from CNG-fuelled vehicles (if present) were calculated. Navigational activities from the port cities are also included in the emissions inventory on the basis of fuel consumption. Delhi has the highest emissions of the cities because it has the largest number of vehicles. According to the Transport Department in Delhi, the total number of vehicles in Delhi is more than the combined total vehicles in Mumbai, Chennai, and Kolkata. Delhi has 85 private cars per 1,000 population versus the

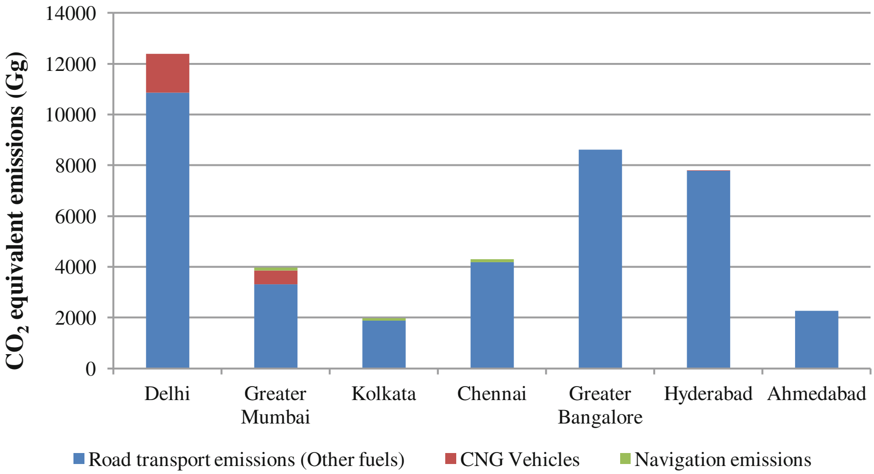

Fig. 5 Carbon dioxide equivalent emissions (CO_2 eq) from the transportation sector

country-wide average of 8 private cars per 1,000 population (SOE 2010). Delhi also had 344,868 CNG vehicles during the year 2009–10 (MoPNG 2010). Emissions resulting from road transportation, including CNG vehicles, and also in the port cities of India are depicted in Fig. 5.

In Delhi during the year 2009–10, total number of registered vehicles was 6,451,883, out of which there were approximately 20 lakhs of cars and jeeps and 40.5 lakhs of motorcycles, including scooters and mopeds. CNG-fuelled vehicles emitted 1527.03 Gg of CO_2 equivalents, whereas the remaining vehicles resulted in 10867.51 Gg of emissions, contributing almost 30 % of the total emissions in this subcategory, which is the highest among all the major cities. This is twice the earlier estimate of 5.35 million tons (5,350 Gg) of CO_2 emissions from road transportation sector in Delhi during the year 2007–08, or emissions of 7,660 Gg using the top–down approach and 8,170 Gg using the bottom-up approach (Dhamija 2010). CNG vehicles are also present in two other cities: Greater Mumbai and Hyderabad. Emissions from CNG vehicles in Mumbai during the year 2009–10 were found to be 531.34 Gg of CO_2 equivalents; for Hyderabad, it was estimated that 21.55 Gg of CO_2 equivalent was emitted from CNG vehicles during the study year. The emission inventories for the transportation sector in all the major cities are given in Table 10.

3.4 GHG Emissions from the Industrial Sector

Emissions were estimated from the major industrial processes that emit considerable GHGs and are located within the city boundaries (Table 11). Electricity consumption in the industrial sector was taken into account, from which the resulting emissions were calculated. Fuel consumption data were also used in a

Table 10 CO_2 equivalent emissions from the transportation sector in different cities

Cities	Road transportation emissions (Gg)		Navigation emissions (Gg)
	Vehicles using fuel other than CNG	CNG vehicles	
Delhi	10867.51	1527.03	_
Greater Mumbai	3320.66	531.34	114.18
Kolkata	1886.60	_	83.06
Chennai	4180.28	_	127.37
Greater Bangalore	8608.00	_	_
Hyderabad	7788.02	21.55	_
Ahmedabad	2273.72	_	_

Table 11 CO_2 equivalent emissions from the industrial sector by city

Cities	Industrial sector emissions (Gg)
Delhi	3049.30
Greater Mumbai	1798.69
Kolkata	2615.84
Chennai	4472.35
Greater Bangalore	2437.03
Hyderabad	1563.14
Ahmedabad	2044.35

few of the industries to estimate the emissions. The iron and steel industry, cement industry, fertilizer plants, and chemical manufacturing are some major industries that release huge amounts of GHGs into the atmosphere during the process. Emissions were calculated from the major polluting industries in city boundaries because the data were not available for small- and medium-scale industries.

Emissions were calculated for ammonia production from the fertilizer industries in Greater Mumbai and Chennai. In Greater Mumbai during the year 2009–10, 654.5 Gg of CO_2 equivalents were emitted from the fertilizer industry. Emissions from the fertilizer industry in Chennai were found to be 223.28 Gg of CO_2 equivalents from the production of ammonia. Emissions were also calculated for the glass industry (Greater Mumbai, Greater Bangalore), paper industry (Kolkata), and petroleum products (Chennai) using the fuel consumption data. Although this study does not present the entire emissions across industrial sectors in a city due to unavailability of data, the major GHG-emitting industries were included, along with the electricity consumption, which constitutes most of the emissions. Figure 6 shows the emissions across different cities.

3.5 GHG Emissions from Agricultural Activities

CH_4 emissions from paddy cultivation and N_2O emissions from soil management are the major sectors responsible for GHG emissions from this sector. Crop residue

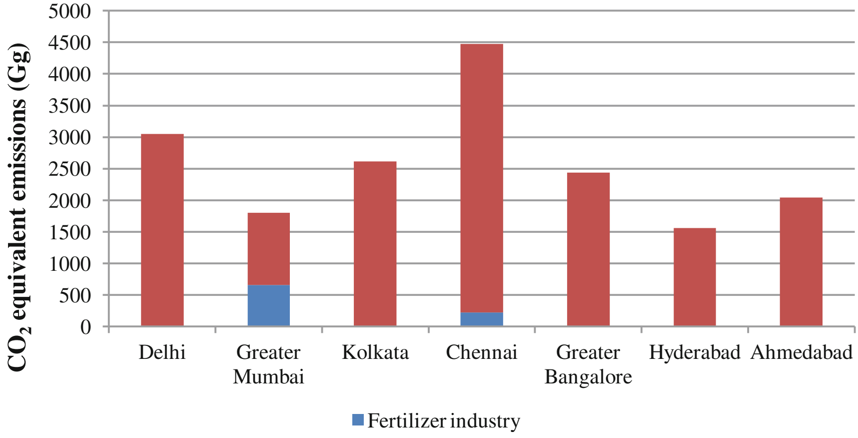

Fig. 6 Carbon dioxide equivalent emissions (CO_2 eq) from the industrial sector

Table 12 CO_2 equivalent emissions from agricultural activities in different cities

Cities	CO_2 equivalent emissions (Gg)		
	Paddy cultivation	Soils	Crop residue burning
Delhi	17.05	248.26	2.68
Greater Mumbai	_	6.95	_
Kolkata	_	10.54	_
Chennai	_	3.73	_
Greater Bangalore	5.10	113.86	_
Hyderabad	_	18.48	_
Ahmedabad	_	38.03	_

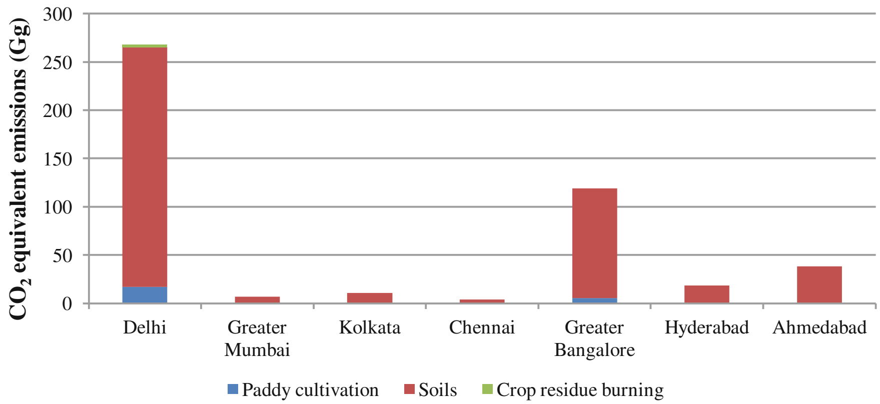

Fig. 7 Carbon dioxide equivalent emissions (CO_2 eq) from agricultural activities

burning is practiced in a few of the northern parts of India, which also releases GHG emissions. In the current study, emission inventory is carried out from these three sectors under agriculture-related activities. Table 12 shows the CO_2 equivalent emissions resulting from agriculture-related activities. Figure 7 shows the pattern of carbon dioxide equivalent emissions in the major cities.

Emissions from paddy cultivation are calculated for two major cities based on the area of paddy fields. Carbon dioxide equivalents were found to be 17.05 Gg in Delhi and 5.10 Gg in Greater Bangalore, respectively. Emissions resulting from the burning of crop residues at the end of the growing year were estimated based on Delhi's emission of 2.68 Gg of CO_2 equivalents. N_2O emissions were converted into CO_2 equivalents, as presented in Table 12. There are no agricultural activities in most of the cities, which indicates decline in agricultural practices as a result of increasing urbanization.

3.6 GHG Emissions from Livestock Management

Enteric fermentation and manure management are the two major activities resulting in the emission of GHGs from animal husbandry. In the present study, emissions from livestock management were carried out to calculate the emissions resulting from enteric fermentation and manure management in the major cities. The livestock population for cities was obtained using the 2003 and 2007 livestock census, from which the number of livestock was extrapolated to the inventory year 2009 (MOA 2003, 2005, 2007, 2008). The emission estimates for the major cities are given in Table 13.

Delhi and Greater Bangalore emitted the highest amounts of greenhouse gases due to animal husbandry. The emissions resulting from enteric fermentation for Delhi and Greater Bangalore were estimated to be 570.57 Gg of CO_2 equivalent and 129.36 Gg of CO_2 equivalents, respectively. Similarly, Delhi and Greater Bangalore emitted 43.09 Gg of CO_2 equivalent and 10.30 Gg of CO_2 equivalent, respectively, making these two cities higher emitters in the livestock management category, among the other cities. Figure 8 shows the emission profile of livestock management for different cities.

3.7 GHG Emissions from the Waste Sector

In the current study, GHG emissions from three major waste sectors were calculated: municipal solid waste, domestic wastewater, and industrial wastewater. CH_4 emissions from municipal solid waste disposal data were obtained from the local city municipality. CH_4 and N_2O emissions were calculated from the domestic

Table 13 CO_2 equivalent emissions from livestock management in different cities

Cities	CO_2 equivalent emissions from livestock management (Gg)	
	Enteric fermentation	Manure management
Delhi	570.57	43.09
Greater Mumbai	18.66	1.38
Kolkata	19.70	1.83
Chennai	7.61	0.55
Greater Bangalore	129.36	10.30
Hyderabad	41.98	3.05
Ahmedabad	93.77	6.66

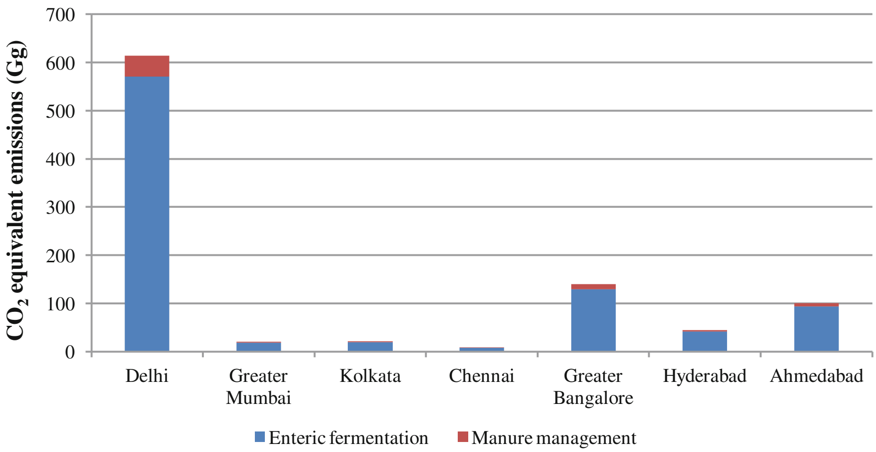

Fig. 8 Carbon dioxide equivalent emissions (CO_2 eq) from livestock management

Table 14 CO_2 equivalent emissions from the waste sector in different cities

Cities	Solid waste disposal		Domestic wastewater		Industrial wastewater (Gg)
	Gg	%	Gg	%	
Delhi	853.19	23.13	1378.75	28.00	_
Greater Mumbai	869.92	23.59	1058.09	21.49	_
Kolkata	535.33	14.51	385.03	7.82	143.84
Chennai	428.27	11.61	394.24	8.01	_
Greater Bangalore	374.73	10.16	759.29	15.42	_
Hyderabad	406.85	11.03	513.56	10.43	_
Ahmedabad	219.89	5.96	434.34	8.82	_

sector. In this study, the industrial wastewater emissions were calculated for only Kolkata city based on the availability of the data. Table 14 shows city wise CO_2 equivalent emissions and their shares in total emissions.

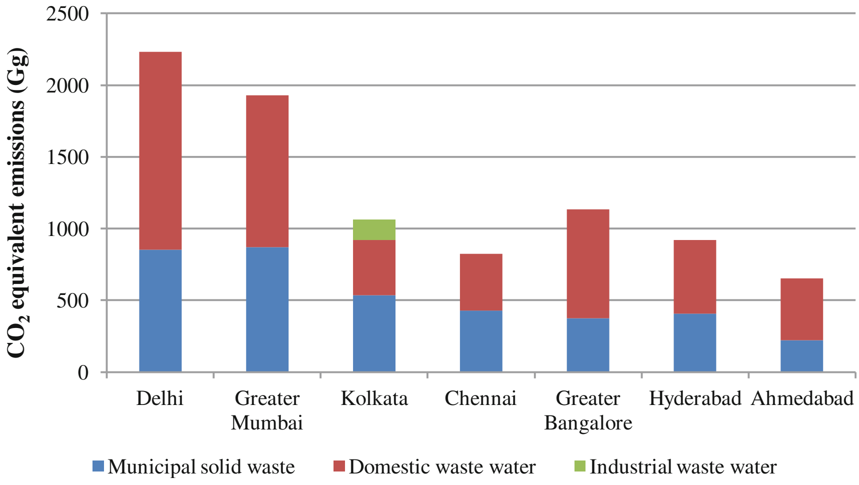

Fig. 9 Carbon dioxide equivalent emissions (CO_2 eq) from the waste sector

From the calculations of the present study, Delhi emits 853.19 Gg of CO_2 equivalents and Greater Mumbai emits 869.92 Gg of CO_2 equivalent using the IPCC (2006) method; together, these cities are responsible for almost 46.7 % of the total emissions occurring from solid waste disposal. The emissions depend on the parameters such as the amount of waste disposed, methane correction factor, degradable organic carbon, and oxidation factor IPCC (2006). Waste disposal in cities is a major source of anthropogenic CH_4 emissions these days. CH_4 and N_2O emissions from domestic water are calculated on the basis of population of the city. From the current inventories, the major emitters from the domestic wastewater sector are Delhi, Greater Mumbai, and Greater Bangalore, which emit 1378.75, 1058.09, and 759.29 Gg of CO_2 equivalents, respectively. Emissions from the industrial wastewater sector in Kolkata emitted 143.84 Gg of CO_2 equivalents during 2009. Waste emission profiles for the major cities are given in Fig. 9.

3.8 Intercity Variations of Carbon Footprint

Economic activity is a key factor that affects GHG emissions. An increase in the economy results in an increase in demand for energy and energy-intensive goods, which will also increase emissions. On the other hand, the growth of a country's economy results in improvements in technologies and promotes the advancement of organizations that focus on environmental protection and mitigation of emissions. In this study, total carbon dioxide equivalent emissions emitted from major Indian cities were compared with their economic activity, measured in terms of GDP. CO_2 equivalent emissions from Delhi, Greater Mumbai, Kolkata, Chennai,

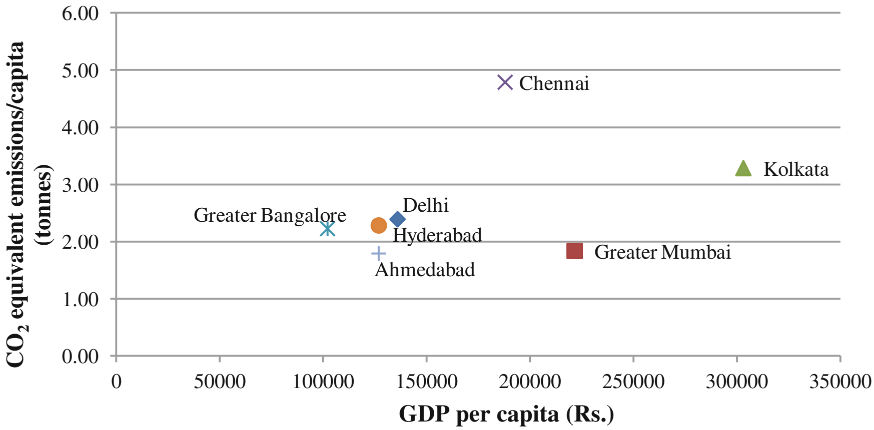

Fig. 10 CO_2 equivalent emissions per capita versus GDP per capita for all cities

Table 15 Values of CO_2 equivalent emissions/capita, GDP/capita and CO_2 equivalent emissions/GDP by city

Cities	CO_2 eq. emissions per capita (tonnes)	GDP per capita (Rs.)	CO_2 eq. emissions per GDP (tonnes CO_2/Lakh Rs.)
Delhi	2.40	136014.76	1.76
Greater Mumbai	1.84	221608.20	0.83
Kolkata	3.29	303187.96	1.08
Chennai	4.79	188020.64	2.55
Greater Bangalore	2.23	102161.49	2.18
Hyderabad	2.29	126936.59	1.80
Ahmedabad	1.80	126870.55	1.42

Greater Bangalore, Hyderabad, and Ahmedabad were found to be 38633.2, 22783.08, 14812.10, 22090.55, 19796.5, 13734.59, and 9124.45 Gg respectively. Figure 10 shows the relationship between carbon dioxide equivalent emissions per capita to GDP per capita.

Table 15 gives the values of carbon dioxide equivalent emissions per capita, GDP per capita, and carbon dioxide equivalent emissions per GDP for the cities.

Chennai emits 4.79 tons of CO_2 equivalent emissions per capita, which is the highest among all the cities, followed by Kolkata, which emits 3.29 tons of CO_2 equivalent emissions per capita. Chennai emits the highest CO_2 equivalent emissions per GDP (2.55 tons CO_2 eq/Lakh Rs.) followed by Greater Bangalore, which emits 2.18 tons CO_2 eq/Lakh Rs. Figure 11 shows the values of carbon dioxide equivalent emissions per GDP and GDP per capita for the cities.

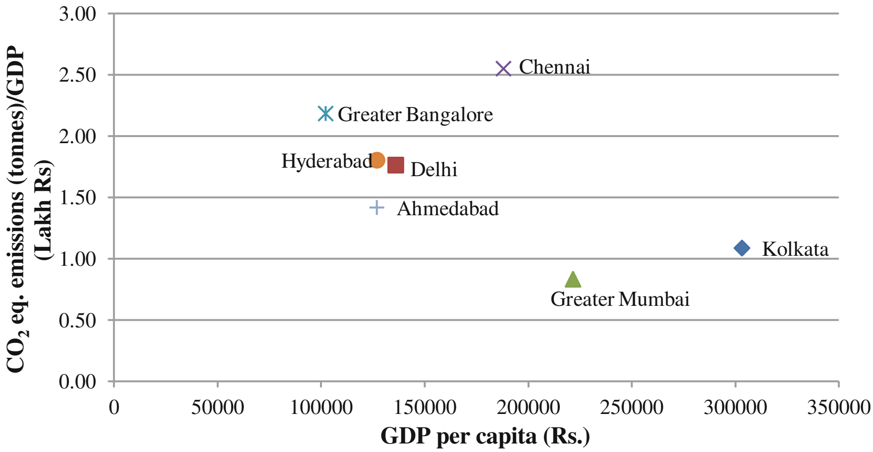

Fig. 11 CO_2 equivalent emissions per GDP versus GDP per capita for all cities

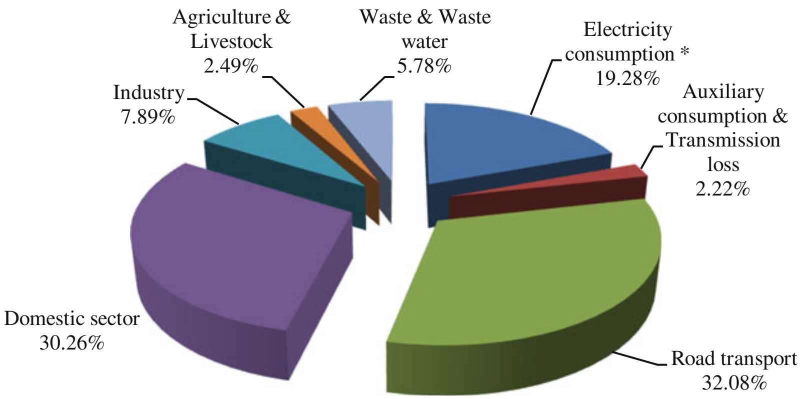

Fig. 12 Carbon dioxide equivalent emissions (Gg) in Delhi

3.9 Carbon Footprint: City and Sector

The aggregation of the carbon footprint of all sectors revealed that carbon emissions in major cities in India ranges from 38633.20 Gg/year (Delhi), 22783.08 (Greater Mumbai), 22090.55 (Chennai), 19796.60 (Greater Bangalore), 14812.10 (Kolkata), to 13734.59 (Hyderabad). Annex 1 details the sector-wise carbon footprint of the major cities in India.

Sector-wise carbon footprint analysis for Delhi (Fig. 12) revealed that the transport sector leads the carbon emissions (32.08 %), followed by the domestic sector (30.26 %) and electricity consumption (19.28 %). Electricity consumption includes public lighting, general purpose, temporary, and colony lighting.

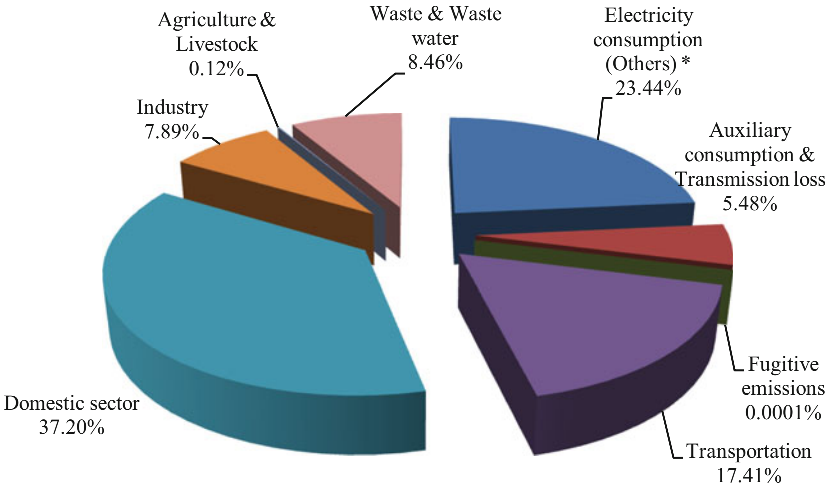

Fig. 13 Carbon dioxide equivalent emissions (Gg) in Greater Mumbai

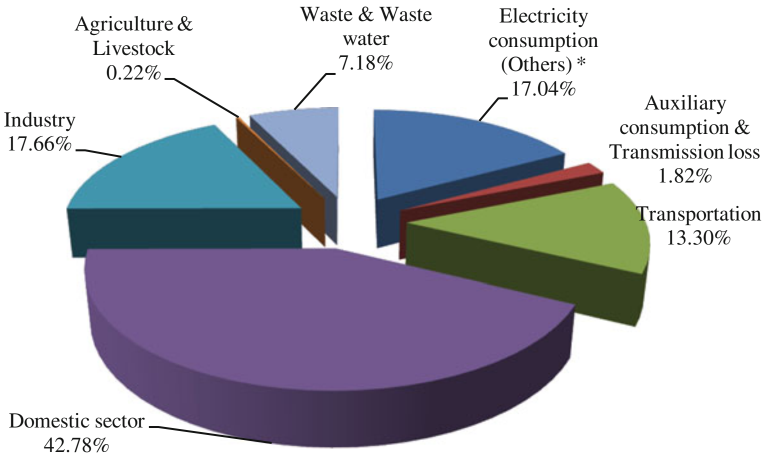

Fig. 14 Carbon dioxide equivalent emissions (Gg) in Kolkata

Figures 13, 14 and 15 depict the sector-wise carbon footprints for Mumbai, Kolkata, and Chennai. In these cities, the domestic sector has the highest carbon footprint, ranging from 42.78 % (Kolkata), 39.01 % (Chennai), to 37.2 % (Greater Mumbai). Next is the transport sector, at 19.50 % (Chennai), 17.41 % (Greater Mumbai), and 13.3 % (Kolkata).

Figures 16 and 17 illustrate the sector-wise carbon emissions for the information technology giants of India—Bangalore and Hyderabad. The lack of

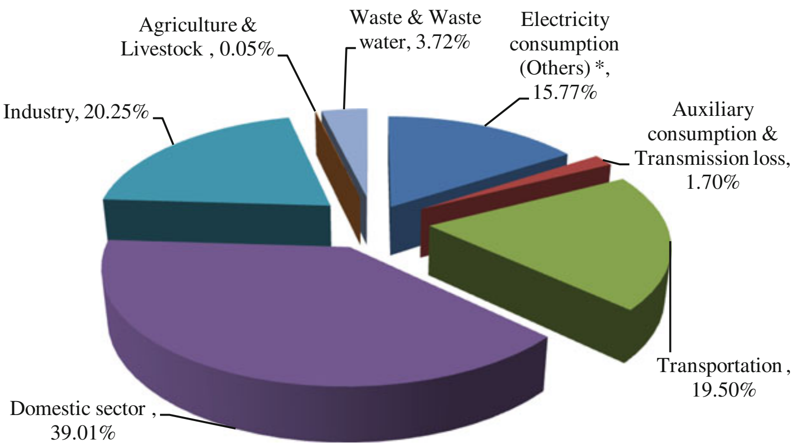

Fig. 15 Carbon dioxide equivalent emissions (Gg) from Chennai in 2009–10

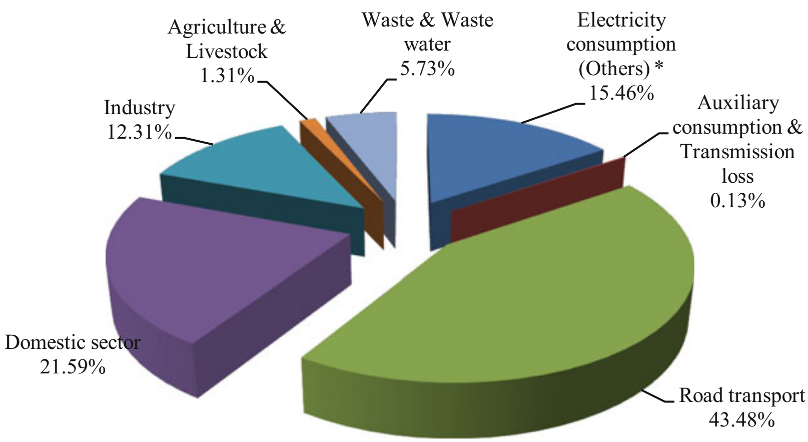

Fig. 16 Carbon dioxide equivalent emissions (Gg) in greater Bangalore

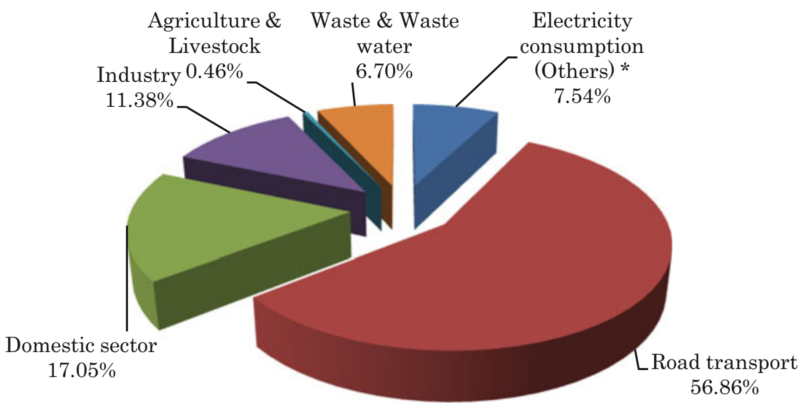

Fig. 17 Carbon dioxide equivalent emissions (Gg) in Hyderabad

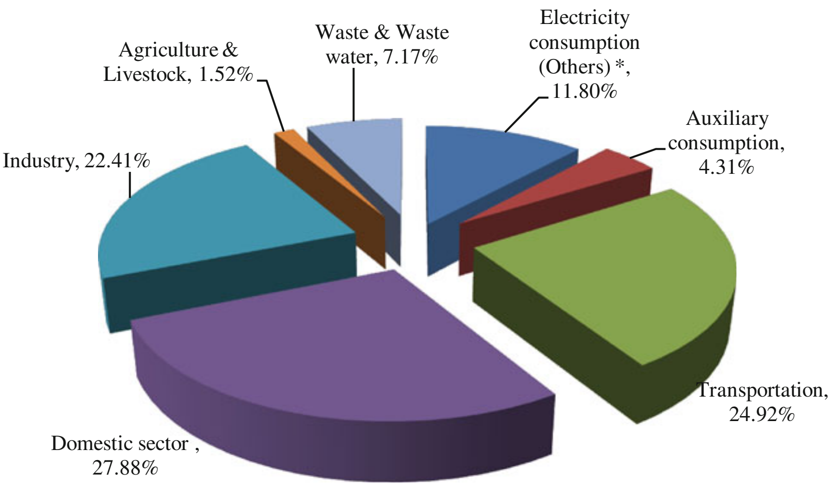

Fig. 18 Carbon dioxide equivalent emissions (Gg) in Ahmedabad

appropriate public transport systems in these cities and haphazard growth due to unplanned urbanization has led to large–scale use of private vehicles in these cities. Emissions from the transport sector range are 43.83 % in Greater Bangalore and 56.86 % in Hyderabad. Figure 18 depicts the carbon footprint of Ahmedabad city, with sector shares ranging from 27.88 % (domestic), 24.92 % (transportation), to 22.41 % (industry).

4 Conclusion

India is currently second most populous country in the world, and it contributes approximately 5.3 % of the total global GHG emissions. Major cities in India are witnessing rapid urbanization. The quality of air in the major Indian cities, which affects the climatic conditions as well as health of the community, is a major environmental concern. Higher levels of energy consumption have contributed to the degradation of the environment. Chennai emits 4.79 tons of CO_2 equivalent emissions per capita—the highest among all the cities—followed by Kolkata, which emits 3.29 tons of CO_2 equivalent emissions per capita. Chennai emits the highest CO_2 equivalent emissions per GDP (2.55 tons CO_2 eq/Lakh Rs.), followed by Greater Bangalore, which emits 2.18 tons CO_2 eq/Lakh Rs.

The carbon footprint of all the major cities in India helps to improve national-level emission inventories. In recent years, the popularity of the carbon footprint has grown, resulting in estimates of greenhouse gas emissions in the major metropolitan global cities and thereby framing regulations to reduce the emissions. The data regarding emissions from different sectors help the policy makers and city planners to devise mitigation strategies focusing on the particular sector, which helps to improve the environmental conditions within the city. Implementation of emission reduction strategies in cities also helps to gain carbon credits in the global markets, which has been an outcome of increased awareness about greenhouse gas emissions. Knowing the carbon footprint of major cities in India sector-wise would help planners in implementing appropriate mitigation measures.

- *Electricity consumption*. The calculation of greenhouse gas emissions from commercial and other sectors (public lighting, advertisements, railways, public water works and sewerage systems, irrigation, and agriculture) shows that energy consumption in the commercial sector is one of the major contributor of emissions in cities; it accounts for 15–24 % of total emissions in cities—except for Hyderabad and Ahmedabad, where it contributes 7.5 and 12 % of the total emissions. Delhi and Greater Mumbai had emissions of 7448.37 and 5341.34 Gg CO_2 equivalents, respectively, during 2009. This study also accounts for emissions from power plants located within the city. The results highlight that energy consumption in the commercial sector in cities is a major source of emissions.
- *Domestic sector*. The study reveals that the domestic sector causes the majority of the emissions in all the major cities due to the use of fossil fuels such as LPG, kerosene, and PNG for cooking purposes. Fossil fuels used for cooking purposes in households cause indoor air pollution. Consumption of electricity in the domestic sector for lighting, heating, and household appliances also share a major portion of emissions. It is calculated that the domestic sector resulted in emissions of 11690.43 Gg of CO_2 equivalents ($\sim$30 % of the total emissions) in Delhi, which is the highest among all the cities, followed by Chennai and Greater Mumbai, which emit 8617.29 Gg ($\sim$39 % of total emissions) and 8474.32 Gg of CO_2 equivalents ($\sim$39 % of total emissions), respectively. GHG emissions from the domestic sector in cities show the scope for cleaner fuels for cooking through the renewable sources, such as solar energy for water heating and other household purposes.
- *Transportation sector*. Road transport is another chief sector causing major emissions in cities. From the results obtained, the major emitters are Delhi and Greater Bangalore, which emit 12394.54 and 8608 Gg of CO_2 equivalents, respectively. The transportation sector is a major source of emissions when city-level studies are carried out. Emissions from CNG vehicles in a few of the cities were calculated, along with fuel consumption for navigation in the port cities. Lesser polluting fuels, such as LPG and CNG, can be made compulsory in major cities, phasing out older and inefficient vehicles; extensive public transport also helps to reduce pollution.
- *Industrial sector*. The industrial sector contributes approximately 10–20 % of the total emissions in all the major cities. In this study, electricity consumption in industries is calculated for all the cities, as well as emissions from major

industries located within the city boundaries. Chennai city was found to be the highest emitter, at 4472.35 Gg of CO_2 equivalents. There are insufficient data for medium- and small-scale industries located within the cities.

- *Agriculture and livestock activities*. Due to increasing urbanization, there are not many agricultural activities and animal husbandry practiced in the major metropolitan cities. This sector accounts for less than 3 % of total emissions among the cities. Delhi and Greater Bangalore emit 961 and 258.6 Gg of CO_2 equivalents due to livestock management and agricultural activities, respectively. The results prove that the agricultural practices are decreasing in cities due to increases in urban growth.
- *Waste sector*. Management and treatment of solid and liquid waste in cities results in emissions. This sector shares 3–9 % of total emissions from the cities. Delhi and Greater Mumbai emit the greatest amounts—2232 and 1928 Gg of CO_2 equivalents—compared with other cities. The waste sector therefore accounts for a considerable amount of greenhouse gas emissions when city-level studies are carried out.

5 Scope of Further Research

- Developing national-level emission factors for processes that have no country-specific emission factors helps to improve the precision of such emission estimations. Data availability for category-wise fossil fuel consumption (commercial, industrial) and for small- and medium-scale industries, along with wastewater treatment data for different years, help to improve the values obtained from these sectors for a particular inventory year.
- Based on the results obtained, policies should be framed to focus on the reduction of emissions from the targeted sector. For example, in cities with higher domestic emissions, the use of cleaner fuels (e.g., LPG, PNG) should be made mandatory, as should the utilization of solar energy for lighting and water heating. For cities with higher transportation emissions, less polluting fuels (e.g. LPG, CNG) may be made compulsory in vehicles such as cars, auto rickshaws, and buses, introducing more public transportation services and phasing out older vehicles. This helps the local authorities to draft regulations resulting in the mitigation of environmental degradation in cities.

Acknowledgement We are grateful to the NRDMS Division, The Ministry of Science and Technology, Government of India; The Ministry of Environment and Forests, Government of India, ISRO-IISc Space Technology Cell, Indian Institute of Science for the financial and infrastructure support. Remote-sensing data were downloaded from public domain (http://glcf.umiacs.umd.edu/data). The latest data of IRS 1D were procured from the National Remote Sensing Centre, Hyderabad

A.1 6 Annexure

Carbon footprint of Delhi

Sector	CO_2 emissions (Gg) 2009–2010	CH_4 emissions (Gg) 2009–2010	N_2O emissions (Gg) 2009–2010	CO_2 equivalent (Gg) 2009–2010
1. *Electricity consumption*				
Nondomestic	5402.6	0.079	0.081	5428.6
Railway traction and Delhi metro rail corporation	456.31	0.007	0.007	458.51
Others[a]	1553.8	0.023	0.023	1561.3
1(a) *Auxiliary consumption and supply losses*	853.55	0.011	0.013	857.69
2. *Road transportation*				
Vehicles using fuels other than CNG	10405	12.77	0.479	10868
CNG vehicles	1371.4	2.99	0.272	1527
3. *Domestic sector*	11639	0.353	0.144	11690
4. *Industrial sector*	3034.7	0.044	0.045	3049.3
5. *Agriculture*				
Paddy cultivation	_	0.682	_	17.05
Soils	_	_	0.833	248.26
Burning of crop residue	_	0.079	0.002	2.68
Electricity	78.92	0.001	0.001	79.3
6. *Livestock management*				
Enteric fermentation	_	22.82	_	570.57
Manure management	_	1.72	0.0002	43.09
7. *Waste*				
Municipal solid waste	_	34.13	_	853.19
Domestic waste water	_	46.07	0.761	1378.8
Total	**34795**	**121.79**	**2.66**	**38633**

Note [a] Others include electricity consumption in worship/hospital, staff, Delhi International Airport Limited, Delhi Jal Board

Carbon footprint of greater Mumbai

Sector	CO_2 emissions (Gg) 2009–2010	CH_4 emissions (Gg) 2009–2010	N_2O emissions (Gg) 2009–2010	CO_2 equivalent (Gg) 2009–2010
1. *Electricity consumption*				
Commercial sector	4031.80	0.071	0.055	4049.85
Others[a]	1285.73	0.023	0.017	1291.49
1(a) *Auxiliary consumption and supply losses*	1242.14	0.024	0.016	1247.54

(continued)

(continued)

Sector	CO_2 emissions (Gg) 2009–2010	CH_4 emissions (Gg) 2009–2010	N_2O emissions (Gg) 2009–2010	CO_2 equivalent (Gg) 2009–2010
1(b) *Fugitive emissions*				
Refinery crude throughput	_	0.0013	_	0.033
2.a. *Road transportation*				
Vehicles using fuels other than CNG	3174.58	3.85	0.168	3320.66
CNG vehicles	471.18	1.12	0.108	531.34
2.b. *Navigation*	113.03	0.010	0.003	114.18
3. *Domestic sector*	8444.48	0.396	0.067	8474.32
4. *Industrial sector*				
Ammonia production	654.50	_	_	654.50
Glass industry	21.09	0.001	0.0002	21.16
Electricity consumption	1118.04	0.020	0.0151	1123.04
5. *Agriculture*				
Soils	_	_	0.023	6.95
6. *Livestock management*				
Enteric fermentation	_	0.746	_	18.66
Manure management	_	0.055	0.000006	1.38
7. *Waste*				
Municipal solid waste	_	34.80	_	869.92
Domestic waste water	_	35.36	0.584	1058.09
Total	**20556.56**	**76.47**	**1.06**	**22783.08**

Note [a] Others include electricity consumption in advertisements, railways, street light, religious, crematorium and burial grounds

Carbon footprint of Kolkata

Sector	CO_2 emissions (Gg) 2009–2010	CH_4 emissions (Gg) 2009–2010	N_2O emissions (Gg) 2009–2010	CO_2 equivalent (Gg) 2009–2010
1. *Electricity consumption*				
Commercial sector	1737.79	0.018	0.027	1746.34
Metro and tramways	104.01	0.001	0.002	104.52
Others[a]	669.64	0.007	0.010	672.93
1(a) *Auxiliary consumption and supply losses*	268.11	0.003	0.004	269.43
2. *Road transportation*	1773.78	1.41	0.260	1886.60
3. *Navigation*	82.22	0.008	0.002	83.06
3(a) *Domestic sector*	6312.22	0.239	0.064	6337.11
4. *Industrial sector*	2603.03	0.027	0.002	2615.84
5. *Agriculture*				
Soils	_	_	0.035	10.54

(continued)

(continued)

Sector	CO_2 emissions (Gg) 2009–2010	CH_4 emissions (Gg) 2009–2010	N_2O emissions (Gg) 2009–2010	CO_2 equivalent (Gg) 2009–2010
6. *Livestock management*				
Enteric fermentation	_	0.788	_	19.70
Manure management	_	0.073	0.000004	1.83
7. *Waste*				
Municipal solid waste	_	21.41	_	535.33
Domestic waste water	_	12.87	0.213	385.03
Industrial waste water	_	5.75	_	143.84
Total	**13550.80**	**42.61**	**0.619**	**14812.10**

Note [a] Others include electricity consumption in educational institutions, hospitals, municipality, public water works and sewerage systems, pumping stations, street lighting, public utilities, sports complex and construction power

Carbon footprint of Chennai

Sector	CO_2 emissions (Gg) 2009–2010	CH_4 emissions (Gg) 2009–2010	N_2O emissions (Gg) 2009–2010	CO_2 equivalent (Gg) 2009–2010
1. *Electricity consumption*				
Commercial sector	2845.19	0.033	0.044	2859.07
Others[a]	621.15	0.007	0.010	624.18
1(a) *Auxiliary consumption and Supply losses*	373.78	0.004	0.006	375.61
2. *Road transportation*	3965.11	6.32	0.192	4180.28
3. *Navigation*	126.09	0.012	0.003	127.37
3(a) *Domestic sector*	8584.11	0.343	0.083	8617.29
4. *Industrial sector*	4452.26	0.059	0.062	4472.35
5. *Agriculture*				
Soils	_	_	0.013	3.73
6. *Livestock management*				
Enteric fermentation	_	0.304	_	7.61
Manure management	_	0.022	0.000002	0.55
7. *Waste*				
Municipal solid waste	_	17.13	_	428.27
Domestic waste water	_	13.17	0.218	394.24
Total	**20967.69**	**37.41**	**0.629**	**22090.55**

Note [a] others include electricity consumption in public lighting and water supply, advertisements, religious, and railway traction

Carbon footprint of greater Bangalore

Sector	CO_2 emissions (Gg) 2009–2010	CH_4 emissions (Gg) 2009–2010	N_2O emissions (Gg) 2009–2010	CO_2 equivalent (Gg) 2009–2010
1. *Electricity consumption*				
Commercial sector	2444.92	0.029	0.037	2456.80
Others[a]	600.54	0.007	0.009	603.46
1(a) *Auxiliary consumption and Supply losses*	24.76	0.001	0.0002	24.85
2. *Road transportation*	8288.55	7.65	0.430	8608.00
3. *Domestic sector*	4256.22	0.170	0.045	4273.81
4. *Industrial sector*	2425.28	0.029	0.037	2437.03
5. *Agriculture*				
Paddy cultivation	_	0.204	_	5.10
Soils	_	_	0.382	113.86
6. *Livestock management*				
Enteric fermentation	_	5.17	_	129.36
Manure management	_	0.411	0.000047	10.30
7. *Waste*				
Municipal solid waste	_	14.99	_	374.73
Domestic waste water	_	25.37	0.419	759.29
Total	**18040.29**	**54.04**	**1.36**	**19796.60**

Note [a] Others include electricity consumption in irrigation and agriculture, street lighting, water works, and Railways

Carbon footprint of Hyderabad

Sector	CO_2 emissions (Gg) 2009–2010	CH_4 emissions (Gg) 2009–2010	N_2O emissions (Gg) 2009–2010	CO_2 equivalent (Gg) 2009–2010
1. *Electricity consumption*				
Commercial sector	866.23	0.013	0.013	870.40
Others[a]	164.95	0.002	0.002	165.74
2. *Road Transportation*				
Vehicles using fuels other than CNG	7488.51	6.60	0.452	7788.02
CNG vehicles	18.64	0.066	0.004	21.55
3. *Domestic sector*	2331.35	0.055	0.030	2341.81
4. *Industrial sector*	1555.82	0.024	0.023	1563.14
5. *Agriculture*				
Soils	_	_	0.062	18.48
6. *Livestock management*				
Enteric fermentation	_	1.68	_	41.98
Manure management	_	0.122	0.00001	3.05

(continued)

(continued)

Sector	CO_2 emissions (Gg) 2009–2010	CH_4 emissions (Gg) 2009–2010	N_2O emissions (Gg) 2009–2010	CO_2 equivalent (Gg) 2009–2010
7. *Waste*				
Municipal solid waste	_	16.27	_	406.85
Domestic waste water	_	17.16	0.284	513.56
Total	**12425.50**	**41.99**	**0.870**	**13734.59**

Note [a] Others include electricity consumption in public lighting, general purpose, temporary and colony lighting

Carbon footprint of Ahmedabad

Sector	CO_2 emissions (Gg) 2009–2010	CH_4 emissions (Gg) 2009–2010	N_2O emissions (Gg) 2009–2010	CO_2 equivalent (Gg) 2009–2010
1. *Electricity consumption*				
Commercial sector	884.52	0.015	0.013	888.73
Others[a]	187.20	0.003	0.003	188.09
1(a) *Auxiliary consumption*	390.93	0.004	0.006	392.85
2. *Road Transportation*	2151.93	3.46	0.118	2273.72
3. *Domestic sector*	2532.60	0.059	0.033	2544.03
4. *Industrial sector*	2034.67	0.034	0.030	2044.35
5. *Agriculture*				
Soils	_	_	0.128	38.03
6. *Livestock management*				
Enteric fermentation	_	3.75	_	93.77
Manure management	_	0.266	0.00003	6.66
7. *Waste*				
Municipal solid waste	_	8.80	_	219.89
Domestic waste water	_	14.51	0.240	434.34
Total	**8181.85**	**30.91**	**0.57**	**9124.45**

Note [a] Others include electricity consumption in water pumping, drainage pumping stations, lighting and temporary supply

References

ADB (2006) Energy efficiency and climate change considerations for on-road transport in Asia. Asian Development Bank, Manila

Adhya TK, Rath AK, Gupta PK, Rao VR, Das SN, Parida KM, Parashar DC, Sethunathan N (1994) Methane emission from flooded rice fields under irrigated conditions, Biol Fertil Soils 18:245–248

Aggarwal PK, Pathak H, Bhatia A, Kumar S (2003) Final report on inventory of nitrous oxide emission from agricultural soils, submitted by Winrock International India on behalf of Ministry of Environment and Forest, Government of India

Alexander M (1961) Introduction to soil microbiology. Wiley, New York, pp 227–244

ALGAS (1998) Asia least cost greenhouse gas abatement strategy project, country case study India. Asian Development Bank, Manila

Andrew JE (2008) What is carbon footprint? An overview of definitions and methodologies. In: Vegetable Industry carbon footprint study – discussion papers and workshop, 26 Sept 2008. Horticulture Australia Limited, Sydney

Anjali S, Devotta S (2007) Indoor air quality of public places in Mumbai India in terms of volatile organic compounds. Environ Monit Assess 133:127–138

ARAI (2007). Emission factor development for Indian vehicles—as a part of ambient air quality monitoring and emission source apportionment studies. Automotive Research Association of India, Project No. AFL/2006-07/IOCL/EmissionFactor Project/Final Rep., Pune

Baidya S, Borken Kleefeld J (2009) Atmospheric emissions from road transportation in India. Energy Policy 37(2009):3812–3822

Balachandran S, Meena BR, Khillare PS (2000) Particle size distribution and its elemental composition in the ambient air of Delhi. Environ Int 26(1–2):49–54

Bandyopadhyay TK, Goyal P, Singh MP (1996) Generation of methane from paddy fields and cattle in India and its reduction at source. Atmos Environ 30(14):2569–2574

Basic Port Statistics of India (2009–10) Transport research wing, Ministry of Shipping, Government of India, 2011

Battle M (1996) Atmospheric gas concentrations over the past century measured in air from firn at South Pole. Nature 383:231–235

Benjamin KS, Marilyn AB (2009) Twelve metropolitan carbon footprints: a preliminary comparative global assessment. Energy Policy 38:4856–4869 (2010)

Berner W, Oeschger H, Stauffer B (1980) Information on the CO_2 cycle from ice core studies. Radiocarbon 22:227–235

Bernstein L, Roy J, Delhotal KC, Harnisch J, Matsuhashi R, Price L, Tanaka K, Worrell E, Yamba F, Fengqi Z (2007) Industry. In: Metz B, Davidson OR, Bosch PR, Dave R, Meyer LA (eds) Climate change 2007: mitigation. Contribution of working group III to the fourth assessment report of the intergovernmental panel on climate change. Cambridge University Press, Cambridge

Bhatia A, Pathak H, Aggarwal PK (2004) Inventory of methane and nitrous oxide emissions from agriculture soils of India and their global warming potential. Curr Sci 87(3):317–324

Bhatia P (2008) Methodologies for measuring the carbon footprint, June 2008, ICTS and climate change, international symposium, London

Bhatia A, Pathak H, Aggarwal PK (2008) Inventory of methane and nitrous oxide emissions from agricultural soils of India and their global warming potential. Curr Sci 87(3):317–324

Bhattacharya S, Mitra AP (1998) Greenhouse gas emissions in India for year 1990, centre on global change. National Physical Laboratory, New Delhi

Blake DR et al (1982) Global increase in atmospheric methane concentrations between 1978 and 1980. Geophys Res Lett 9:477–480

Bouwman AF (1990) Exchange of greenhouse gases between terrestrial ecosystems and the atmosphere. In: Bouman AF (ed) Soils and the greenhouse effect. Wiley, New York, pp 61–127

Bronson KF, Neue HU, Singh U, Abao EB Jr (1997) Automated chamber measurements of methane and nitrous oxide flux in a flooded rice soil: I. Residue, nitrogen and water management. Soil Sci Soc Am J 61:981–987

BSI (2008) Publicly available specification 2050. Specification for the assessment of the life cycle greenhouse gas emissions of goods and services, British Standards Institute

Butler JH, Battle M, Bender ML, Montzka SA, Clark AD, Saltzman ES, Sucher CM, Severinghaus JP, Elkins JW (1999) A record of atmospheric halocarbons during the twentieth century from polar firn air. Nature 399:749–755

Carbon Trust (2007a) Carbon footprint measurement methodology, version 1.1. The Carbon Trust, London http://www.carbontrust.co.uk. Accessed 27 Feb 2007

Carbon Trust (2007b) Carbon footprinting. An introduction for organizations. Available online at http://www.carbontrust.co.uk/publications/publicationdetail.htm?productid=CTV033. Accessed 5 May 2008

Census of India (2001). Houses, household amenities and assets: India, States and Union Territories—2001 census, Office of the Registrar General, Census of India, Government of India, New Delhi www.censusindia.net/2001housing/housing_tables_main.html

Census of India (2011) Cities having population 1 lakh and above, Census. http://censusinida.gov.in

Chakraborty N, Sarkar GM, Lahiri SC (2000) Methane emission from rice paddy soils, aerotolerance of methanogens and global thermal warming. Environmentalist 20:343–350

Chakraborty N, Mukherjee I, Santra AK, Chowdhury S, Chakraborty S, Bhattacharya S, Mitra AP, Sharma C (2007) Measurement of CO_2, CO, SO_2 and NO emissions from coal based thermal power plants in India. Atmos Environ 42(6):1073–1082 (February 2008)

Chattopadhyay S, Gupta A, Ray S (2009) Municipal solid waste management in Kolkata, India. Waste Manage 29(4):1449–1458

Chelani AB, Devotta Sukumar (2007) Air quality assessment in Delhi: before and after CNG as fuel. Environ Monit Assess 125:257–263

Chun M, Mei-ting J, Xiao-chun Z, Hong-yuan L (2011) Energy consumption and carbon emissions in a coastal city in China, Procedia Environ Sci 4:1–9

Cook E (1971) The flow of energy in an industrial society. Energy and power; a scientific American book. W. H. Freeman and Co. San Francisco, pp 83–91

Courchene TJ, Allan JR (2008) Climate change: the case of carbon tariff/tax. Policy Options 3:59–64

CPCB (2007) Transport fuel quality for year 2005. Central Pollution Control Board, Government of India, New Delhi

Das Anjana, Jyoti Parikh (2004) Transport scenarios in two metropolitan cities in India: Delhi and Mumbai. Energy Convers Manage 45(15–16):2603–2625

Delmas RJ, Ascencio JM, Legrand M (1980) Polar ice evidence that atmospheric CO_2 20,000 yr BP was 50 % of present. Nature 284:155–157

Dhakal Shobhakar (2009) Urban energy use and carbon emissions from cities in China and policy implications. Energy Policy 37(11):4208–4219

Dhamija (2010) Executive summary of inventorization of greenhouse gases—sources and sinks in Delhi, climate change agenda for Delhi 2009–12, Government of Delhi

Dlugokencky EJ, Masarie KA, Lang PM, Tans PP (1998) Continuing decline in the growth rate of the atmospheric methane burden. Nature 393:447–450

EEA (2009) EMEP/EEA Air pollutant emission inventory guidebook. European Environment Agency, Copenhagen

EPA (2010) Methane and nitrous oxide emissions from natural sources. United States Environmental Protection Agency

Etheridge DM (1996) Natural and anthropogenic changes in atmospheric CO_2 over the last 1000 years from air in Antarctic ice and firn. J Geophys Res 101:4115–4128

Fadel MEI, Massoud M (2001) Methane emissions from wastewater management. Environ Pollut 114(2):177–185

Fan Y, Liu L-C, Wu G, Tsai H-T, Wei Y-M (2007) Changes in carbon intensity in China: empirical findings from 1980–2003. Ecol Econ 62(3–4):683-691

Ferry JG (1992) Biochemistry of methanogenesis. CRC CRic Rev Biochem Mol Biol 27:473–503

Fraser PJ, Khalil MAK, Rasmussen RA, Crawford AJ (1981) Trends of atmospheric methane in the southern hemisphere. Geophys Res Lett 8:1063–1066

Garg A, Shukla PR (2002) Emission inventory of India. Tata McGraw Hill, New Delhi, pp 84–89

Garg A, Sumana B, Shukla PR, Dadhwal VK (2001) Regional and sectoral assessment of greenhouse gas emissions in India. Atmos Environ 35:2679–2695

Garg A, Shukla PR, Manmohan K, Deepa M (2004) Indian methane and nitrous oxide emissions and mitigation flexibility. Atmos Environ 38:1965–1977

Garg A, Shukla PR, Manmohan K (2006) The sectoral trends of multi gas emissions inventory of India. Atmos Environ 40:4608–4620

Garg A, Bhushan K, Shukla PR (2011) Methane emissions in India: Sub regional and sectoral trends. Atmos Environ 45(28):4922–4929

Gaur AC, Neelkantan S, Dargan KS (1984) Organic manures, 2nd edn. Indian Council of Agricultural Research, New Delhi

Ghose Mrinal K, Paul R, Banerjee SK (2004) Assessment of the impacts of vehicular emissions on urban air quality and its management in Indian context: the case of Kolkata. Environ Sci Policy 7(4):345–351

Ghosh S, Majumdar D, Jain MC (2003) Methane and nitrous oxide emissions from an irrigated rice of North India. Chemosphere 51(3):181–195

Global Footprint Network (2007), Ecological footprint: overview, global footprint network, http://www.footprintnetwork.org/gfn_sub.php?content=footprint_overview

Goldemberg J (1995) Energy needs in developing countries and sustainability. Science 269:1058–1059

Goyal P, Rama Krishna TV (1998) Various methods of emission estimation of vehicular traffic in Delhi. Transp Res Part D 3:309–317

Graedel TE, McRae JE (1980) On the possible increase of atmospheric methane and carbon monoxide concentrations during the last decade. Geophys Res Lett 7:977–979

Granli T, Bockman OC (1994) Nitrous oxide from agriculture. Norw J Agric Sci 12:128

Gupta R (2005) Oxford climate change action plan. Oxford city council environment scrutiny committee

Gupta AK, Subhankar N, Mukhopadhyay UK (2006) Characterisation of PM10, PM2.5 and benzene soluble organic fraction of particulate matter in an urban area of Kolkata, India. Environ Monit Assess 115:205–222

Gupta PK, Arvind KJ, Koul S, Sharma P, Pradhan V, Vandana G, Sharma C, Nahar S (2007) Methane and nitrous oxide emission from bovine manure management practices in India, Environ Pollut 146(1):219–224

Gupta PK, Gupta V, Sharma C, Das SN, Purkait N, Adhya TK, Pathake H, Ramesh R, Baruah KK, Venkatratnam L, Gulab Singh CSP Iyer (2009) Development of methane emission factors for Indian paddy fields and estimation of national methane budget. Chemosphere 74(4):590–598

Gurjar BR, van Aardenne JA, Lelieveld J, Mohan M (2004) Emission estimates and trends (1990–2000) for megacity Delhi and implications. Atmos Environ 38(33):5663–5681

Gurjar BR, Nagpure AS, Prashant K, Nalin S (2010) Pollutant emissions from road vehicles in mega-city Kolkata, India: past and present trends. J Air Pollut Control 10(2):18–30

Hammond G (2007) Time to give due weight to the 'carbon footprint' issue. Nature 445(7125):256

Harnisch J, Eisenhauer A (1998) Natural CF_4 and SF_6 on Earth. Geophys Res Lett 25:2401–2404

Hasselmann K (1993) Optimal fingerprints for the detection of time dependent climate change. J Clim 6:1957-1971

Hawksworth J Hoehn T, Tiwari A (eds) (2009) Pricewaterhouse coopers UK economic outlook

Hingane LS, Rupa Kumar K, Ramana Murthy BV (1985) Long-term trends of surface air temperature in India. Int J Climatol 5:521–528

Hoornweg D, Sugar L, Gomez CLT (2011) Cities and greenhouse gas emissions: moving forward. Environ Urbanization 23:207–227

Houghton JT, Callander BA, Varney SK (eds) (1992) Climate change 1992, the supplementary report to the ipcc scientific assessment, intergovernmental panel on climate change. Cambridge University Press, Cambridge

HRA (2008) House Rent Allowance, Sixth Central Pay Commission, Ministry of Finance, Government of India

http://unfccc.int/ghg_data/items/3825.php Global warming potentials: Climate change 1995, the science of climate change: summary for policymakers. Technical Summary of the working group I report, p 22, 1995. (http://unfccc.int/ghg_data/items/3825.php)

http://www.iso.org/iso/iso_catalogue/catalogue_tc/catalogue_detail.htm?csnumber=38382. Accessed 5 Dec 2012

http://www.iso.org/iso/catalogue_detail?csnumber=38381. Accessed 9 Dec 2012

Humphrey CR, Buttel FR (1984) Environment energy and society. Wadsworth Publishers, Belmont

INCCA (2010) India: Greenhouse Gas Emissions (2007). Indian Network for Climate Change Assessment (INCCA), The Ministry of Environment and Forests, Government of India

Indermuhle A (1999) Holocene carbon-cycle dynamics based on CO_2 trapped in ice at Taylor Dome Antarctica. Nature 398:121–126

IPCC (1996) Report of the twelfth season of the Intergovernmental Panel on Climate Change, Mexico City, 11–13 Sept 1996

IPCC (1997) Revised 1996 IPCC guidelines for national greenhouse gas inventories. In: Houghton JT, Meira Filho LG, Lim B, Treanton K, Mamaty I, Bonduki Y (eds)

IPCC (1998) Principles governing IPCC work. Approved at the fourteenth session, Vienna (1–3 October 1998)

IPCC (2000). Good practice guidance and uncertainty management in national greenhouse gas inventories. In: Penman J, Kruger D, Galbally I, Hiraishi T, Nyenzi B, Emmanul S, Buendia L, Hoppaus R, Martinsen T, Meijer J, Miwa K Tanabe K (eds) Published for the IPCC by the Institute for Global Environmental Strategies, Japan. ISBN 4-88788-000-6

IPCC (2001a) Climate change 2001: the scientific basis. contribution of working group I to the third assessment report of the intergovernmental panel on climate change. In: Houghton JT, Ding Y, Griggs DJ, Noguer M, van der Linden PJ, Dai X, Maskell K, Johnson CA (eds) Cambridge University Press, Cambridge, p 881

IPCC (2001b) Climate change 2001: the scientific basis. Contribution of working group I to the third assessment report of the intergovernmental panel on climate change In: Houghton JT et al (eds) Cambridge University Press, Cambridge p 881

IPCC (2003) Good practice guidance for land use, land-use change and forestry. In: Jim P, Michael G, Taka H, Thelma K, Dina K, Riitta P, Leandro B, Kyoko M, Todd N, Kiyoto T, Fabian W (eds) Institute for Global Environmental Strategies (IGES) for the IPCC ISBN 4-88788-003-7

IPCC (2006) IPCC guidelines for national greenhouse gas inventories, prepared by the national greenhouse gas inventories programme. In: Eggleston HS, Buendia L, Miwa K, Ngara T, Tanabe K (eds) IGES, Japan

IPCC (2007a) Climate change 2007: synthesis report. contribution of working groups I, II and III to the fourth assessment report of the intergovernmental panel on climate change. In: Core Writing Team: Pachauri RK, Reisinger A (eds) IPCC, Geneva, p 104

IPCC (2007b) Contribution of working group III to the fourth assessment report of the intergovernment panel on climate change. Mertz B, Davidson OR, Bosch PR, Dave R, Meyer LA (eds) Climate change 2007: mitigation. Cambridge University Press, Cambridge

ISO (2006a) ISO 14064-1:2006. Greenhouse gases part 1: specification with guidance at the organization level for quantification and reporting of greenhouse gas emissions and removals

ISO (2006b) ISO 14064-2:2006. Greenhouse gases part 2: specification with guidance at the project level for quantification, monitoring and reporting of greenhouse gas emission reductions or removal enhancements

Jain MC, Kumar S, Wassmann R, Mitra S, Singh SD, Singh RP, Singh R, Yadav AK, Gupta S (2000) Methane emissions from irrigated rice fields in Northern India (New Delhi). Nutr Cycl Agroecosyst 58:75–83

Jalihal SA, Reddy TS (2006) CNG: an alternative fuel for public transport. J Sci Ind Res 65:426–431

Jalihal SA, Ravinder K, Reddy TS (2005) Traffic characteristics in India. In: Proceedings of the Eastern Asia society for transportation studies, volume 5, pp 1009–1024

Jessica A (2008) Report- what is carbon footprint, 19 February 2008. The Edinburg Centre for Carbon Management

Kampschreur MJ, Temmnik H, Kleerebezem R, Jetten MSM, van Loosdrecht MCM (2009) Nitrous oxide emission during wastewater treatment. Water Res 43(17):4093–4103

Keeling CD (1961) The concentration and isotopic abundances of carbon dioxide in rural and marine air. Geochim Cosmochim Acta 24:277–298

Keeling CD (1998) Rewards and penalties of monitoring the Earth. Ann Rev Energy Environ 23:25–82

Kennedy C, Steinberger J, Gasson B, Hansen Y, Hillman T, Havranek M, Pataki D, Phdungsilp A, Ramaswami A, Mendez GV (2009) Greenhouse gas emissions from global cities. Environ Sci Technol 2009(43):7297–7302

Kennedy C, Steinberger J, Gasson B, Hansen Y, Hillman T, Havranek M, Pataki D, Phdungsilp A, Ramaswami A, Mendez GV (2010) Methodology for inventorying greenhouse gas emissions from global cities. Energy Policy 38(2010):4828–4837

Khalil MAK, Rasmussen RA (1988) Nitrous oxide: trends and global mass balance over the last 3000 years. Ann Glaciol 10:73–79

Khosa MK, Sidhu BS, Benbi DK (2010) Effect of organic materials and rice cultivars on methane emission from rice field. J Environ Biol 31:281–285

Koyama T (1963) Gaseous metabolism in lake sediments and paddy soils and the production of atmospheric methane and hydrogen. J Geophys Res 68:3971–3973

Kulkarni P, Chandra V (2000) Atmospheric polycyclic aromatic hydrocarbons in Mumbai, India. Atmos Environ 34(2000):2785–2790

Kumar S (2003) Cleaner production technology and bankable energy efficiency drives in fertilizer industry in India to minimize greenhouse gas emissions—case study. In: Greenhouse gas control technologies—sixth international conference 2003, p 1031–1036

Kumar U, Jain MC, Pathak H, Kumar S, Majumdar D (2000) Nitrous oxide emissions from different fertilizers and its mitigation by nitrification inhibitors in irrigated rice. Biol Fertil Soils 32(2000):474–478

Kumar S, Gaikwad SA, Shekdar AV, Kshirsagar PS, Singh RN (2004) Estimation method for national methane emission from solid waste landfills. Atmos Environ 38:3481–3487

Kumar S, Bhattacharya JK, Vaidya AN, Chakrabarti T, Devotta S, Akolkar AB (2009) Assessment of the status of municipal solid waste management in metro cities, state capitals, class I cities and class II towns in India: An insight. Waste manage 29:883–895

Larsen HN, Hertwich EG (2010) Identifying important characteristics of municipal carbon footprints. Ecol Econ 70:60–66

Latha K, Madhavi V, Krishna P, Badarinath KVS (2003) Aerosol characteristics and radiative forcing over industrial areas of urban environment—a case study from Hyderabad and its environs. IGU J Indian Geophys Union 7(1):25–29

Le Treut H, Somerville R, Cubasch U, Ding Y, Mauritzen C, Mokssit A, Peterson T, Prather M (2007) Historical overview of climate change. In: Solomon S, Qin D, Manning M, Chen Z, Marquis M, Averyt KB, Tignor M, Miller HL (eds) Climate change 2007: The physical science basis. contribution of working group I to the Fourth assessment report of the intergovernmental panel on climate change. Cambridge University Press, Cambridge

Loganathan D, Kamatchiammal S, Ramanibai R, Jayakar Santhosh D, Saroja V, Indumathi S (2011) Status of groundwater at Chennai city, India. Indian J Sci Technol 4(5):566–572

Lu Y, Huang Y, Zou J, Zheng X (2006) An inventory of N_2O emissions from agriculture in China using precipitation-rectified emission factor and background emission. Chemosphere 65(11):1915–1924

Machida T (1995) Increase in the atmospheric nitrous oxide concentration during the last 250 years. Geophys Res Lett 22:2921–2924

Majumdar D, Kumar S, Pathak H, Jain MC, Kumar U (2000) Reducing nitrous oxide emission from rice field with nitrification inhibitors. Agric Ecosyst Environ 81(2000):163–169

Matthew E (1999) Greenhouse warming potential model. Based J Chem Educ 76:1702–1705

MCI (2008) Motor Coach Industries (MCi's) go green glossary, viewed 15th Sept 2008. http://www.mcicoach.com/gogreen/greenGlossary.htm

McKinnon AC, Piecyk MI (2010) Logistics 2050: moving freight by road in a very low carbon world. Logistics Research Centre, Heriot-Watt University, Edinburgh

McMichael AJ, Woodruff R, Hales S (2006) Climate change and human health: present and future risks. The Lancet 367(9513):859–869

MFL (2010), Madras Fertilizers Limited, 2009–10

MiEF (2004) India's Initial National Communication to the United Nations Framework Convention on Climate Change. Ministry of Environment and Forests, Government of India, New Delhi, pp 38–41

Mitra AP (1992) Greenhouse gas emission in India: 1991 methane campaign, science report no. 2. Council of Scientific and Industrial Research and Ministry of Environment and Forests, New Delhi

Mitra AP, Bhattacharya S (2002) Climate change and greenhouse gas inventories: projections, impacts and mitigation strategies. In: Shukla PR, Sharma SK, Ramana PV (eds) Climate change and India: issues, concerns and opportunities. Tata McGraw-Hill Publishing Company Limited, New Delhi

Mitra S, Jain MC, Kumar S, Bandyopadhyay SK, Kalra N (1999a) Effect of rice cultivars on methane emission. Agric Ecosyst Environ 73(3):177–183

Mitra S, Jain MC, Kumar S, Bandyopadhyay SK, Kalra N (1999b) Effect of rice cultivars on methane emission. Agric Ecosyst Environ 73(3):177–183

Mittal ML, Sharma C (2003) Anthropogenic emissions from energy activities in India: generation and source characterization (part II: emissions from vehicular transport in India). http://www.osc.edu/research/pcrm/emissions/India_Report_1Pagelayout.pdf

Mitra AP, Sharma C (2002) Indian aerosols: present status. Chemosphere 49(9):1175–1190

MMRDA (2008) Mumbai Urban Infrastructure Project—Mumbai Metropolitan Region Development Authority

MOA (2003) Livestock Census 1997 and 2003

MOA (2005) 17th Indian Livestock Census,—All India Summary Report, Department of Animal Husbandry and Dairing, Ministry of Agriculture, Government of India

MOA (2007) 18th Indian Livestock Census,—All India Summary Report. Department of Animal Husbandry and Dairying, Ministry of Agriculture, Government of India

MOA (2008) Agricultural Statistics at a Glance (2008) Directorate of Economics and Statistics, Department of Agriculture and Cooperation (DAC), Ministry of Agriculture, Government of India

Moomaw WR (1996) Industrial emissions of greenhouse gases. Energy Policy 24(10–11):951–968

MoPNG (2010) Statistics on Petroleum and Natural Gas, 2009–10

MoUD (2009) Area, population and density of cities and towns of India—2001, area and density—metropolitan cities, town and country planning organization. Ministry of Urban Development, Government of India

Murthy NS, Panda M et al (1997a) Economic development, poverty reduction and carbon emissions in India. Energy Econ 19(3):327–354

Murthy NS, Panda M et al (1997b) Economic growth, energy demand and carbon dioxide emissions in India: 1990–2020. Environ Dev Econ 2(2):173–193

Naqvi SMK, Sejian V (2011) Global climate change: role of livestock. Asian J Agric Sci 3(1):19–25

Narasimha R, Sant G, Rajan SC (2009) An overview of Indian energy trends: low carbon growth and development challenges. Prayas Energy Group, India

NATCOM (2004) India's Initial National Communication to the UNFCC. Ministry of Environment and Forests, Government of India

NEERI (2005) Assessment of status of municipal solid waste management in metro cities, state capitals, class—I cities and class—II towns. NEERI, Nagpur

Neftel A (1982) Ice core sample measurements give atmospheric CO_2 content during the Past 40,000 Yr. Nature 295:220–223

Neftel A, Moor E, Oeschger H, Stauffer B (1985) Evidence from polar ice cores for the increase in atmospheric CO_2 in the past 2 centuries. Nature 315:45–47

Nelson GC, Richard R, Siwa M, Tingju Z, Xiaoli L, Puja J (2009) Greenhouse gas mitigation: issues for indian agriculture. International Food Policy Research Institute

Nicholas S (2006) Stern review on the economics of climate change. UK Treasury

North GR, Kim K-Y (1995) Detection of forced climate signals part II: simulation results. J Clim 6:409–417

Pachauri S (2004) An analysis of cross-sectional variations in total household energy requirements in India using micro survey data. Energy Policy 32(15):1723–1735

Pachauri S, Spreng D (2002) Direct and indirect energy requirements of households in India. Energy Policy 30(6):511–523

Pachauri RK, Sridharan PV (1998) In: Pachauri RK, Batra RK (eds) Directions innovations and strategies for harnessing action for sustainable development. TERI, New Delhi

Pandey D, Agarwal M, Pandey JS (2011) Carbon footprint: current methods of estimation. Environ Monit Assess 178:135–160

Parashar DC, Rai J, Gupta PK, Singh N (1991) Parameters affecting methane emission from paddy fields. Indian J Radio Space Phys 20:12–17

Parashar DC, Gupta PK, Rai J, Sharma RC, Singh N (1993) Effect of soil temperature on methane emission from paddy fields. Chemosphere 26(1–4):247–250

Parashar DC, Mitra AP, Sinha SK, Gupta PK, Rai J, Sharma RC, Singh N, Kaul S, Lal G, Chaudhary A, Ray HS, Das SN, Parida KM, Rao SB, Kanung SP, Ramasami T, Nair BU, Swamy M, Gupta SK, Singh AR, Saikia BK, Barua AKS, Pathak MG, Iyer CPS, Gopalakrishnan M, Sane PV, Singh SN, Banerjee R, Sethunathan N, Adhya TK, Rao VR, Palit P, Saha AK, Purkait NN, Chaturvedi GS, Sen SP, Sen M, Sarkar B, Banik A, Subbaraya BH, Lal S, Venkatramani S (1994) Methane budget from Indian paddy fields. In: Minami K, Mosier A, Sass RL (eds) CH_4 and N_2O: global emissions and controls from rice fields and other agricultural and industrial sources. Tsukuba, Japan: NIAES Series 2. pp 27–39

Parikh KJ (2001) Poverty-environment Nexus. Int J Glob Environ Issues 1(1):2001

Parikh KJ, Manoj KP, Murthy NS (1997) Consumption patterns by income groups and carbon dioxide implications for India: 1990–2010. Int J Energy Issues 9:237–255

POST (2006) Carbon footprint of electricity generation. POST note 268, October 2006, Parliamentary Office of Science and Technology, London. http://www.parliament.uk/documents/upload/postpn268.pdf

Qader MR (2009) Electricity consumption and GHG emissions in GCC countries. Energies 2009(2):1201–1213

Raghuvansi SP, Chandra A, Raghav AK (2006) Carbon dioxide emissions from coal based power generation in India. Energy Convers Manag 47(4):427–441

Ramachandra TV, Bachamanda S (2007) Environmental audit of municipal solid waste management. Int J Environ Technol Manage 7(3/4):369–391

Ramachandra T, Shwetmala V (2009) Emissions from India's transport sector: state wise synthesis, atmospheric environment. doi:10.1016/j.atmosenv.2009.07.015

Ramachandra T, Shwetmala V (2012) Decentralised carbon footprint analysis for opting climate change mitigation strategies in India. Renew Sustain Energy Rev 16(8):5820–5833

Ramachandra T, Uttam Kumar V (2010) Greater Bangalore: emerging urban heat, Island. GIS Dev 14(1). (http://www.gisdevelopment.net/application/urban/sprawl/Greater-Bangalore-Emerging-Urban-Heat-Island.htm)
Ramachandra TV, Aithal BH, Sanna DD (2012) Insights to urban dynamics through landscape spatial pattern analysis. Int J Appl Earth Obs Geoinf 18:329–343
Ramanathan V (1975) Greenhouse effect due to chlorofluorocarbons: Climatic implications. Science 190:50–52
Ramanathan R, Parikh JK (1999) Transport sector in India: an analysis in the context of sustainable development. Transp Policy 6(1):35–45
Rao MN, Reddy BS (2007) Variations in energy use by Indian households: an analysis of micro level data. Energy 32(2):143–152
Rao JK, Shantaram MV (2003) Soil and water pollution due to open landfills. In: Proceedings of the sustainable landfill management workshop. 3–5 Dec 2003 Anna University, Chennai, pp 27–38
Ravindra K, Eric W, Sushil KT, Suman M, van Grieken R (2006) Assessment of air quality after the implementation of CNG as fuel in public transport in Delhi, India. Environ Monit Assess 115(1–3):405–417
Rawat Manju A, Ramanathan L (2011) Assessment of methane flux from municipal solid waste (MSW) landfill areas of Delhi, India. J Environ Prot 2:399–407
RCF (2010) 32nd Annual Report, Rashtriya Chemical and Fertilizers Limited, 2009–10
Reddy BS, Srinivas T (2009) Energy use in Indian household sector-an actor oriented approach. Energy 34(8):992–1002
Reddy MS, Boucher O, Venkataraman C (2002) Seasonal carbonaceous aerosol emissions from open biomass burning in India. Bull IASTA 14:239–243
Road Transport Year Book (2007–2009) Transport Research Wing, Ministry of Road Transport and Highways, Government of India, 2011
Sass RL, Fisher FM Jr (1998) Methane from irrigated rice cultivation. In: Parashar DC, Sharma C, Mitra AP (eds) Global environment chemistry. pp 77–94
Schlesinger ME, Ramankutty N (1994) An oscillation in the global climate system of period 65–70 years. Nature 360:330–333
Schumacher K, Sathaye J (1999) India's cement industry: productivity, energy efficiency and carbon emissions, energy analysis program. Lawrence Berkeley National Laboratory, Berkely
Schutz H, Holzapfel-Pschorn A, Conrad R, Rennenberg H, Seiler W (1989) A three-year continuous record on the influence of daytime season and fertilizer treatment on methane emission rates from an Italian rice paddy field. J Geophys Res 94:16405–16416
Seiler W, Holzapfel-Pschorn A, Conrad R, Scharfe D (1984) Methane emission from rice paddies. J Atmos Chem 1:241–268
Shafik N, Bandyopadhyay S (1992) Economic growth and environmental quality: time series and cross-country evidence. In: Policy research working paper series 904, The World Bank
Sharma C, Pundir R (2008) Inventory of greenhouse gases and other pollutants from the transport sector: Delhi. Iran J Environ Health Sci Eng 5(2):117–124
Sharma C, Gupta PK, Parashar DC (1995) Nitrous oxide estimates from paddy fields and forests in India. Indian J Radio Space Phys 24:311–313
Sharma SK, Bhattacharya S, Garg A (2009) Greenhouse gas emissions from India: a perspective. Curr Sci 90:326–333
Sharma SK, Asim C, Pinaki S, Subhashis B, Anil S, Pradeep KD, Ajay KS, Suman M, Arti B, Madhu M, Rajesh K, Jha CS, Murthy MSR, Ravindranath NH, Jayant KB, Karthik M, Bhattacharya S, Chauhan R (2011) Greenhouse gas inventory estimates for India. Curr Sci 101:405–415
Singh A, Gangopadhyay S, Nanda PK, Bhattacharya S, Sharma C, Bhan C (2008) Trends of greenhouse gas emissions from the road transport sector in India. Sci Total Environ 390:124–131
Singhal KK, Mohini M, Jha AK, Gupta PK (2005) Methane emission estimates from enteric fermentation in Indian livestock: Dry matter intake approach. Curr Sci 88(1):119–127

Sinha SK (1995) Global methane emission from rice paddies: excellent methodology but poor extrapolation. Curr Sci 68(6):643–646

Sikdar PK, Singh A (2009) Transportation sector: energy and GHG emissions. In: Proceedings of the seminar on energy and GHG emissions. Seminar on energy for sustainable infrastructure development and lifestyle. Consulting engineers india, New Delhi, 23–24 February 2009

SOE—State of Environment Report for Delhi (2010) Department of Environment and Forests, Government of Delhi

Steele LP (1996) Atmospheric methane, carbon dioxide, carbon monoxide, hydrogen, and nitrous oxide from Cape Grim air samples analyzed by gas chromatography. In: Francey RJ, Dick AL, Derek N (eds) Baseline atmospheric program Australia, 1994–95. Bureau of Meteorology and CSIRO Division of Atmospheric Research, Melbourne, pp 107–110

Stern D, Common M, Barbier E (1996) Economic growth and environmental degradation: the environmental kuznets curve and sustainable development. World Dev 24:1151–1160

Swamy M, Bhattacharya S (2006) Budgeting anthropogenic greenhouse gas emission from Indian livestock using country-specific emission coefficients. Curr Sci 91(10):1340–1353

Talyan V, Dahiya RP, Anand S, Sreekrishnan TR (2007) Quantification of methane emission from municipal solid waste disposal in Delhi. Resour Conserv Recycl 50(3):240–259

TEDDY (2006, 2011) TERI Energy Data Directory and Yearbook. The Energy and Resource Institute, New Delhi

Vijay B, Mehta VM (2010) Atmospheric particulate pollutants and their relationship with meteorology in Ahmedabad. Aerosol Air Qual Res 10:301–315

Wakdikar S (2002) Compressed natural gas: a problem or a solution? Curr Sci 82(1):25–29

Weiss RF (1981) The temporal and spatial distribution of tropospheric nitrous oxide. J Geophys Res 86:7185–7195

Weisser D (2007) A guide to life-cycle greenhouse gas (GHG) emissions from electric supply technologies. Energy 32(9):1543–1559

World Bank http://worldbank.org. Accessed 20th Aug 2013

Wright LA, Coello J, Kemp S, Williams I (2011) Carbon footprinting for climate change management in cities. Carbon Manage 2(1):49–60

Yamulki S, Jarvis SC, Owen P (1999) Methane emission and uptake from soils as influenced by excreta deposition from grazing animals. J Environ Qual 28:676–682

The Use of Carbon Footprint in the Wine Sector: Methodological Assumptions

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Abstract Wine production is an important economic sector in many countries worldwide. In addition, its sales and consumption are steadily augmenting on an annual basis. This has increased the interest by stakeholders and consumers in the environmental sustainability of wine production practices. Despite the wide range of environmental dimensions that are monitored through environmental management tools, worldwide concerns related to greenhouse gas emissions and their effect on global warming have boosted the analysis of a single score indicator to monitor these emissions: carbon footprint (CF). In fact, due to the important consequences that climate change is expected to have on wine appellations and regions, CF has proliferated in this sector in recent years. The aim of this study is to provide a critical review on the application of CF to the wine sector based on peer-reviewed publications, focusing on the controversial methodological assumptions and the level of granularity of the life cycle inventory. Finally, a series of potential advancements in the application of CF to the wine sector will be assessed and discussed.

Keywords LCA · Life Cycle Inventory · Vinification · Viticulture · Wine production

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S. S. Muthu (ed.), *Assessment of Carbon Footprint in Different Industrial Sectors, Volume 2*, EcoProduction, DOI: 10.1007/978-981-4585-75-0_9,

1 Introduction

The production of wine is considered to be one of the most ancient forms of agriculture in human history. Evidence of viticulture has been found as far back as the Neolithic period (McGovern et al. 1996). Viticulture has historically developed throughout Europe. In fact, wines arriving from this geographical area are commonly referred to as "Old World" wines. In contrast are the "New World" wines—those arriving from relatively modern areas of production, such as the Southern Hemisphere (South Africa, New Zealand, Australia, Chile, Peru, or Argentina), the United States and Canada, and most of the Asian continent (e.g. China or Iran).

Old World vineyards represent just over 50 % of the global surface area used for viticulture activities, as shown in Fig. 1. Spain, France, and Italy (and, to a lesser extent, Portugal) are the countries in the Old World with a highest surface area linked to winemaking; in the New World, Turkey and the United States lead the rankings. Nevertheless, it should be noted that the relative weight of Old World wine on an international scale has gradually decreased in recent decades. In addition, global vineyards have experienced a slight decrease in surface area (OIV 2013).

In terms of production, 60 % of the wine produced worldwide still arrives from Old World appellations and vineyards. Nevertheless, in a similar way to what was described in terms of surface area, wine production has also experienced a dwindling tendency in the past few years (Fig. 2). Interestingly, these values are in opposition to wine consumption trends worldwide; the latter has shown a substantial increase in the past decade (OIV 2013). The reason behind these opposing trends is linked to the optimization of production stocks, which are increasingly exported to meet demand in other areas (see Fig. 3).

Operations linked to the viticulture phase are highly variable, depending on a wide range of issues such as climatic conditions, soil characteristics, and altitude. In addition, the current tendency throughout wine regions to homogenize viticulture and vinification operations within one single appellation has led to the appearance of a series of common standards with which winemakers must comply. Nevertheless, despite this homogenization in terms of operational inputs, it has become common to see viticulture practice divided according to the operations related to plant protection and fertilization. Hence, many studies in the field of viticulture distinguish between conventional wine production and organic wine production (Gabzdylova et al. 2009).

The main characteristic of conventional viticulture, when compared with organic viticulture, is the fact that there are no legal restrictions regarding the use of fertilizing and plant protection agents (European Commission 2007, 2008, 2012). Another important characteristic of conventional viticulture systems is the use of machinery for most operational activities. Finally, despite advocating for certain quality standards, which are usually regulated by the appellations that manage specific wine types or areas, conventional viticulture prioritizes obtaining high yield rates.

Organic viticulture does not use mineral fertilizers on vineyards; it also strictly limits the synthetic substances that may be used as plant protection agents. Within

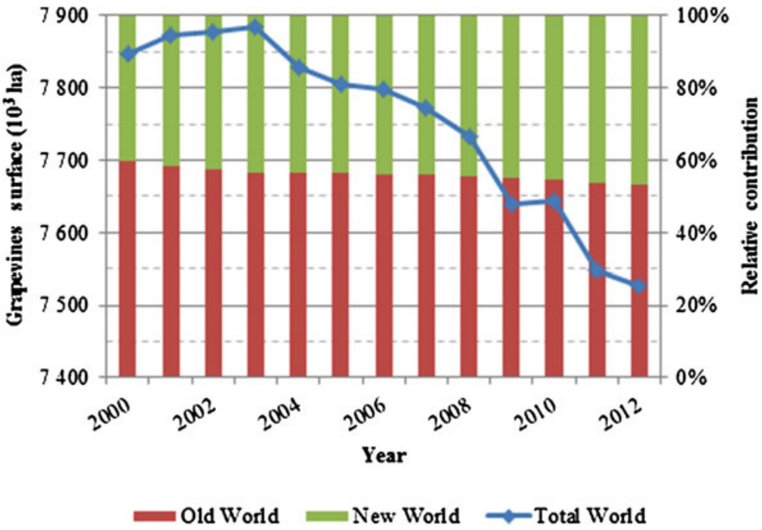

Fig. 1 Worldwide distribution of vineyards by geographical location. Adapted from OIV (2013)

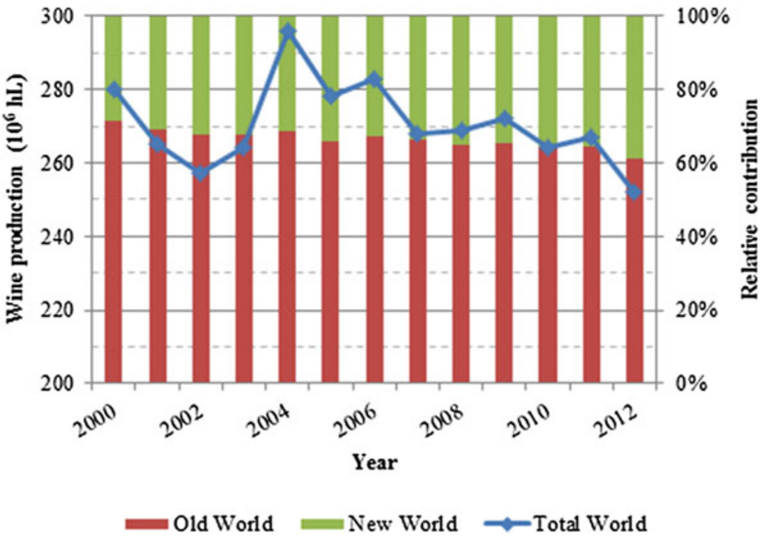

Fig. 2 Worldwide distribution of wine production by geographical location. Adapted from OIV (2013)

organic wine, an interesting subcategory is biodynamic viticulture (Villanueva-Rey et al. 2013). The latter is even more restrictive than regular organic wine sites, seeking a complete harmonization of the vineyards with their surrounding eco-systems and using a series of biodynamic preparations to treat the vines (Lotter

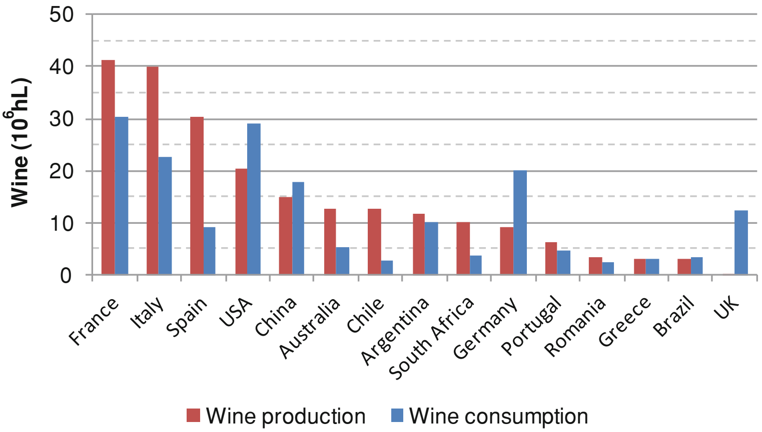

Fig. 3 Absolute production and consumption of wine in a selection of countries. Adapted from OIV (2013)

2003; Masson 2009). These cultivation sites, unlike conventional viticulture, are aimed at prioritizing grape and wine quality, as well as seeking an environmentally friendly approach to winemaking, rather than enhancing productivity. Organic and biodynamic viticulture are currently experiencing a strong proliferation, with many new and old wine farms promoting a change in operational activities (Gabzdylova et al. 2009). In fact, many stakeholders see in this transition an opportunity to improve their sales and attain a better position in the wine market.

Nevertheless, it is often argued that the shift to organic or biodynamic viticulture practices does not guarantee a higher environmental sustainability of wine products; these practices only focus on the products that are applied to the vineyards for vine protection or fertilizing, rather than applying a more integrated concept of sustainability aimed at reducing operational inputs throughout the supply chain (Venkat 2012). In this context, a series of environmental management tools have arisen in the last few decades with the objective of providing an integrated environmental assessment of products, processes, or services. Life Cycle Assessment (LCA) was the first of these methods to be internationally regulated through the ISO standards (ISO 2006a, b). To date, this methodology has been applied to a considerable amount of wine processes, including wine farms, appellations, viticulture practices, and supply chains (Rugani et al. 2013). Despite the relative importance of plant protection agents and fertilizers in the viticulture stage of winemaking, it has been shown that many other activities throughout the production and supply chain lead to important environmental burdens.

Although LCA studies provide environmental information for a cluster of environmental dimensions, referred to as impact categories, in many cases some of these categories are analyzed individually due to the particular interest that they may generate in the production system under study (Udo de Haes 2006; Weidema et al. 2008). Moreover, current worldwide environmental concerns, such as water scarcity and global warming, have created increasing interest in using assessment methods that address these impacts specifically. Consequently, it is common to see life-cycle studies using single indicators, such as carbon footprint (CF), to measure emissions linked to climate change or water footprint to monitor the stress of water supply due to anthropogenic activities (Ridoutt et al. 2009; Weidema et al. 2008).

More specifically, CF has experienced an exponential proliferation in recent years due to the international concerns regarding climate change, as well as other derived factors, such as consumer awareness or the willingness of stakeholders to enhance product transparency and market quota through CF-oriented campaigns and certifications (Weidema et al. 2008; Laurent et al. 2012). In fact, most studies on CF in the wine sector have focused on allotting anthropogenic greenhouse gas (GHG) emissions to the wine production and supply chain operations for the identification of hot spots throughout the life-cycle of wine products or for communication purposes, mainly oriented towards stakeholders and consumers (Aranda et al. 2005; Benedetto 2013; Rugani et al. 2013).

The aim of this chapter is to provide an in-depth analysis of the main methodological assumptions that should be taken into account when applying the CF method to the wine sector. More specifically, this chapter provides a step-by-step description of the CF single-score methodology as applied to wine based on literature reviews. Special attention will be paid to controversial issues in life-cycle thinking, such as the fixation of the system boundaries, the definition of the goal and scope, functional unit (FU) choice, allocation, and the selection of different assessment methods. In addition, a detailed description of Life Cycle Inventory (LCI) items is provided, analyzing the pertinence of including them in CF studies. Finally, a set of understudied potentials of life cycle thinking as applied to the wine sector are analyzed and discussed.

Section 2 presents the main methods used to perform a CF assessment of viticulture and vinification products. Section 3 provides an analysis of the main wine CF results available to date in the literature. Section 4 delves into the main challenges that CF studies face in order to increase their impact in the scientific community and improve their utility for stakeholders and in decision-making processes. Finally, Sect. 5 wraps up with the main conclusions of the chapter.

2 Methods

2.1 Environmental Management Tools: Holistic or Single Score?

2.1.1 Life Cycle Assessment in Viticulture and Vinification

LCA is an internationally standardized environmental management tool that has been repeatedly used to study the environmental profile of products and services in recent decades (ISO 2006a, b). From a viticulture and vinification perspective, its

Table 1 List of peer-reviewed publications addressing case studies and methodological issues in the CF of wine

Study	Country	Viticulture management	Grapes/wine type	Methodology applied
Notarnicola et al. (2003)	Italy	Conventional	Unspecific	LCA
Aranda et al. (2005)	Spain	Conventional	Unspecific	LCA
Ardente et al. (2006)	Italy	Conventional	Unspecific	POEMS/LCA
Gonzales et al. (2006)	France	Conventional and organic	Red wine	LCA
Niccolucci et al. (2008)	Italy	Conventional and organic	Red wine	Ecological footprint
Pizzigallo et al. (2008)	Italy	Conventional and organic	Red wine	LCA and emergy
Carballo-Penela et al. (2009)	Spain	Conventional	Unspecific	Carbon footprint
Kavargiris et al. (2009)	Greece	Conventional and organic	Pink wine	Greenhouse gases and energy
Gazulla et al. (2010)	Spain	Conventional	Red wine	LCA
Notarnicola et al. (2010)	Italy	Conventional	Red wine	LCA
Barry (2011)	New Zealand	Conventional	Unspecific	LCA
Bosco et al. (2011)	Italy	Conventional	Red and white wine	Carbon footprint
Comandaru et al. (2012)	Romania	Conventional	Not specified	LCA and water footprint
Pattara et al. (2012)	Italy	Organic	Red wine	Carbon footprint
Point et al. (2012)	Canada	Conventional	White and red wine	LCA
Vázquez-Rowe et al. (2012a)	Spain	Conventional	White wine	LCA
Vázquez-Rowe et al. (2012b)	Spain	Conventional	White grapes	LCA and DEA
Vázquez-Rowe et al. (2013a)	Spain, Italy, and Luxembourg	Conventional and organic	White and red grapes	LCA
Benedetto (2013)	Italy	Conventional	White wine	LCA
Neto et al. (2013)	Portugal	Conventional	White wine	LCA
Villanueva-Rey et al. (2013)	Spain	Conventional and biodynamic	White grapes	LCA

LCA: Life cycle assessment; *DEA:* Data Envelopment Analysis

application is more recent, with the first studies dating from 2003. Nevertheless, LCA has shown an important proliferation, as depicted in Table 1.

The geographical distribution of these studies is generally in Old World regions, mainly Italy (Niccolucci et al. 2008; Pattara et al. 2012; Benedetto 2013), Spain (Vázquez-Rowe et al. 2012a, b; Villanueva-Rey et al. 2013), Luxembourg (Vázquez-Rowe et al. 2013a), and Portugal (Neto et al. 2013). Some studies have also arisen in New World areas, such as Canada (Point et al. 2012) and New Zealand (Barry 2011). Nevertheless, it should be noted that most of the studies focus on single wine appellations (Point et al. 2012; Neto et al. 2013), individual wineries (Vázquez-Rowe et al. 2012a; Benedetto 2013), or a group of viticulture sites (Vázquez-Rowe et al. 2012b). Only a few studies have attempted to provide specific characterization values beyond the appellation level. Vázquez-Rowe et al. (2013a), who provided a CF calculation on a national level for Luxembourg, made a cross-country comparison between Luxembourg, Italy, and Spain. In addition, a review on wine CF attempted to obtain a worldwide average value for GHG emissions linked to the consumption of a bottle of wine, by aggregating the CF results available in the literature (Rugani et al. 2013). Moreover, these results were scaled up to the entire worldwide production and consumption of wine worldwide, concluding that the wine industry represents approximately 0.3 % of total GHG emissions (Rugani et al. 2013).

Regarding the different viticulture practices, most available studies have focused on conventional practices (Notarnicola et al. 2003; Carballo-Penela et al. 2009; Bosco et al. 2011; Vázquez-Rowe et al. 2012b) or on the comparison between organic and conventional farms (Pizzigallo et al. 2008; Kavargiris et al. 2009; Vázquez-Rowe et al. 2013a). Furthermore, a recent study by Villanueva-Rey et al. (2013) analyzed the environmental impacts associated with three different viticulture practices in the same appellation, including a biodynamic cultivation site, in order to understand the trade-offs in terms of critical impact categories depending on the selected practices.

Finally, a wide range of existing wine types have been examined using either LCA or CF studies, although some contributions do not state specifically what wine type was analyzed. Nevertheless, most studies have focused on analyzing either red or white wines (Ardente et al. 2006; Benedetto 2013; Villanueva-Rey et al. 2013) of medium price range. However, Vázquez-Rowe et al. (2013a) also examined sparkling wine and an expensive red wine with long aging periods, and Neto et al. (2013) analyzed the environmental impacts related to Portugal's *vinho verde*.

2.1.2 Carbon Footprint

Carbon footprint (CF) analysis in viticulture and vinification has been mainly related to the extraction of the global warming potential (GWP) impact category from the CML baseline 2000 LCA assessment method (Frischknecht et al. 2007) or using the Intergovernmental Panel on Climate Change (IPCC) assessment method (IPCC 2007). Moreover, to date, most studies have developed CF calculations within the framework of the ISO 14040 standard (ISO 2006a, b). Nevertheless, in recent years, a considerable number of CF protocols have arisen, including PAS 2050:2011 (BSI 2011), the Greenhouse Gas Protocol (Initiative GGP 2011), Bilan Carbone (ADEME 2010), and the specific ISO standard for carbon footprinting, ISO 14067 (ISO 2013). Moreover, the OIV has developed its own standards (OIV 2011).

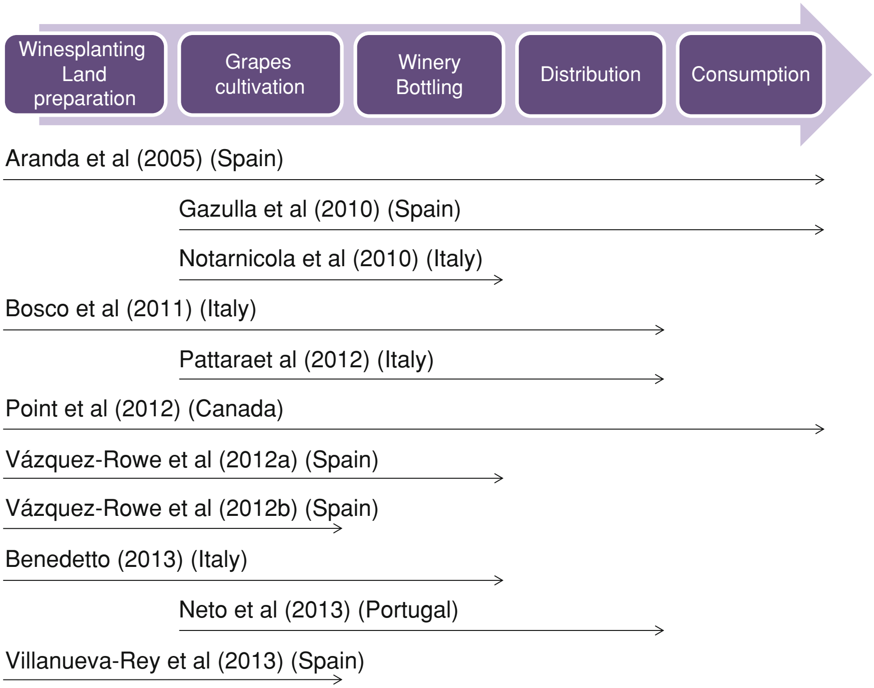

Fig. 4 Scope of the study in selected wine CF studies

2.2 Scope of Wine LCA and CF Studies

The scope of existing studies is highly variable. In most cases, as can be observed in Fig. 4, studies have focused overwhelmingly on the viticulture and vinification stages of the supply chain—despite the fact, as stated by Rugani et al. (2013), that these two stages are not necessarily the one's that contribute the most to the total environmental impact. The vineyard planting phase, previous to the viticulture stage, is not always included in these studies. In fact, a general observation throughout the available literature is that the initial phases of land preparation, vine nursing, and vine planting are treated with care or directly through omission on most studies, suggesting that data availability in these stages of the process are scarce.

Regarding the perspective undertaken in these studies, an attributional approach was used in all of them (Rugani et al. 2013). Attributional approaches in life-cycle studies take into consideration the environmental impacts (in the case of CF, GHG emissions exclusively) in a steady-state condition of the system under assessment, without considering the interaction that may occur with other interrelated production systems. In contrast to the attributional perspective, the more recent consequential (prospective) approach aims to measure the environmental consequences linked to changes in the production system, rather than monitoring the direct emissions, based on a series of market-, temporal-, or spatial-driven constraints and/or changes, among other potential shocks on the system under study (UNEP 2011; Vázquez-Rowe et al. 2013b, c).

2.3 System Boundaries

The establishment of temporal boundaries for grape production, given its annual characteristics, is different than for seasonal crops. Figure 5 shows a schematic representation of the system boundaries for the viticulture, vinification, and bottling phases of wine production, up to the gate of the farm ready for transportation, distribution, and consumption, including the most relevant operational activities that are undertaken throughout the production chain.

Certain elements within these boundaries, such as the infrastructure in the vinification stage, are repeatedly disregarded in most studies (except in the case of Gazulla et al. 2010 and Vázquez-Rowe et al. 2013a). A similar situation occurs with the vine trellis, although recent publications are increasingly inclined to include this element (Vázquez-Rowe et al. 2012a, b) because the wide variety of materials used for this type of infrastructure (e.g. wood, concrete, granite) have been shown to imply important differences in GHG emissions and also in other environmental aspects, such as land use impacts (Villanueva-Rey et al. 2013).

Machinery throughout these stages is also usually excluded when it comes to the production and transportation of the machinery itself, but included in terms of the use phase (e.g., diesel or electric consumption). Finally, vine nursing, land preparation, and vine planting are phases prior to the viticulture stage that are not always included within the system boundaries. For instance, no studies in the literature report an LCI for vine nursing. In the case of land preparation and vine planting, however, there is a tendency in the literature to include these phase if the vineyards have been planted recently, while excluded them from old cultivation sites (Villanueva-Rey et al. 2013).

To sum up, all elements disregarded from the system boundaries seem to have two common denominators: (i) it is difficult to obtain feasible data for these processes, as in the case of vine nursing (Vázquez-Rowe et al. 2012a); (ii) the scope and aims of the study allowed the exclusion of these stages (Neto et al. 2013; Rugani et al. 2013). Nevertheless, it should be noted that cut-off criteria fixed by the ISO standards for LCA are not usually followed in most CF studies because the criteria are mainly linked to data availability, rather than after a detailed inventory collection phase (ISO 2006b).

2.4 Function and Functional Unit

The functions of most wine LCAs and CFs reported to date are basically oriented towards the environmental certification of the analyzed product (i.e. the bottle of

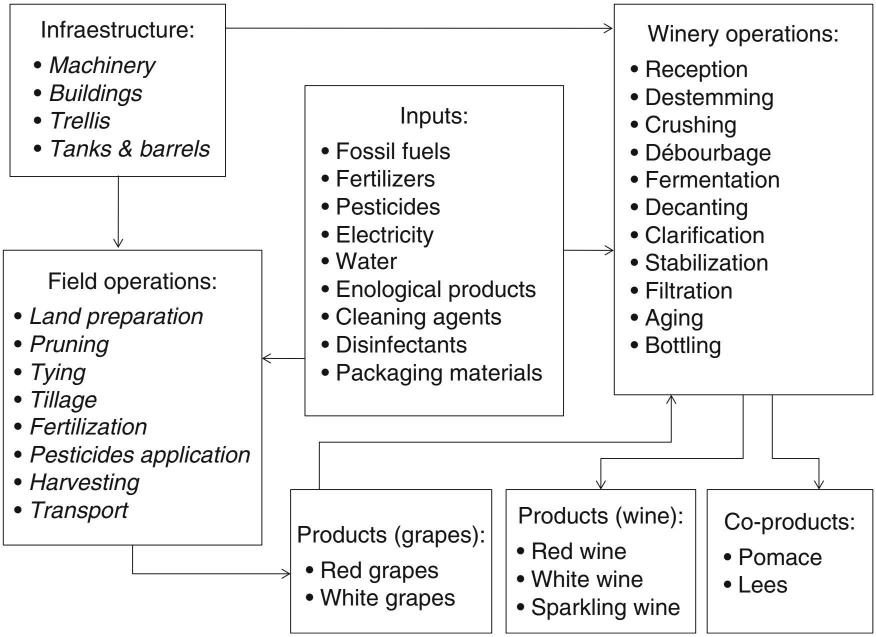

Fig. 5 Schematic representation of a typical wine production system. Operational inputs or processes that are commonly excluded from the inventories in wine CF studies are indicated by *italics*

wine) or for an in-depth analysis of the main hot spots and, thereafter, improvement actions throughout the supply chain. Therefore, an overwhelming majority of the analyzed studies use the same FU to report the environmental profile of wine products: one standard wine package (usually a 750-mL glass) that is the main format used for sales (Aranda et al. 2005; Gazulla et al. 2010; Petti et al. 2010; Bosco et al. 2011; Point et al. 2012; Vázquez-Rowe et al. 2012a). Other studies use different volumes of wine (Gonzales et al. 2006), whereas Notarnicola et al. (2010) used the percentage in alcohol volume and the hedonistic value of wine. Nevertheless, it should be noted that many studies only focused on the viticulture stage (Vázquez-Rowe et al. 2012b; Villanueva-Rey et al. 2013). In these particular studies, however, the FU selected was the amount of grapes needed to produce 750 mL of wine, rather than the selection of other options, such as cultivated surface.

After the analysis of the different functions and FUs available in the literature, it appears that the utility of wine CF and LCA has been limited to communication of results based on the final marketable product: wine bottles (Rugani et al. 2013). Therefore, it seems as if integral assessments of wine farms, appellations, and wine regions are still missing in the available literature, constituting an interesting field for future studies.

2.5 Importance of the Life Cycle Inventory stage in Wine CF

Most of the studies represented in Table 1 base their LCI structure on the guidelines provided by ISO 14040 and 14044 (ISO 2006a, b). In this section, however, a detailed review on the different specific inventory items that have been provided in the literature is examined, as well as a discussion on how the overall depth of the inventory could be improved through, for example, the inclusion of understudied operational items.

2.5.1 Operational Inputs in Grape Production

- *Fossil fuels and related emissions.* Diesel has been highlighted as the main fossil fuel used in the viticulture phase of winemaking. In fact, its production and combustion account for an important fraction of the GHG emissions in this stage of the production chain (Pattara et al. 2012; Vázquez-Rowe et al. 2012a; Villanueva-Rey et al. 2013). Furthermore, it should be noted that it is necessary to account for the specific activities in which diesel is being used, in order to be able to tackle feasible and realistic improvement actions to reduce environmental burdens while increasing efficiency. In fact, a recent study conducted in the *Rías Baixas* appellation in Northwestern Spain, which focused exclusively on the pre-vinification stages of wine production (but excluded vine nursing), showed that 60 % of diesel consumption in this appellation is linked to the use of plant protection agents. Nevertheless, it is important to bear in mind that Atlantic wines, in most cases, need a higher dose of plant protection agents in the viticulture stage than Mediterranean wines or other wines in areas with dryer climates, increasing the interventions and, therefore, diesel consumption and costs. Finally, it should be noted that a majority of the revised literature uses the EMEP-Corinair guideline (EMEP-Corinair 2006) or its latest updates (EMEP-Corinair 2009) to estimate the different emissions from agricultural machinery based on the provided emission factors (Vázquez-Rowe et al. 2012a; Villanueva-Rey et al. 2013).
- *Mineral fertilizers and associated emissions.* These are usually enrichers that have been produced synthetically to provide a specific balance of primary nutrients (i.e. NPK), secondary nutrients (e.g. calcium, magnesium, or sulfur), and oligoelements (e.g. manganese, iron, zinc, copper, etc.). In addition, in many cases they are used to correct certain deficiencies in the physico-chemical characteristics of soils. For instance, this may be the case in granitic areas in which the pH of the soil must be corrected to avoid toxicity in the roots due to aluminum (Roux et al. 1988) or in cases where there is a competence between bases and, therefore, a correction of nutrient deficits is needed. The inclusion of these fertilizing agents has been shown to have an important impact in terms of GHG emissions, not only due to the high energy intensity processes undertaken in the production stage, as well as the transport of the goods to the wine farms,

but also due to field emissions when applied on the vineyards. For the latter emissions, the methods that have been most widely used to estimate N_2O emissions are those suggested by Brentrup et al. (2000) and the IPCC (2006). Carbon dioxide emissions from the liming process are mainly calculated through the IPCC methodology. Finally, it should be noted that fertilizers based on urea should also consider the specific GHG emissions linked to this particular fertilizing scheme (IPCC 2006).

- *Organic fertilizers and associated emissions.* The main objective of using these types of fertilizers is linked to the improvement of certain physio-chemical characteristics of the soil, such as texture, structure, and cationic exchange capacity. In the case of organic and biodynamic viticulture, except in very specific situations, vine growers are obliged to use these products as the only way to nourish the soil with new nutrients (European Commission 2012). The origin of this type of fertilizer is usually very variable, such as compost or guano, or residues from animal, plant, or urban origin. In viticulture, pelletized compost from livestock origin is one of the predominant products used due to an easy spreading process. When these processes are included in the LCI of a CF study, it is important to identify their origin (agricultural by-product, residue, etc.), as well as the emissions linked to their maturation as a compost. However, given the difficulty of tracing back these processes, there is a tendency to omit this process from the system boundaries, including the emissions due to the spreading of compost exclusively (Villanueva-Rey et al. 2013). GHG field emissions are estimated based on the nitrogen content of the compost to obtain accurate N_2O emissions. The most commonly used methodologies to estimate these emissions, given the high costs linked to on-site measuring, are those proposed by Brentrup et al. (2000) and the IPCC (2006).
- *Infrastructure and trellis.* Trellis for vine support in vineyards can vary vastly between regions due to the variety of materials that can be used. In fact, traditionally the materials used in vineyard trellis are strongly related to the availability of natural resources in the specific wine region. Nevertheless, in recent decades, trellises have shown an important proliferation, with many being constructed with steel, granite, or concrete-based materials. In addition, a series of materials of plastic origin are used in a complementary manner in many vineyards to aid the growth of the vines. Therefore, from an LCI perspective, it is important to understand the combination of materials that make up the trellis in order to build adequate unit processes, as well as the reposition time for these elements. Nevertheless, it should be noted that the GHG emissions linked with trellis have been shown to be relatively low compared with other activities, operations, and units in the viticulture stage (Vázquez-Rowe et al. 2013a). However, some exceptions include cases such as biodynamic wine, in which the lower GHG emissions associated with other activities provides higher relative importance to the trellis system (Villanueva-Rey et al. 2013).
- *Electricity.* Direct electricity consumption in the viticulture stage is mainly related to machinery that does not use diesel or gasoline as the energy carrier.

This is usually the case for pruning machinery, certain plant protection agent operations, and water pumping.

- *Water.* Water consumption for irrigation, which is most commonly done in New World appellations or areas in Europe with water scarcity problems, has important impacts on the water footprint. However, in terms of GHG emissions and CF, its importance is limited to scenarios in which water canalization to the vineyards is linked to anthropogenic activities, such as water pumping in wells or other supply chains.
- *Machinery and machinery maintenance.* The machinery that is used in viticulture practices is in many cases associated with the use of a tractor, although this is sometimes limited by orographic characteristics in certain appellations and regions. Moreover, in recent years, the use of all-terrain vehicles is increasing. Although the inclusion of machinery in the LCIs of grape and wine production is not widely done in the existing literature, studies in which these elements were included followed the ecoinvent® v2.2 guidelines (Nemecek and Kägi 2007). Finally, regarding maintenance, the ecoinvent® recommendations are usually followed due to the difficulty in obtaining feasible primary data.
- *Plant protection agents* (*pesticides*). The use of pesticides in viticulture activities is done to control cryptogamic diseases and plagues that attack vines. The number of interventions can vary enormously between wine regions. For instance, due to their climatic characteristics, Atlantic regions along Western Europe need a higher number of interventions for cryptogamic diseases, such as downy or powdery mildew. The inclusion of pesticides in the LCI in CF studies is important from a production perspective. In fact, despite the importance of their direct emissions to water and air in terms of toxicity impacts, these emissions lack any relevance from a global warming perspective because they are not GHG emissions. The specific unit processes that should be included in the LCI when using the ecoinvent® database are limited to clusters in plant protection agent families, due to the difficulty to trace the production chain for all the active substances used in synthetic pesticides. In fact, Vázquez-Rowe et al. (2013a) discussed the high variability in terms of environmental impacts regarding GHG emissions if different methodological assumptions are considered. For instance, the use of the unit process *unspecified pesticides* in the foreground system of the LCI can render substantially different CF results (in most cases, higher GHG values; see Figure 1 in Vázquez-Rowe et al. 2013a) than using the specific unit processes for plant protection agent families. Another crucial point when classifying and inventorying plant protection agents is the computation of inorganic substances, such as copper- and sulfur-based compounds. These compounds usually have been included in LCIs based on specific processes for compounds of organic synthesis origin. However, Villanueva-Rey et al. (2013), when studying biodynamic viticulture practices in Northwestern Spain, highlighted the fact that these compounds become highly important in the overall contribution to the viticulture stage in biodynamic systems due to the low contributions in terms of fertilization, interventions using

diesel, or the use of other types of plant protection agents. Therefore, in an attempt to gain precision, Villanueva-Rey et al. (2013) included sulfur-based products as subproducts derived from refined oil processes, whereas copper-based products were modelled based on alternative copper compounds.

2.5.2 Vinification, Bottling, and Packaging Stages

- *Fossil fuels and related emissions.* Throughout the vinification stage, there are numerous activities and specific machineries that require the use of a wide range of fossil fuels (e.g., propane, gasoline, diesel, natural gas), including operations such as heating, electricity generation, water pumping, and the use of certain machinery. The emissions derived from these processes, in a similar way to those occurring in the viticulture stage, may be estimated based on the proposed emissions factor by EMEP-Corinair or IPCC (EMEP-Corinair 2006, 2009; IPCC 2006).
- *Electricity.* The consumption of electricity during this phase of the production chain will depend on the characteristics of the wine farm. The GHG emissions derived from this consumption are related to the main sources of energy of the wine farm: the electricity mix of the country where the farm is located will, in most cases, be the main or exclusive carrier of the consumption. ecoinvent® provides detailed unit and system processes to implement country electricity mixes in the LCI. Version 2.2 of the database provided detailed data for European countries and a selection of other countries worldwide for 2006, whereas version 3.0 includes 90 % of worldwide electricity production for more recent years (ecoinvent® 2013). Nevertheless, Vázquez-Rowe et al. (2012a), with the aim of providing individual CF values for different harvest years, adapted the unit processes of ecoinvent® to the actual years under study to obtain more accuracy in the final results. Finally, it is important to remark that many wine farms worldwide are advocating for a wider presence of renewable energy in their premises. This has led, for instance, to the construction of photovoltaic panels in many wine farms as an alternative source of electricity production.
- *Machinery and maintenance.* The use of different materials to maintain the quality of the wine-making process machinery is necessary to quantify in the LCI. Bottling machines, presses, barrels, etc., are all items that should be analyzed in detail when delving into the inventory items of these processes.
- *Packaging materials.* Most LCA and CF studies available in literature highlight the importance of glass bottling in the overall GHG emissions of wine production (Point et al. 2012; Vázquez-Rowe et al. 2012a). In fact, an important improvement action suggested in many studies is the substitution of glass bottles with other types of packaging materials (Neto et al. 2013; Point et al. 2012), although this option entails important constraints in terms of consumer acceptance (Lockshin et al. 2009). In addition, in their study of a wide range of wine products, Vázquez-Rowe et al. (2013a) demonstrated a strong correlation

($r^2 = 0.77$) between the weight of the wine glass bottle and the GHG emissions in the bottling and packaging stage of the product. Finally, other packaging elements, some of which have been widely analyzed in LCA studies as applied to wine (i.e. cork stoppers; Rives et al. 2011), appear to have a minimal relative contribution to the overall CF of wine products, as long as glass bottling is maintained.

- *Water.* Water use intensity can be relatively high in the vinification stage due to the cleaning processes of the machinery, barrels, etc. For example, Vázquez-Rowe et al. (2012a) reported the use of 9 L of water per 750-mL bottle of *Ribeiro* white wine.
- *Wastewater.* The wine industry produces a significant amount of wastewater, although this issue has not been highly discussed in available studies. Nevertheless, it should be noted that very few wine farms worldwide have their own wastewater treatment plant. Consequently, further analysis on the role of wastewater arriving from wine farms in wastewater treatment plants should be performed to understand their actual relevance in terms of GHG emissions.
- *Chemical products.* Chemical products used in this stage of the process can be divided into two main blocks: (1) cleaning products, such as detergents, soaps, and other cleaning agents; and (2) products that are integral to the production of wine, such as refrigerant agents used in the barrels or substances used in enology to improve or preserve the quality of the wine, such as sulfites and bentonite (Vázquez-Rowe et al. 2012a).
- *Waste production and management.* Most of the residues that are created in this phase are plant residues derived from harvesting that are discarded prior or during the vinification process. Therefore, 80–85 % of these residues are of organic origin, such as grape pomace (62 %), lees (14 %), stalk (12 %), or dewatered sludge (12 %), according to Ruggieri et al. (2009). Some of these substances, such as grape pomace, are treated as by-products in the wine production process, whereas others (e.g. stalks) are considered residues directly. The remaining nonorganic residues originated throughout the life-cycle are treated as regular nonorganic residues. Most of the latter are generated in the packaging stage (e.g. glass, paper, cardboard, or plastics).
- *Wine aging.* Different wines can be classified by the aging period after the vinification process. The aging process is crucial in a perishable product such as wine, because the complex chemical reactions that occur in terms of sugar, acid, and phenolic compounds (e.g. tannins) content can have an important influence on quality indicators, such as aroma, taste, or color, and finally determine the price and market niche of a specific bottle. The aging process of wine has been specifically analyzed by a recent publication (see figure 2 in Vázquez-Rowe et al. 2013a), demonstrating a strong correlation between aging time and CF values ($r^2 = 0.83$). The reasons behind increased GHG emissions with more maturation months is related to higher demand of capital goods in some cases (infrastructure demands, electricity consumption), and in others to a higher demand for enologic products added to the wine.

2.5.3 Distribution

The distribution of the final product (i.e. wine bottle) to wholesalers, retailers, and consumers has a crucial role in the GHG emissions of the entire supply chain (Aranda et al. 2005; Gazulla et al. 2010; Pattara et al. 2012; Point et al. 2012; Neto et al. 2013). Although this observation is not exclusive to wine products because other products from the agri-food sector present high overall contributions in this stage of the supply chain (Vázquez-Rowe et al. 2012c; Ziegler et al. 2013), it should be noted that wine products are very commonly exported, at least when medium- and high-quality brands were analyzed. The main producers, such as Spain, Italy, South Africa, Australia, and Chile, export more than 50 % of the national production (OIV 2013). In fact, some studies (Aranda et al. 2005; Gazulla et al. 2010; Neto et al. 2013) have noted that the final CF of wine products increases considerably if they are destined to exportations to other countries. Having said this, and in relation with recent studies that deconstruct the myth around the concept of *food miles* (Weber and Matthews 2008), the transportation mode is usually the key aspect that determines whether the impacts in the distribution phase are high or not (Weber and Matthews 2008; Point et al. 2012; Ziegler et al. 2013). Therefore, as mentioned by Point et al. (2012), marine freighting of wine exports can in many cases imply lower emissions than regional truck freighting or air freighting to remote areas. Consequently, a balanced selection of transport type and target market distances appears to be the main factor that is going to influence the impacts in this stage. Finally, it is important to bear in mind that some wines need to be freighted under controlled conditions in terms of temperature and humidity, which is usually related to an increase in the energy intensity of the transportation.

3 Analysis of CF Results

3.1 Identification of the Main Hot Spots in Terms of CF

In those studies in which the entire production chain has been analyzed from a CF perspective, the viticulture stage has been shown to have with highest relative impact, ranging from 16 to 40 % in the cross-appellation study performed by Vázquez-Rowe et al. (2013a) to more than 50 % in other individual case studies (Vázquez-Rowe et al. 2012a; Neto et al. 2013). Nevertheless, other studies have noted that if preparation of the cultivation land is considered to be an independent stage prior to the viticulture activities, it would be the most considerable contribution to GHG emissions (Benedetto 2013). Having said this, it is important to clarify that in many studies the preparation of the land and the plantation of the vines is implicitly included in the viticulture phase (Vázquez-Rowe et al. 2012a; Villanueva-Rey et al. 2013), while in others the transparency on this matter tends to be low. Therefore, future studies should focus on providing transparent and reproducible stage delimitations in the agricultural phase to provide clarity to stakeholders and other LCA practitioners.

GHG emissions in the viticulture stage vary considerably between wine regions. This is mainly linked to climatic characteristics and vine needs in terms of protection agents and fertilizers. More specifically, a study conducted by Vázquez-Rowe et al. (2013a) observed that appellations in Northwestern Spain were presenting higher GHG-linked impacts than appellations in other, dryer, European regions, such as Tuscany, Sardinia, and Luxembourg (see Table 2). One of the main observations was that this increase in emissions was partially related to a higher level of interventions in the fields, due to an increased need to add pesticides through the year. In fact, CF values in Northwestern Spain are in accordance with those observed in other Atlantic regions, such as North Portugal (Neto et al. 2013) and Nova Scotia (Point et al. 2012), with closer climatic conditions.

In the postharvesting stages, bottling is the main contributor to the CF of wine production, due to the high energy intensity in the production of glass (Aranda et al. 2005; Bosco et al. 2011; Point et al. 2012; Vázquez-Rowe et al. 2012a; Benedetto 2013). Figure 6 presents a schematic representation of the main sources of GHG emissions that can be found throughout the wine production chain, divided into direct emissions (emissions that are actually linked to the operations themselves) and indirect emissions (related to the embedded emissions in the background processes of the production of elements needed in this production chain).

3.2 Improvement Actions

A set of improvement actions are usually recommended in the available literature to reduce the carbon-related emissions in the production and supply chain of wine products. Obviously, these improvement actions are aimed at tackling those processes, summarized in Fig. 6, that imply highest relative contributions. Hence, in the viticulture stage, these actions are linked to reducing diesel consumption in the interventions (without endangering the quality of the grapes), thus requiring more labor in the vineyards. In addition, the use of more efficient motors and machinery, as well as changes in the dimensions of the machinery in order to adapt to the size of the fields, are also important methods to reduce GHG emissions in this stage.

A reduction in the amount of fertilizers used (organic and mineral) and the amount of plant protection agents can also contribute to an important reduction in emissions. Current viticulture practices are known to be highly dependent on the use of fertilizers, but adequate dosage controls and more accurate dose timing (avoiding nutrient runoff) would allow a substantial reduction of the amounts needed and of the on-field emissions. Another possibility is to reduce the dependence on mineral fertilizers to take advantage of the organic residues generated in the postharvesting processes (Ruggieri et al. 2009). Pruning residues are also an interesting source of organic matter for the soil, as well as dead vine stumps

Table 2 Comparison of key methodological issues, results, and conclusions for a selection of studies in the wine production sector

Study	System boundaries	Functional unit (FU)	Carbon footprint[a]	Business-to-business carbon footprint[b]	Environmental hot spots	Key conclusion
Ardente et al. (2006)	Viticulture–Winery–Distribution	1 bottle of wine (750 mL)	1.60	–	Glass bottle, transport, fertilizers and pesticides	Interorganizational environmental management for the production processes
Gazulla et al. (2010)	Viticulture–Winery–Distribution	1 bottle of wine (750 mL)	1.01–1.02	0.93	Fertilizers and glass bottle	Measure of the economic return on environmental investment
Bosco et al. (2011)	Viticulture–Winery–Distribution–Waste management	1 bottle of wine (750 mL	0.63–1.28	0.52–1.00	Fertilizers, pesticides, and glass bottle	Use of fertilizers, N_2O emissions, and grapes yield
Pattara et al. (2012)	Viticulture–Winery–Distribution	1 bottle of wine (750 mL)	1.29	1.20	Glass bottle	Need for further studies to validate a CF calculator tool
Point et al. (2012)	Viticulture–Winery–Distribution–Consumption	1 bottle of wine (750 mL)	3.22	1.61	Consumer shopping trip, diesel, and glass bottle	Choices in food production and consumption vary environmental burdens
Vázquez-Rowe et al. (2012a)	Viticulture–Winery	1 bottle of wine (750 mL)	2.64–3.21	2.64–3.21	Organic fertilizer, diesel, and glass bottle	Impact depends on harvest year
Benedetto et al. (2013)	Viticulture–Winery	1 bottle of wine (750 mL)	1.64	1.64	Diesel and glass bottle	A switch to lighter bottles and biodiesel
Neto et al. (2013)	Viticulture–Winery–Distribution	1 bottle of wine (750 mL)	2.91	2.68	Diesel, fertilizers and glass bottle	Optimization the dosage of fertilizers

[a] CF reported in kg CO_2-eq/FU. This value refers to the entire life-cycle that was identified in the original publications
[b] CF with a business-to-business approach, including greenhouse gas emissions up to the gate of the winery

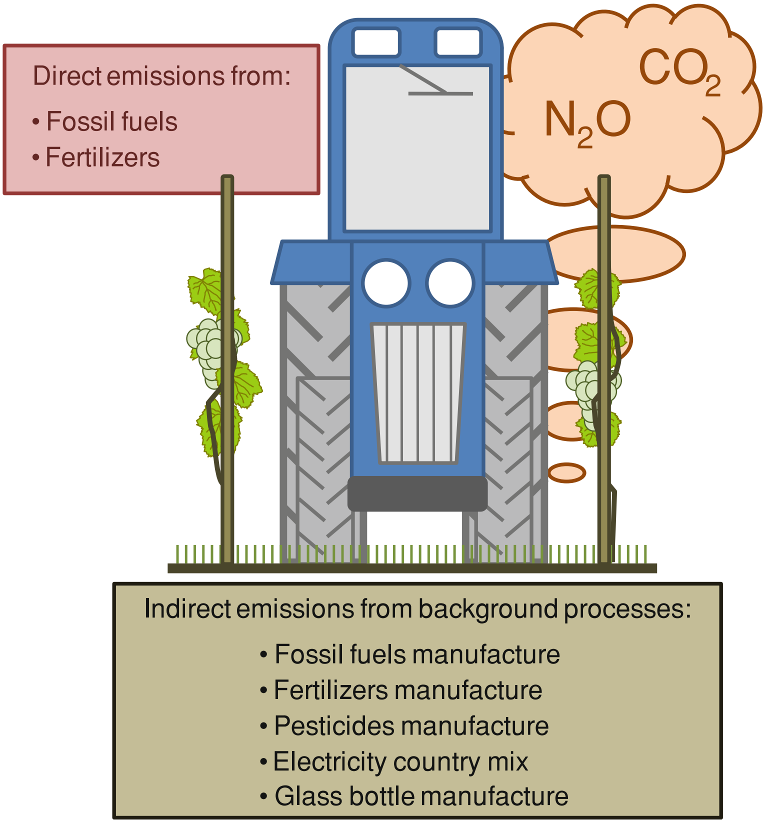

Fig. 6 Main sources of environmental impact observed in wine CF studies

(unless these stumps suffered from diseases). Finally, the way in which the fertilizer is applied can condition the final emissions. For instance, drip irrigation can allow a reduction in direct emissions after application, but it can also lower diesel consumption because the machinery interventions are reduced (García et al. 2012).

The plantation of cover crops in vineyards has been shown to improve the characteristics of the soil in terms of porosity, structure, and texture; it can increase the organic matter content and reduce erosion processes or the runoff of nitrates, therefore fixating nitrogen to the soil. However, they imply an increase in interventions to control their growth (Nicholls et al. 2001; Gómez et al. 2011).

Improvement actions when using pesticides are limited to respecting the security periods indicated by the producers, an exhaustive control of potential diseases through daily monitoring and adequate dosage. Only through the combination of these actions will there be an actual reduction in the use of pesticides and, therefore, in the number of machinery interventions.

The main improvements highlighted in the literature for the vinification stage are related to changes in the energy carrier or minimization of electricity consumption. One option that has been implemented already in some wine farms across Europe is the inclusion of renewable energy infrastructure within the farm, such as profiting from fallow land or the roof of the farm to install solar panels (Smyth and Russell 2009).

The bottling stage has been identified as the main carrier of the GHG emissions in the postharvesting production chain (Rugani et al. 2013). Therefore, existing studies (Aranda et al. 2005; Point et al. 2012; Vázquez-Rowe et al. 2012a) indicate the necessity to reduce the average weight of wine bottles or change the packaging material of the container. However, it should be noted that a reduction in the weight of glass bottles is limited in many cases to legislative constraints, and there is obviously a minimum threshold beyond which the bottle could imply risks during freighting or for the consumer. In fact, this situation is especially important in sparkling wine bottles, whose weight is substantially higher than for other types of wine (Vázquez-Rowe et al. 2013a). Vázquez-Rowe et al. (2012a) examined the difference in GHG emissions between green and white glass bottles used for bottling in a winery in Northwestern Spain. Despite green bottles showing a lower CF, the difference between the two was too low to be considered significant.

Changes in the packaging material used as a container for the wine have been proposed in many LCA and CF studies. However, it should be noted that substituting glass bottles by other materials, such as polyethylene terephthalate, may be a constraint in terms of consumer and stakeholder acceptance, despite the important reduction in CF (Lockshin et al. 2009). In fact, Lockshin et al. (2009) concluded that materials other than glass may trigger the perception that the purchased wine is of lower quality. Furthermore, Ghidossi et al. (2012) pointed out the fact that the use of materials other than glass may alter the quality and/or the organoleptic characteristics of wine. Therefore, given the market and acceptance constraints that changes in the packaging materials may involve, ecodesign appears to be an important option to involve scientists, stakeholders and consumers in common actions to attain compromises regarding future changes in strategy (González-García et al. 2011).

Improvement actions during distribution will be mostly linked to the use efficient transport modes, the optimization of the loading capacity of trucks and other freighting alternatives, and the selection of slow transportation options (marine freighting rather than air freighting). In addition, Aranda et al. (2005) recommended that the bottling and packaging of the wine should be done at destination rather than at the winery. However, this scheme would go against the standards and regulations approved by most European appellations. In contrast, certain wine farms from Australia, New Zealand, or South Africa have started freighting their wine in big containers, realizing the bottling stage at the destination (Cimino and Marcelloni 2012).

Finally, even though the postdistribution stages of the wine supply chain have not been the focus of this chapter, it is important to mention that the way in which consumers, wholesalers, and retailers behave when handling the product can imply strong variations in the final CF of the wine product. Although these variations have not been the focus of any wine CF study, consumer scenarios have proven to be relevant in other food and beverage products (Vázquez-Rowe et al. 2013d, e, 2014). For example, a high shelf time of the products at the retailer or the refrigeration of white wine during the entire supply chain may contribute to increase the CF of wine products considerably. In a similar way, losses of the product at the consumer stage have been quantified by Kounina et al. (2012), ranging from approximately 2 to 5 %.

3.3 The Role of Harvest Yield on Final CF Results

Beyond the main product derived from winemaking (i.e. wine), there are multiple by-products exiting the production system. Therefore, in many cases (always depending on the system function), it is necessary to allocate the co-products (Notarnicola et al. 2003). However, allocation, to date, has not been discussed in detail in most wine CF studies because the entire burden has been usually assigned to the main product (i.e. wine production). In other cases, the assessment was limited to the preharvesting phases (Vázquez-Rowe et al. 2012b) and, therefore, no allocation was necessary.

In their study, Gazulla et al. (2010) allocated the environmental burdens to the different co-products based on their economic value. This allocation strategy derived a 98 % allocation for the wine bottle, while the other co-products had an irrelevant role in the overall system. A study by Bosco et al. (2011) chose a mass allocation to allot the GHG emissions between the wine and stalk, skin, and pip products. This implied that wine CF impacts were approximately 25–30 % lower than if 100 % of the impacts were allocated entirely to the wine product. Finally, other studies, such as Notarnicola et al. (2003), excluded the treatment of byproducts (i.e. compost from rasps or tartaric acid from marc) from the system boundaries.

According to a review by Rugani et al. (2013), the use of allocation in wine CF has been understudied due to the lack of adequate information to allow an expansion of the system boundaries. In fact, given the lack of information in many cases regarding the alternative production systems for byproducts such as pomace, lees, or press syrup, system expansions were not considered to be feasible in these processes (Gazulla et al. 2010).

The natural variability in terms of productivity of agricultural products leads to different harvest yields on an interannual basis. Furthermore, this variability was shown to have an important influence on LCA and CF studies with a product-oriented function and FU (Ramos et al. 2011). As shown in a set of publications (Moreira et al. 2011; Vázquez-Rowe et al. 2012a; Villanueva-Rey et al. 2013), harvest yield can have an important effect on the final CF values in the wine sector, especially considering that the preharvesting stages of grapes can constitute over 50 % of the GHG emissions of the entire production chain, as mentioned in

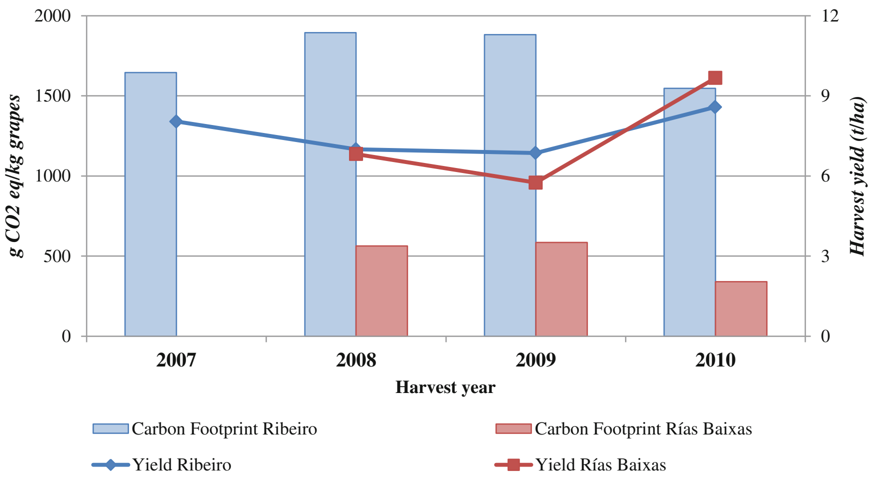

Fig. 7 Interannual changes in harvest yield and CF for the viticulture stage of two wine appellations in Northwestern Spain (adapted from Moreira et al. 2011 and Vázquez-Rowe et al. 2012a)

Sect. 3.1. Vázquez-Rowe et al. (2012a), after analyzing a wine farm in Northwestern Spain, identified changes in GHG emissions as high as 10 % on an annual basis, mainly from changes in harvest yield but also to different plant protection agent interventions from year to year due to climatology. Nevertheless, it should be noted that when the preharvesting stages are examined independently, interannual variation can be as high as 25–30 %. Figure 7 shows the changes in harvest yield and CF value for two studies conducted in Northwestern Spain in the period 2007–2010 (viticulture stage only).

Interannual changes in the postharvesting stages of the production chain are more difficult to detect from a product-oriented perspective because variations in operational inputs will be strongly correlated to the amount of grapes entering the wine farm in a given year, decreasing or increasing based on the current availability of the raw product. Nevertheless, studies in which the function of the system was to analyze the overall GHG emissions of the wine farms would have to consider this important factor. Finally, it should be mentioned that the minor interannual changes observed in these stages have been reported to be linked to changes in the electricity mix of a country from year to year (Vázquez-Rowe et al. 2012a). However, strong changes in the energy carrier in this stage (e.g. inclusion of photovoltaic panels in the wine farm) could potentially create relevant GHG emissions variability.

4 Challenges and Future Perspectives in the Use of CF in the Wine Sector

4.1 The Inclusion of Biogenic CO_2 in CF Estimations

The inclusion of biogenic CO_2 is quite a discussed and controversial topic in CF studies. In fact, the viticulture and winemaking stages present photosynthesis processes that capture CO_2, while the fermentation processes in the vinification phase emit CO_2. In addition, microbial activity in the soil of the vineyards can have a relative impact on the final fixation or emission of CO_2. To date, most studies have excluded CO_2 fixation in vineyards (Notarnicola et al. 2010; Pattara et al. 2012; Bosco et al. 2011; Neto et al. 2013; Point et al. 2012; Vázquez-Rowe et al. 2012a, b). In the vinification stage, studies such as Neto et al. (2013) include the emissions derived from the fermentation processes, while the majority of studies still exclude these biogenic emissions (Bosco et al. 2011; Pattara et al. 2012; Point et al. 2012; Vázquez-Rowe et al. 2012a). Rugani et al. (2013) estimated, based on a joint aggregation of available wine CF studies, that biogenic CO_2 was important in the vinification stage, representing approximately 20 % of the total emissions. However, this relative CF value is down to 2–3 % if the entire production chain is considered (Colman and Päster 2009; Neto et al. 2013).

In addition, it is also worth highlighting the fact that, until recently, most GHG emissions standards excluded biogenic CO_2 emissions from the calculation. However, the GHG Protocol Product Standard and the revised version of PAS 2050 have started to consider these emissions (BSI 2011). Nevertheless, it remains clear that further research must be done, especially in the viticulture stage, to understand the importance of biogenic CO_2 emissions in the overall impact of the wine production process (Rugani et al. 2013; Villanueva-Rey et al. 2013).

4.2 Climate Change

Viticulture, in the same way as many other agricultural crops, does not escape the consequences of global warming, a phenomenon believed to be responsible for increasing extreme climatic events in recent years (Tate 2001; Jones et al. 2005; Mira de Orduña 2010). Therefore, it is expected that changes in rainfall and increases in mean temperature will have important effects on wine regions and appellations (Team et al. 2007). A broad range of studies have analyzed the expected consequences that climate change will have on grape production and, therefore, on the characteristics and quality of the wine products. Mira de Orduña (2010), for instance, observed changes in grape composition and in vine phenology, which will eventually affect the vinification stage because enologists will have to correct these changes. Nevertheless, despite the fact that wine production is very sensitive to global warming, it is expected that many vineyards will be relocated within the same wine region in order to maintain the basic characteristics of the wine (Anderson et al. 2008). Other actions, such as changes in the trellis system or in the timing or style of certain operational activities (e.g. pruning, irrigation management, row orientation) will also be enforced to adapt to changes without having to relocate vineyards.

Despite this brief summary of the threats of climate change on viticulture and winemaking, it is important to bear in mind that there are numerous outcomes to this ongoing process that are expected to continue for the next century. For instance, global warming may imply an opportunity within the wine market for wineries that decide to undertake organic and biodynamic practices as a way to enhance their social responsibility and gain market quota. However, a series of changes in the market beyond the individual wine market may be expected, such as reductions in wine consumption due to increases in beverages that are usually consumed with higher ambient temperatures, such as beer (Lenten and Mossa 1999). Consequently, as will be discussed in Sect. 4.3, CF and more widely LCA can be very useful methodologies to monitor the environmental consequences of changes in the wine sector system through the application of a consequential perspective rather than the commonly used attributional approach.

4.3 The Use of a Consequential Life-cycle Perspective

As mentioned, the use of an attributional perspective in wine CF can be extremely useful for a wide range of utilities and applications. However, when it comes to making a series of projection in terms of environmental sustainability, and despite the usefulness of future scenario modelling in attributional CF studies, a consequential approach would allow linking a series of factors that are exogenous to the production system, but ultimately affect it, to monitor the environmental consequences due to changes in the production system (Weidema 2003; UNEP 2011). Therefore, this approach would target policy and strategic decisions, rather than environmental accounting and communication, by assessing the environmental consequences of changes affecting a particular production system at the meso- or macro-scale (Vázquez-Rowe et al. 2013c).

The application of a consequential perspective in wine CF therefore would be useful mainly at high decisional levels, such as the European Union's Common Agricultural Policy, to understand the environmental consequences of changes in the overall surface dedicated to vineyards or in domestic production through time, guiding policy decisions. Other possible applications include the macro-scale calculation of environmental consequences linked to the inclusion of new marketable products in the wine sector; the equilibria between wine products and other products that compete in the same market; the effects of expanding New World wines on the consumption patterns on a local, national, or international level; and the analysis of the effects that limiting factors or agents, such as policies or climate change, may have on appellations and wine regions through time.

Consequential CF and/or LCA is yet to be applied to the wine sector, and its use within other agri-food products is very limited, usually linked to the inclusion and environmental effects of bioenergy products competing with food production sites (Vázquez-Rowe et al. 2013c). Therefore, it remains to be seen which will be the specific structure of consequential studies as applied to the wine sector in terms of system delimitation or modelling of the expected changes (Marvuglia et al. 2013).

4.4 Rebound Effects

Rebound effects (REs) are defined as economic activities that generally appear (but in some cases may cease) due to an increase in production efficiency (Hertwich 2005; Sorrell and Dimitropoulos 2008). When analyzing wine production systems and other industrial ecology systems, CF practitioners tend to propose a series of improvement actions to reduce the carbon emissions of the specific process under study. These reductions, as mentioned in Sect. 3.2, are usually attained through two main mechanisms: (i) advocating for an optimization of operational inputs by maximizing efficiency in terms of, for example, pesticide or fertilizer use; or (ii) using more efficient machinery and processes. However, wine CF studies have never reflected on the fact that a reduction in either of these two mechanisms usually implies financial changes in the wineries, either through a reduction of costs due to the optimization of operational inputs or through an increase in costs due to the new machinery acquired. These changes in revenues influence the capacity of the winery to provide new investments or to delve into new market strategies, such as increasing or lowering the consumer's purchase price of the wine bottles.

Consequently, the intensity of REs may determine the actual final capacity of an improvement action to produce a final decrease in GHG emissions. In fact, ignoring the effects or the existence of REs throughout the wine life-cycle may lead to an overestimation or underestimation of the real effects that novel sustainable technologies actually generate. Hence, it is important to identify how the markets react to these new improvements throughout the production and supply chain, as well as how stakeholders and consumers may react to the appearance of new wine products in the market (Benedetto et al. 2014). Although REs are yet to be evaluated in wine CF studies, a consequential approach appears to be the most appropriate one to grasp the market equilibria that guide and engender REs (Benedetto et al. 2014).

5 Conclusions

This chapter has provided an in-depth analysis on the current state-of-the-art of CF in the wine sector. A detailed evaluation of a set of methodological assumptions observed in previous studies served as a start to delve into the current milestones that have been achieved in this field of research and how these have helped to answer specific research questions. Finally, based on the state-of-the-art, future challenges in the use and further potential for wine CF were examined and discussed.

Despite the strong proliferation of wine CF studies in recent years, most studies have limited the scope to environmental communication to stakeholders and consumers, as well as the analysis of possible improvement actions to lower the carbon dependence of the wine industry. However, a top-down perspective, in which appellations and/or wine regions are examined as a whole, that incorporates a more complex modelling perspective, such as a consequential approach or REs, is still lacking in wine CFs. Such an approach will definitely be necessary for studies to upgrade their utility in terms of policy-making and strategic guidance to appellations and wineries.

References

ADEME (2010) Guide méthodologique version 6.1 du Bilan Carbone—Objectifs et principes de comptabilisation

Anderson K, Findlay C, Fuentes S, Tyerman S (2008) Viticulture, wine and climate change. Garnaut climate change review. University of Adelaide, Australia. http://www.garnautreview.org.au/ca25734e0016a131/WebObj/01-HViticulture/$File/01-H%20Viticulture.pdf. Accessed on December 2012

Aranda A, Zabalza I, Scarpellini S (2005) Economic and environmental analysis of the wine bottle production in Spain by means of life cycle assessment. Int J Agric Res Gov Ecol 4:178–191. doi:10.1504/IJARGE.2005.007199

Ardente F, Beccali G, Cellura M, Marvuglia A (2006) POEMS: a case study of an Italian wine-producing firm. Environ Manage 38:350–364

Barry MT (2011) Life cycle assessment and the New Zealand wine industry: a tool to support continuous environmental improvement. M.Sc. Dissertation. Massey University, Wellington, NZ

Benedetto G (2013) The environmental impact of a Sardinian wine by partial life cycle assessment. Wine Econ Policy 2:33–41. doi:10.1016/j.wep.2013.05.003

Benedetto G, Rugani B, Vázquez-Rowe I (2014) Rebound effects due to economic choices when assessing the environmental sustainability of wine. Food Policy. Under review

Bosco S, Di Bene C, Galli M et al (2011) Greenhouse gas emissions in the agricultural phase of wine production in the Maremma rural district in Tuscany, Italy. Ital J Agron 6:93–100. doi:10.4081/ija.2011.e15

Brentrup F, Küsters J, Lammel J, Kuhlmann H (2000) Methods to estimate on-field nitrogen emissions from crop production as an input to LCA studies in the agricultural sector. Int J Life Cycle Assess 5:349–357. doi:10.1007/BF02978670

BSI (2011) PAS 2050: 2011. Specification for the assessment of life cycle greenhouse gas emissions of goods and services. British Standards Institution

Carballo-Penela A, García-Negro MC, Doménech Quesada JL (2009) A methodological proposal for corporate carbon footprint and its application to a wine-producing company in Galicia, Spain. Sustainability 1:302–318

Cimino M, Marcelloni F (2012) Enabling traceability in the wine supply chain. In: Anastasi G, Bellini E, Di Nitto E, Ghezzi C, Tanca L, Zimeo E (eds) Methodologies and technologies for networked enterprises, vol 7200, pp 397–412. doi:10.1007/978-3-642-31739-2_20

Colman T, Päster P (2009) Red, white and "green": the cost of carbon in the global wine trade. J Wine Resour 20:15–26. doi:10.1080/09571260902978493

Comandaru IM, Bârjoveanu G, Peiu N, Ene SA, Teodosiu C (2012) Life cycle assessment of wine: focus on water use impact assessment. Environ Eng Manag J 11:533–543

ecoinvent® (2013) An introduction to the new features and data. www.ecoinvent.org

EMEP-Corinair (2006) EMEP/EEA air pollutant emission inventory guidebook, 2006. EEA

EMEP-Corinair (2009) EMEP/EEA air pollutant emission inventory guidebook, 2009. EEA

European Commission (2007) Council regulation on organic production and labelling of organic products and repealing regulation (EEC) No 2092/91 (EC) No 834/2007 of 28 June

European Commission (2008) Council regulation laying down detailed rule for the Implementation of Council Regulation (EC) No 834/2007 on organic production and labelling of organic products with regard to organic production, labelling and control (EC) No 889/2008 of 5 September

European Commission (2012) Amending regulation (EC) No 889/2008 laying down detailed rules for the implementation of council regulation (EC) No 834/2007, as regards detailed rules on organic wine. (EC) No 203/2012 of 8 March

Frischknecht R, Jungbluth N, Althaus HJ, et al (2007) Implementation of life cycle impact assessment methods. ecoinvent® report No. 3, v2.0. Swiss Centre for Life Cycle Inventories, Dübendorf

Gabzdylova B, Raffensperger JF, Castka P (2009) Sustainability in the New Zealand wine industry: drivers, stakeholders and practices. J Clean Prod 17:992–998. doi:10.1016/j.jclepro.2009.02.015

García García J, Martínez-Cutillas A, Romero P (2012) Financial analysis of wine grape production using regulated deficit irrigation and partial-root zone drying strategies. Irrigation Sci 30:179–188. doi:10.1007/s00271-011-0274-4

Gazulla C, Raugei M, Fullana-i-Palmer P (2010) Taking a life cycle look at crianza wine production in Spain: where are the bottlenecks? Int J Life Cycle Assess 15:330–337. doi:10.1007/s11367-010-0173-6

Ghidossi R, Poupot C, Thibon C et al (2012) The influence of packaging on wine conservation. Food Control 23:302–311. doi:10.1016/j.foodcont.2011.06.003

Gómez JA, Llewellyn C, Basch G, Sutton PB, Dyson JS, Jones CA (2011) The effects of cover crops and conventional tillage on soil and runoff loss in vineyards and olive groves in several Mediterranean countries. Soil Use Manage 27:504–512. doi:10.1111/j.1475-2743.2011.00367.x

Gonzales A, Klimchuk A, Martin M (2006) Life cycle assessment of wine production process. Finding relevant process efficiency and comparison to eco-wine production. Royal Institue of Technology, Stockholm

González-García S, Silva FJ, Moreira MT et al (2011) Combined application of LCA and eco-design for the sustainable production of wood boxes for wine bottles storage. Int J Life Cycle Assess 16:224–237. doi:10.1007/s11367-011-0261-2

Hertwich G (2005) Consumption and the rebound effect. An industrial ecology perspective. J Ind Ecol 9:85–98. doi:10.1162/1088198054084635

Initiative GGP (2011) Product life cycle accounting and reporting standard. 150

IPCC (2006) IPCC guidelines for national greenhouse gas inventories, vol 4: Agriculture, forestry and other land use

IPCC (2007) Climate change 2007—the physical science basis. Contribution of Working Group I to the 4th IPCC Assessment Report

ISO (2006a) ISO 14040. Environmental management—life cycle assessment—principles and framework. International Organization for Standardization

ISO (2006b) ISO 14044. Environmental management—life cycle assessment—requirements and guidelines. International Organization for Standardization

ISO (2013) ISO 14067—Carbon footprint of products—requirements and guidelines for quantification and communication. http://www.iso.org/iso/iso_catalogue/catalogue_tc/catalogue_detail.htm?csnumber=59521

Jones GV, White MA, Cooper OR, Storchmann K (2005) Climate change and global wine quality. Clim Change 73:319–343. doi:10.1007/s10584-005-4704-2

Kavargiris SE, Mamolos AP, Tsatsarelis CA et al (2009) Energy resources' utilization in organic and conventional vineyards: energy flow, greenhouse gas emissions and biofuel production. Biomass Bioenerg 33:1239–1250. doi:10.1016/j.biombioe.2009.05.006

Kounina A, Tatti E, Humbert S, Pfister R, Pike A, Ménard JF, Loerincik Y, Jolliet O (2012) The importance of considering product loss rates in life cycle assessment: the example of closure systems for bottled wine. Sustainability 4:2673–2706. doi:10.3390/su4102673

Laurent A, Olsen SI, Hauschild MZ (2012) Limitations of carbon footprint as indicator of environmental sustainability. Environ Sci Technol 46:4100–4108. doi:10.1021/es204163f

Lenten LJA, Mossa I (1999) Modelling the trend and seasonality in the consumption of alcoholic beverages in the United Kingdom. Appl Econ 31:795–804. doi:10.1080/000368499323760

Lockshin L, Mueller S, Louviere J et al (2009) Development of a new method to measure how consumers choose wine. Wine Ind J 24:37–43

Lotter DW (2003) Organic agriculture. J Sustain Agr 21:59–128. doi:10.1300/J064v21n04_06

Marvuglia A, Benetto E, Rege S, Jury C (2013) Modelling approaches for consequential life cycle assessment (C-LCA) of bioenergy: critical review and proposed framework for biogas production. Renew Sust Energ Rev 25:768–781. doi:10.1016/j.rser.2013.04.031

Masson P (2009) Biodinámica: guía práctica para agricultores y aficionados. Editorial Fertilidad de la Tierra, Estella, Navarra

McGovern PE, Fleming SJ, Katz SH (1996) The origins and ancient history of wine. Gordon and Breach Publishers, Amsterdam

Mira de Orduña R (2010) Climate change associated effects on grape and wine quality and production. Food Res Int 43:1844–1855. doi:10.1016/j.foodres.2010.05.001

Moreira MT, Vázquez-Rowe I, Villanueva-Rey P, Feijoo G (2011) The importance of timeline analysis in viticulture. A case study based on Rías Baixas production area (NW Spain). Proceeding from LCA XI Instruments for Green Futures Markets

Nemecek T, Kägi T (2007) Life cycle inventories of agricultural production systems. ecoinvent® report No. 15, 353 pages

Neto B, Dias AC, Machado M (2013) Life cycle assessment of the supply chain of a Portuguese wine: from viticulture to distribution. Int J Life Cycle Assess 1–13. doi:10.1007/s11367-012-0518-4

Niccolucci V, Galli A, Kitzes J et al (2008) Ecological Footprint analysis applied to the production of two Italian wines. Agr Ecosyst Environ 128:162–166. doi:10.1016/j.agee.2008.05.015

Nicholls CI, Parrella MP, Altieri MA (2001) Reducing the abundance of leafhoppers and thrips in a Northern California organic vineyard through maintenance of full season floral diversity with summer cover crops. Agr Forest Entomol 2:107–113. doi:10.1046/j.1461-9563.2000.00054.x

Notarnicola B, Tassielli G, Nicoletti M (2003) LCA of wine production. In: Mattsonn B, Sonesson U (eds) Environmentally-friendly food processing. Woodhead Publishing Ltd., Cambridge, pp 306–326

Notarnicola B, Tassielli G, Settanni E (2010) Including more technology in the production of a quality wine: the importance of functional unit. In: Proceedings of the 7th international conference on LCA in the agri-food sector, vol 1, pp 235–240

OIV (2011) General principles of the OIV greenhouse gas accounting protocol (GHGAP) for the vine and wine sector. Resolution OIV-CST 431-2011. General Assembly of Member States, Montpellier, France

OIV (2013) International Organisation of Vine and Wine. Statistical report on world vitiviniculture

Pattara C, Raggi A, Cichelli A (2012) Life cycle assessment and carbon footprint in the wine supply-chain. Environ Manage 49:1247–1258. doi:10.1007/s00267-012-9844-3

Petti L, Ardente F, Bosco S, De Camillis C, Masotti P, Pattara C, Raggi A, Tassielli G (2010) State of the art of life cycle assessment (LCA) in the wine industry. In: Proceedings of 7th

international conference on life cycle assessment in the agri-food sector, Bari, Italy, pp 493–498
Pizzigallo ACI, Granai C, Borsa S (2008) The joint use of LCA and emergy evaluation for the analysis of two Italian wine farms. J Environ Manage 86:396–406. doi:10.1016/j.jenvman.2006.04.020
Point E, Tyedmers P, Naugler C (2012) Life cycle environmental impacts of wine production and consumption in Nova Scotia, Canada. J Clean Prod 27:11–20. doi:10.1016/j.jclepro.2011.12.035
Ramos S, Vázquez-Rowe I, Artetxe I et al (2011) Environmental assessment of the Atlantic mackerel (Scomber scombrus) season in the Basque Country. Increasing the timeline delimitation in fishery LCA studies. Int J Life Cycle Assess 16:599–610. doi:10.1007/s11367-011-0304-8
Ridoutt B, Eady S, Sellahewa J, Simons L, Bektash R (2009) Water footprinting at the product brand level: case study and future challenges. J Clean Prod 17:1228–1235. doi:10.1016/j.clepro.2009.03.002
Rives J, Fernández-Rodríguez I, Rieradevall J, Gabarrell X (2011) Environmental analysis of the production of natural cork stoppers in Southern Europe (Catalonia e Spain). J Clean Prod 19:259–271. doi:10.1016/j.clepro/2010.01.007
Roux EG, Peisach M, Pineda CA, Pougnet MAB (1988) The toxic effect of aluminium in vines. J Radioanal Nucl Ch Ar 120:97–104. doi:10.1007/BF02037855
Rugani B, Vázquez-Rowe I, Benedetto G, Benetto E (2013) A comprehensive review of carbon footprint analysis as an extended environmental indicator in the wine sector. J Clean Prod 54:61–77. doi:10.1016/j.jclepro.2013.04.036
Ruggieri L, Cadena E, Martínez-Blanco J et al (2009) Recovery of organic wastes in the Spanish wine industry. Technical, economic and environmental analyses of the composting process. J Cle Prod 17:830–838. doi:10.1016/j.jclepro.2008.12.005
Smyth M, Russell J (2009) From graft to bottle—analysis of energy use in viticulture and wine production and the potential for solar renewable technologies. Renew Sust Energ Rev 13:1985–1993. doi:10.1016/j.rser/2009.01.007
Sorrell S, Dimitropoulos J (2008) The rebound effect: microeconomic definitions, limitations and extensions. Ecol Econ 65:636–649. doi:10.1016/j.ecolecon/2007.08.013
Tate AB (2001) Global warming's impact on wine. J Wine Res 12:95–109. doi:10.1080/09571260120095012
Team CW, Pachauri RK, Reisinger A (2007) IPCC, 2007: Climate change 2007: synthesis report. Contribution of Working Groups I, II and III to the 4th Assessment Report of the Intergovernmental Panel on Climate Change. p 104
Udo de Haes HA (2006) Life-cycle assessment and the use of broad indicators. J Ind Ecol 10:5–7. doi:10.1162/jiec.2006.10.3.5
UNEP (2011) Global guidance principles for life cycle assessment databases a basis for greener processes and products. Shonan guidance principles. Life Cycle Initiative, SETAC and United Nations Environment Programme
Vázquez-Rowe I, Moreira MT, Feijoo G (2012a) Environmental assessment of frozen common octopus (*Octopus vulgaris*) captured by Spanish fishing vessels in the Mauritanian EEZ. Marine Policy 36:180–188. doi:10.1016/j.marpol.2011.05.002
Vázquez-Rowe I, Villanueva-Rey P, Iribarren D et al (2012b) Joint life cycle assessment and data envelopment analysis of grape production for vinification in the Rías Baixas appellation (NW Spain). J Clean Prod 27:92–102. doi:10.1016/j.jclepro.2011.12.039
Vázquez-Rowe I, Villanueva-Rey P, Moreira MT, Feijoo G (2012c) Environmental analysis of Ribeiro wine from a timeline perspective: harvest year matters when reporting environmental impacts. J Environ Manage 98:73–83. doi:10.1016/j.jenvman.2011.12.009
Vázquez-Rowe I, Rugani B, Benetto E (2013a) Tapping carbon footprint variations in the European wine sector. J Cleaner Prod 43:146–155. doi:10.1016/j.jclepro.2012.12.036

Vázquez-Rowe I, Marvuglia A, Rege S, Benetto E (2013b) Applying consequential LCA to support energy policy: land use change effects of bioenergy production. Sci Total Environ. doi:10.1016/j.scitotenv.2013.10.097

Vázquez-Rowe I, Rege S, Marvuglia A, Thénie J, Haurie A, Benetto E (2013c) Application of three independent consequential LCA approaches to the agricultural sector in Luxembourg. Int J Life Cycle Assess 18:1593–1604. doi:10.1007/s11367-013-0604-2

Vázquez-Rowe I, Villanueva-Rey P, Moreira MT, Feijoo G (2013d) The role of consumer purchase and post-purchase decision-making in sustainable seafood consumption. A Spanish case study using carbon footprinting. Food Policy 41:94–102. doi:10.1016/j.foodpol.2013.04.009

Vázquez-Rowe I, Villanueva-Rey P, Mallo J, De la Cerda JJ, Moreira MT, Feijoo G (2013e) Carbon footprint of a multi-ingredient seafood product from a business-to business perspective. J Clean Prod 44:200–210. doi:10.1016/j.jclepro.2012.11.049

Vázquez-Rowe I, Villanueva-Rey P, Hospido A, Moreira MT, Feijoo G (2014) Life cycle assessment of European pilchard (*Sardina pilchardus*) consumption. A case study for Galicia (NW Spain). Science of the Total Environment. Accepted for publication

Venkat K (2012) Comparison of twelve organic and conventional farming systems: a life cycle greenhouse gas emissions perspective. J Sust Agr 36:620–649. doi:10.1080/10440046.2012.672378

Villanueva-Rey P, Vázquez-Rowe I, Moreira MT, Feijoo G (2013) Comparative life cycle assessment in the wine sector: biodynamic vs. conventional viticulture activities in NW Spain. J Clean Prod. doi:10.1016/j.jclepro.2013.08.026

Weber CL, Matthews HS (2008) Food-miles and the relative climate impacts of food choices in the United States. Environ Sci Technol 42:3508–3513. doi:10.1021/es702969f

Weidema BP (2003) Market information in life cycle assessment. Environmental Project No. 863. Danish Environmental Protection Agency, Copenhagen, DK. http://www2.mst.dk/udgiv/publications/2003/87-7972-991-6/pdf/87-7972-992-4.pdf. Accessed on December 2012

Weidema BP, Thrane M, Christensen P, Schmidt J, Løkke S (2008) Carbon footprint—a catalyst for life cycle assessment? J Ind Ecol 12:3–6. doi:10.1111/j.1530-9290.2008.00005.x

Ziegler F, Winther U, Skontorp-Hognes E, Emanuelsson A, Sund V, Ellingsen H (2013) The carbon footprint of Norwegian seafood products. J Ind Ecol 17:103–116. doi:10.1111/j.1530-9290.2012.00485.x